Journal of Land Use Science
Publication details, including instructions for authors and subscription information:
http://www.tandfonline.com/loi/tlus20

Assessing ecological and social uncertainty in the evaluation of land-use impacts on ecosystem services

Fabien Quétier [a b] , Sandra Lavorel [a b] , Sophie Daigney [a b] & Jacqueline de Chazal [a b]

[a] Laboratoire d'Ecologie Alpine, Université Joseph Fourier, Grenoble Cedex, France

[b] Station Alpine Joseph Fourier, Université Joseph Fourier, Grenoble Cedex, France

Version of record first published: 19 Aug 2009.

To cite this article: Fabien Quétier , Sandra Lavorel , Sophie Daigney & Jacqueline de Chazal (2009): Assessing ecological and social uncertainty in the evaluation of land-use impacts on ecosystem services, Journal of Land Use Science, 4:3, 173-199

To link to this article: http://dx.doi.org/10.1080/17474230903036667

Journal of Land Use Science
Vol. 4, No. 3, September 2009, 173–199

Taylor & Francis
Taylor & Francis Group

Assessing ecological and social uncertainty in the evaluation of land-use impacts on ecosystem services

Fabien Quétier[a,b,$,*], Sandra Lavorel[a,b], Sophie Daigney[a,b†] and Jacqueline de Chazal[a,b‡]

[a]*Laboratoire d'Ecologie Alpine, Université Joseph Fourier, Grenoble Cedex, France;* [b]*Station Alpine Joseph Fourier, Université Joseph Fourier, Grenoble Cedex, France*

Ecosystem services provided by semi-natural agro-ecosystems in Europe are vulnerable to land-use change. Typically, vulnerability assessments include uncertainty around the future political and economic context of land-use systems through the use of scenarios. Here, we use a novel indicator sequence to assess the acceptability of scenario impacts on ecosystem properties to social representations of ecosystem services in a case study area. The sequence can incorporate multiple sources of uncertainty, and we highlight the key role of plant dispersal in driving ecosystem service provision by comparing outcomes from different methods for projecting scenario impact on ecosystem properties. By varying which properties underly valued ecosystem services and by comparing how people could adapt their future demand for ecosystem services we also incorporate social uncertainty. We find that social representations most negatively affected by litter accumulation are the most vulnerable.

Keywords: complexity; modelling; ecosystem services; evaluation; uncertainty; central French Alps; *Festuca paniculata* grasslands

Introduction

Changes in the intensity and spatial patterns of land-use can lead to the loss of important ecosystem services (Vitousek, Mooney, Lubchenco and Melillo 1997; Foley *et al*. 2005; Díaz, Fargione, Chapin and Tilman 2006). In Western Europe, many semi-natural ecosystems and the ecosystem services they provide are related to socially and economically marginal agricultural production systems that are very dependent on public subsidies. This makes these ecosystem services particularly vulnerable to possible evolutions of public policies regarding agriculture and the environment, as well as rural development policies and the organization of agricultural markets (Baldock *et al*. 2002; Dobbs and Pretty 2004; Rounsevell *et al*. 2006).

Future scenarios describing these possible evolutions are central to assessing vulnerability (e.g. Peterson, Cumming and Carpenter 2003b; Verburg, Rounsevell and Veldkamp 2006b; Westhoek, van den Berg and Bakker 2006; Eickhout, van Meijl, Tabeau and van Rheenen 2007). It is on the basis of such scenarios that impacts on society in general

Present address: [$]*Instituto Multidisciplinario de Biología Vegetal, CONICET – Universidad Nacional de Córdoba, Casilla de correo 495, 5000, Córdoba, Argentina;* [†]*ADEME Limousin, 38 ter Avenue de la Libération, BP 20259, 87007 Limoges Cedex, France;* [‡]*NationalCentre for Epidemiology and Population Health, The Australian National University, Canberra, ACT, 0200, Australia*
*Corresponding author. E-mai: fabien.quetier@ecosystem-services.org

ISSN 1747-423X print/ISSN 1747-4248 online

DOI: 10.1080/17474230903036667
http://www.informaworld.com

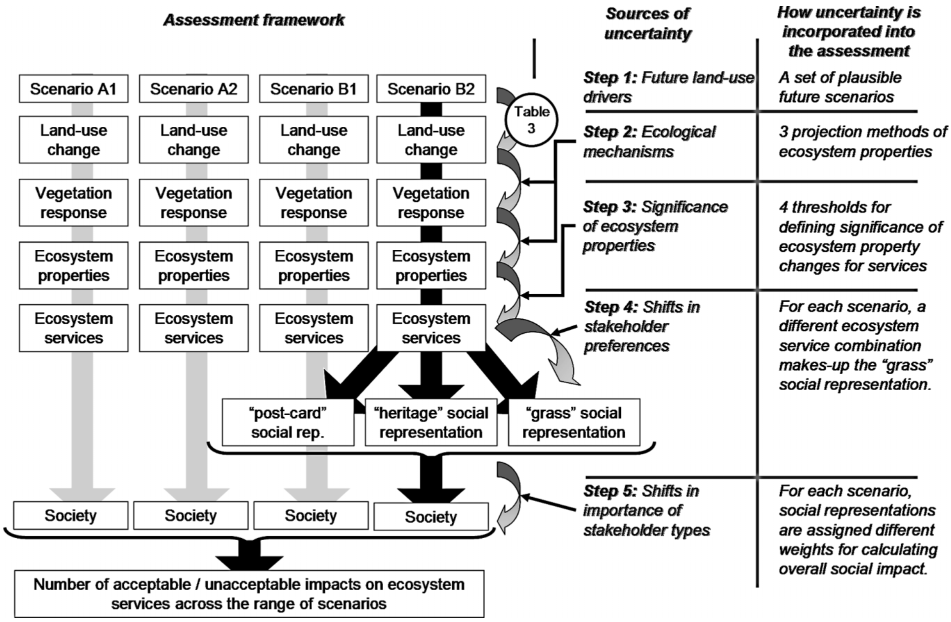

Figure 1. Illustration of the assessment framework. The information flow is illustrated on the left-hand side: scenario-based land-use changes are successively translated into land-cover (vegetation) change, changes in ecosystem properties and ecosystem service impacts before being evaluated as acceptable (positive or neutral impacts) or unacceptable (negative impacts) to different social representations. These evaluations are then grouped to provide overall social evaluations of scenario acceptability. Finally, integrating these evaluations across scenarios incorporates exposure into the vulnerability assessment. On the right-hand side, steps 1–5 detail which sources of uncertainty were incorporated into the assessment and how this was done.

(or stakeholders in particular) are then projected (e.g. Baker *et al*. 2004; Rounsevell, Ewert, Reginster, Leemans and Carter 2005; Verburg *et al*. 2006b; Sheate, Partidário, Byron, Bina and Dagg 2008). Formulating scenarios allows for the voluntary incorporation of uncertainty related to the indeterminacy of future developments (Peterson *et al*. 2003b – for a meta-analysis in a land-use context, see Busch 2006). On the other hand, it is often argued that uncertainties derived from the projection of impacts – i.e. related to projection models themselves – should be minimized (Rounsevell *et al*. 2006 but see Castella, Pheng Kam, Dinh Quang, Verburg and Thai Hoanh 2007).

For projecting land-use change impacts on ecosystems, models based on current relationships between ecosystem properties and past and present land-use (static models, e.g. Quétier, Thébault and Lavorel 2007b) presuppose that these relationships will remain valid in the future. To overcome this limitation, mechanistic models can be developed but limited knowledge of ecological mechanisms often makes uncertainty unavoidable (e.g. Schumaker, Ernst, White, Baker and Haggerty 2004; Quétier, Lavorel, Thuiller and Davies 2007a; Dorrough, Vesk and Moll 2008). Similarly, uncertainties are also unavoidable when making projections of future societal response to scenarios and scenario impacts (e.g. when evaluating adaptive capacity – Castella and Verburg 2007). In particular, evaluating future scenarios on the basis of current stakeholder preferences for a set of ecosystem services requires that these stay the same and that the relationships between ecosystem properties and the services they provide also remain unmodified (e.g. Parkins and MacKendrick 2007).

In this paper, we explicitly consider uncertainties related to (1) the projection of land-use change impacts on ecosystems (ecological mechanisms), (2) how ecosystems are valued (the relationships between ecosystem properties and ecosystem services) and (3) social adaptation (Which ecosystem services will be most important to particular stakeholders or groups of stakeholders in the future?). We propose to appraise the individual and combined effects of these three sources of uncertainty on the assessment of land-use change impacts on ecosystem services in a case study landscape of the central French Alps dominated by semi-natural grasslands.

We use an assessment framework first proposed by de Chazal, Quétier, Lavorel and van Doorn (2008) and built as an indicator sequence whose successive steps are illustrated in Figure 1. The assessment framework is built around the projection and subsequent social evaluation of changes in the ecosystem properties underlying ecosystem service provision. These projections are made using quantitative linkages between ecosystem properties and vegetation (land-cover) response to land-use. In this study, these linkages are made easier by using a functional approach to describe and model vegetation response to land-use change. Rather than on species composition, such an approach is based on plant functional diversity indices such as the distribution of plant trait values in the vegetation (e.g. Díaz *et al*. 2007; Quétier *et al*. 2007a).

On the basis of social survey data, projected changes in ecosystem properties are subsequently interpreted in terms of ecosystem service provision, and the latter provides input for a social evaluation of each scenario's impacts. The framework recognizes that not all stakeholders share a common evaluation, and it allows any stakeholder classification to be used for differentiating their views. In this study, we used social representations, identified through social surveys (Rivoal 2004; Quétier 2006), as a way of describing the various ways in which people relate to ecosystems in the case study site. The social evaluations of the different stakeholders are finally combined to estimate an overall acceptability of each scenario. Scenarios could then be ranked based on their acceptability, either between or across the different social representations. Although we recognize that society cannot (or should not) be restricted to the sum of its component interest groups, our assessment framework aims primarily to identify potential winners and losers rather than identifying a consensus (O'Brien and Leichenko 2000, 2003).

It is also important to point out that because of its focus on ecosystem services, the assessment framework and the results obtained in this study ignore many social and economic components that people would take into account when making an overall evaluation of scenarios (such as economic development or social justice). The importance of these components was explored by presenting the scenarios to a range of stakeholders in the case study site, both residents and visitors, during individual interviews and focus group discussions (Daigney 2005). Their opinions offer an interesting point of comparison for our results from the point of view of those who will have to adapt to – and in part determine – future land-use change in the study area. We used the same social representations to interpret and categorize the preferences they expressed as for the analysis, making the two approaches comparable.

Acknowledging ecological and social uncertainties in assessing ecosystem impacts of land-use change scenarios

We assessed the effects of the three sources of uncertainty introduced above by (1) considering three methods of increasing complexity for projecting land-use change effects on ecosystem properties, (2) by applying four increasing detection thresholds below which a change in ecosystem property is considered insignificant and finally (3) by making the importance of various ecosystem services or social representations consistent with each scenario's storyline. These sources of uncertainty operate at different steps in the indicator sequence, as illustrated in Figure 1. The assessment framework is flexible enough to allow for different types of relations between land-use, biodiversity, ecosystem properties, ecosystem services and stakeholders to be combined at each one of these steps (de Chazal *et al*. 2008).

Uncertainty related to ecological mechanisms

Land-use and land-cover change models have been very successful in linking socio-economic drivers with changing patterns in land-cover across multiple spatial and temporal scales (Verburg, Kok, Pontius and Veldkamp 2006a) and have been particularly useful in making scenario-based projections (e.g. Rounsevell *et al*. 2005; Alcamo *et al*. 2006). However, these models need to more explicitly incorporate ecological mechanisms operating at finer scales, within landscapes, in particular if they are to help bridge the gap between land-cover and ecosystem services (Chhabra *et al*. 2006; Díaz *et al*. 2007). Assessing land change impacts on ecosystem service provision requires a systematic approach where ecological and socio-economic systems interact (e.g. Peterson, Carpenter and Brock 2003a) and where ecological uncertainties, resulting from spatio-temporal variation in environmental factors or ignorance as to which ecological processes should be represented in the models, will flow through to socio-economic outcomes and thus land-use decisions (see Dorrough *et al*. 2008 for an example of coupled socio-economic and ecological models in grasslands). Acknowledging such uncertainty is now considered the best practice in environmental modelling and forecasting (e.g. Sutherland and Watkinson 2001; Thuiller 2004; Refsgaard, van der Sluijs, Hojberg and Vanrolleghem 2007).

To incorporate ecological uncertainty, we compared three projection methods of increasing complexity for projecting ecosystem property response to land-use change scenarios (step 2 in Figure 1). The first method involved directly extrapolating field data on ecosystem properties measured in different grassland types (Quétier *et al*. 2007b). In contrast, the second and third methods combined a vegetation dynamics model that simulates changes in plant functional type abundance and hence functional diversity with statistical models predicting the impact of these changes on relevant ecosystem properties (these models are detailed in Quétier 2006 and Quétier *et al*. 2007a). Unlike the first method, these included a dynamic component. The third method was a modification of the second, whereby dispersal limitations were imposed on a particular plant functional type: large unpalatable tussock grasses. These have a strong effect on community structure, functional diversity and ecosystem properties (Quétier *et al*. 2007a,b). Comparing method 1 with methods 2 and 3 enabled us to evaluate how much was gained by including ecological mechanisms related to the dynamics of plant functional diversity in our assessment. Comparing methods 2 and 3 enabled us to assess how important dispersal, a landscape-level ecological mechanism that is poorly known, can be for projecting land-use change impacts on ecosystems. All three projection methods were obtained on the basis of field data (Quétier *et al*. 2007a,b).

Uncertainty related to perceptions of ecosystem properties

Considering a change in ecosystem property as significant or not involves judgement, and variations in judgement add uncertainty to the assessment of scenario impacts. Firstly, the significance of projected changes in ecosystem properties might be decided on the basis of current scientific methodologies [e.g. the precision allowed by projection methods or measurement apparatus, sampling methods – see Clapham, Aguilar and Hatch (2008) for an example in wildlife management]. These are likely to change in the future, shifting the boundary between significant and insignificant impacts. Secondly, it is not easy to decide how much change in each ecosystem property is relevant or significant to a stakeholder without a complex quantitative assessment of how they perceive these properties and their effects on ecosystem service provision. As an example, is a 1% change in Simpson's index of plant diversity significant for a nature reserve manager? These stakeholder perceptions are also likely to evolve with time. To explore this source of uncertainty, we applied three detection thresholds to projected changes in ecosystem properties whereby properties whose responses were below the threshold were considered as unmodified (i.e. undetected or insignificant) in the following steps of the assessment (step 3 in Figure 1).

Uncertainty related to social adaptations

We explored two facets of adaptive capacity that generate uncertainty in assessing scenario impacts on ecosystem services and the resulting vulnerability. The first facet recognizes that depending on the context each scenario offers, stakeholders might change the combination of ecosystem services they benefit from and thus modify their criteria for evaluating scenarios. The second facet involves possible shifts in the relative importance of different stakeholder groups and their ecosystem services in the overall evaluation of scenario impacts, depending on which scenario unfolds. Testing for this last effect allowed us to investigate a form of adaptive capacity – that of changing social representations of the environment – which is rarely touched upon in assessments of vulnerability (e.g. Metzger, Rounsevell, Acosta-Michlik, Leemans and Schröter 2006; Schröter *et al*. 2006). We investigated the effects of these two sources of uncertainty independently.

Material and methods

Study site

The study site is set on the south facing slopes of the valley above the village of Villar d'Arène in the central French Alps (45.04°N, 6.34°E). Climate is sub-alpine with a mean annual rainfall of 956 mm and mean monthly temperatures at 1650 m (lowest point in the study site) ranging between −2.6°C in January and 13°C in July.

When the local population was at its peak around 1830 (Rousset 1977), arable fields used to cover the lower slopes (1650–2000 m) and large expanses of natural grasslands were cut for hay between 1800 and 2500 m. As mountain agriculture lost ground to more profitable activities and rural exodus drained the population away to the cities (from the 1830s until present), former arable fields were abandoned and subsequently converted to grasslands used for hay or grazing. The resulting landscape is dominated by grassland ecosystems that are still used by a small but nevertheless active farming community based on sheep and cattle rearing for lamb and steer production. Former arable land has given the landscape a unique distinctiveness in the form of terraced slopes extending up to 2000 m a.s.l. This 'heritage' of the former land-use system makes this cultural landscape the focus of numerous preservation efforts including subsidies to the remaining farmers (Parc National des Ecrins 2004). Moreover, the contrast between the wilderness of the glaciated Meije massif and the gentle grassland landscape of the opposing south-facing slope is one of the main attractions to Villar d'Arène. Indeed, tourism has taken over agriculture as the dominant economic activity (Parc National des Ecrins 2004). The site is located in the buffer zone of the Ecrins National Park and receives many thousands of visitors annually attracted by opportunities for outdoor recreation, the grandiose mountain setting, its abundant and diverse fauna and flora as well as the traditional or authentic character of villages and farmlands.

Ecosystem services

Perceptions of ecosystem services were elicited using 45 semi-guided interviews. Rather than a predefined list of questions, an interview guide was used to lead the interview from a description of the overall landscape through to a discussion of more detailed descriptions of the local grasslands. We found that local people and visitors referred to a variety of ecosystem services and liabilities from local grasslands (Rivoal 2004; Quétier 2006; Quétier *et al*. 2007a). Farming systems are heavily constrained by winter fodder requirements, and all machinery-accessible grasslands are cut for hay at present. Relative to winter fodder stocks, available grazing area is not limiting in current farming systems. Rather, shepherds recognize that selective grazing favours large unpalatable tussock grasses. Grass quantity is thus an important ecosystem service for hay whilst grass quality (palatability and nutritional value) is an important ecosystem service for grazing. Plant diversity contributes to landscape aesthetics. It also contributes to biodiversity conservation objectives of the Ecrins National Park (Parc National des Ecrins 2004). Some people worry about signs of inappropriate 'stewardship' that diminish cultural heritage value. All local people acknowledge that inappropriate land-use practices can increase snow gliding and associated avalanche and land-slide risk (e.g. Newesely, Tasser, Spadinger and Cernusca 2000; Tasser, Mader and Tappeiner 2003). Relationships between these ecosystem services and underlying ecosystem properties (Table 1) are discussed in Quétier *et al*. (2007a).

Social representations

Social representations do not represent social groups, organizations or individuals. Rather, they correspond to spontaneous and self-consistent descriptions of the grasslands, their

Table 1. Relationships between ecosystem properties, ecosystem services and social representations.

Ecosystem properties	Ecosystem services	Social representations
AGB* in mown grasslands	+ Grass for hay	'grass'
Sward height	+ Grass for grazing	
Lower Leaf tensile strength		
Abundance of legumes	+ Forage quality	
Leaf crude protein content		
Simpson's biodiversity index	+ Flowering diversity	'post-card'
	+ Plant diversity	
		'heritage'
Spring litter in un-mown plots	– Appropriate stewardship	
	+ Snow-gliding risk	Shared concern

Note: The '+' and '–' signs indicate the direction of the ecosystem properties' effects on ecosystem services. Notice that two ecosystem properties contribute to grass for grazing (through palatability, which determines how much can be grazed) and forage quality and that Simpson's biodiversity index contributes to both flower diversity (for its beauty) and plant diversity (for conservation). Snow-gliding risks were recognized by the majority of those interviewed and is not considered part of a social representation in particular. *AGB is above-ground biomass.

landscape and their uses, which can be shared between individuals. On the basis of interviews with local residents and visitors, these representations were associated to different combinations of the ecosystem services described above (Table 1): The 'grass' social representation focused on grasslands as a natural resource to be used for livestock. In the 'post-card' representation, grasslands were only one minor component of the wider mountain landscape, but the aesthetic contribution of biodiversity was nevertheless recognized. The 'heritage' representation focused on grasslands as a natural and cultural heritage to be conserved. For a more in-depth description of these social representations see Rivoal (2004) and Quétier (2006).

Assessment framework and selected vulnerability indicator

The framework uses a set of matrices to translate input data into output data through an indicator sequence described in Figure 1 (de Chazal *et al*. 2008). For each scenario, we first made quantitative projections of scenario impacts on vegetation and ecosystem properties using the three different projection methods (step 2, illustrated in Figure 1 and described in the corresponding subsection). We used qualitative relationships (detailed in Table 1) to translate these scenario impacts on ecosystem properties into impacts on ecosystem services (step 3 in Figure 1). Quantitative values for ecosystem services are often difficult to establish (Díaz *et al*. 2007), generating uncertainty in assessments. We therefore translated quantitative projections of ecosystem properties (i.e. rates of change) into qualitative impacts on ecosystem services (i.e. directions of change, thus losing some information). When two or more ecosystem properties were relevant to an ecosystem service (e.g. both sward height and palatability are relevant for 'grass for grazing' – Table 1), only the biggest absolute change was taken into account (i.e. the bigger increase in one property cancels out the smaller decrease in the other). The framework could accommodate more complex relationships between ecosystem properties and services, but it would be less tractable for the purpose of presenting the approach.

Subsequent steps in the vulnerability assessment were based on the qualitative projections of ecosystem service change obtained. Each of these was evaluated as positive, neutral or negative for each social representation as summarized in Table 1 (step 4 in Figure 1). This forms the basis for the social evaluation of each scenario's impacts on ecosystem services, first from the point of view of each social representation and secondly considered across social representation to approximate an overall social evaluation of the impacts (step 5 in Figure 1).

The framework proposed by de Chazal *et al*. (2008) was designed to allow for multiple perspectives and can generate a number of indicators. We chose to use 'acceptability ratios', i.e. the ratio of acceptable to unacceptable impacts on selected ecosystem services (e.g. if provision of one ecosystem service is improved while that of two other services is negatively affected by a scenario the scenario's acceptability ratio will be 1:2). Selected ecosystem services refer for example to those that make up a social representation. This means that acceptability ratios give an indication of how much more likely it is that people holding this representation will benefit from acceptable impacts rather than suffer from unacceptable ones.

Acceptability ratios would typically be calculated across the whole range of scenarios, in order to capture exposure (Turner *et al*. 2003). This involves summing the number of acceptable (positive and neutral) and the number of unacceptable (negative) scenario impacts on a social representation's selection of ecosystem services under all four scenarios (de Chazal *et al*. 2008). Social representations with the lowest ratio of acceptable to unacceptable impacts will be considered most vulnerable to changes in ecosystem service provision (hereafter simply referred to as vulnerable). Here we also calculated acceptability ratios across projection methods, detection thresholds or alternative combinations of ecosystem services and social representations thereby incorporating the various sources of uncertainty explored in this study into the assessment.

Step 1: Scenarios

Land-use change scenarios were based on local projections of European-wide land-use change scenarios described in Rounsevell *et al*. (2006) and formulated on the basis of global storylines (Nakicenovic and Swart 2000). They opposed a global (type 1 scenario) and a regional (type 2 scenario) future, on the one hand, and a materialist (type A scenario) and an environmentally and socially conscious (type B scenarios) future, on the other. Rounsevell *et al*. (2006) related the global storylines to relevant socio-economic drivers of land-use change at the European scale. In collaboration with local and regional stakeholders of farming, rural development and nature conservation interests, we built on their results to formulate four contrasting land-use scenarios for the case study site of Villar d'Arène (Daigney 2005; Quétier 2006). As argued by Busch (2006) and Audsley *et al*. (2006), we considered that the European-scale scenarios provided by Rounsevell *et al*. (2006) offered a set of contrasting salient and credible (not implausible) futures on which to apply our framework. Involving stakeholders in downscaling European-level scenarios guaranteed that these were locally not implausible (i.e. they were credible) but also legitimate (e.g. Hulse, Branscomb and Payne 2004). We do recognize however that involving local stakeholders could lead to more conservative (less extreme or salient) storylines (e.g. Dockerty, Lovett, Appleton, Bone and Sünnenber 2006).

In this study, A1 was a global and materialist scenario where agro-pastoral land-use is abandoned as financial support to marginal agriculture disappears and European agricultural markets open to international trade (Table 2). Grass for hay thus ceases to be an ecosystem service under A1 as no one benefits from it. A2 was a regional and materialist scenario where financial support dwindles but European agricultural markets go local, giving a premium to local dairy and meat produce. This drives intensification of grass management on the most productive and accessible parts of the study landscape (former arable fields that are currently mown). B1 was a global environmentally and socially conscious scenario where subsidies for mowing-for-biodiversity increase, allowing mowing to continue. The B2 scenario was a regional environmentally and socially conscious future where agricultural subsidies promote self-reliance in local communities as a solution to environmental issues. This favours

Table 2. Land-use change scenarios used to map coherent combinations of disturbance and soil resources across the simulated landscape.

		Future land-use			
Past land-use	Current land-use	A1	A2	B1	B2
Cultivated	Fertilized and mown	None	Fertilized and mown	Mown	Fertilized and mown
	Mown	None	Fertilized and mown	Mown	Fertilized and mown
	Extensively grazed	None	None	None	Fertilized and mown
Mown	Mown	None	None	Mown	Mown
	Extensively grazed	None	None	None	None

sustainable stewardship of the grassland resource through fertilization and mowing. Table 2 illustrates the combined distribution of fertilization and mowing relative to past-ploughing for future scenarios and the current land-use patterns (see also Quétier *et al*. 2007a).

Step 2: Uncertainty related to ecological mechanisms

The formulated land-use scenarios provided us with consistent and not implausible combinations of resources, disturbance and land-use legacies (Table 2) for modelling land-use change effects on ecosystem services at the landscape scale (Tscharntke, Klein, Kruess, Steffan-Denwenter and Carsten 2005). On the basis of these combinations, we generated projections of changes in vegetation characteristics (species composition and plant functional diversity) and ecosystem properties using three different methods. These methods included increasingly complex ecological mechanisms, and we compared the vulnerability outcomes each one generated to explore their importance in the context of a vulnerability assessment. We also incorporated uncertainty related to ecological mechanisms by calculating acceptability ratios across projection methods.

The first projection method directly associated scenario changes in grassland management with values of ecosystem properties measured in the field by Quétier *et al*. (2007b). These authors describe how grasslands in Villar d'Arène have responded to changes in management (including former ploughing) using a state and transition (S&T) model. S&T models offer a flexible conceptual tool for integrating data on community composition, plant traits and ecosystem properties in the context of management-mediated successional dynamics (Briske, Fuhlendorf and Smeins 2003).

Plant traits provide a mechanistic link between vegetation shifts in response to management and ecosystem properties (Díaz *et al*. 2007; Quétier *et al*. 2007a,b). Methods 2 and 3 incorporated such links in the form of predictive statistical models based on field data collected across a broad range of management practices by Quétier *et al*. (2007a). These authors used a trait-based plant classification in a landscape-modelling platform to model community dynamics under the same scenarios as used in this study and then used the predictive models to project ecosystem properties.

Methods 2 and 3 differed in that the latter includes a constraint on the dispersal of large unpalatable late-successional grasses (such as *Festuca paniculata*) (Quétier *et al*. 2007a). These grasses are favoured under decreased land-use intensity and have a disproportional influence on key ecosystem properties such as litter accumulation and palatability, which are relevant to a number of ecosystem services (Table 1, see also Quétier *et al*. 2007a,b). This makes these grasses key ecosystem service providers (*sensu* Kremen 2005). Because these grasses are limited to never-ploughed grasslands in the current landscape, scenario impacts on these ecosystem properties and thus ecosystem services will be very affected by the ability of these species to disperse across the landscape (Quétier 2006). We assessed the effect of ecological uncertainties concerning the dispersal ability of late-successional grasses by comparing projections made with methods 2 (unconstrained dispersal across the whole area) and 3 (local dispersal only) described above.

Step 3: Uncertainty related to perceptions of ecosystem properties

Detection thresholds were set at 1, 5 and 10% change in ecosystem property as compared with present (thresholds were applied to each ecosystem property in step 3 of Figure 1). We compared vulnerability outcomes obtained with different thresholds to explore the effects of differences in how much change in ecosystem properties might be considered significant in the future. By calculating acceptability ratios across thresholds, we thus incorporated uncertainty concerning the perception of these ecosystem properties.

Steps 4 and 5: Uncertainty related to social adaptations

Scenarios lead to important changes in the dominant production systems and associated management of grasslands (Quétier *et al*. 2007a). As farming systems change, the combination of ecosystem services (forage production, grazing) farmers value will evolve, shifting for example between forage quality (e.g. nutritious and easily digestible early growth that is only available in small quantities) and quantity (e.g. late harvests that allow for a larger accumulation of biomass but of lesser quality). The trade-off between grass quantity and quality is one of the key ecosystem service trade-offs in mountain grassland production systems (Fleury 1994; Méot, Hubert and Lasseur 2003).

We explored this avenue for adaptation of the ‘grass’ representation to a new production context by using different weights for different ecosystem services when calculating acceptability ratios for the ‘grass’ social representation (step 4 in Figure 1): In the A1 scenario, ‘grass for grazing’ was given double the importance of ‘forage quality’ (Table 3a). Conversely, in the A2 scenario where milk production requires more quality fodder, the opposite weighting was used (Table 3a). In B1, grasslands are cut for nature conservation and ‘forage quality’ is not an important ecosystem service, but in B2, hay will feed resident flocks through the winter and must therefore combine both large quantities of ‘grass for hay’ and high ‘forage quality’ whereas ‘grass for grazing’ is not limiting and is thus less valued in the ‘grass’ social representation (Table 3a).

As presented and discussed in Rounsevell *et al*. (2006), the global and European-level storylines used in this study are associated to different dominant conceptions of the environment: It is seen as either a resource to be used (A scenarios) or a heritage to be conserved (B scenarios). Locally, under one or the other type of scenario, the relative weight of the three social representations in the socio-political arena could shift to favour one or the other (for example, based on the number of individual citizens who share a representation). To explore the effect of such shifts, we assigned different weights to each social representation under different scenarios (Table 3b) when calculating acceptability ratios across representations (step 5 in Figure 1).

Table 3. Uncertainty from social adaptations: (a) Weighting of different ecosystem services (columns) in calculating acceptability of scenarios (lines) for the ‘grass’ social representation under the adaptation hypothesis. The reference assessment used equal weights (1) for all ecosystem services in each scenario; (b) Weighting of social representations (columns) used to calculate acceptability of scenarios (lines) across all stakeholders under the adaptation hypothesis. In the reference assessment all social representations received equal weight (1) in each scenario.

(a)		Relative importance of ecosystem services for the ‘grass’ social representation			(b)	Relative importance of social representations in overall scenario evaluation		
Ecosystem services		Grass for hay	Grass for grazing	Forage quality	Social rep.	‘Grass’	‘Post-card’	‘Heritage’
Scenarios	A1	0	2	1	A1	2	1	0
	A2	1	1	2	A2	2	0	1
	B1	1	0	0	B1	0	2	1
	B2	2	1	2	B2	0	1	2

In scenarios A, the environment is an economic resource to be exploited through farming or tourism. Thus the 'heritage' social representation could be expected to lose out to the 'grass' and 'post-card' representations in A1 and to a lesser extent in A2 given the scenario's focus on regional identities and small tourism industry. In the B scenarios, the environment is to be conserved. In the global perspective of scenario B1, its natural heritage value (related to its biological richness recognized in the 'post-card' social representation) dominates whereas in B2, it also includes a substantial cultural component that relates to the 'heritage' representation.

Benchmark scenario evaluation by stakeholders themselves

Scenarios were discussed in detail with local and regional stakeholders while being formulated and once agreed upon, they were presented to both residents and visitors in the case study site during the summer of 2005. We used focus group discussions and individual interviews, initiating them with a brief presentation of farming systems and their role in managing the grassland landscape (using pictures of the landscape and the different types of grasslands – Daigney 2005). After presenting the study objectives, European-level scenarios were presented and debated (Which is best? and Why?) before further debating their possible local consequences. Our aim was not to represent a population of stakeholders. Rather, we aimed to cover a broad range of perspectives, ranging from the occasional visitor to the native resident, including farmers themselves (Daigney 2005).

For each scenario, we classified stated arguments into one or more of the three social representations described above. Those relating to the overall landscape or 'the environment' in general were related to the 'post-card' representation. Those relating directly to farming or farmers were classified as 'grass' while those on the area's cultural and natural (both biodiversity and landscape) value were classified as 'heritage'. On the basis of the data gathered through these focus group discussions, we were able to rank the scenarios and compare the ranking to those obtained using our assessment framework. Evidently, many people did not judge the scenarios on their impacts on grasslands. Rather, they would concern themselves for the socio-economic development of the village community or the future of human relations. Such arguments could not be classified into the social representations as they do not relate to grasslands and the services they provide, but they were essential to put our conclusions into perspective, by incorporating scenario components not directly related to ecosystem services, such as concerns for social justice, economic development or local identities.

Results

Vulnerability assessment and uncertainty related to ecological mechanisms

Scenario A1 always had a negative impact on all ecosystem services except forage quality using method 3 (Table 4). Scenarios A2 and B1 generally had a negative impact, but some positive effects on grass for grazing and grass for hay, or on forage quality (Table 4). In scenario A2, methods 1 and 3 projected opposite impacts: Method 1 projected an increase in sward height (+7.5%), which was not compensated for by the decrease in palatability (–0.78%), resulting in an increase in grass for grazing (Table 4a). On the contrary, in projections with method 3, sward height increased less than palatability decreased (+5.7 and −5.8%, respectively), which led to decreased grass for grazing (Table 4c). Under scenario B1, projections with method 1 also included less litter accumulation in the spring, thus lowering snow-gliding risk and appealing to those who care for good stewardship of the grasslands (Table 4a). Overall, all three projection methods thus led to similar ecosystem service projections (Table 4).

On the basis of scenario impacts on ecosystem services (Table 4), we calculated acceptability ratios for each social representation under each scenario (Table 5). Again, we found that all three projections give broadly similar results: scenarios were consistently ranked in the same order by the three methods: A1 = A2 < B1 < B2 (Table 6).

Table 4. Scenario impacts on ecosystem services as projected using methods 1–3.

	(a) Method 1 (no threshold)				(b) Method 2 (no threshold)				(c) Method 3 (no threshold)			
	Scenarios											
Ecosystem services	A1	A2	B1	B2	A1	A2	B1	B2	A1	A2	B1	B2
Grass for hay	N/A	–	–	+	N/A	+	+	+	N/A	+	+	+
Grass for grazing	–	+	–	+	–	–	–	+	–	–	+	+
Forage quality	–	–	–	–	–	–	+	+	+	+	+	+
Flowering diversity	–	–	–	+	–	–	–	+	–	–	–	–
Plant diversity	–	–	–	+	–	–	–	+	–	–	–	–
Stewardship	–	–	+	±	–	–	–	–	–	–	–	–
Snow-gliding risk	+	+	–	±	+	+	+	+	+	+	+	+

Note: Grass for hay is not applicable (N/A) to the A1 scenario as only transhumant flocks graze the landscape in this scenario. '+', '+/–' and '–' signs indicate positive, neutral and negative impacts respectively (in the case of snow-gliding, increased risk is evidently undesirable). Cells with acceptable ratios (i.e. positive or equal to 1) are shaded.

Table 5. Acceptability ratios for each social representation (SR), and across representations, in each scenario, and across scenarios, obtained using projection methods 1–3 (with no detection thresholds) and across all methods, thereby incorporating uncertainty regarding ecological mechanisms.

			Method 1	Method 2	Method 3	
			No detection threshold applied			Across methods
Scenarios	A1	Grass	1/2	1/2	2/1	4/5
		Post-card	1/1	1/1	1/1	1/1
		Heritage	0/2	0/2	0/2	0/2
		Across SR	2/5	2/5	3/4	1/2
	A2	Grass	1/2	1/2	2/1	4/5
		Post-card	1/1	1/1	1/1	1/1
		Heritage	0/2	0/2	0/2	0/2
		Across SR	2/5	2/5	3/4	1/2
	B1	Grass	0/3	2/1	3/0	5/4
		Post-card	1/1	1/1	1/1	1/1
		Heritage	1/1	0/2	0/2	1/5
		Across SR	2/5	3/4	4/3	3/4
	B2	Grass	2/1	3/0	3/0	8/1
		Post-card	2/0	2/0	1/1	5/1
		Heritage	2/0	1/1	0/2	1/1
		Across SR	6/1	6/1	4/3	16/5
Across scenarios		Grass	4/8	7/5	5/1	7/5
		Post-card	5/3	5/3	1/1	7/5
		Heritage	3/5	1/7	0/8	1/5
		Across SR	3/4	13/15	1/1	39/45

Note: Cells with acceptable ratios (i.e. positive or equal to 1) are shaded.

Table 6. Scenario ranking for each social representation (SR) and across representations for methods 1–3 and across methods, without applying detection thresholds or obtained across detection thresholds.

	No thresholds	Across thresholds
Method 1	No thresholds	Across thresholds
Grass	A1 = A2 < B1 < B2	A1 < A2 = B1 < B2
Post-card	A1 = A2 = B1 < B2	A1 = A2 < B1 < B2
Heritage	A1 = A2 < B1 < B2	A1 = A2 < B1 < B2
Across SR	A1 = A2 = B1 < B2	A1 < A2 < B1 < B2
Method 2	No thresholds	Across thresholds
Grass	A1 = A2 < B1 < B2	A1 < A2 < B1 < B2
Post-card	A1 = A2 = B1 < B2	A1 < A2 = B1 < B2
Heritage	A1 = A2 = B1 < B2	A1 < A2 = B1 < B2
Across SR	A1 = A2 < B1 < B2	A1 < A2 < B1 < B2
Method 3	No thresholds	Across thresholds
Grass	A1 = A2 < B1 = B2	A1 = A2 < B1 = B2
Post-card	A1 = A2 = B1 = B2	A1 = A2 = B1 < B2
Heritage	A1 = A2 = B1 = B2	A2 < A1 = B1 < B2
Across SR	*A1 = A2 < B1 = B2*	*A2 < A1 = B1 < B2*
Across methods	No thresholds	Across thresholds
Grass	A1 = A2 < B1 < B2	A1 = A2 < B1 < B2
Post-card	A1 = A2 = B1 < B2	A1 < A2 < B1 < B2
Heritage	A1 = A2 < B1 < B2	A1 = A2 < B1 < B2
Across SR	A1 = A2 < B1 < B2	A1 < A2 < B1 < B2

Calculating acceptability ratios across scenarios enabled us to assess the relative vulnerability of the three social representations. For example, using method 2, we found that the 'grass' representation was more likely to benefit from acceptable rather than suffer from unacceptable impacts (seven positive and neutral impacts against five negative impacts, Table 5), whereas the 'heritage' representation was more likely to be affected negatively (7) than to see acceptable changes in ecosystem services (1), making it the most vulnerable social representation in this case (Table 5). The 'post-card' representation had an intermediate acceptability ratio of 5:3 (Table 5).

Results show that under scenarios A1, A2 and B2, acceptability ratios of the different social representations (as well as across all three of them) do not differ significantly between methods (except for the 'grass' representation under A1 where method 3 gives a more favourable outcome and the 'heritage' representation under B2 where methods 2 and 3 give increasingly favourable outcomes – Table 5). Under scenario B1, however, the different projection methods generate contrasting acceptability ratios for the 'heritage' and 'grass' social representations: increasing complexity of ecological mechanisms leads to more favourable outcomes for the 'grass' social representation (increased grass for grazing) but less favourable for the 'heritage' representation (increased litter). These differences between projection methods are also reflected when calculating ratios across social representations.

Incorporating ecological uncertainty by calculating acceptability ratios across projection methods led to very different outcomes for scenarios A1, A2 and B1, on the one hand (unacceptable), and the B2 scenario, on the other (acceptable). It did not however modify the fact that under B1, the 'grass' social representation is more likely to benefit from favourable impacts than the 'heritage' representation is (Table 5).

Uncertainty related to perceptions of ecosystem properties

Increasing detection thresholds led to an increase in the number of ecosystem services considered as not impacted reflecting a small magnitude in projected changes in ecosystem services (see Appendix 1 for detailed results per method and threshold). As an example, with a 1% detection threshold using method 1, the environmental B1 and B2 scenarios no longer had any impact on ecosystem services other than those related to forage production. A 10% threshold was required to obtain the same outcome with method 3. Due to large scenario impacts on litter accumulation (up to +168% using method 2), the ‘heritage’ social representation remained negatively affected even at the 10% threshold and thus remained the most vulnerable social representation whichever projection method was used or detection threshold was applied (see Appendix 1).

Overall, increasing detection thresholds thus translated into making scenarios more acceptable as less and less ecosystem services were negatively impacted. This was more the case for the ‘post-card’ and ‘heritage’ than for the ‘grass’ social representation, thus decreasing the vulnerability of the former compared with the latter (i.e. across scenarios – see Appendix 1).

By calculating acceptability ratios across detection thresholds we incorporated uncertainty regarding how much change in ecosystem property is significant for a given ecosystem service into the vulnerability assessment (Table 7, for detailed results per ecosystem service, see Appendix 2). A comparison of Tables 5 and 7 shows how a larger share of positive or neutral impacts on ecosystem services decreases vulnerability. However, incorporating uncertainty related to the perception of ecosystem properties did not change the scenario rankings obtained without a detection threshold: A2 A1 $<$ B1 $<$ B2 (Table 6).

Table 7. Acceptability ratios for each social representation (SR), and across SR, in each scenario, and across scenarios, obtained using projection methods 1–3 across detection thresholds (and thus integrating uncertainty related to how ecosystem properties are perceived). Acceptability ratios in the rightmost column were calculated across projection methods as well as detection thresholds and thus also integrate uncertainty related to ecological mechanisms.

			Method 1	Method 2	Method 3	
			Across detection thresholds			Across methods
Scenarios	**A1**	Grass	6/6	4/8	9/3	**9/8**
		Post-card	6/2	5/3	6/2	**17/7**
		Heritage	2/6	1/7	3/5	**1/3**
		Across SR	1/1	5/9	9/5	**3/2**
	A2	Grass	5/7	5/7	9/3	**9/8**
		Post-card	6/2	6/2	6/2	**3/1**
		Heritage	2/6	2/6	2/6	**1/3**
		Across SR	9/8	9/8	3/2	**7/6**
	B1	Grass	5/7	8/4	12/0	**16/7**
		Post-card	7/1	6/2	6/2	**19/5**
		Heritage	7/1	2/6	3/5	**1/1**
		Across SR	7/3	4/3	3/1	**2/1**
	B2	Grass	10/2	12/0	12/0	**17/2**
		Post-card	8/0	8/0	7/1	**23/1**
		Heritage	8/0	6/2	5/3	**19/5**
		Across SR	13/1	13/1	6/1	**32/3**
Across scenarios		Grass	6/5	3/2	7/1	**2/1**
		Post-card	27/5	25/7	25/7	**4/1**
		Heritage	3/2	1/2	2/3	**4/5**
		Across SR	**9/4**	**11/8**	**5/2**	**11/6**

Note: Cells with acceptable ratios (i.e. positive or equal to 1) are shaded.

Uncertainty related to social adaptations

Changing the relative contribution of the different ecosystem services in the make-up of the 'grass' social representation (Table 3a) had little incidence on how scenarios were ranked for that representation. Its acceptability was only modified under the B1 scenario whose impacts on grass for grazing and forage quality projected with method 1 were considered irrelevant for the conservation objectives the B1 scenario imposes on the grassland managers. Relative to the other social representations, differential weighting of ecosystem services increased acceptable impacts and led to a neutral acceptability ratio (results not shown).

Applying differential weighting of social representations in calculating acceptability ratios across social representations enabled us to incorporate uncertainty related to social change into the vulnerability assessment (Table 8). This led to important changes in scenario ranking. For example, comparing Table 8a and b, we see that whichever projection method is used, the least acceptable scenario changes from being A1 to A2. In fact, using projection method 3, incorporating social adaptations reversed scenario preferences to make A1 the most likely scenario to produce favourable outcomes.

Scenario acceptability as evaluated by stakeholders themselves

Overall, local and regional stakeholders favoured the more regional scenarios. They argued that marginal farming systems cannot compete with lowland farming and regional specificity is important so they can market high-quality products with a strong regional label to survive. Nevertheless, some stakeholders did mention the importance of remaining open to exchanges with other regions. Although interest in environmental questions varied greatly

Table 8. Acceptability ratios projected across all social representations considered (a) equal; (b) adapting their relative importance to the scenario (Table 3b) or (c) integrating uncertainty from social adaptations by computing both into the calculation of acceptability ratios. Ratios were calculated for each scenario and projection method as well as across scenarios and projection methods.

		Method 1	Method 2	Method 3	Across methods
		No detection threshold			
(a) All social representations considered equal					
Scenarios	A1	2/5	2/5	3/4	1/2
	A2	2/5	2/5	3/4	1/2
	B1	2/5	3/4	4/3	3/4
	B2	6/1	6/1	4/3	16/5
Across scenarios		12/16	13/15	14/14	39/45
(b) Importance of social representations weighted according to scenario					
Scenarios	A1	3/5	3/5	5/3	11/13
	A2	2/6	2/6	4/4	1/2
	B1	3/3	2/4	2/4	7/11
	B2	6/0	4/2	1/5	11/7
Across scenarios		14/14	11/17	12/16	37/47
(c) Integrating both weighted and not-weighted outcomes					
Scenarios	A1	5/10	5/10	8/7	2/3
	A2	4/11	4/11	7/8	1/2
	B1	5/8	5/8	6/7	16/23
	B2	12/1	10/3	5/8	9/4
Across scenarios		13/15	3/4	13/15	19/23

Note: Cells with acceptable ratios (i.e. positive or equal to 1) are shaded.

Table 9. Interpretation through social representations of preferences expressed by interviewed stakeholder for each scenario. Considerations not related to ecosystem properties were also taken into account to make for an overall evaluation.

Expressed scenario preferences (from stakeholder interviews)				
	Scenarios			
Social representations	A1	A2	B1	B2
'grass'	−	−	−	±
'post-card'	−	−	+	+
'heritage'	−	−	±	+
Considerations not related to ecosystem properties	−	+	−	+
Overall evaluation	−	±	±	+

Note: Cells with acceptable scenario outcomes (i.e. positive or neutral impacts) are shaded.

between stakeholders, the issue was always raised. Stakeholders with strong ties to agriculture (e.g. farmer unions, local administrations – Daigney 2005) favoured scenario A2 where farmers could live from farming whereas others (e.g. CEMAGREF, CIPRA, extension offices) favoured a mix of A2 and B2 where environmental issues would have greater importance. Thus, our discussions with regional stakeholders showed that they favour regional scenarios for maintaining marginal farming systems in the mountains.

Residents and visitors to the grasslands of Villar d'Arène followed these preferences (Table 9, see also Quétier 2006). On the basis of their arguments, we found that scenarios A1 and A2 were both unacceptable to all three social representations. In these scenarios, the 'post-card' and 'heritage' social representations are expressions of concern for the possible degradation of the landscape's beauty and the loss of the cultural heritage represented by former cultivated terraces in the landscape. Worries concerning shrub encroachment and the diminishing number of farmers were related to the 'grass' representation. Opinions on the environmental B1 and B2 scenarios were more nuanced. B1 and B2 were acceptable to the 'post-card' and 'heritage' social representations as they both preserve the overall landscape, but scenario B1 was unacceptable for the 'grass' representation as grasslands lose their role as a source of forage for livestock and farmers turn into care-takers of the landscape or disappear. B2, on the other hand, was acceptable in that it maintains a productive use of the grasslands, but unacceptable in the constraints it imposes on farmers (Table 9). Preserving terraces and maintaining a local farming community made scenario B2 acceptable to the 'heritage' representation whereas B1 was only partially acceptable because it lacks authenticity.

In terms of vulnerability, the arguments used by interviewees suggest that the 'post-card' and 'grass' social representations were the most and least vulnerable, respectively (Table 9). However, it is important to note that most interviewees did not base their preferences on scenario impacts on grasslands and the ecosystem services they provide. Rather, they were concerned with maintaining a dynamic and authentic village community in Villar d'Arène. These considerations led them to strongly prefer scenario B2 and reject A1 (Table 9).

Discussion

Uncertainty from ecological mechanisms

Comparing projection methods allowed us to discuss the effects of taking specific ecological mechanisms into account on projections of scenario impacts on ecosystem services. Our results show that scenario impacts on ecosystem services differ between the three methods compared (Table 4). These differences are related to the abundance of large long-lived and unpalatable tussock grasses. Dynamic methods project that these grasses will spread across a larger share of the study landscape than with method 1. For the same reason, imposing a dispersal constraint on these grasses (method 3) also modified projections.

Overall, the likelihood of net negative impacts is higher when these grasses are allowed to disperse freely across the whole landscape (ratio of 13:15) than when they are restricted to their current range (higher ratio of 1:1) (Table 5). Because ecosystem properties such as accumulated litter or palatability are essentially driven by the abundance of these grasses, the 'heritage' social representation is always the most vulnerable, whichever combinations of projection methods and detection thresholds are applied. Our results thus show that uncertainties concerning the dispersal of large perennial tussock grasses such as *Festuca paniculata* could have considerable effect on ecosystem services, confirming their status as key ecosystem service providers (Kremen 2005).

Extrapolating from the current state of the grasslands (method 1, see Quétier *et al*. 2007b) gives the most similar results to those obtained through stakeholder interviews (Tables 6 and 9). This could reflect stakeholders' understanding of grassland dynamics in the area, as grasslands in this model were classified following their past and present management by farmers (Quétier *et al*. 2007b). Together with the strong effects of tussock grass dispersal on ecosystem service provision, this finding suggests that ecological mechanisms that operate on longer time scales and/or broader spatial scales must be taken into account for projecting land-use change impacts on ecosystem properties and services (e.g. Parkins and MacKendrick 2007; Thuiller *et al*. 2008).

All three projection methods used here ranked scenarios A1, A2, B1 and B2 from least to most acceptable (Table 6) and this ranking is consistent with that obtained from stakeholder interviews (Table 9). They unanimously rejected the A1 scenario, both for the consequences they imagined in terms of scenery, farming and heritage, and for the local community. B2 was the preferred scenario because it preserves the area's scenery and heritage and the local community. As we found in our assessment, interviews showed that B2 did not necessarily lead to positive changes, yet was the preferred scenario. Scenario A1, on the other hand, clearly has negative impacts and was rejected.

Uncertainty related to perceptions of ecosystem properties

Despite the generalities discussed above, projection methods do vary in the magnitude of projected impacts as demonstrated through our exploration of detection threshold effects. Our analysis shows that above a 5% detection threshold, 'environmental' scenarios (B1 and B2) have no effects on ecosystem services related to biodiversity. Small impacts on biodiversity mean these are not necessarily detectable by those concerned (Quétier *et al*. 2007a). When integrating across the different methods, these generalities were conserved (Table 5). We can thus conclude that, for example, the negative impacts of 'economic' (A) scenarios on social representations are robust to ecological uncertainty.

Acceptability ratios were very sensitive to detection thresholds (Tables 5 and 7). Using method 2, the 'grass' social representation was the least vulnerable below a 1% threshold (results not shown). The 'post-card' social representation was the least vulnerable at a 10% threshold because its corresponding ecosystem services were unaffected (neutral impact) as the detection threshold rose. In fact, biodiversity does not seem to be very sensitive to land-use change in Villar d'Arène. However, public policies aiming at its conservation (e.g. type B scenarios) would benefit the 'heritage' value of the grasslands and could thus be used to maintain the provision of a broad range of other ecosystem services in this type of landscape.

Across social representations, applying detection thresholds did not modify scenario ranking but shifted acceptability of scenarios A1, A2 and B1 from negative to positive (Tables 5 and 7). Scenario B2 remained acceptable across detection thresholds. This suggests that B2 is more acceptable than A1 and A2 are unacceptable.

The large negative impacts of scenarios A1 and A2 on litter accumulation are large and less sensitive to detection thresholds, which highlights again the key role of *Festuca paniculata* dispersal.

Uncertainty related to social adaptations

We also show that on the basis of qualitative linkages between ecosystem properties and ecosystem services, differential weighting of ecosystem services and social representations had little effect on scenario ranking (the only exception being with method 3 and a 10% detection threshold), acceptability or vulnerability. However, taking into account social adaptations did decrease vulnerability of the 'grass' social representation (Table 10). Differential weighing of quantitative, rather than qualitative, ecosystem service projections would certainly have had stronger effects, but the latter require considerably more data than were available, either from social surveys (e.g. Hunziker and Kienast 1999) or from field measurements such as monitoring the behaviour of grazing flocks (e.g. Meuret 1997).

Scenario acceptability as evaluated by stakeholders themselves

Our assessment framework focuses exclusively on ecosystem services yet the scenarios used in this assessment include economic, social and cultural considerations that extend well beyond the study sites' scenic appeal or its grasslands (e.g. Pereira, Queiroz, Pereira

Table 10. Acceptability ratios projected for the 'grass' social representation with (a) and without (b) adapting its ecosystem service make-up (see Table 3a). Ratios were calculated for each scenario and projection method as well as across scenarios and projection methods.

		Method 1	Method 2	Method 3	
		No detection threshold			Across methods
(a) Ecosystem services with same weights under all scenarios					
Scenarios	A1	1/2	1/2	2/1	4/5
	A2	1/2	1/2	2/1	4/5
	B1	0/3	2/1	3/0	5/4
	B2	2/1	3/0	3/0	8/1
Across scenarios		4/8	7/5	5/1	8/7
(b) Ecosystem services weighted according to scenario					
Scenarios	A1	1/2	1/2	2/1	4/5
	A2	1/2	1/2	2/1	4/5
	B1	2/1	3/0	3/0	8/1
	B2	2/1	3/0	3/0	8/1
Across scenarios		6/6	8/4	10/2	2/1
(c) Integrating both weighted and not-weighted outcomes					
Scenarios	A1	1/2	1/2	2/1	4/5
	A2	1/2	1/2	2/1	4/5
	B1	1/2	5/1	6/0	13/5
	B2	2/1	6/0	6/0	8/1
Across scenarios		5/12	5/3	5/1	5/3

Note: Cells with acceptable ratios (i.e. positive or equal to 1) are shaded.

and Vicente 2005). In the face of these considerations, preserving an authentic local community in Villar d'Arène was unanimously desired. It is associated with the creation of local jobs, autonomy in decision-making and ownership of the land and the flocks. Although one farmer and a few visitors feared isolation, this did not change their opinion against the more global scenarios and regional scenarios were preferred. Opposition to A1-type scenarios was previously found in other scenario-based studies of rural change such as *MedAction* (Kok, Patel, Rothman and Quaranta 2006; Kok, Biggs and Zurek 2007) and *Bioscene* (Marty, Guignier, Caplat, Lepart and Lhuillier 2005; Soliva *et al*. 2008). This opposition seems to reflect common fears of the consequences of free-market globalization. Buchecker, Hunziker and Kienast (2003) find similar results in the Swiss Alps and comment that the appeal of the 'traditional village community' is still the main reference for ideal landscape planning. In such a community, all (or nearly all) contribute to land-use and landscape planning through the land they own. Global scenarios, where strong links to the land are lost, go against this idealized reference.

Opinions were more varied concerning scenarios A2 and B1. Scenario B1 preserves scenery, terraces and varied grasslands but ignores the local community; individual farmer's freedoms and authenticity is sacrificed to environmental efficiency. In contrast, scenario A2 does not preserve scenery or heritage but allows farmers to live off their land and favours economic self-sufficiency of the area. It is interesting to note that fewer social and environmental fears emerge concerning the A2 than for the A1 scenario. Interviewees believe local entrepreneurs would do less harm than outsiders, making A2 more acceptable than A1. We also found that most interviewees often preferred a mix of the scenarios we offered them and that they designated A1 and B1 as more likely, underlining the difference between desired and anticipated futures. The method we used in our assessment is similar to that of a standard impact assessment study, where external drivers affect a relatively powerless local community. An alternative normative approach, focusing on desired futures, could have led to very different outcomes and perhaps given more details on each social representation's preferences rather than emphasizing the 'nightmare' A1 scenario (Kok *et al*. 2006, 2007).

We found the 'heritage' social representation to be the most vulnerable, whichever projection method or detection threshold were used to project scenario impacts on ecosystem properties (Tables 5 and 7). With a low detection threshold, acceptability ratios showed that the 'grass' representation was the least vulnerable. It encompasses a wider range of ecosystem properties and ecosystem services than the other two social representations and was thus more likely to benefit from compensations between ecosystem services, for example, between grass for hay and grass for grazing in scenario A2 (Table 4). These results are very different from the relative vulnerability assessed through our interviews (Table 9). Residents and visitors of Villar d'Arène viewed the 'grass' representation as more often affected negatively (three of four scenarios, the fourth being neutral) while the 'post-card' representation was the least vulnerable. This could be due to the 'grass' representation being more strongly associated with the 'traditional village community' as described above. The difference could also result from the broad interpretation we gave to the social representations when analysing interviews where scenario impacts were not limited to changes in ecosystem properties as they were in our assessment framework (e.g. anything to do with 'nature' or 'scenery' would go to the 'post-card' representation).

Taking into account scenario, impacts beyond ecosystem services would require an in-depth analysis not only of the local human – environment system but also of the system's insertion in a wider, social, economic and political context and the possibilities it offers for stakeholders to adapt. Stakeholders are knowledgeable about such possibilities in the short term but they are rarely made explicit (e.g. Pereira *et al*. 2005). This can make inter-site comparisons difficult as contexts are not shared. Considerations other than land-use impacts on ecosystem services might be better revealed using a normative approach, where stakeholders formulate desirable or undesirable futures.

Conclusions

Appropriate analytical frameworks must be developed to assess and manage the trade-offs between maintaining the long-term capacity of ecosystems to provide ecosystem services and human needs in the short term (Ramankutty *et al*. 2006; Bennet and Balvanera 2007). The framework used in this study provides such a tool. It was designed to assess the vulnerability of ecosystem services using a combination of quantitative ecological models and qualitative social survey data and is built around an indicator sequence, linking drivers of change, system properties and impacts on stakeholder categories. The indicator sequence is both transparent and flexible, which allows the framework to integrate multiple variants of the indicator sequence and thereby incorporate additional sources of uncertainty to those offered by scenario formulation. We were able to compare and integrate outcomes from three different methods for projecting scenario impact on ecosystem properties and highlighted the key role of dispersal in driving ecosystem service provision in the case study landscape. We were also able to incorporate social uncertainties in scenario evaluation, concerning either the significance of ecological impacts or the possible social adaptations through changes in social representations of the grassland ecosystems.

In addition to incorporating uncertainty, the indicator sequence used here also allowed for multiple perspectives on the value of ecosystem services. This addresses the fact that too few vulnerability assessments identify what and whose values characterize the study system (Eakin and Leurs 2006). Here, we explicitly considered stakeholder differences (through their social representations) and we could identify winners and losers: the ‘heritage’ social representation was the most vulnerable. We also found that with only a few exceptions, scenarios can be ranked as follows: A1 $<$ A2 $<$ B1 $<$ B2 and that this ranking is shared across social representations, as well as by direct assessment by stakeholders.

The framework used in this study improved our assessment of vulnerability through the greater transparency and flexibility of the indicator sequence used to project scenario impacts on ecosystems and the multiple values assigned to ecosystem properties and services. Such improvements should be applied more widely in investigating land-use change impacts on ecosystem services and human well-being.

Acknowledgements

This research was carried out as part of the EU funded project VISTA (Vulnerability of Ecosystem Services to Land Use Change in Traditional Agricultural Landscapes – EVK2-2001-000356) and the DIVHERBE project of the ACI – ECOGER program of CNRS. It would not have been possible without the tireless data-gathering efforts of many students and colleagues, the patience of the people of Villar d’Arène and the logistical support of the Station Alpine Joseph Fourier and its staff. Specific thanks go to Pierre Liancourt and Ian Davies for work on LAMOS, Aurélie Thébault for data handling and preparation, Yannick Prebay for thoughtful discussions on the assessment framework, Serge Aubert for facilitating contact with local and regional stakeholders. We are grateful to Anette Reenberg and Lars Jørgensen for organizing the third LaSyS Workshop in Copenhagen, Denmark, and to successive referees for their help in improving the manuscript.

References

Alcamo, J., Kok, K., Busch, G., Priess, J.A., Eickhout, B., Rounsevell, M., Rothman, D.S., and Heistermann, M. (2006), “Searching for the Fugure of Land: Scenarios from the Local to Global

Scale," in *Land-Use and Land-Cover Change – Local Processes and Global Impacts, IGBP Global Change Series*, eds. E.F. Lambin and H. Geist, Cambridge, UK: Cambridge University Press, pp. 137–155.
Audsley, E., Pearn, K.R., Simota, C., Cojocary, G., Koutsidou, E., Rounsevell, M.D.A., Trnka, M., and Alexandrov, V. (2006), "What can Scenario Modelling Tell us About Future European Scale Agricultural Land Use, and What no?," *Environmental Science and Policy*, 9, 148–162.
Baker, J., Hulse, D.W., Gregory, S.V., White, D., van Sickle, J., Berger, P.A., Dole, D., and Schumaker, N.H. (2004), "Alternative Futures for the Willamette River Basin, Oregon," *Ecological Applications*, 14(2), 313–324.
Baldock, D., Dwyer, J., and Sumpsi Vinas, J.M. (2002), "Environmental Integration and the CAP," A Report to the European Commission, DG Agriculture, Institute for European Environmental Policy, London, 113 pp.
Bennet, E.M., and Balvanera, P. (2007), "The Future of Production Systems in a Globalized World," *Frontiers in Ecology and the Environment*, 5(4), 191–198.
Briske, D.D., Fuhlendorf, S.D., and Smeins, F.E. (2003), "State-and-Transition Models, Thresholds and Rangeland Health: A Synthesis of Ecological Concepts and Perspectives," *Rangeland Ecology and Management*, 58, 1–10.
Buchecker, M., Hunziker, M., and Kienast, F. (2003), "Participatory Landscape Development: Overcoming Social Barriers to Public Involvement," *Landscape and Urban Planning*, 64, 29–46.
Busch, G. (2006), "Future European Agricultural Landscapes – What can we Learn from Existing Quantitative Land Use Scenario Studies?," *Agriculture, Ecosystems and Environment*, 114, 121–140.
Castella, J.-C., Pheng Kam, S., Dinh Quang, D., Verburg, P.H., and Thai Hoanh, C. (2007), "Combining Top-Down and Bottom-Up Modelling Approaches for Land Use/Cover to Support Public Policies: Application to Sustainable Management of Natural Resources in Northern Vietnam," *Land Use Policy*, 24, 531–545.
Castella, J.-C., and Verburg, P.H. (2007), "Combination of Process-Oriented and Pattern-Oriented Models of Land-Use Change in a Mountain Area of Vietnam," *Ecological Modelling*, 202, 410–420.
Chhabra, A., Geist, H., Houghton, R.A., Haberl, H., Braimoh, A.K., Vlek, P.L.G., Patz, J., Xu, J., Ramankutty, N., Coomes, O., and Lambin, E.F. (2006), "Multiple Impacts of Land-Use/Cover Change," in *Land-Use and Land-Cover Change – Local Processes and Global Impacts, IGBP Global Change Series*, eds. E.F. Lambin and H. Geist, Cambridge, UK: Cambridge University Press, pp. 71–116.
Clapham, P.J., Aguilar, A., and Hatch, L.T. (2008), "Determining Spatial and Temporal Scales for Management: Lessons from Whaling," *Marine Mammal Science*, 24(1), 183–201.
Daigney, S. (2005), "Adaptations d'une petite agriculture marginale à des scénarios contrastés, Villar d'Arène, Hautes-Alpes," Thesis, Ecole Nationale Supérieure Agronomique de Rennes, Rennes, France.
de Chazal, J., Quétier, F., Lavorel, S., and van Doorn, A. (2008), "Including Multiple Differing Stakeholder Values into Vulnerability Assessments of Socio-Ecological Systems," *Global Environmental Change*, 19(2), 306–315.
Diaz, S., Fargione, J., Chapin, III, F.S., and Tilman, D. (2006), "Biodiversity Loss Threatens Human Well-Being," *PLoS Biology*, 4 (8), 1300–1305.
Díaz, S., Lavorel, S., de Bello, F., Quétier, F., Grigulis, K., and Robson, M.T. (2007), "Incorporating Plant Functional Diversity Effects in Ecosystem Service Assessments," *Proceedings of the National Academy of Science*, 104(52), 20684–20689.
Dobbs, T.L., and Pretty, J.N. (2004), "Agri-Environment Stewardship Schemes and 'Multifunctionality'," *Review of Agricultural Economics*, 26, 220–237.
Dockerty, T., Lovett, A., Appleton, K., Bone, A., and Sünnenberg, G. (2006), "Developing Scenarios and Visualisations to Illustrate Potential Policy and Climatic Influences on Future Agricultural Landscapes," *Agriculture, Ecosystems and Environment*, 114, 103–120.
Dorrough, J., Vesk, P.A., and Moll, J. (2008), "Integrating Ecological Uncertainty and Farm-Scale Economics When Planning Restoration," *Journal of Applied Ecology*, 45, 288–295.
Eakin, H., and Leurs, A.M. (2006), "Assessing the Vulnerability of Social–Environmental Systems," *Annual Review of Environment and Resources*, 31, 365–394.
Eickhout, B., van Meijl, H., Tabeau, A., and van Rheenen, T. (2007), Economic and Ecological Consequences of Four European Land Use Scenarios," *Land Use Policy*, 24, 562–575.

Fleury, P. (1994), “Le diagnostic agronomique des végétations prairiales et son utilisation dans la gestion des exploitations agricoles,” Thèse de doctorat, Institut National Polytechnique de Lorraine, Nancy, France.

Foley, J.A., Defries, R., Asner, G.P., Barford, C., Bonan, G., Carpenter, S.R., Chapin, F.S., Coe, M.T., Daily, G.C., Gibbs, H.K., Helkowski, J.H., Holloway, T., Howard, E.A., Kucharik, C.J., Monfreda, C., Patz, J.A., Prentice, I.C., Ramankutty, N., and Snyder, P.K. (2005), “Global Consequences of Land Use,” *Science*, 309, 570–574.

Hulse, D.W., Branscomb, A., and Payne, S.G. (2004), “Envisioning Alternatives: Using Citizen Guidance to Map Future Land and Water Use,” *Ecological Applications*, 14(2), 325–341.

Hunziker, M., and Kienast, F. (1999), “Potential Impacts of Changing Agricultural Activities on Scenic Beauty – A Prototypical Technique for Automated Rapid Assessment,” *Landscape Ecology*, 14, 161–176.

Kok, K., Biggs, R., and Zurek, M. (2007), Methods for Developing Multiscale Participatory Scenarios: Insights from Southern Africa and Europe,” *Ecology and Society*, 13(1), Article no 8 [online]. http://www. ecologyandsociety.org/vol12/iss1/art8/.

Kok, K., Patel, M., Rothman, D.S., and Quaranta, G. (2006), “Multi-Scale Narratives From an IA Perspective: Part II. Participatory Local Scenario Development,” *Futures*, 38, 285–311.

Kremen, C. (2005), “Managing Ecosystem Services: What do we Need to Know About their Ecology?” *Ecology Letters*, 8, 468–479.

Marty, P., Guignier, C., Caplat, P., Lepart, J., and Lhuillier, C. (2005), “To Reconcile Agriculture and Biodiversity? Appraisal of the Sustainability of Four Scenarios of Land Use. The Case of a Mediterranean Upland,” *Biodiversity Conservation and Sustainable Development in Mountain Areas of Europe. The Challenge of Interdisciplinary Research*, in eds. J. Pantis, S. Sgardelis and T. Kallimanis, Bioscene Conference, 20–24 September 2005, Ioannina, Greece.

Méot, A., Hubert, B., and Lasseur, J. (2003), “Organisation of the Pastoral Territory and Grazing Management: Joint Modelling of Grazing Management Practices and Plant Cover Dynamics,” *Agricultural Systems*, 76, 115–139.

Metzger, M.J., Rounsevell, M.D.A., Acosta-Michlik, L., Leemans, R., and Schröter, D. (2006), “The Vulnerability of Ecosystem Services to Land Use Change,” *Agriculture, Ecosystems and Environment*, 114, 69–85.

Meuret, M. (1997), “Préhensibilité des aliments chez les petits ruminants sur parcours en landes et sous-bois,” *Productions Animales*, 10, 391–401.

Nakicenovic, N., and Swart, R., (eds). Emission Scenarios. A special report of Working Group III of the Intergovernmental panel on Climate Change, Cambridge University Press, Cambridge (2000), 1–612.

Newesely, C., Tasser, E., Spadinger, P., and Cernusca, A. (2000), “Effects of Land-Use Changes on Snow Gliding Processes in Alpine Ecosystems,” *Basic and Applied Ecology*, 1, 61–67.

O’Brien, K.L., and Leichenko, R.M. (2000), “Double Exposure: Assessing the Impacts of Climate Change Within the Context of Economic Globalization,” *Global Environmental Change*, 10, 221–232.

——— (2003), “Winners and Losers in the Context of Global Change,” *Annals of the Association of American Geographers*, 93(1), 89–103.

Parc National des Ecrins (2004), “Projet de programme d’aménagement 2005–2010 du Parc national des Ecrins,” Internal Report, Parc National des Ecrins, Gap, France.

Parkins, J.R., and MacKendrick, N.A. (2007), Assessing Community Vulnerability: A Study of the Mountain Pine Beetle Outbreak in British Columbia, Canada,” *Global Environmental Change*, 17, 460–471.

Pereira, E., Queiroz, C., Pereira, H.M., and Vicente, L. (2005), “Ecosystem Services and Human Well-Being: A Participatory Study in a Mountain Community in Portugal,” *Ecology and Society*, 10, Paper no 14 [online]. http://www.ecologyandsociety.org/vol10/iss2/art14/.

Peterson, G.D., Carpenter, S.R., and Brock, W.A. (2003a), “Uncertainty and the Management of Multistate Ecosystems: An Apparently Rational Route to Collapse,” *Ecology*, 84(6), 1403–1411.

Peterson, G.D., Cumming, G.S., and Carpenter, S.R. (2003b), “Scenario Planning: A Tool for Conservation in an Uncertain World,” *Conservation Biology*, 17(2), 358–366.

Quetier, F. (2006), “Vulnérabilité des écosystèmes semi-naturels européens aux changements d’utilisation des terres. Application aux prairies subalpines de Villar d’Arène, France,” PhD Thesis Ecole Nationale Supérieure Agronomique de Montpellier, Montpellier, France.

Quétier, F., Lavorel, S., Thuiller, W., and Davies, I. (2007a), "Plant Trait-Based Assessment of Ecosystem Service Sensitivity to Land-Use Change in Mountain Grasslands," *Ecological Applications*, 17(8), 2377–2386.

Quétier, F., Thébault, A., and Lavorel, S. (2007b), "Plant Traits in a State and Transition Framework as Markers of Ecosystem Response to Past and Present Land-Use in Subalpine Grasslands," *Ecological Monographs*, 77, 33–52.

Ramankutty, N., Graumlich, L., Achard, F., Alves, D., Chhabra, A., Defries, R.S., Foley, J.A., Geist, H., Houghton, R.A., Goldewijk, K.K., Lambin, E.F., Millington, A., Rasmussen, K., Reid, R.S., and Turner, II, B.L. (2006), "Global Land-Cover Change: Recent Progress, Remaining Challenges," in *Land-Use and Land-Cover Change – Local Processes and Global Impacts, IGBP Global Change Series*, eds. E.F. Lambin and H. Geist, Cambridge, UK: Cambridge University Press, pp. 9–39

Refsgaard, J.C., van der Sluijs, J.P., Hojberg, A.L., and Vanrolleghem, P.A. (2007), "Uncertainty in the Environmental Modelling Process – A Framework and Guidance," *Environmental Modelling and Software*, 22(11), 1543–1556.

Rivoal, F. (2004), "Etude des usages et représentations d'un paysage agro-pastoral de montagne. Exemple du versant adret de la commune de Villar d'Arène," Thesis, Institut National d'Horticulture d'Angers, Angers, France.

Rounsevell, M.D.A., Ewert, F., Reginster, I., Leemans, R., and Carter, T.R. (2005), "Future Scenarios of European Agricultural Land Use II: Projecting Changes in Cropland and Grassland," *Agriculture, Ecosystems and Environment*, 107, 117–135.

Rounsevell, M.D.A., Reginster, I., Araújo, M.B., Carter, T.R., Dendoncker, N., Ewert, F., House, J.I., Kankaanpää, S., Leemans, R., Metzger, M.J., Schmit, C., Smith, P., and Tuck, G. (2006), "A Coherent Set of Future Land-Use Change Scenarios for Europe," *Agriculture, Ecosystems and Environment*, 114, 57–68.

Rousset, P.-L. (1977), *Au Pays de la Meije*, Grenoble, France: Editions Didier Richard, pp. 414.

Schröter, S., Cramer, W., Leemans, R., Prentice, C.I., Araújo, M.B., Arnell, N.W., Bondeau, A., Bugmann, H., Carter, T.R., Gracia, C.A., de la Vega-Leinert, A.C., Erhard, M., Ewert, F., Glendining, M., House, J.I., Kankaanpää, S., Klein, R.J.T., Lavorel, S., Lindner, M., Metzger, M.J., Meyer, J., Mitchell, T.D., Reginster, I., Rounsevell, M., Sabaté, S., Sitch, S., Smith, B., Smith, J., Smith, P., Sykes, M.T., Thonicke, K., Thuiller, W., Tuck, G., Zaehle, S., and Zierl B. (2006), "Ecosystem Service Supply and Vulnerability to Global Change in Europe," *Science*, 314, 257–258.

Schumaker, N.H., Ernst, T., White, D., Baker, J., and Haggerty, P. (2004), "Projecting Wildlife Responses to Alternative Future Landscapes in Oregon's Willamette Basin," *Ecological Applications*, 14(2), 381–400.

Sheate, W., Partidário, M., Byron, H., Bina, O., and Dagg, S. (2008) "Sustainability Assessment of Future Scenarios: Methodology and Application to Mountain Areas of Europe," *Environmental Management*, 41, 282–299.

Soliva, R., Rønningen, K., Bella, I., Bezák, P., Cooper, T., Egil Flø, B., Marty, P., and Potter, C. (2008), "Envisioning Upland Futures: Stakeholder Responses to Scenarios for Europe's Mountain Landscapes," *Journal of Rural Studies*, 24, 56–71.

Sutherland, W.J., and Watkinson, A.R. (2001), "Policy Making Within Ecological Uncertainty: Lessons from Badgers and GM Crops," *Trends in Ecology and Evolution*, 16(5), 261–263.

Tasser, E., Mader, M., and Tappeiner, U. (2003), "Effects of Land Use in Alpine Grasslands on the Probability of Landslides," *Basic and Applied Ecology*, 4, 271–280.

Thuiller, W. (2004), "Patterns and Uncertainties of Species' Range Shifts Under Climate Change," *Global Change Biology*, 10, 2020–2027.

Thuiller, W., Albert, C., Araújo, M.B., Berry, P.M., Cabeza, M., Guisan, A., Hickler, T., Midgley, G.F., Paterson, J., Schurr, F.M., Sykes, M.T., and Zimmermann, N.E. (2006), "Predicting Global Change Impacts on Plant Species' Distributions: Future Challenges," *Perspectives in Plant Ecology, Evolution and Systematics*, 9, 137–152.

——— (2008), "Predicting Global Change Impacts on Plant Species' Distributions: Future Challenges," *Perspectives in Plant Ecology, Evolution and Systematics*, 9, 137–152.

Tscharntke, T., Klein, A.M., Kruess, A., Steffan-Denwenter, I., and Carsten, T. (2005), "Landscape Perspectives on Agricultural Intensification and Biodiversity–Ecosystem Service Management," *Ecology Letters*, 8, 857–874.

Turner, B.L., Kasperson, R.E., Matson, P., McCarthy, J.J., Corell, R.W., Christensen, L., Eckley, N., Kasperson, J.X., Luers, A., Martello, M.L., Polsky, C., Pulsipher, A., and Schiller, A. (2003), "A Framework for Vulnerability Analysis in Sustainability Science," Proc. Natl. Acad. Sci. U.S.A., 100, 8074–8079.

Verburg, P.H., Kok, K., Pontius, Jr., R.G., and Veldkamp, A. (2006a), "Modeling Land-Use and Land-Cover Change," in *Land-Use and Land-Cover Change – Local Processes and Global Impacts, IGBP Global Change Series*, eds. E.F. Lambin and H. Geist, Cambridge, UK: Cambridge University Press, 117–135.

Verburg, P.H., Rounsevell, M.D.A., and Veldkamp, A. (2006b), "Scenario Based Studies of Future Land Use in Europe," *Agriculture Ecosystems and Environment*, 114, 1–6.

Vitousek, P.M., Mooney, H.A., Lubchenco, J., and Melillo, J.M. (1997), "Human Domination of Earth's Ecosystems," *Science*, 277, 494–499.

Westhoek, H., van den Berg, M., and Bakker, J. (2006), "Development of Land Use Scenarios for European Land Use", *Agriculture, Ecosystems & Environment,* 114, 7–20.

Appendix 1

Qualitative projections of ecosystem service change under scenarios A1, A2, B1 and B2 obtained using methods 1–3 for projecting scenario impacts on ecosystem properties, in combination with 4 four detection thresholds (none, 1%, 5%, 10%) below which changes in ecosystem properties are considered non significant and ignored.

	(a) Method 1 (no threshold)				(b) Method 1 (1% threshold)				(c) Method 1 (5% threshold)				(d) Method 1 (10% threshold)			
Ecosystem services	Scenarios															
	A1	A2	B1	B2	A1	A2	B1	B2	A1	A2	B1	B2	A1	A2	B1	B2
Grass for hay		–	–	+		–	–	+		–	+/–	+/–		–	+/–	+/–
Grass for grazing	–	+	–	+	–	+	–	+	–	+	–	+	+/–	+/–	+/–	+/–
Forage quality	–	–	–	–	–	–	–	–	–	–	+/–	+/–	+/–	+/–	+/–	+/–
Flowering diversity	–	–	–	+	–	–	+/–	+/–	+/–	+/–	+/–	+/–	+/–	+/–	+/–	+/–
Plant diversity	–	–	–	+	–	–	+/–	+/–	+/–	+/–	+/–	+/–	+/–	+/–	+/–	+/–
Stewardship	–	–	+	+/–	–	–	+/–	+/–	–	–	+/–	+/–	–	–	+/–	+/–
Snow-gliding risk	+	+	–	+/–	+	+	+/–	+/–	+	+	+/–	+/–	+	+	+/–	+/–
	(e) Method 2 (no threshold)				(f) Method 2 (1% threshold)				(g) Method 2 (5% threshold)				(h) Method 2 (10% threshold)			
Ecosystem services	Scenarios															
	A1	A2	B1	B2	A1	A2	B1	B2	A1	A2	B1	B2	A1	A2	B1	B2
Grass for hay		+	+	+		+	+	+		+	+	+/–		+/–	+/–	+/–
Grass for grazing	–	–	–	+	–	–	–	+	–	–	–	+	–	–	–	+
Forage quality	–	–	+	+	–	–		+	–	–		+				+

(*Continued*)

							+/–				+/–		+/–	+/–	+/–	
Flowering diversity	–	–	–	+	–	–	–	+	–	+/–	+/–	+/–	+/–	+/–	+/–	+/–
Plant diversity	–	–	–	+	–	–	–	+	–	+/–	+/–	+/–	+/–	+/–	+/–	+/–
Stewardship	–	–	–	–	–	–	–	–	–	–	–	+/–	–	–	–	+/–
Snow-gliding risk	+	+	+	+	+	+	+	+	+	+	+	+/–	+	+	+	+/–

	(i) Method 3 (no threshold)				(j) Method 3 (1% threshold)				(k) Method 3 (5% threshold)				(l) Method 3 (10% threshold)			
Ecosystem services	Scenarios															
	A1	A2	B1	B2	A1	A2	B1	B2	A1	A2	B1	B2	A1	A2	B1	B2
Grass for hay		+	+	+		+	+	+		+	+	±		+	±	±
Grass for grazing	–	–	+	+	–	–	+	+	–	–	+	+	±	±	±	+
Forage quality	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+
Flowering diversity	–	–	–	–	–	–	–	±	±	±	±	±	±	±	±	±
Plant diversity	–	–	–	–	–	–	–	±	±	±	±	±	±	±	±	±
Stewardship	–	–	–	–	–	–	–	–	–	–	–	±	±	–	±	±
Snow-gliding risk	+	+	+	+	+	+	+	+	+	+	+	±	±	+	±	±

Appendix 2

Postive (1), neutral (0) and negative (–1) changes in ecosystem service provision across scenarios, projection method and detection thresholds. These changes are used as a basis for calculating scenario acceptability ratios for each social representation.

Postive (1), neutral (0) and negative (−1) changes in ecosystem service provision across scenarios, projection method and detection thresholds		Method 1				Method 2				Method 3				Across methods			
		None	1%	5%	10%	None	1%	5%	10%	None	1%	5%	10%	None	1%	5%	10%
Scenario A1	Grass social representation																
	Grass for hay	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Grass for grazing	−1	−1	−1	0	−1	−1	−1	−1	−1	−1	−1	0	−1	−1	−1	−1/3
	Forage quality	−1	−1	−1	0	−1	−1	−1	−1	1	1	1	1	−1/3	−1/3	−1/3	0
	Post-Card social representation																
	Flowering diversity	−1	−1	0	0	−1	−1	−1	0	−1	−1	0	0	−1	−1	−1/3	0
	Heritage social representation																
	Plant diversity	−1	−1	0	0	−1	−1	−1	0	−1	−1	0	0	−1	−1	−1/3	0
	Appropriate stewardship	−1	−1	−1	−1	−1	−1	−1	−1	−1	−1	−1	0	−1	−1	−1	−2/3
Scenario A2	Grass social representation																
	Grass for hay	−1	−1	−1	−1	1	1	1	0	1	1	1	1	1/3	1/3	1/3	0
	Grass for grazing	1	1	1	0	−1	−1	−1	−1	−1	−1	−1	0	−1/3	−1/3	−1/3	−1/3
	Forage quality	−1	−1	−1	0	−1	−1	−1	0	1	1	1	1	−1/3	−1/3	−1/3	1/3
	Post-Card social representation																
	Flowering diversity	−1	−1	0	0	−1	−1	0	0	−1	−1	0	0	−1	−1	0	0
	Heritage social representation																
	Plant diversity	−1	−1	0	0	−1	−1	0	0	−1	−1	0	−1	−1	−1	0	−1/3
	Appropriate stewardship	−1	−1	−1	−1	−1	−1	−1	−1	−1	−1	−1	0	−1	−1	−1	−2/3
Scenario B1	Grass social representation																
	Grass for hay	−1	−1	0	0	1	1	1	0	1	1	1	0	1/3	1/3	2/3	0
	Grass for grazing	−1	−1	−1	0	−1	−1	−1	−1	1	1	1	0	−1/3	−1/3	−1/3	−1/3
	Forage quality	−1	−1	0	0	1	0	0	0	1	1	1	1	1/3	0	1/3	1/3
	Post-Card social representation																
	Flowering diversity	−1	0	0	0	−1	−1	0	0	−1	−1	0	0	−1	−2/3	0	0
	Heritage social representation																
	Plant diversity	−1	0	0	0	−1	−1	0	0	−1	−1	0	0	−1	−2/3	0	0
	Appropriate stewardship	1	0	0	0	−1	−1	−1	−1	−1	−1	−1	0	−1/3	−2/3	−2/3	−1/3
Scenario B2	Grass social representation																
	Grass for hay	1	1	0	0	1	1	0	0	1	1	0	0	1	1	0	0
	Grass for grazing	1	1	0	0	1	1	1	1	1	1	1	1	1	1	2/3	2/3

	Forage quality	–1	–1	0	0	1	1	1	1	1	1	1	1	1/3	1/3	2/3	2/3
	Post-Card social representation																
	Flowering diversity	1	0	0	0	1	1	0	0	–1	0	0	0	1/3	1/3	0	0
	Heritage social representation																
	Plant diversity	1	0	0	0	1	1	0	0	–1	0	0	0	1/3	1/3	0	0
	Appropriate stewardship	0	0	0	0	–1	–1	0	0	–1	–1	0	0	–2/3	–2/3	0	0
Across scenarios	Grass social representation																
	Grass for hay	–1/4	–1/4	–1/4	–1/4	3/4	3/4	1/2	0	3/4	3/4	1/2	1/4	1/9	1/9	0	0
	Grass for grazing	0	0	–1/4	0	–1/2	–1/2	–1/2	–1/2	0	0	0	1/4	–0	–0	–0	–0
	Forage quality	–1	–1	–1/2	0	–1/4	–1/4	0	1	1	1	1	0	–0	0	0	
	Post-Card social representation																
	Flowering diversity	–1/2	–1/2	0	0	–1/2	–1/2	–1/4	0	–1	–3/4	0	0	–1/6	–1/7	–0	0
	Heritage social representation																
	Plant diversity	–1/2	–1/2	0	0	–1/2	–1/2	–1/4	0	–1	–3/4	0	–1/4	–1/6	–1/7	–0	–0
	Appropriate stewardship	–1/4	–1/2	–1/2	–1/2	–1	–1	–3/4	–3/4	–1	–1	–3/4	0	–1/5	–1/5	–1/6	–1/9