Integrating Ecosystem Services in Land Use Planning: Concepts and Applications

Davide Geneletti

CID Research Fellow and Graduate Student
Working Paper No. 54
January 2012

Working Papers
Center for International Development
at Harvard University

Integrating Ecosystem Services in Land Use Planning: Concepts and Applications

Davide Geneletti

Abstract

Despite the attention that research on ecosystem services has attracted in recent years, its use to support real-life decision-making processes is still very limited, especially at the planning level. Land use and spatial planning result in actions that may affect the distribution and quality of a wide range of ecosystem services. Hence, decisions in this sector would benefit from systematic considerations of their effects on ecosystem services. This working paper is structured in two parts. Part I aims at providing insights on why it is important to promote ecosystem service-inclusive land use planning processes, and on how to do it by taking advantages of existing procedures, such as Strategic Environmental Assessment (SEA). Possible actions to include relevant information in the various planning and SEA stages are presented. The advantages are then discussed by considering both the characteristics of ecosystem services approaches and the criteria of good-quality planning and SEA. Finally, four main challenges are identified, concerning scoping, scale issues, trade-offs and indicators, respectively. To provide a contribution to meeting these challenges, Part II presents an empirical application in land use planning. Specifically, the study aims at exploring how the implementation of different land use zoning policies can affect the future provision of a set of ecosystem services (water purification, soil conservation, habitat for species, carbon sequestration and timber production). Firstly, land use scenarios associated to different policies were constructed. Subsequently, the effects of the land use scenarios on the provision of the selected ecosystem services were assessed in a spatially explicit way, by using state-of-the-art modeling tools. Finally, a set of metrics was developed to compare scenarios and assess trade-offs. The results illustrate the importance of taking into account the spatial arrangement of land uses produced by different policies. The paper concludes by discussing the potential contribution of the approach to support spatial planning.

Keywords: spatial planning, ecosystem services, trade-offs, land use change, scenario analysis, strategic environmental assessment

JEL subject codes: Q01, Q56, Q57, Q24

Citation, Context, and Program Acknowledgements

This paper may be cited as:

Geneletti, Davide. "Integrating Ecosystem Services in Land Use Planning: Concepts and Applications." CID Research Fellow and Graduate Student Working Paper No. 54. Center for International Development at Harvard University, January 2012.

It is available at http://www.hks.harvard.edu/centers/cid/publications/research-fellow-graduate-student-working-papers/cid-research-fellow-and-graduate-student-working-paper-no.-54.
Professor William Clark has approved this paper for inclusion in the working paper series.
Comments are welcome and may be directed to the author, davide.geneletti@ing.unitn.it.

Davide Geneletti wrote this paper while a Giorgio Ruffolo Research Fellow in the Sustainability Science Program at Harvard University's Center for International Development. He was on leave from the Faculty of Engineering of the University of Trento, where he is a researcher and lecturer. His main fields of expertise are environmental assessment, land use planning, biodiversity conservation and spatial decision analysis. He has contributed to research, cooperation and capacity-building projects in Europe, Latin America, Southern Asia and sub-Saharan Africa. He serves on the editorial board of four scientific journals in the area of environmental assessment and planning. He received his degrees from the University of Milano (MSc in Environmental Sciences) and the Free University of Amsterdam (PhD).

The Sustainability Science Program at Harvard's Center for International Development harnesses the University's strengths to promote the design of institutions, policies, and practices that support sustainable development. The Program addresses the challenge of sustainable development by: advancing scientific understanding of human-environment systems; improving linkages between research and policy communities; and building capacity for linking knowledge with action to promote sustainability. The Program supports major initiatives in policy-relevant research, faculty research, training of students and fellows, teaching, and outreach. See http://www.hks.harvard.edu/centers/cid/programs/sustsci.

Author Acknowledgements

William C. Clark (Harvard University) contributed to the design of the methodology presented in Part II. Matteo Gavagnin helped in running the InVEST models. Fernando Peña (Laboratorio de Planificación Territorial, Universidad Católica de Temuco) provided most of the GIS baseline data, as well as valuable insights on the study region. Finally, Cristian Bonacic and Nicolás Gálvez (Fauna Australis Wildlife Laboratory, Pontifícia Universidad Católica de Chile) kindly made available the wildlife data.

This work was conducted while the author was a Giorgio Ruffolo Fellow in the Sustainability Science Program at Harvard University's Center for International Development. Support of CID and the Italian Ministry for the Environment, Land and Sea is gratefully acknowledged.

Table of Contents

Part I: Concepts

Integrating information about ecosystem services in strategic environmental assessment of spatial plans

1. Introduction

Research on ecosystem services has attracted a lot of attention in recent years and has become one of the most significant and fastest evolving areas in environmental sciences and ecological economics. It has been defined as a “cornerstone of sustainability science” for its focus on the interaction between nature and society (Clark and Dickson, 2003). The publication of the Millennium Ecosystem Assessment (MA, 2005) fuelled a number of studies aimed at analyzing and quantifying the importance of ecosystems for human well-being with a view toward supporting better decisions regarding the sustainable use of Earth’s resources. Recent research has revealed the possibilities for measuring and projecting the effects of policy choices on the structure and processes of ecosystems and the services they provide (Carpenter et al., 2009; Tallis and Polasky, 2009). However, despite this increasing body of literature, the use of ecosystem service concepts to support real-life decision making processes is still limited. As Daily et al. (2009) put it: “The Millennium Ecosystem Assessment advanced a powerful vision for the future, and now it is time to deliver”.

The narrow use of ecosystem services information affects especially the planning level of decision making, whereas more applications are found at the policy level (e.g., payment for ecosystem services policies, see Jack et al., 2008 for a review). In particular, spatial planning decisions would benefit from systematic considerations of their effects on ecosystem services. Spatial planning aims at “creating a more rational territorial organization of land uses and the linkages between them, to balance demands for development with the need to protect the environment and to achieve social and economic development objectives” (EC, 1997). Key issues in spatial planning concern land and resource use, the physical organization of space and the integration of sectoral strategies (agriculture, nature protection, transportation, tourism development, etc.). Spatial planning eventually results in actions that may affect the distribution, quality and use of a wide range of ecosystem services, and that are instrumental to their conservation and enhancement (TEEB, 2010, p.106). Hence, it is crucial to use information on ecosystem services to support planning processes.

The internalization of ecosystem service concerns into spatial planning should take advantage of existing procedures to support plan making. Strategic Environmental Assessment (SEA) is particularly suited to this purpose. SEA refers to a “range of analytical and participatory approaches that aim to integrate environmental considerations into policies, plans and programmes and evaluate the interlinkages with economic and social considerations” (OECD, 2006). One of the key SEA tasks consists in supporting the development of policies, plans and programmes by assessing the environmental impacts that are likely to results from their execution. A recent review of influential cases highlighted the topicality of ecosystem service-inclusive SEA, but also the fact that no methodological reference exists in the literature (Slootweg and van Beukering, 2008).

This paper aims at providing insights on why it is important to promote ecosystem service-inclusive SEA processes and on how to do it, with particular emphasis on spatial planning. Section 2 elaborates on the SEA process and on the inclusion of ecosystem services throughout it. Section 3 illustrates the advantages related to the integration of ecosystem service information in SEA, whereas Section 4 presents and discusses some of the open challenges. Finally, conclusions are drawn in Section 5.

2. Key SEA stages and relevant ecosystem service information

To ensure a proper consideration of environmental impacts, risks and opportunities from the early decisional stages onwards, SEA needs to be fully integrated into the planning process (see Partidário, 2007a for a thorough discussion on this). Only in this way can SEA contribute to decision making, from the very preliminary identification of the plan's scope and objectives, until the final implementation and monitoring. Hence, SEA must be flexible and able to adapt to the planning context, which is very different among countries (in terms of content, level of detail, timing, consultation with stakeholders, etc.) and among planning tiers (national, regional, etc). A number of guidance documents have been produced over the years to tailor the application of SEA to different decision-making contexts (e.g., OECD, 2006; Partidário, 2007b). For these reasons, a standard and internationally accepted procedure does not exist for SEA, even though some key common stages can be identified.

In very general terms, and for the purpose of illustrating the key SEA stages, the planning process has been decomposed into four main operational moments: defining the scope and the objectives of the plan, identifying suitable actions to achieve such objectives, drafting and refining the plan, and finally implementing it. In practice, these stages are often not organized in a linear sequence and interactions and feedback loops are common. For example, the objectives of a plan may be revised after a discussion on possible options to achieve them has taken place. SEA is a parallel process, which aims at supporting each of these stages by providing insights on the implications of the plan's strategies and decisions, as well as by broadening the plan's scope to ensure that appropriate sustainability objectives are taken into account (Bonde and Cherp, 2000; Partidário, 2007a). Table 1 lists the typical SEA activities linked to each of the four stages and provides a description of possible analysis to include information on ecosystem services. Public participation is an essential element of both planning and SEA, however it was not listed in Table 1 because ideally it is carried out in different moments of the process, rather than representing a separate stage.

In the first stage, the role of ecosystem service information consists mainly in supporting the identification of the most pressing issues related to the territory under analysis, as well as to the scope and content of the plan. During this phase, information can be collected and processed to understand opportunities and constraints related to the conservation and use of ecosystem services, for instance by generating graphs or maps of production and fruition patterns. These data can also be compared with information on population density and socio-economic conditions to perform a screening of critical sites. For example, GIS-based analysis could provide information on populated areas with and without suitable access to a given service (Geneletti et al., 2007).

In the second stage, the spatial plan begins to take shape, and proposals for land use changes, infrastructure development or new regulations are made. Ecosystem service information can be used to identify constraints to development and to perform land suitability analysis. The effects of alternative actions can be tested by predicting changes in important ecosystem services and evaluating them in biophysical and/or monetary terms. The pros and cons of the different alternatives can be highlighted by unveiling trade-offs among ecosystem services, locations (where are changes likely to occur?) and beneficiaries (who wins and who loses?).

During the third stage the plan is drafted and then completed. SEA performs important tasks such as the assessment of the cumulative effects (i.e., the combined effects of all the actions of the plan, as well as of external driving forces) on key ecosystem services and the comparison of the new proposal with alternative proposals (e.g., no plan, execution of the existing plan). This can be done by generating future land use scenarios that make different assumptions about key drivers (population growth, climate change, market prices, etc.). Land use scenarios can be constructed in a qualitative or quantitative and spatially-explicit form (Geneletti, 2011a), depending on the availability of data and on the level of detail of the plan. The results of these analyses are used to suggest revisions and mitigations in an iterative fashion until the plan reaches its final form. Mitigations may include measures to limit the negative impacts of the plan on ecosystem services, but also measures directed, for instance, at reducing the dependency of the plan's objectives on ecosystem services that may become scarce in the future.

Finally, during the implementation stage, the purpose of SEA is to monitor the execution of the plan's actions and strategies, as well as the evolution of the environmental and socio-economic context. This allows us to test whether the plan is achieving its objectives and whether patterns of use and production of services are evolving as expected. Adaptive management practices can be included in monitoring in order to steer the plan's implementation whenever required.

Table 1. Planning stages associated with SEA activities and with examples of actions to include relevant information on ecosystem services throughout the process.

Planning stages	SEA activities	Actions to include information on ecosystem services (ES)
Defining the scope and the objectives of the plan	• Describe environmental baseline • Identify environmental and sustainability objectives relevant to the territory being planned • Identify other relevant plans and policies and test for consistency	• Identify what ES the plan's objectives depends upon or affect (see also Table 3) • Map areas of production and fruition of key ES (including analysis of beneficiaries and stakeholders) • Collect data on spatial and temporal trends • Analyze issues of scale and spatial relationships (see also section 4)
Identify actions to achieve the objectives	• Propose and compare alternative actions, possibly in different scenario conditions • Predict and assess environmental effects • Support the selection of the preferred options • Test for consistency among plan's actions	• Track the direct and indirect drivers of changes in ES by paying particular attention to the foreseen land use changes • Test the effects of different options on ES by quantifying changes whenever possible and evaluating them in biophysical and/or monetary terms • Make trade-offs and synergies among ES explicit, considering both the production of services (where are they likely to increase/decrease?) and their use by different groups of beneficiaries (who wins and who loses?)
Drafting the plan, revision and final approval	• Suggest mitigations • Assess overall impact of the plan and suggest mitigations • Write SEA report	• Suggest solutions to reduce the impact of the plan on critical ES • Suggest solutions to reduce the dependency of the plan on critical ES • Assess cumulative effects on the ES under different future scenarios
Implementation	• Monitoring and follow-up	• Verify if patterns of use and production of ES are evolving as expected, and suggest adaptive management strategies

3. Advantages of ecosystem service-inclusive SEA

Using SEA for the purpose of including ecosystem services in planning is appropriate for a number of reasons. Firstly, SEA provides a window of opportunity to formally mainstream ecosystem services into decisions at the strategic level. This is because SEA has a legal basis in several dozen countries around the world. In Europe, for example, it is mandatory under the European Commission SEA Directive (Directive 2001/42/EC). Additionally, a growing number of countries (mostly in the developing world), international organizations and NGOs are applying SEA-type processes (Dalal-Clayton and Sadler, 2005).

Secondly, the scenario-analysis approach used by several studies on ecosystem services is consistent with the typical framework adopted in impact assessment, and hence can be easily applied in SEA. This approach is based on the analysis of expected changes in the distribution, value and fruition of services following the implementation of a given strategy or proposal, as opposed to the "static" evaluation of services (see for instance Balmford et al., 2008, p.9; Nelson et al., 2009; Birch et al., 2011). In SEA, baseline and policy scenarios are often used to understand the possible future status of a system with or without a proposed strategic action, and under different assumptions about uncertainty factors. This is very similar to many scenario analyses conducted for ecosystem services at various spatial scales (Willemen et al., 2010; Carpenter et al., 2006). SEA is essentially an "exercise in futuring" (Duinker and Greig, 2007), aimed at providing support to decision processes undertaken under conditions of uncertainty and scarcity of information. For this reason, scenario analysis is commonly listed in the SEA toolbox (OECD, 2006; Therivel, 2004), and its use is advocated in a number of scientific papers (Zhu et al., 2010; Noble, 2008).

Table 2. Examples of contributions of ecosystem services information to the quality of SEA, associated with the six characteristics of good SEA processes (as defined in IAIA, 2002).

Characteristics of a good-quality SEA process	**Contribution of ecosystem services (ES) information**
Integrated	• ES inherently address the interrelationships between biophysical and socio-economic aspects • The analysis of ES-related scale issues facilitates the interaction with relevant plans and policies at different decision-making tiers
Sustainability-led	• ES approaches explicitly link changes in ecosystems and biodiversity with effects on human wellbeing. Hence, ES-inclusive SEA processes extend beyond the assessment of biophysical and environmental factors only, and promote plans that are more sustainable
Focused	• ES approaches offer a key to read the most important interactions between human society and the environment, identifying issues that are important for the specific decision-making context
Accountable	• Analysis of expected future trends in ES under different scenario conditions can be used to document how sustainability issues were taken into account, and to justify planning choices
Participative	• Information on ES by definition requires the identification of beneficiaries and stakeholders, paving the way to more participative SEA processes
Iterative	• The analysis of ES can be included, in different forms, throughout the whole process (see Table 1), so as to provide information on the expected impacts of plan's choices during the different "decision windows" of the planning process

Thirdly, exploring methods to include ecosystem services in SEA is instrumental to the development of integrated approaches to assess the sustainability of proposed plans. In many contexts, SEA is increasingly including social and economic effects, and it is largely seen (both in the scientific literature and in practical applications) as a key entry point to sustainability assessment (Pope et al., 2004; Dalal-Clayton and Sadler, 2011). Obviously, a proper consideration of the effects of spatial plans on ecosystem services cannot be limited to the analysis of the biophysical environment but must include key socio-economic issues. As a matter of fact, addressing ecosystem services implies addressing the beneficiaries of such services and their characteristics (spatial distribution, socioeconomic status, contribution of services to wellbeing, etc). One last consideration is that the integration of information of ecosystem services can be highly beneficial to the SEA process, enhancing its quality. This concept is illustrated in Table 2, where examples of possible contribution of ecosystem services to good-quality SEA are provided, by referring to the list of SEA performance criteria developed by IAIA (2002).

4. Overview of the main challenges

Four main challenges to the development of ecosystem service-inclusive SEA can be identified. The first one concerns the selection of key ecosystem services from the very extensive lists developed in the literature. In order for SEA to be effective in influencing plan making, the number of services included in the analysis should be kept to a minimum, by considering only the ones that are relevant to the specific decision problems addressed by the plan, and to the characteristic of the area. This selection can be performed by identifying the services that are required for the implementation of the spatial plan and the services that the plan will affect (see OECD, 2008 p. 11). The first group refers to services upon which the plan depends. For instance, if the plan aims at promoting nature-based tourism, the achievement of this depends upon cultural services such as the aesthetic value of unspoiled landscapes. The second group refers to services that will be positively or negatively affected by the plan. Achieving the objectives of the plan may trigger drivers that in turn will alter the quality, quantity and/or spatial distribution of a given (bundle of) service(s). These drivers can be of a direct nature (e.g., physical interventions such as the choice of location of land uses) or an indirect one (policies that may affect the way in which society makes use of ecosystem services, such as for instance the ones that regulate access to recreation areas).

As an illustration of this concept, Table 3 presents the results of an empirical analysis of the possible relationships between the objectives of an exemplary spatial plan and ecosystem services. For each objective, the first column identifies the ecosystem services required for its achievement. For instance, the development of the horticulture sector relies on soil formation and retention. The table specifies if such dependency may extend beyond the boundary of the area being planned, hence requiring a broader scale analysis. For example, the regulation of water to support horticulture may depend upon water-use decisions taken outside the region. The second column identifies situations where the achievement of the objective will have a positive/negative effect on the ecosystem services. For instance, the protection of natural areas is bound to contribute to soil formation and retention, but it may reduce recreation opportunities. This type of analysis helps to set the context for SEA (see Table 1, first row), by identifying critical interactions that deserve to be studied in more detail. For instance, potential inconsistencies among plan’s objectives exist whenever the achievement of one objective relies on a given service, which in turn can be affected by a different objective. These situations can be detected by looking at each row of Table 3 (see, for instance, the case of water regulation and supply). This analysis is useful to define and revise the objectives of the plan, to suggest suitable stakeholder groups to be consulted (i.e., beneficiaries and users of the services affected in different ways by the plan), as well as to understand where further data and investigation are needed (e.g., quantification of service provision and fruition expressed in biophysical and/or economical units). Much of the quality of the overall SEA depends on this specific stage, so the open challenge is to perform it in a comprehensive way and early enough in the planning process.

The second challenge concerns scale. A spatial plan focuses on a geographically bounded area. Ecosystem services are provided and used at different spatial scales, and those scales may be much broader than the boundaries of a particular planning effort. The differences between the area that is being planned and the area that is being affected in terms of ecosystem services complicate the process of predicting the effects of spatial plans. Recent papers have attempted to examine the various scales at which services are provided and used (Hein et al., 2006), and to classify possible types of spatial relationships between the area of a service production and the area of use (Fisher et al., 2009; Costanza, 2008). In SEA for spatial planning, a proper recognition of services and stakeholders must be performed in order to understand situations where benefits accrue at one scale, but costs are borne at another. As an illustration of this concept, in Table 3 services whose production involves also areas outside the boundary of the region being planned are highlighted in bold. The challenge consists in understanding scale issues and pushing the analysis further to unveil the degree of dependency on outside conditions that characterize the services that are required to achieve the objectives of the plan. This will enable identification of other relevant plans and policies at the different tiers (national, regional, local, etc.) whose contents and regulations must be taken into account during SEA to exploit synergies and reduce inconsistencies.

The third challenges concerns trade-offs among ecosystem services. Spatial plan policies may change the relative mix of ecosystem services within a region by trading off the increase in one service with the decrease in another one. It has been observed that such trade-offs can be an explicit choice, but can also arise without awareness (Rodriguez et al., 2006). Research on ecosystem service trade-offs and on how to make them explicit in planning and decision making is limited (Carpenter et al., 2009). Land use planning is about resolving conflicts on competing demand for limited resources and uneven distribution of costs and benefits. Hence, trade-offs represent a pivotal issue. Develop methods to systematically analyze the main trade-offs between a range of selected ecosystem services and to link them to plans' policies and decisions is still an open challenge. The literature has mainly focused on the trade-offs between the production of services (Raudsepp-Hearne et al., 2010; Nelson et al., 2009), with limited efforts directed toward understanding the implications of such trade-offs for different groups of beneficiaries, characterised by different needs and levels of dependency of such services (Geneletti, 2011b).

Finally, there is the problem of selecting a manageably small set of indicators of ecosystem services that serve the needs of planners. Recently, modeling tools have been developed to spatially represent the distribution of multiple ecosystem services and highlight trade-offs (Tallis and Polasky, 2009). These tools are based on the generation of land use/land cover scenarios, and represent a promising path towards the inclusion of ecosystem services in spatial planning. However, the process of analyzing multiple ecosystem functions in different scenario conditions is associated with extraordinarily high amounts of information. It is unclear to what extent these tools can be used in actual spatial planning settings, with all the constraints that those settings involve. The open challenge consists in finding ways to present information to decision makers in a manageable way, possibly by creating new indicators that consist of combinations of existing ecosystem services indicators and indicators traditionally used in land use planning (e.g., indicators of land suitability for different uses).

5. Conclusions

This paper discussed why and how SEA may be useful for bringing information on ecosystem services to bear in planning decisions. The paper focused on spatial and land use planning, being the area where the effects of plan's decisions on service provision and use are perhaps more evident and straightforward. However, most considerations can be extended to planning processes in other sectors, such as tourism, water, energy, etc. Four open challenges to the implementation of effective ecosystem service-inclusive SEA were identified, concerning scoping, scale issues, trade-offs and indicators, respectively. These challenges can be addressed by carrying out pilot applications in real planning contexts, and by taking advantage of the data, tools and methods for ecosystem service representation and modeling that are becoming increasingly available.

Table 3. Exemplary analysis of the possible relationships between the objectives of a spatial plan and ecosystem services. For each objective, the first column identifies the ecosystem services required for its achievement. The bold font indicates that such dependency may extend beyond the boundary of the area being planned, hence requiring a broader scale analysis. The second column identifies situations where the achievement of the objective will have a positive/negative effect on the ecosystem services. The objectives were selected from the ones of the regional spatial plan of The Araucania, Chile. Ecosystem services were selected from Costanza et al. (1997).

		Spatial plan's objectives															
		Increase competitiveness in the nature tourism sector		Increase competitiveness in the timber sector		Increase competitiveness in the aquaculture sector		Increase competitiveness in the horticulture sector		Reduce exposure to natural risks		Protect areas with high natural value		Respect and promote cultural and ethnical diversity		Promote polycentric development	
		Depend	Affect	Depend	Affect	Depend	Affect	Depend	Affect	Depend	Affect	Depend	Affect	Depend	Affect	Depend	Affect
Ecosystem services	Climate regulation				+												
	Water regulation and supply				-	**x**		**x**					+				-
	Waste treatment						-		-							**x**	
	Soil formation			x				x					+				
	Erosion control			x				x					+				
	Raw materials				+								-				
	Cultural	x			-		-						+	**x**			
	Recreation	x					-						-				+
	Food production				-		+		+								
	Disturbance regulation				-		-			**x**			+				
	Refugia	x	-				-				+	x	+				-

Part II: Applications

Modeling the effect of spatial planning policies on future ecosystem services in The Araucania, Chile

1. Introduction

Ecosystem services are the benefits human populations derive from the ecosystems, such as goods and products (e.g., fresh water, fuel), regulation of natural processes (e.g., climate, flooding, erosion), and nonmaterial benefits (e.g., recreation, aesthetic enjoyment) (Daily, 1997). Research on ecosystem services has attracted a lot of attention in recent years, and has become one of the fastest-evolving areas of investigation at the intersection of environmental and social sciences. Fisher et al. (2009) reported an exponential growth of scientific publications in this field over the last decade. Research on ecosystem services has been defined a "cornerstone of sustainability science" for its focus on the interaction between nature and society (Clark and Dickson, 2003). The Millennium Ecosystem Assessment (MA, 2005) made this interaction explicit by documenting the rate of degradation of many services, and the associated negative consequences for human well-being. The publication of the MA fuelled a number of studies aimed at understanding trends in ecosystem services distribution to eventually support decisions concerning the sustainable use of Earth's resources (Carpenter et al., 2009).

Land use conversions rank among the most significant drivers of change in ecosystem services worldwide, affecting human well-being and threatening the survival of other species (Foley et al., 2005; Nelson et al., 2006; Metzger et al., 2006). Hence, predicting the effects of land use decisions on ecosystem services has emerged as a crucial need in land management (DeFries et al 2004; NRC, 2005). Land use choices typically produce trade-offs by increasing the provision of some services at the expenses of others (Rodriguez et al., 2006; Bennett et al., 2009). For example, Raudsepp-Hearne et al. (2010) showed strong landscape-scale trade-offs between the production of goods and commodities and regulating and cultural services (e.g., nutrient retention and nature appreciation) for a study region in Canada. Nelson et al. (2010) compared the trade-offs between crop production and important regulating services in a set of global-level land use projections. Several other papers explored the interactions between multiple ecosystem services and land use/land cover change at different scales (Chan et al. 2006; Egoh et al. 2008; Naidoo et al. 2008; Willemen et al., 2010). Clearly, these approaches require spatially explicit representations of ecosystem services. To this purpose, new GIS-based methods and tools have been recently developed (Grêt-Regamey et al., 2008; Beier et al., 2008; Tallis and Polasky, 2009; Nelson et al., 2009).

A common feature in studies addressing the relationship between land use and ecosystem services is the use of scenario analysis to represent and compare possible futures. From the publication of the MA onwards, scenarios have been frequently employed in both conceptual frameworks (Carpenter et al., 2006; Balmford et al., 2008) and empirical applications (Nelson et al., 2009; Polasky et al., 2011; Birch et al., 2010). In these exercises, future scenarios are constructed by making different assumptions on the underlying driving forces, and hence on the magnitude of land use change processes. For instance, Polasky et al. (2011) built different land use scenarios by varying the rates of urban, agricultural and forestry expansion. Less attention has been devoted so far to the effects on ecosystem services of different spatial arrangements of land uses.

The effects of spatial patterns on ecological processes represent one of the central themes of landscape ecology (Turner, 1989). Scientists have tried to understand these effects for a broad range of ecosystem services, such as habitat provision (Dale et al., 1994; Villard et al., 1999), pollination (Brosi et al., 2008) and water purification (Lee et al., 2009). According to a recent article, the capacity of a land system to deliver ecosystem services is determined by the kind, magnitude and spatial patterns of land uses (Turner, 2010). Research on land use change and ecosystem services has widely addressed the first two attributes but has disregarded so far the third one, spatial patterns. To the best of our knowledge, none of the studies presented in the literature considered scenarios that differ only in the distribution of land use types, rather than in their relative proportions. This gap inspired the present paper and its central research question: What are the effects of different land use patterns on the provision of multiple ecosystem services?

This question is addressed by focusing on land use patterns induced by spatial planning tools. In many regions of the world, spatial plans are the principal policy instrument affecting the distribution of land uses within a jurisdiction (Willis and Keller, 2007). The paper presents case-study research aimed at empirically exploring how the implementation of different land use zoning policies can affect the future provision of ecosystem services. The term 'land use zoning policy' is used here to indicate regulations concerning permitted, prohibited or preferred land uses. The design of such policies is often a core issue in spatial planning, and they represent one of the most tangible elements of a plan (Geneletti, 2011). The method is based on the generation of future land use scenarios that simulate the implementation of different zoning policies. Scenarios are constructed by holding constant the rate of land use change processes but varying their location according to the different policies. The effects of the land use scenarios on selected ecosystem services are then modeled and compared through a set of metrics. The study area is The Araucanía (southern Chile), a region rich in natural resources, but affected by widespread poverty, relatively low performance in development indicators and reliance on the conservation of ecosystem services to support rural livelihoods. The application of the methodology to the case study addresses three more specific questions: What are the effects of different zoning policies on future land uses within the region? How do these affect the conservation of ecosystem services? What are the empirical patterns of trade-offs among ecosystem services associated with the different policies? By answering these questions, the paper aims also at illustrating the potential contribution of the approach to support spatial planning.

2. Study area, selected ecosystem services and zoning policies

The Araucanía, one of Chile's 15 regions, covers an area of 31,842 Km^2 and has a population of 890,000, about a third of which is distributed in rural areas (Figure 1). It is considered to be among the regions with the highest natural capital of the country, with 12 national parks and reserves (mainly distributed mainly on the Andes Mountain Range), and almost 30% of the area covered by native forest (Gobierno de Chile, 2009). The whole region falls within one of the World's 25 Biodiversity Hotspots: the Chilean winter rainfall-Valdivian forests, characterized by a high number of endemic species (Myers et al., 2000). The Araucanía is also among Chile's poorest areas, with a relatively low performance in development indicators and 27% of people living below the poverty line (Mideplan, 2009). Part of the rural population, especially in the coastal area, relies on subsistence farming, fishing or timber collection. The region is the heartland of the indigenous Mapuche people, who currently represent 23% of the population. In the last three decades, the region has experienced a rapid growth of monoculture conifer plantations (almost exclusively made of *Pinus radiata* and *Eucalyptus sp*.), at the expenses of native forest, marginal agricultural fields and grasslands. Remaining native forests are concentrated almost exclusively on the Coastal and Andes mountain ranges (CONAF et al. 1999; Gobierno de Chile, 2009). Another, and more recent, trend in land use is the development of market-oriented forms of agriculture, such as berry production (ODEPA and CIREN, 2006). These changes in land cover and land use brought about a number of environmental problems, including water scarcity, water pollution, soil erosion and biodiversity loss (Echeverria et al., 2006; Lara et al., 2009).

The characteristics of the study area drove the selection of the ecosystem services considered in this study, which include water purification, soil retention, carbon sequestration, provision of habitat for species and timber production. Water purification refers to the capacity of ecosystems to mitigate water pollution by retaining some non-point source pollutants. Soil retention refers to the capacity to keep soil in place, so avoiding erosion. Carbon sequestration is the capacity of plants and soil to store and accumulate carbon. The habitat service is the capacity of ecosystems to supply living space for species, hence supporting biodiversity. Besides being extremely relevant to the land use change processes that characterize the study area, these ecosystem services were also selected because they represent a mix among different categories of services, namely regulating (water purification, soil retention, carbon sequestration), supporting (habitat for species) and provisioning (timber production) services. This allows us to understand a broad suite of ecosystem responses to land use changes and to assess the associated trade-offs (DeFries et al., 2004).

In Chile, a new form of spatial plan at regional level has been recently introduced (the Plan Regional de Ordenamiento Territorial, PROT). This differs from existing plans because, among other things, it promotes the development of a land use zoning policy that defines the preferred land uses for the whole territory, rather than for urban areas only. This zoning policy aims at giving spatial representation to regional development strategies by specifying where the strategies' objectives are to be achieved and through which uses of land (Subdere, 2010). The PROT has not been developed yet for The Araucanía region, even though preliminary analysis and planning tools exist (PRDUyT, 2005). The information contained in these documents was employed in this study to draft possible land use zoning policies for the region. These were used to test the proposed method. The purpose was simply to formulate plausible zoning for the regional context, in the light of the existing information, rather than to develop actual alternatives. The first zoning policy (ZP0) is represented by the zero-alternative, or "laissez-faire"approach where no constraints or preferences are set concerning the location of land use change processes. The second policy (ZP1) was inspired by the indications contained in the existing land use plan on the preferred and prohibited uses of different land units. The third policy (ZP2) was designed by merging indications resulting from a survey on land suitability conducted at municipal scale (PRDUyT, 2005). The policies subdivide the region into zones, specifying for each zone the land uses that are preferred, prohibited or disfavored. A sketch representing the policies is presented in Figure 2.

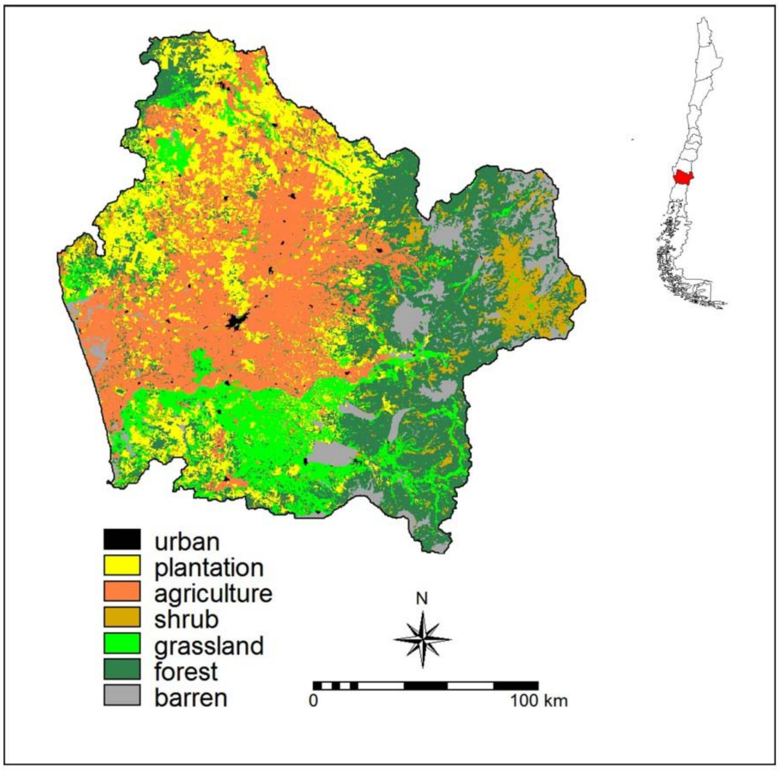

Figure 1. Location of The Araucanía region in Chile (top right) and main land uses

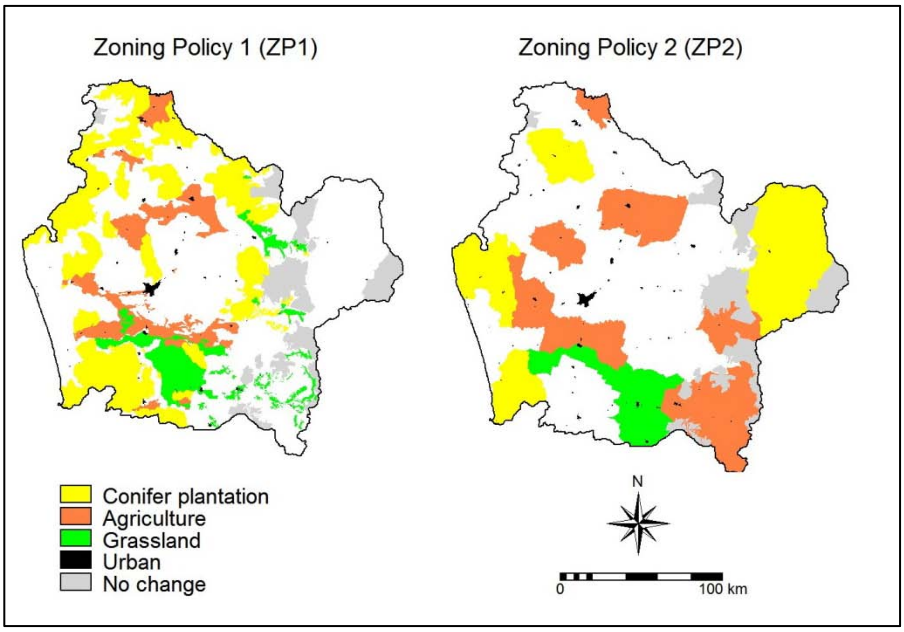

Figure 2. Zoning policy 1 and 2 (for better visualization, only preferred land uses are represented)

3. Methods

The first step of the method consisted of the construction of land use scenarios associated with the three zoning policies previously described (Section 3.1). Subsequently, the effects of the land use scenarios on the provision of the selected ecosystem services were assessed in a spatially explicit way, by using state-of-the-art modeling tools (Section 3.2). Finally, a set of metrics was developed to compare scenarios and assess trade-offs (Section 3.3).

3.1 Development of land use scenarios

Map representations of land use scenarios were generated through spatial modeling in a GIS. In particular, empirical mathematical models of past change processes were applied to project maps of land use at a given time. The approach consisted of four main stages: land use change analysis and modeling, projecting future land use quantities, setting incentives and constraints according to the different zoning policies, and finally projecting future land use scenarios. All operations were carried out through the Land Change Modeler (LCM) tool implemented in the software package IDRISI Taiga (Eastman, 2009).

The input to the first stage was represented by two land use maps of 1994 and 2007, which were available with a 30-m spatial resolution. After having harmonized their legend, the two maps were processed and compared to detect land use changes. A transition matrix was built, showing the amount of land that was converted from each land use to any other use. Consistently with the objective of the study, the modeling of future land uses focused only on the main man-induced transitions, as opposite to transitions due to natural processes (e.g., from shrubs to forest). In particular, the following six transitions were considered: agriculture to urban, grassland to urban, forest to agriculture, and forest/grassland/agriculture to conifer plantation. For each transition, a mathematical sub-model was developed. Sub-models are based on the combination of user-selected explanatory variables, using the binomial logistic regression and the Maximum Likelihood method (more information on this procedure can be found in Aldrich and Nelson, 1984, and Clark and Hosking, 1986). In short, each sub-model determines how the different variables explain a specific land use transition that occurred in the period 1994–2007, and on the basis of this information is able to generate a map of "transition potential" for a given time in the future. The variables were selected from the existing regional database or generated from such a database through GIS analysis (e.g., reclassification, distance calculation). For example, the sub-model for the agriculture-to-plantation transition included the following variables: elevation, slope, aspect, soil type, proximity to infrastructure and urban areas, and proximity to existing plantations. Prior to their use in the sub-models, the explanatory power of variables was tested using Cramer's V, in order to discard variables that showed low association with land uses (Eastman, 2009).

In the second stage, the rate of future land use transitions was set. Two rates of change were considered in this study to illustrate the effects of the zoning policies in different conditions, hence expanding the scope of their comparison. The first one ("Rate 1") was derived from a Markov Chain prediction (Bell, 1974), which assumes that the type and rate of future land use transitions will be equivalent to the ones that occurred in the past. Using the results of the land use change analysis, the Markov Chain determines exactly how much land would be expected to transition from each land use category to every other category in a given time in the future. The second rate ("Rate 2") was computed by increasing by 50% the Markovian rate of the considered land use transitions. This was to simulate a condition characterized by a more rapid increase in conifer plantations, urban areas and agriculture, considered as a possible future trend for the area.

In the third stage, maps of constraints/incentives were formulated to reflect the implementation of the different zoning policies. For each zoning policy and each land use transition, a map of constraints/incentives was built by classifying the zones into the following categories: absolute constraints (value zero), constraints (value 0.5), unconstrained areas (value 1) and incentives (value 2). In the LCM, these maps are multiplied by the transition potentials maps developed in the first stage. As a result, absolute constraints become areas where the land use transition under consideration cannot occur, whereas constraints and incentives areas are, respectively, disfavored and favored for that transition. Finally, land use scenarios were generated for the three zoning policies, under the two assumptions concerning the future rate of land use change processes. The final time horizon was set to 2050, even though scenarios were developed also for two intermediate times (2020 and 2035) in order to gain a better understanding of temporal trends. Hence, a total of 18 land use scenarios was constructed (see Figure 3). Operationally, scenarios are generated in LCM by an algorithm that first looks through all transitions and creates a list of host classes (classes that will lose some amount of land) and a list of claimant classes (classes that will acquire land) for each host (see Eastman, 2009 for more details). The land quantities for each class are determined as described in stage two. A multi-objective allocation process is then run to assign land for all claimants of a host class, using as a reference the maps of transition potential developed in stage two (and then adjusted according to the incentives/constraints maps, as per stage three). The results of the allocation of each host class are then overlaid to produce the final land use map (full details on the allocation procedure can be found in Eastman et al., 1995). Therefore, at any time horizon, land use scenarios associated with the three policies are characterized by constant quantities of each land use type, but different spatial patterns.

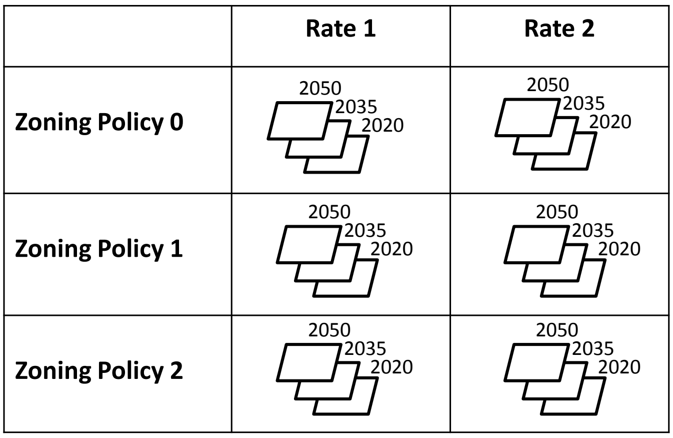

Figure 3. Diagram of the set of land use scenarios resulting from the combination of the three zoning policies and the two rates of change, at three different time horizons

3.2 Ecosystem service modeling

The provision of the selected ecosystem services was modeled across the 18 different land use scenarios and also for the 2007 land use map, which was used as a baseline for comparison. Modeling of carbon sequestration, water purification and soil retention was performed through a recently developed set of GIS-based models, the Integrated Valuation of Ecosystem Services and Tradeoffs tool (InVEST) (Tallis and Polasky, 2009). Habitat provision was modeled using MaxEnt (Phillips et al., 2004). Finally, timber production was modeled by considering the land area covered by plantations. An overview of the models and of the input data that were used in this study follows. A full description of the models, including information on their assumptions and limitations, can be found in Tallis et al. (2011) and Phillips et al. (2004, 2006).

The carbon sequestration model aggregates the amount of carbon stored in four carbon pools: aboveground biomass (living plant material above the soil, such as trunks and branches), belowground biomass (root systems), soil organic matter and dead organic matter (litter and dead wood). The model allows also inclusion of a fifth carbon pool, namely the harvested wood products (e.g., firewood or charcoal), which represents the biomass removed through harvest. However, this pool was not considered here because data concerning harvesting intensity and frequency across the study region were lacking. The basic assumption of the model is that each land use type has a fixed storage level, so that changes over time are due to transition from one land use type to another, disregarding other effects, such as natural succession of vegetation. This assumption is acceptable for the purpose of this work, which aims at assessing differences caused by land use changes, rather than attempting to measure absolute values. Data on carbon stored in each of the four pools for each land use type were taken from published reference data (IPCC, 2006), by using information on the climate domain of the study region. The outputs of this model are maps that represent the amount of carbon (in Mg per grid cell) that will be stored under the different land use scenarios.

The water purification service refers to the capacity of ecosystems to mitigate water pollution by retaining some non-point source pollutants through the action of vegetation and soil. The InVEST model uses data on runoff, land use, nutrient loading and filtration rate to determine the nutrient retention capacity of every grid cell in a given land use scenario. Nitrogen was selected as target nutrient in this study. The model first calculates average runoff from each cell. Subsequently, it estimates how much nitrogen leaves each cell using appropriate export coefficients. Finally, it determines how much of this load is retained by the downstream cells, and eventually how much pollutant reaches waterways from each cell. The latter output (expressed in kg of N per cell) is the one used in this study to model the effect of the different land use scenarios.

The soil conservation service refers to the capacity of vegetation to keep soil in place. For any given land parcel, the service is modeled by comparing the erosion rate on that parcel with the rate that would occur if no vegetation were present. Erosion rates are predicted by using the Universal Soil Loss Equation (USLE, Wischemeier and Smith, 1978), which accounts for land use, soil type, rainfall intensity and topography. The model also estimates how much of the sediment eroded is trapped by downstream vegetation. The final outputs are maps containing the total sediment retained by each cell (sediment retained on the cell itself + sediment removed from the loadings of the upstream cells) under the different scenarios.

The habitat provision service is the capacity of ecosystems to supply living spaces for species. In this study, it was modeled by selecting the kodkod cat *Oncifelis guigna* as a target species. *O. guigna*, one of the world’s smallest and least known felids, has a very small geographical distribution, restricted to a narrow strip of temperate forests within southern Chile and Argentina (Sanderson et al., 2002). Its habitat includes forests and semi-open areas, and it is strongly associated with Araucaria native forests, which are endemic in the study region (Dunstone et al., 2002). *O. guigna* is considered endangered in Chile, its survival being threatened by forest loss and fragmentation (Acosta-Jamett et al., 2003). Presence data of *O. guigna* in the study region, collected over the last few years, were available. These data, in combination with GIS layers of environmental and land use variables thought to influence species distribution (vegetation type, elevation, slope, distance from water bodies, distance from roads, etc), were used to model the habitat of *O. guigna*. Modeling was performed using MaxEnt, a species distribution model based on the ecological niche concept (Phillips et al., 2004). MaxEnt is a machine-learning model based on the maximum entropy concept that provides as a result maps showing the probability gradient for the potential distribution of a species in a 0-1 range. Further details on the tool and its algorithms can be found in Phillips et al. (2006; 2004). The selection of MaxEnt was suggested by recent studies that showed its good performance in comparison with other models (Gontier et al, 2010; Hernandez et al., 2006). Additionally, MaxEnt works with only species presence data, while other models require also absence data, which were not available for the study area. The model was run for the 2007 baseline and for the 18 land use scenarios.

Finally, commercial timber production was estimated by using the land area of plantations as a proxy. This index is commonly used to assess the productive capacity of forests (Haynes, 2003). The actual estimate of the volume of harvested timber would have required spatially-distributed data on harvest level and cycle, which were not available. The proposed proxy is adequate to the purpose of this work, which aims at assessing changes from baseline conditions, as opposite to absolute biophysical or monetary quantification of services.

3.3 Metrics for policies comparison

The following metrics were developed to compare the effects of the zoning policies on the selected ecosystem services:

1. Percentage of change in service provision across the region with respect to the 2007 baseline;
2. Percentage of the landscape where the service is preserved. In any given cell, a service is considered preserved if its 2050 value is at least equivalent to 90% of its 2007 value;
3. Percentage of the landscape where the service is degraded. In any given cell, a service is considered degraded if its 2050 value is less than 50% of its 2007 value.

The combined used of these metrics aim at providing further understanding on the performance of the different policies. For example, they help to discriminate policies with good overall performance but severe local degradation of services, from policies with relatively lower performance but more homogeneous effects across the landscape.

4. Results

The transition matrix resulting from the analysis of the land use change between 1994 and 2007 is presented in Table 1. A simplified legend is used to highlight transitions of interest for this study. Conifer plantations increased by about 32%, at the expenses of agricultural land, forest and grassland in that order. Urban areas increased by 24%, mainly as a result of the encroachment of agriculture and grasslands. Native forest experienced a total reduction of about 4%, due to conversion to agriculture and plantations. As anticipated in Section 3, these were the transitions addressed here. Other land use changes are either minor, due to natural processes, or partly explained by classification errors affecting the land use maps (e.g., plantation to forest and forest to grassland/shrub) (Gobierno de Chile, 2009). These results were used to compute the two rates of future land use changes, which in turn were used to project the size of the different land uses in 2020, 2035 and 2050 (Figure 4).

The land use scenarios resulting from the modeling are presented in Figure 5 (for conciseness only the 2050 scenarios are shown). A visual comparison of the scenarios shows the effects of the different zoning policies. ZP0 causes plantations to first grow in the northwestern sector, and then gradually expand to the South in a more scattered and patchy fashion. ZP1 produces an intrusion of plantations mainly in the central sector of the region, generating a more fragmented agricultural landscape. Under ZP2, plantations develop along a ring from the northern to the southwestern sectors of the region, leaving an agricultural core in the middle. As to changes in the forest cover, ZP0 and ZP1 tend to encroach native forest with similar patterns, affecting mainly the southeastern sector, whereas ZP2 causes forest loss especially in the northeastern areas.

The results of the ecosystem services modeling are presented in the maps of Figure 6 (for 2050 and Rate 1 only, spatial patterns being similar in the other scenarios) and in the graphs of Figure 7. The latter figure shows the trends of the five ecosystem services through time under the three zoning policies and for the two rates of change (metric 1, Section 3.3.). The values reported in the diagrams are obtained by summing up the value of the service in each cell across the maps and then normalizing by the 2007 values. The habitat provision service represents an exception in that the values were obtained by counting the cells with probability gradient above 0.5 (i.e., by looking at how many locations have a highly suitable habitat, rather than at the average habitat suitability). A sensitivity analysis was run by also testing different thresholds (0.75 and 0.85), but this did not change the relative performance of the policies. As expected, all policies produced the same general trend: a decline in water purification, soil retention and habitat provision (triggered by the progressive conversion to less natural land covers), and an increase in carbon sequestration and timber production (promoted by the expansion of conifer plantations). Timber production does not differ among policies because the adopted indicator measures the size of plantation areas. Carbon sequestration is also very similar because it is strongly related to the size of different land use types. The spatial patterns of carbon sequestration and timber production are clearly associated with the land use changes that occurred in the different scenarios (compare Figure 5 and 6). However, the decline in water purification and soil retention is the result of watershed-scale phenomena, and also affects areas not directly involved in land use changes (e.g., the central sector of the coastal strip and part of the pre-cordillera).

Table 1. Transition matrix showing land use changes (in thousands of hectares) that occurred between 1994 and 2007 (the bold font indicates transitions addressed in this study)

2007 **1994**	**Urban**	**Plantation**	**Agriculture**	**Grassland/scrub**	**Forest**	**Other**
Urban	10.48	0.04	0.00	0.00	0.00	0.03
Plantation	0.25	545.52	0.00	0.00	38.82	2.57
Agriculture	**1.74**	**107.69**	856.89	0.00	0.00	0.00
Grassland/scrub	**0.43**	**54.28**	0.00	394.68	0.00	2.36
Forest	0.12	**63.05**	**14.24**	13.97	820.71	0.00
Other	0.10	7.78	0.00	0.00	9.64	236.55

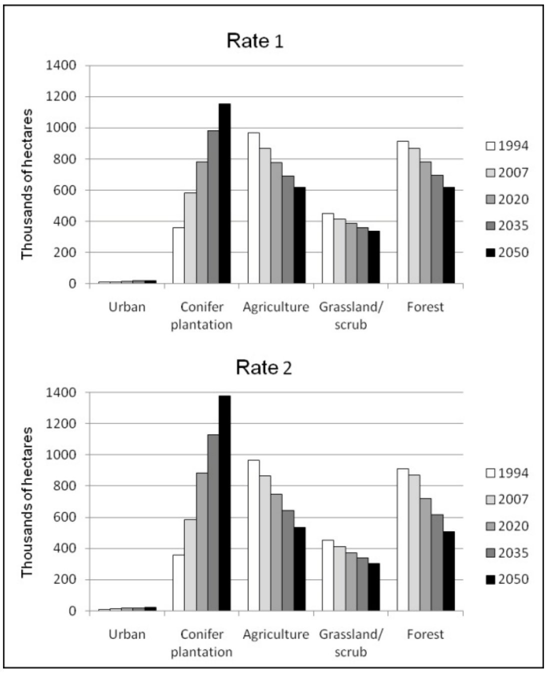

Figure 4. Land use projections under the two rates of land use change

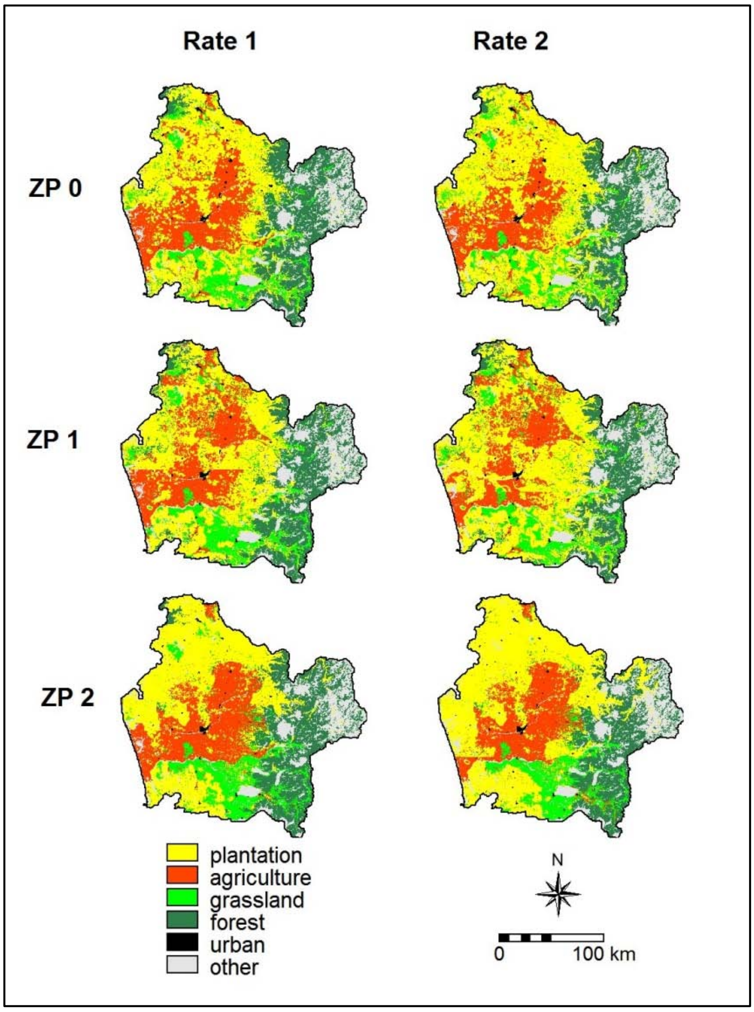

Figure 5. The land use scenarios generated for the three zoning policies and the two rates of land use change (time horizon: 2050)

The diagrams of Figure 7 show that the gradient of the decline in water purification, soil retention and habitat provision depends upon the land use zoning policy. The results indicate that ZP1 has the smallest impact in the conservation of the two regulating services of water filtration and soil retention, as well as the supporting service of habitat provision. Differences between the best and the worst performing policies are on the order of 5% for water and soil, and 10% for habitat. These differences are comparable in magnitude to the differences that characterize, for the same policy, the two rates of land use changes (compare solid and dotted lines in Figure 7). For example, the implementation of ZP0 assuming Rate 1 generates in 2050 the same decline in water filtration as the implementation of ZP1 assuming Rate 2 (hence increasing by 50% the rate of land conversion to urban areas and plantations). This is even more evident for soil retention and habitat: ZP1 assuming Rate 2 causes less reduction in these services than ZP2 assuming Rate 1. The trade-off diagrams represented in Figure 8 offer a clear picture of the dominance of ZP1 over ZP0 and ZP2. The first set of diagrams (Figure 8 a-c) focuses on the trade-offs between provisioning (timber) and regulating and supporting services (soil retention, water purification, habitat provision). The second set (Figure 8 d-f) illustrates trade-offs between regulating services that occur at different spatial scales: changes in soil retention and water purification (which can be considered regional-scale services) are compared with changes in carbon sequestration, a global-scale service. All diagrams show that ZP1 is the most efficient policy: for the same levels of timber production or carbon sequestration, it preserves more habitat, soil and clean water.

Finally, Table 2 presents metrics 2 and 3. These were computed only for water, soil and habitat, as they are not relevant to the other two services. According to these metrics, the ranking of the policies is more variable. However, SP1 is characterized by the best, or very close to best, performance in terms of both minimizing areas where the three services are highly-degraded (metric 3), and maximizing areas where they are well preserved (metric 2). The only exception is represented by the preservation of habitat, for which SP2 has a much better performance.

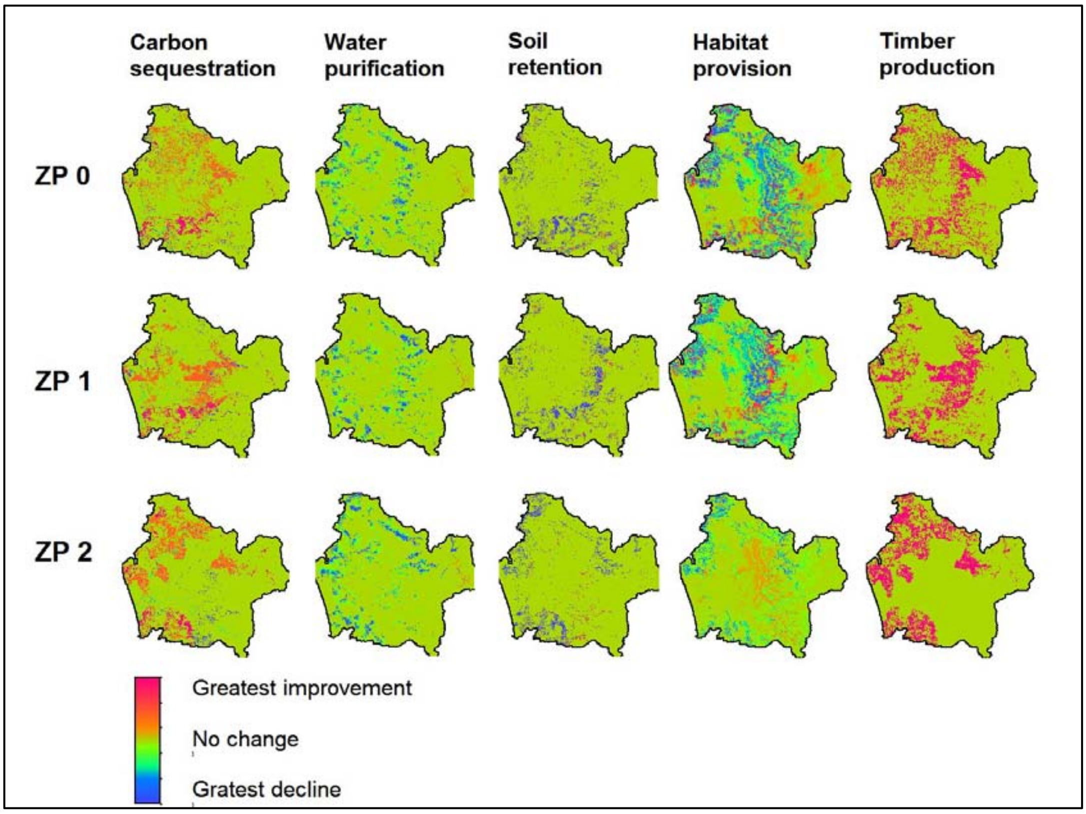

Figure 6. Results of the ecosystem service modeling for Rate 1 (time horizon: 2050)

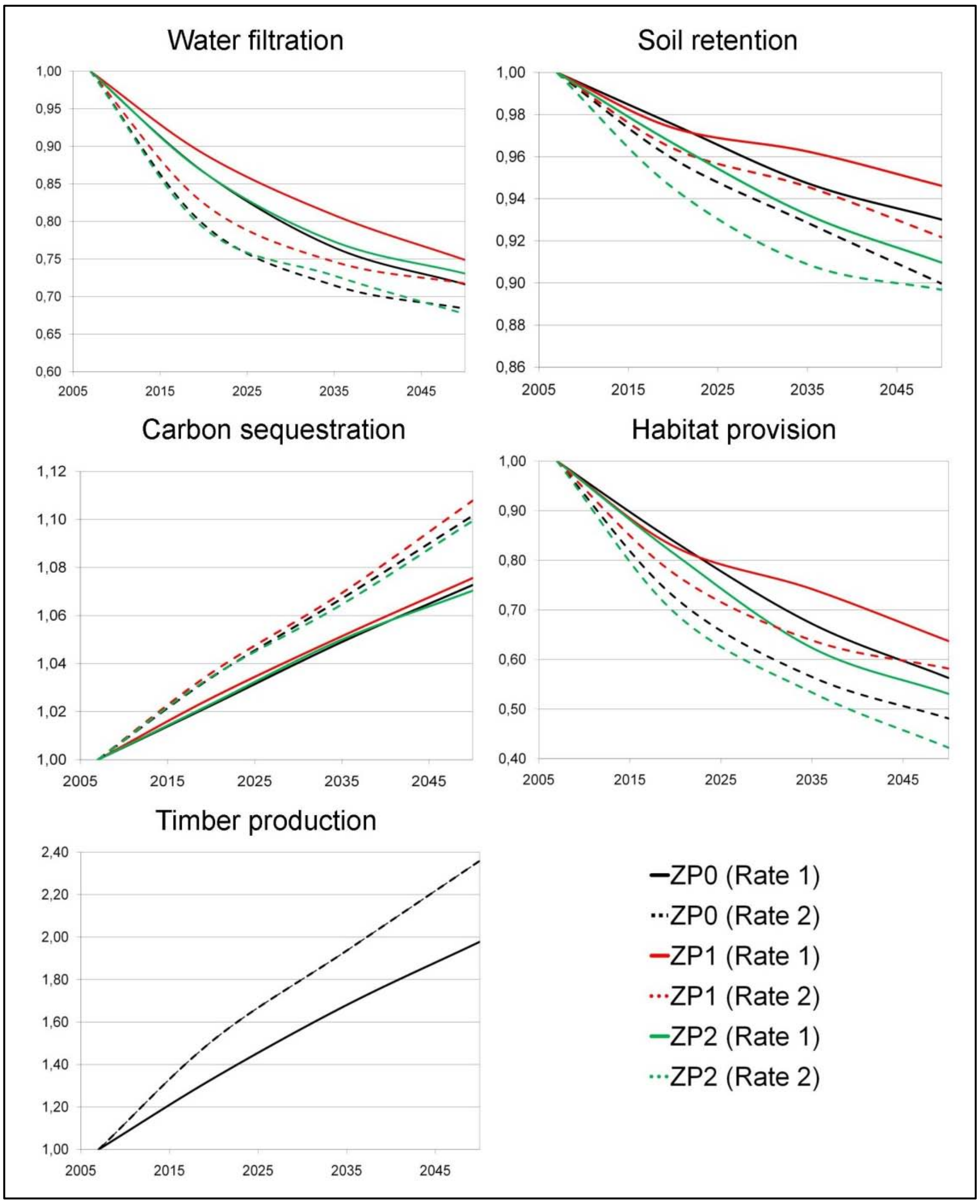

Figure 7. Trends of the five ecosystem services through time under the three zoning policies

Table 2. Results obtained by applying metric 2 (percentage of the landscape where the service is preserved) and metric 3 (percentage of the landscape where the service is degraded).

	Water filtration [% of area]		Soil conservation [% of area]		Habitat provision [% of area]	
	Preserved	**Degraded**	**Preserved**	**Degraded**	**Preserved**	**Degraded**
ZP0	29.92	19.59	76.12	4.58	24.73	14.98
ZP1	43.78	15.12	82.96	4.75	26.93	10.39
ZP2	44.09	16.93	72.85	10.44	43.37	28.93

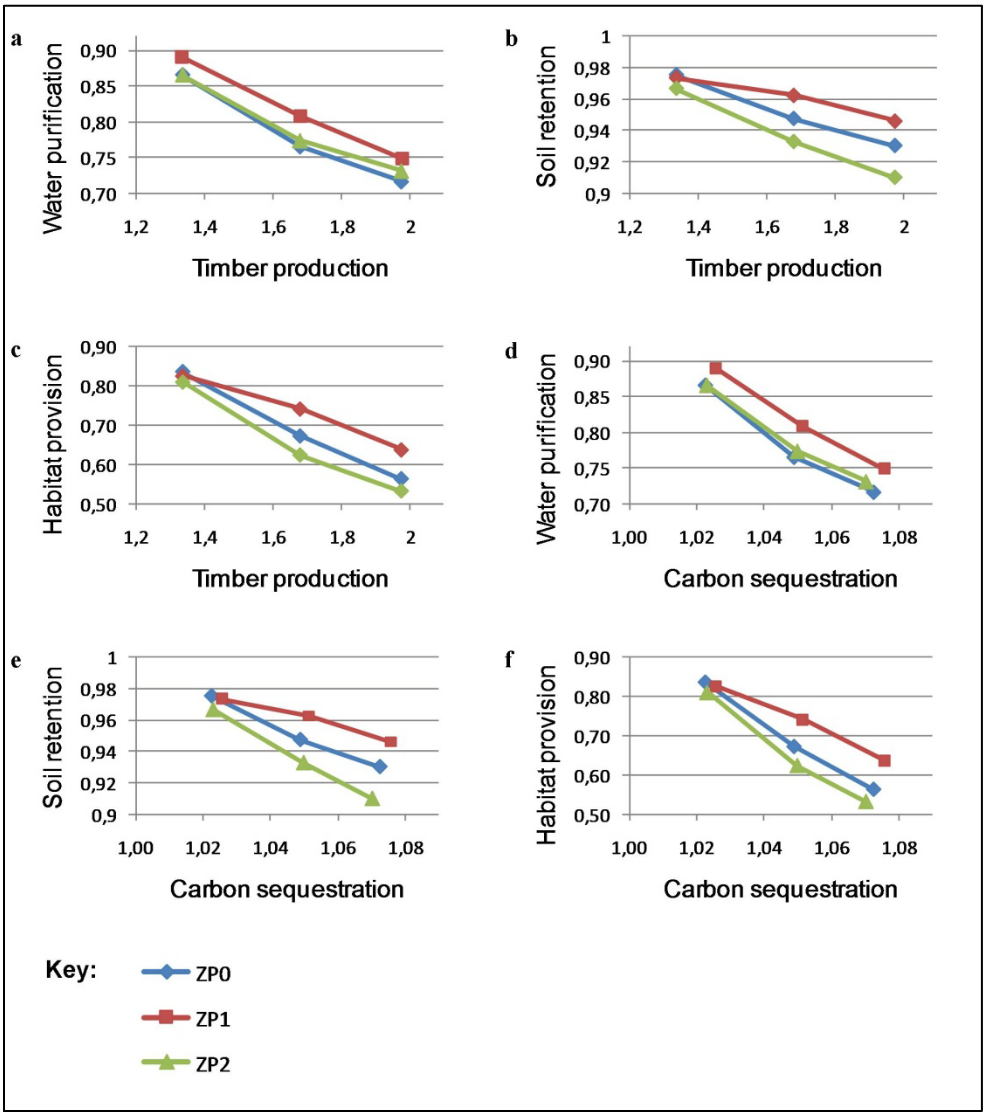

Figure 8. Diagrams of the trade-offs between pairs of ecosystem services. a-c represent trade-offs between provisioning and regulating/supporting services. d-f represent trade-offs between regulating services that occur at regional and global scale.

5. Discussion and conclusions

This paper applied spatially-explicit models to generate land use scenarios associated with different zoning policies and to predict their effect on a set of ecosystem services. The research contributes to the growing literature on the interactions between land use changes and the provision of multiple ecosystem services (Polasky et al., 2011; Nelson et al., 2010). Consistent with an emerging consensus in this literature (Birch et al., 2011), the approach compared the effects of alternative policy actions, rather than performing a static analysis of current service provision. In typical scenario exercises, future land uses are projected by making different assumptions on the driving forces, or on their intensity. In this study, we assumed that the driving forces are identical, but we changed their effects by imposing spatial constraints/incentives on the location of possible land use conversions. The results illustrated the importance of taking into account the spatial arrangement of land uses produced by different policies.

In line with the literature on scenario analysis, scenarios have been used in this study to describe what could happen, rather than to predict what will happen, or even what is likely to happen (Shearer, 2005). In particular, the use of empirical modeling allowed building land use scenarios that are realistic because they are based on the analysis of the factors that have driven past land use transitions (e.g., slope, soil, proximity to roads). In this way, scenarios represent the possible effects “on the ground” of the different zoning policies. This is particularly relevant at the regional level, where land use zoning aims to provide indications on the general allocation of land uses, rather than specifying the future use of each land parcel (Geneletti et al., 2007). The use of two different rates of future land use change processes allowed us to compare the magnitude of the effects on the ecosystem services attributable to changes in the size of land uses with the one attributable to changes in their pattern. The results indicate that, for this case study, spatial configuration of land uses is as an important factor as their size. This suggests that the analysis of land use patterns deserves attention, and that this information should be included in scenario exercises aimed to support spatial planning. By using the two different rates, we also wanted to test to what extent the relative performance of the policies was influenced by the rate of the land use change processes. One final consideration concerning the land use scenarios developed here is that, in reality, land use change processes and zoning policies are not likely to be independent from each other, as it can be argued that the policies themselves drive, at least to some extent, the intensity of future changes (especially if the zoning schemes are particularly restrictive). However, keeping them independent was instrumental to the purpose of the paper, which was to contribute to fill a gap identified in the current literature, by testing the effects of different arrangement of land uses on ecosystem services.

The research made use of InVEST, a state-of-the-art GIS tool to model the future provision of multiple ecosystem services. The use of InVEST has been demonstrated in several scientific papers over the last years (Nelson et al., 2009 and 2010; Polasky et al., 2011; Kareiva et al., 2011), and this tool currently offers the most complete suite of models for the spatial representation of services. Although a discussion on the models’ performance and limitations goes beyond the objectives of this paper, the reliability of the results is obviously an issue of concern, particularly if we need to choose among policies whose differences is quite narrow (in some case less than 10%, as shown in Figure 7). An obvious strategy to improve the reliability of the results would be to improve the input data, which in this research were limited to those data that are typically available in a regional planning setting. To test the feasibility of the approach in actual settings, only existing data generated by local authorities and research institutions were used, complemented by available global-scale information (e.g., on carbon storage in different climate domains). Fieldwork can be performed to calibrate and refine such data. Additionally, sensitivity analysis can be run to test the robustness of the ranking of the policies, and to determine the most critical input variables for the different ecosystem service models.

More generally, the modeling of ecosystem services performed here can be improved in different ways. First of all, the effects of climate change can be included by using available predictions to model the future provision of services in a dynamic way (i.e., by changing precipitation and temperature through time). The reduction in precipitation and the increased frequency of droughts, in particular, are issues of concern in the study area. Further improvements in the scenarios consist of considering also the natural evolution of ecosystems (e.g., scrub to forest transition and transition among different forest types), as well as important parameters, such as timber harvesting frequency in different parcels. However, this requires extensive fieldwork to produce more detailed data on land use and land cover. The timber production model can be improved by considering issues of accessibility and the fact that the linear relationship between production and plantation area might break down for small plantations areas, due to edge effects (Balmford et al., 2008). One final note concerns the set of ecosystem services addressed here. The study can be complemented by modeling additional important services, such as water yield, i.e., the amount of available fresh water. This service was originally included in the set but discarded during the analysis due to lack of data. Water yield, and its seasonal fluctuation, is critically associated with land use changes occurring in the study region, and particularly with the replacement of native forest with conifer plantation (Lara et al., 2009; Little et al., 2009). Its inclusion in the study would provide a more comprehensive picture of the effects of the different zoning policies.

Spatial planning eventually results in actions that may affect the provision of a wide range of ecosystem services and that are instrumental to their conservation and enhancement (TEEB, 2010). Hence, spatial planning decisions would benefit from a systematic consideration of their effects on ecosystem services. Despite the increasing body of literature, the use of ecosystem service concepts to support real-life decision-making processes is still limited, and there is a need to move “from conceptual framework and theory to practical integration of ecosystem services into decision making, in a way that is credible, replicable, scalable and sustainable” (Daily and Matson, 2008). This research presented an approach to include information of ecosystem services in a specific aspect of spatial planning, namely the comparison of land use zoning policies. This comparison was characterized by two elements that are further discussed next: the use of multiple metrics to express the effects on ecosystem services and the representation of the trade-offs among services.

The literature has emphasized that studies in ecosystem services should not focus only on the average level of services, because local “bottlenecks” may be very important in constraining service supply and use (see for instance van Jaarsveld et al. 2005). By building on this concept, additional metrics were computed in this study to describe effects such as the degradation and preservation of services across the landscape. The inclusion of these metrics offers a broader picture of the consequences of the different policies. However, the proposed metrics are meant to be exemplary only, and further or different metrics can be developed, ideally to answer case-specific questions. Analogously, in this study the thresholds that define the preservation and degradation conditions were arbitrarily set. These thresholds can be revised and adapted in the light of available scientific evidence and/or by considering policy objectives. For example, the importance of maintaining portions of the landscape where a given service is well-preserved can be suggested by the state of conservation and/or the strategic relevance of the service at different scales (e.g., at national level). Finally, as shown in this research, the computation of similar metrics requires spatially-explicit representation of service provision. We consider the availability of this information to be a key element to mainstream ecosystem services in decision making.

Spatial planning is about resolving conflicts on competing demand for limited resources and uneven distribution of costs and benefits. Hence, the analysis of trade-offs associated to planning choices represents a pivotal issue. By driving future land use changes, spatial planning decisions affect the relative mix of ecosystem services within a region, determining trade-offs among them. Such trade-offs can be an explicit choice but can also arise without awareness (Rodriguez et al., 2006). Typically, trade-offs involving regulating services and trade-offs across space (e.g., when benefits accrue locally, but the cost are borne elsewhere) are less visible, hence less likely to be properly addressed. This study generated a representation of the trade-offs between provisioning and regulating services, and between regulating services that occur at different spatial scales. The resulting trade-offs curves are useful to identify inefficient alternatives, but also to highlight conditions such as “small loss-big gain” (when a small reduction in one service has major benefit for another service), or vice-versa (DeFries et al., 2004). This information provides valuable support to planning by narrowing the scope of potential decisions.

One last note addresses a potential further improvement of this research, currently under development: the analysis of how different policies affect the fruition of services by beneficiaries. Policies may differ not only in terms of changing the flow of services across the landscape, but also in being more or less effective in providing the services where they are most needed and used. A conceptual framework needs to be built to link changes in services with expected effects on the well-being of beneficiaries (for instance in terms of material assets, sufficient food, safety, health, etc.). This can be achieved by combining the output of this study with spatially-resolved socioeconomic variables (e.g., population density, livelihood systems, poverty indicators) that estimate the appropriation of services by people. In this way, the scope of the policy comparison can be expanded to include the analysis of trade-offs among different groups of beneficiaries, characterised by different needs and levels of dependency on the selected services.

References

Acosta-Jamett, G., Simonetti, J., Bustamante, R., Dunstone, N. 2003. Metapopulation approach to assess survival of Oncifelis guigna in fragmented forests of central Chile: A theoretical model. J Neotrop Mammal 10:217-229.

Aldrich, J.H., Nelson, F.D. 1984. Linear probability, logit, and probit models. Sage University Paper series on Quantitative Applications in the Social Sciences. Sage, Newbury Park, CA.

Balmford A., Rodrigues A., Walpole M., et al. 2008. Review on the Economics of Biodiversity Loss: Scoping the Science. European Commission. Available at: http://ec.europa.eu/environment/nature/biodiversity/economics/pdf/scoping_science.pdf.

Beier, C. M., Patterson, T. M., Chapin, F. S. 2008. Ecosystem services and emergent vulnerability in managed ecosystems: A geospatial decision-support ool. Ecosystems 11 (6): 923-938.

Bell, E. 1974. Markov analysis of land use change—an application of stochastic processes to remotely sensed data. Socio-Eco Plan Sci. 8 (6): 311-316.

Bennett, E.M., Peterson, G.D., Gordon, L.J. 2009. Understanding relationships among multiple ecosystem services. Ecol Lett 12 (12): 1394-404.

Birch, J. C., Newton, A. C., Aquino, C. A., Cantarello, E., Echeverría, C., Kitzberger, T., et al. 2010. Cost-effectiveness of dryland forest restoration evaluated by spatial analysis of ecosystem services. Proc Natl Acad Sci USA 107 (50): 21925-21930.

Bonde, J., Cherp, A. 2000. Quality review package for strategic environmental assessments of land use plans. Impact Assess Proj Apprais 18 (2): 99-110.

Brosi, B.J., Armsworth, P.R., Daily, G.C. 2008. Optimal design of agricultural landscapes for pollination services. Conserv Lett 1 (1): 27-36.

Carpenter, S. R., Bennett, E.M., Peterson, G.D. 2006. Scenarios for ecosystem services: An overview. Ecol Soc 11 (1).

Carpenter, S.R., Mooney, H.A., Agard, J., Capistrano, D., Defries, R.S., Díaz, S., et al. 2009. Science for managing ecosystem services: Beyond the Millennium Ecosystem Assessment. Proc Natl Acad Sci USA 106 (5): 1305-12.

Chan, K.M.A., Shaw, M.R., Cameron, D.R., Underwood, E.C., Daily, G.C. 2006. Conservation planning for ecosystem services. PLoS biol 4 (11): e379.

Clark, W.A., Hosking, P.L. 1986. Statistical Methods for Geographers. John Wiley & Sons, New York.

Clark, W.C., Dickson, N.M. 2003. Sustainability science: The emerging research program. Proc Natl Acad Sci USA 100 (14): 8059-8061.

CONAF, CONAMA, BIRF, Universidad Austral de Chile, Pontificia Universidad Católica de Chile y Universidad Católica de Temuco. 1999. Catastro y Evaluación de los Recursos Vegetacionales Nativos de Chile. Informe Nacional con Variables Ambientales. Santiago de Chile, 88 p.

Costanza, R. 2008. Ecosystem services: Multiple classification systems are needed. Biol Conserv 141: 350-352.

Costanza, R., d'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon. B. et al. 1997. The value of the world's ecosystem services and natural capital. Nature 387:253-260.

Daily, G.C. (Ed). 1997. Nature's Services: Societal Dependence on Natural Ecosystems. Island Press, Washington, DC, 392 pp.

Daily, G.C., Matson, P.A. 2008. Ecosystem services: From theory to implementation. Proc Natl Acad Sci USA 105 (28): 9455-6.

Daily, G.C., Polasky, S., Goldstein, J., Kareiva, P.M., Mooney, H.A., Pejchar, L. et al. 2009. Ecosystem services in decision making: Time to deliver. Front Ecol Environ 7 (1): 21-28.

Dalal-Clayton, B., Sadler, B. 2011. Sustainability Appraisal: A Sourcebook and Reference Guide to International Experience. Earthscan, London.

Dalal-Clayton, B., Sadler, B. 2005. Strategic Environmental Assessment: A Sourcebook and Reference Guide to International Experience. Earthscan, London.

Dale, V.H., Pearson, S.M., Offerman, H.L., Neill, R.V.O. 1994. Relating patterns of land-use change to faunal biodiversity in the Central Amazon. Conserv Biol 8 (4): 1027-1036.

DeFries, R.S., Foley, J.A., Asner, G.P. 2004. Land use choices: Balancing human needs and ecosystem function. Front Ecol Environ 2 (5): 249-257.
Duinker, P.N., Greig, L.A. 2007. Scenario analysis in environmental impact assessment: Improving explorations of the future. Environ Impact Assess Rev 3:206-219.
Dunstone, N., Durbin, L., Wyllie, I., Freer, R., Jamett, G.A., Mazzolli, M., et al. 2002. Spatial organization, ranging behaviour and habitat use of the kodkod (Oncifelis guigna) in southern Chile. J Zool 257 (1): 1-11.
Eastman, J.R. 2009. IDRISI Taiga. Guide to GIS and Image Processing. Clark Labs. Clark University. Worcester, MA.
Eastman, J.R., Jin, W., Kyem, P.A.K., Toledano, J. 1995. Raster procedures for multi-criteria/multi-objective decisions. Photogramm Eng Rem S 61 (5): 539-547.
Echeverria, C., Coomes, D., Salas, J., Rey Benayas, J., Lara, A., Newton, A. 2006. Rapid deforestation and fragmentation of Chilean temperate forests. Biol Conserv 130 (4): 481-494.
Egoh, B., Reyers, B., Rouget, M., Richardson, D., Lemaitre, D., Vanjaarsveld, A. 2008. Mapping ecosystem services for planning and management. Agr Ecosyst Environ 127 (1-2): 135-140.
European Commission (EC). 1997. Compendium of European Planning Systems. Regional Development Studies Report 28. Office for Official Publications of the European Communities, Luxembourg.
Fisher, B., Turner, R., Morling, P. 2009. Defining and classifying ecosystem services for decision making. Ecol Econ 68 (3): 643-653.
Foley, J.A., Defries, R., Asner, G.P., Barford, C., Bonan, G., Carpenter, S.R., et al. 2005. Global consequences of land use. Science 309 (5734): 570-4.
Geneletti, D., Bagli, S., Napolitano, P., Pistocchi, A. 2007. Spatial decision support for strategic environmental assessment of land use plans. A case study in southern Italy. Environ Impact Assess 27 (5): 408-423.
Geneletti, D. 2011. Environmental assessment of spatial plan policies through land use scenarios: A study in a fast-developing town in rural Mozambique. Environ Impact Assess doi:10.1016/j.eiar.2011.01.015
Gobierno de Chile. 2009. Catastro de uso del suelo y vegetación. Monitoreo y actualización region de La Araucanía. Ministerio de Agricultura, Santiago de Chile.
Gontier, M., Mörtberg, U., Balfors, B. 2010. Comparing GIS-based habitat models for applications in EIA and SEA. Environ Impact Assess 30 (1): 8-18.
Grêt-Regamey, A., Bebi, P., Bishop, I.D., Schmid, W.A. 2008. Linking GIS-based models to value ecosystem services in an Alpine region. J Environ Manage 89 (3): 197-208.
Haynes, R. 2003. An analysis of the timber situation in the United States: 1952 to 2050. U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. General Technical Report PNW-GTR-560. Portland, Oregon.
Hein, L., van Koppen, K., de Groot, R.S., van Ierland, E.C. 2006. Spatial scales, stakeholders and the valuation of ecosystem services. Ecol Econ 57:209-228.
Hernandez, P.A., Graham, C.H., Master, L.L., Albert, D.L. 2006. The effect of sample size and species characteristics on performance of different species distribution modeling methods. Ecography 29:773–85.
IAIA (International Association for Impact Assessment). 2002. Strategic environmental assessment. Performance criteria. Special publication series No. 1.
IPCC (The Intergovernmental Panel on Climate Change). 2006. IPCC Guidelines for National Greenhouse Gas Inventories. Prepared by the National Greenhouse Gas Inventories Programme, Eggleston H.S., Kareiva, P., Tallis, H., Ricketts, Taylor H., Daily, G.C., and Polasky, S. 2011. Natural Capital. Theory and Practice of Mapping Ecosystem Services. Oxford University Press, Oxford, UK, p.432.
Jack, B.K., Kousky, C., Katharine, K., Sims, R.E. 2008. Designing payments for ecosystem services: Lessons from previous experience with incentive-based mechanisms. Proc Natl Acad Sci USA 105 (28): 9465-70.
Lara, A., Little, C., Urrutia, R., McPhee, J., Álvarez-Garretón, C., Oyarzún, C., et al. 2009. Assessment of ecosystem services as an opportunity for the conservation and management of native forests in Chile. Forest Ecol Manag 258 (4): 415-424.

Lee, S.-W., Hwang, S.-J., Lee, S.-B., Hwang, H.-S., Sung, H.-C. (2009). Landscape ecological approach to the relationships of land use patterns in watersheds to water quality characteristics. Landscape Urban Plan 92 (2): 80-89.
Little, C., Lara, A., McPhee, J., Urrutia, R. (2009). Revealing the impact of forest exotic plantations on water yield in large scale watersheds in South-Central Chile. Journal Hydrol 374 (1-2): 162-170.
MA (Millennium Ecosystem Assessment). 2005. Ecosystems and human well-being: The assessment series (four volumes and summary). Island Press, Washington, DC.
Metzger, M.J., Rounsevell, M. D., Acosta-Michlik, L., Leemans, R., Schröter, D. 2006. The vulnerability of ecosystem services to land use change. Agr Ecosyst Environ 114(1), 69-85.
Mideplan (Ministerio de Planificación y Cooperación), 2009. Encuesta de Caracterización Socioeconómica Nacional (CASEN). Gobierno de Chile. Available at: http://www.mideplan.gob.cl/casen/.
Myers, N., Mittermeier, R.A., Mittermeier, C.G., Fonseca, G.A, Kent, J. 2000. Biodiversity hotspots for conservation priorities. Nature 403 (6772): 853-8.
Naidoo, R, Balmford, A., Costanza, R., Fisher, B., Green, R.E., Lehner, B., et al. 2008. Global mapping of ecosystem services and conservation priorities. Proc Natl Acad Sci USA 105 (28): 9495-500.
Nelson, G.C., Bennett, E., Berhe, A.A., Cassman, K., DeFries, R., Dietz, T, et al. 2006. Anthropogenic drivers of ecosystem change: An overview. Ecol Soc 11 (2).
Nelson, E., Mendoza, G., Regetz, J., Polasky, S., Tallis, H., Cameron, Dr., et al. 2009. Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. Front Ecol Environ 7 (1): 4-11.
Nelson, E., Sander, H., Hawthorne, P., Conte, M., Ennaanay, D., Wolny, S., et al. 2010. Projecting global land-use change and its effect on ecosystem service provision and biodiversity with simple models. PLoS ONE, 5 (12): e14327.
Noble, B.F. 2008. Strategic approaches to regional cumulative effects assessment: A case study of the Great Sand Hills, Canada. Impact Assess Proj Apprais 26 (2): 78-90.
NRC (National Research Council). 2005. Valuing ecosystem services: Toward better environmental decision-making. National Academies Press, Washington, DC, 277 p.
ODEPA and CIREN. 2006. Catastro Frutícola IX Región de la Araucanía. Actualización 2006. Oficina de Estudios y Políticas Agrarias y Centro de Información de Recursos Naturales. Santiago de Chile.
OECD (Organisation for Economic Development and Co-operation). 2008. Strategic environmental assessment and ecosystem services. Advisory notes. Available at www.seataskteam.net/guidance.php.
OECD (Organisation for Economic Development and Co-operation). 2006. Applying strategic environmental assessment. Good practice guidance for development co-operation. DAC Guidelines and Reference series. Available at www.seataskteam.net/guidance.php.
Partidário, M.R. 2007a. Scale and associated data — What is enough for SEA needs? Environ Impact Assess 27:460-478.
Partidário, M.R. 2007b. Strategic Environmental Assessment Good Practice Guidance: Methodological Guidance. Lisboa: Agência Portuguesa do Ambiente.
Phillips, S.J., Dudík, M., Schapire, R.E. 2004. A maximum entropy approach to species distribution modelling. Proceedings of the twenty-first international conference on machine learning. Banff, Alberta: ACMPress. p. 83.
Phillips, S.J., Anderson, R.P., Schapire, R.E. 2006. Maximum entropy modeling of species geographic distributions. Ecol Model 190:231–59.
Polasky, S., Nelson, E., Pennington, D., Johnson, K.A. 2011. The impact of land-use change on ecosystem services, biodiversity and returns to landowners: A case study in the state of Minnesota. Environ Res Econ 48 (2): 219-242.
Pope, J., Annandale, D., Morrison-Saundersm A. 2004. Conceptualizing sustainability assessment. Environ Impact Assess Rev 24 (6): 595-616.
PRDUyT. 2005. Actualización Plan Regional de Desarrollo Urbano Región de La Araucanía. Laboratorio de Planificación Territorial, Universidad Católica de Temuco.

Raudsepp-Hearne, C., Peterson, G.D., Bennett, E.M. 2010. Ecosystem service bundles for analyzing tradeoffs in diverse landscapes. Proc Natl Acad Sci USA 107 (11): 5242-7.
Rodríguez, J.P., Beard, T.D., Bennett, E.M., Cumming, G.S., Cork, S.J., Agard, J., et al. 2006. Trade-offs across space, time, and ecosystem services. Ecol Soc 11 (1).
Sanderson, J., Sunquist, M.E., Iriarte, A.W. 2002. Natural history and landscape-use of Guignas (Oncifelis guigna) on Isla Grande de Chiloé , Chile. J Mammal 83 (2): 608-613.
Shearer, A.W. 2005. Approaching scenario-based studies: Three perceptions about the future and considerations for landscape planning. Environ Plann B 32 (1): 67-87.
Slootweg, R., van Beukering, P. 2008. Valuation of ecosystem services and strategic environmental assessment. Lessons from influential cases. Netherlands Commission for Environmental Assessment, Utrecht.
Subdere. 2010. Plan Regional de Ordenamiento Territorial. Contenido y procedimiento. Gobierno de Chile, Subsecretaría de Desarrollo Regional y Administrativo.
Tallis, H., Polasky, S. 2009. Mapping and valuing ecosystem services as an approach for conservation and natural-resource management. Ann NY Acad Sci 1162: 265-83.
Tallis, H.T., Ricketts, T., Guerry, A.D., Nelson, E, Ennaanay, et al. 2011. InVEST 2.1 beta User's Guide. The Natural Capital Project, Stanford.
TEEB 2010. TEEB for Local and Regional Policy Makers Report. The economics of ecosystems and biodiversity. Available at: http://www.teebweb.org/ForLocalandRegionalPolicy/LocalandRegionalPolicyMakersChapterDrafts/tabid/29433/Default.aspx
Therivel, R. 2004. Strategic Environmental Assessment in Action. Earthscan, London.
Turner, M.G. 1989. Landscape ecology: The effects of pattern on process. Annu. Rev Ecol Syst 20: 171-197.
Turner II, B.L. 2010. Sustainability and forest transitions in the southern Yucatán: The land architecture approach. Land Use Policy 27 (2): 170-179.
van Jaarsveld, A. S. , Biggs, R., Scholes, R.J., Bohensky, E., Reyers, B., Lynam, T., et al. 2005. Measuring conditions and trends in ecosystem services at multiple scales: The Southern African Millennium Ecosystem Assessment (SAfMA) experience. Philos Tr Soc Lon B 360 (1454): 425-41.
Villard, M.A., Trzcinski, M. K., Merriam, G. 1999. Fragmentation effects on forest birds: Relative influence of woodland cover and configuration on landscape occupancy. Conserv Biol 13 (4): 774-783.
Willemen, L., Hein, L., Verburg, P.H. 2010. Evaluating the impact of regional development policies on future landscape services. Ecol Econ 69 (11): 2244-2254.
Willis, M.R., Keller, A.A. 2007. A framework for assessing the impact of land use policy on community exposure to air toxics. J Environ Manage 83 (2): 213-27.
Wischmeier, W.H., Smith, D.D. 1978. Predicting rainfall erosion losses: A guide to conservation planning. U.S. Dept. of Agriculture. Agriculture Handbook 537. Washington, DC.
Zhu, Z., Bai, H., Xu, H., Zhu, T. 2010. An inquiry into the potential of scenario analysis for dealing with uncertainty in strategic environmental assessment in China. Environ Impact Assess Rev. doi:10.1016/j.eiar.2010.02.001.