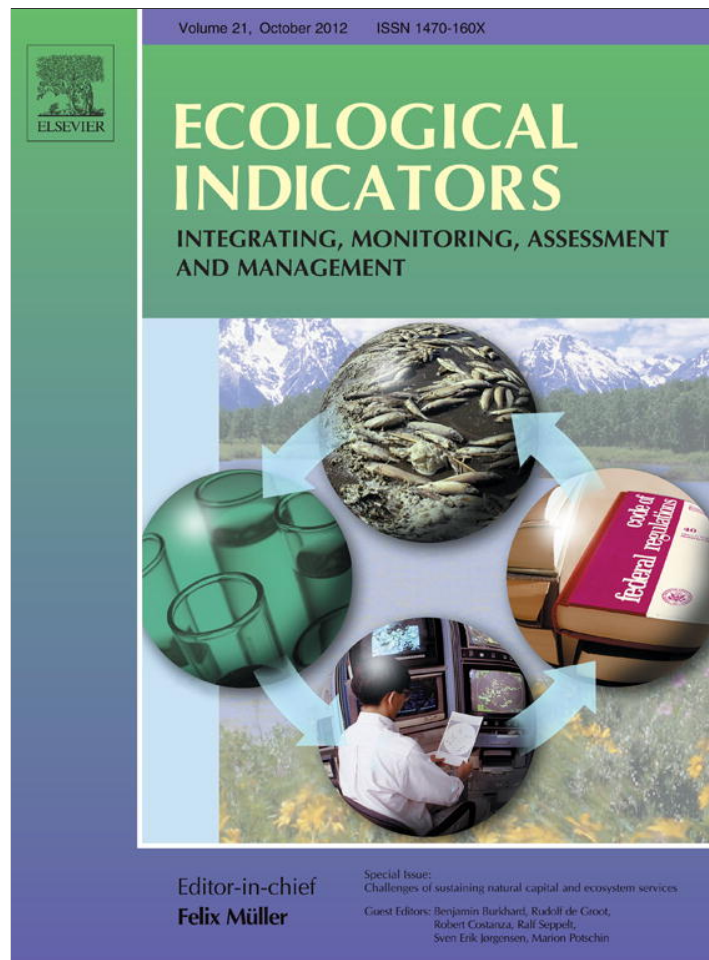


Provided for non-commercial research and education use.
Not for reproduction, distribution or commercial use.



This article appeared in a journal published by Elsevier. The attached copy is furnished to the author for internal non-commercial research and education use, including for instruction at the authors institution and sharing with colleagues.

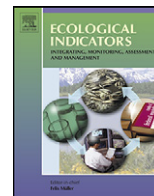
Other uses, including reproduction and distribution, or selling or licensing copies, or posting to personal, institutional or third party websites are prohibited.

In most cases authors are permitted to post their version of the article (e.g. in Word or Tex form) to their personal website or institutional repository. Authors requiring further information regarding Elsevier's archiving and manuscript policies are encouraged to visit:

<http://www.elsevier.com/copyright>

Contents lists available at [SciVerse ScienceDirect](http://www.elsevier.com/locate/ecolind)

Ecological Indicators

journal homepage: www.elsevier.com/locate/ecolind

Indicators of ecosystem service potential at European scales: Mapping marginal changes and trade-offs

Roy Haines-Young^a, Marion Potschin^{a,*}, Felix Kienast^b^a Centre for Environmental Management, School of Geography, University of Nottingham, NG7 2RD, United Kingdom^b Swiss Federal Research Institute WSL, Zürcherstrasse 111, 8903 Birmensdorf, Switzerland

ARTICLE INFO

Keywords:

Ecosystem service mapping
Ecosystem service indicator
Trade-off analysis
Marginal analysis
Land use change

ABSTRACT

This study develops an approach to mapping indicators of the potential of ecosystems to supply ecosystem services, and the impact of changes in land cover and use upon them. The study focuses on the EU-25 plus Switzerland and Norway, and develops the methodology proposed by Kienast et al. (2009), which uses expert- and literature-driven modelling methods. The methods are explored in relation to mapping and assessing four of the ecosystem services: “Crop-based production”; “Wildlife products”; “Habitat diversity”; and “Recreation”. The potential to deliver services is assumed to be influenced by (a) land-use, (b) net primary production, and (c) bioclimatic and landscape properties such as mountainous terrain, adjacency to coastal and wetland ecosystems, as well as adjacency to landscape and nature protection zones.

The novel aspect of this work is an analysis of whether the historical and the projected land use changes for the periods 1990–2000, 2000–2006, and 2000–2030 are likely to be supportive or degenerative in the capacity of ecosystems to deliver ecosystem services; we refer to these as ‘marginal’ or incremental changes. The latter are assessed by using land account data for 1990–2000 and 2000–2006 (LEAC, EEA, 2006) and EURURALIS 2.0 land use scenarios for 2000–2030. The results are reported at three spatial reporting units, i.e. (1) the NUTS-X regions, (2) the bioclimatic regions, and (3) the dominant landscape types. All mapped output has been compared with independently generated continent-wide assessments (maps of ecosystem services or environmental parameters/indicators), which revealed that the straight-forward binary links work satisfactorily and generate plausible geographical patterns. This conclusion mainly holds for provisioning services. At the continental scale, the selected input parameters are thus valid proxies which can be used to assess the medium-term potential of landscapes to provide ecosystem services.

For a subset of NUTS-x regions for which change trajectories for 1990–2000, 2000–2006 and 2000–2030 are available, trade-offs between the four services have been analyzed using cluster analyses. The latter allowed us to simultaneously analyze the state of the four services in year 2000 and the individual trajectories of each service over three time periods. As a result we obtained seven regions with distinct trade-off patterns. To our knowledge this is one of the first continental-wide analyses where land use trajectories are taken into account to construct an indicator to estimate the balance between a set or bundle of ecosystem services. The relationship between the outputs of this work and the development of rapid assessment and accounting frameworks is discussed.

Crown Copyright © 2011 Published by Elsevier Ltd. All rights reserved.

1. Introduction

Although the assessment of ecosystem services is currently the focus of intense policy interest (ten Brink, 2011; European Commission, 2011), there is often a lack of empirical information about service flows and how they are changing over time. These gaps in our knowledge arise from both the complexity of measuring ecosystem service outputs directly and from the fact that existing

monitoring systems were not designed to deliver such information. As a result we are forced to rely on either proxy measures derived from empirical data (indicators) or modelled estimates.

The difficulties of assessing changes in ecosystem service outputs pose particular problems for those seeking to develop integrated methods of economic and environmental accounting, the success of which depends on establishing clear relationships between economic activities and ecosystem functioning. Much work has focused on trying to capture relationships through the development of ‘production’ and ‘value’ functions (e.g. Daily and Matson, 2008; Kienast et al., 2009; Tallis et al., 2008; Tallis and Polasky, 2009), that model the link between ecosystem service

* Corresponding author. Tel.: +44 115 8467398; fax: +44 115 9515249.
E-mail address: marion.potschin@nottingham.ac.uk (M. Potschin).

outputs and the sets of biophysical and management factors that influence them. Tallis and Polasky (2011) note, however, that while the production function concept holds promise, recent approaches differ in their complexity and the current challenge is to determine both reliability and suitability in different decision making contexts. Given this background, the aim of this study is to examine how sets of functional relationships for services can be developed in a robust and credible way that is relevant to the land accounting framework developed by the European Environment Agency (EEA, 2006) and others. These accounting methods enable information on land cover stock and change to be described systematically at a range of spatial scales, and in particular, enable data on land cover and use change from different sources to be brought together in an integrated and consistent way. In more general terms this paper also examines how land cover and use data can be used for the development of a multi-criteria approach to monitoring changes in ecosystem service potential, defined as the *capacity* of land to deliver a range of ecosystem services at continental scales, in order to identify where significant changes in natural capital might be taking place.

In the discussion that follows we consider both 'land cover' and 'land use', and take the former to be defined primarily by the biophysical characteristics of a given parcel and the latter the utility that people derive from it. Although conceptually these two ideas define distinct characteristics of land, in practice the classification systems used for many data sources conflate cover and use descriptions, and so the two are not always so easy to separate (Jansen and Di Gregorio, 2002; Comber, 2008; Haines-Young, 2009; Seppelt et al., 2012). Clearly in identifying how particular land cover types might support particular services, an understanding of how people use the land or ascribe some purpose to it is fundamental. Thus we deliberately refer to 'cover' and 'use' in this analysis as dictated by context.

2. Methodological context

2.1. Background

Cowling et al. (2008) have argued that we need to develop an 'operational model' for mainstreaming ecosystem services in decision making, and that such a model must include methods for making and combining biophysical, social and valuation assessments. Social assessments, they suggest, are used to help understand the values and priorities of the people who have the capacity to manage an ecosystem service or who benefit from it, and so ideally should precede biophysical assessments. Within this context biophysical assessments can then supply information about the types and location of the biophysical features that affect the capacity to generate ecosystem services, the flows of services over space and time in relation to beneficiaries, and the impacts of changes in land and water on service delivery. Valuation assessments then take place, according to Cowling et al. (2008) at the 'intersection' of the biophysical and the social components. Other commentators (e.g. Haines-Young and Potschin, 2010; De Groot, 2010) have also sought to describe the relationships between ecosystem structure and function and the values and benefits derived by people, in terms of the 'cascade model'. This framework seeks to identify the key elements that need to be considered in any comprehensive ecosystem assessment and shows how ecosystem services sit at the interface of the biophysical and social components of a social–ecological system (see also Potschin and Haines-Young, 2011). According to this model an ecosystem function only becomes a service if a beneficiary can be identified (cf. Boyd and Banzhaf, 2007; Fisher et al., 2009; Busch et al., 2012).

Although the work of Cowling et al. (2008) is indicative of what is required to bring ecosystem assessments into the mainstream, these authors do not make an explicit link to environmental accounting methods which can potentially provide much of the biophysical information that is required. In fact, the arguments of Cowling et al. (2008) echo many of these surrounding the role of land accounts for integrated economic and environmental accounting (e.g. SEEA 2003, see United Nations, 2003). The latter which sees them as complementary but distinct from economic assessments, providing the physical basis for understanding how the underlying resource base changes over time. While monetary values may eventually be attached to such changes within the SEEA framework it is generally accepted that biophysical accounts can provide much of the information that we need to make a judgement about the extent to which current trends are sustainable. The use of land accounts would also go some way to addressing the critique of current approaches to biophysical assessments made by Cowling et al. (2008), who argue that they tend to lack social context. Accounting frameworks do not seek to describe the world in its totality, but rather describe what are considered to be the most important or relevant aspects in a specific problem or decision making context.

There are an increasing number of studies that attempt to map landscape functionality or to link landscape properties to the potential output of ecosystem goods and services and the physical behaviour of landscapes and catchments (e.g. Bindraban et al., 2000; Leibowitz et al., 2000; Wu et al., 2003; Peterseil et al., 2004; Wrba et al., 2004; Fohrer et al., 2005; De Groot, 2006; Egoh et al., 2008; Naidoo et al., 2008; Verburg et al., 2008; Willemsen et al., 2008; Nelson et al., 2009; Burkhard et al., 2009; Maes et al., 2011). A recent mini-review is provided by Bolliger and Kienast (2010). Although they conclude that landscape functions can now be regarded as a 'powerful tool to assess the potential of landscapes to deliver ecosystem services in a changing environment' (Bolliger and Kienast, 2010, p. 3), they suggest that recent work still has a number of limitations. It has, for example, tended to focus on provisioning and regulating services, rather than cultural aspects. Furthermore, few studies have looked at the issue of trade-offs between landscape functions and the need to make multi-functional assessments, particularly at broad spatial scales.

A difficulty of these recent debates arises, however, in relation to terminology, and in particular the way the term 'function' is used, when associated with the terms ecosystem and landscape (cf. Jax, 2005; Bastian et al., 2012). In this and our earlier paper (and following the cascade model) we use the term 'ecosystem function' to refer to those properties of an ecosystem that give rise to a service. The term 'landscape function' would therefore have a similar, but more specific meaning, referring to the capacities of land (and especially land cover mosaics) to generate, or give rise to, a service. In either case, we argue, that the relationships between the ecosystem or land characteristics and service output can be modelled by some relationship, which others have referred to variously as 'production' or 'value' functions. When the employed in this way the term 'function' is being used in its mathematical sense to refer to a calculation device that assigns a unique output value to one or more inputs of a specified type. Nevertheless the intention of both usages is clear – to understand ecosystem services in terms of a set of cause–effect relationships. In the discussion that follows we will use the term landscape function to focus specifically on the subset of ecosystem functions that are related to land (Nedkov and Burkhard, 2012; Burkhard et al., 2012).

The studies by Burkhard et al. (2009) and Kienast et al. (2009) illustrate some key aspects of recent approaches to modelling landscape functions. Both use expert-based look-up tables to link different aspects of land cover and use with the potential to generate different kinds of service output. While Burkhard et al. (2009) propose a single, generic look-up table, Kienast et al. (2009) use a set

of context variables, such as biogeographical region, altitude, slope and proximity to urban areas to modify the pattern and strength of linkages. To some extent the difference in approaches reflect the varying scale perspectives of the two studies; Burkhard et al. (2009) were focussed more at the regional levels, while Kienast et al. (2009) were concerned with a continental-wide assessment. Nevertheless the contrast does illustrate the importance of better understanding what determines the strength and pattern of these functional linkages. Indeed, Burkhard et al. (2009) recognise the hypothetical nature of their matrix, and emphasise that it may need to be modified as conditions change from one place to another and new knowledge develops.

Willemen et al. (2008) also recognise the utility of rule-based approaches for mapping and quantifying landscape functions but suggest that other methods are applicable, depending on the types of data available. These include linking landscape functions to land cover or policy defined areas, and assessing landscape functions with empirical models using spatial indicators. The first of these is, they suggest, appropriate where landscape functions are 'completely observable', as in the case, say, of residential land or intensive livestock production. By contrast, the approach based on empirical models or spatial indicators is most suitable in situations where the location and extent of particular functions as to be inferred through some proxy; thus while 'tourism' cannot be directly observed, a number of metrics can be used to estimate its level, and multivariate models may even be constructed based on empirically observed relationships. For these authors the 'rule based' approach is confined to those situations where there is no information on function location and extent. They go on to show how these approaches can be used to make an assessment of eight landscape functions in a small area the Gelderse Vallei, a transitional rural area in the Netherlands.

The study by Peterseil et al. (2004) also illustrates some variations on the approaches described above, in their work on ecological sustainability of Austrian agricultural landscapes. They use rule-based approaches to link landscape structure and pattern to their sustainability indicator, but the rules were based on fuzzy set theory, and so made a more probabilistic association between input and output variables for each landscape type. They also employ an alternative approach based on a statistically derived empirical function, that measured the intensity of land use (hemerobiotic state), and in terms of mapped output found it showed close correspondence with the rule-based method.

Studies based explicitly on the use of modelled landscape functions include those of Chan et al. (2006), Naidoo et al. (2008) and Nelson et al. (2009). All used information on spatial variation in land cover and use as key inputs. The first two examined the coincidence of regions of high service output and zones of importance for biodiversity conservation, while the later looked at changing patterns of service output under a range of future scenarios. These studies like those of Lesta et al. (2007), for example, use GIS techniques to map the potential of different areas to generate service output. Although quantitative in character these assessments are, like the rule-based work discussed earlier, fundamentally hypothetical, in that they use a range of surrogates or proxies to map or estimate variations in service output.

With the availability of systems such as InVEST¹ (see for example Tallis and Polasky, 2009) and MIMES² it is likely that the use of model-based methods for mapping landscape functions will increase, and that a wider range of decision support tools will become available. The problem with these methods is that the models themselves are often sector or process specific and do not easily

allow the linkages between ecosystem services to be considered. Clearly, by referencing outputs to a common spatial framework these methods allow the coincidence landscape functions can be examined, and potential 'hot-spots' to be identified. However, the issues of trade-offs at the functional level are more difficult to address, and the impacts of different drivers of change on bundles of services is more difficult to determine.

We suggest that while there has been considerable progress in the use of rule- and model-based assessments of landscape functions and their links to the output of ecosystem goods and services, there is still a pressing need to find ways of describing the marginal or incremental changes in service output resulting from modifications of land cover and use, or other environmental change. The issue of marginal change is critical in making an economic assessment of the impacts of alternative policy or management strategies on ecosystem services because it defines the conditions under which such analyses are appropriate (Fisher et al., 2008).³ More generally the issue of marginality is important in terms of understanding in physical terms, the trade-offs and synergies that exist within the bundles of services that particular places deliver, and the shifts that have occurred as the result of past changes, or which might occur under plausible future scenarios (see for example, Tallis et al., 2008). The need for such analysis of marginal or incremental change has, for example, recently been demonstrated in the *UK National Ecosystem Assessment*.⁴ The study successfully brought together a large body of empirical data to establish the current status of ecosystem services at the national scale and the impacts of past trends. However the work also exposed the problem that we still lack sufficient understanding of the functional relationships between land cover and use, and ecosystem services so that the impacts of different drivers of change can be examined through the development of scenarios or by the comparison of different policy or management options (Haines-Young et al., 2011). These problems are even more pressing as we look to continental or global scale studies, and in particular how we might cope with a situation in which decision makers require regular updates when integrated economic and environmental accounting systems are more common place.

2.2. Approach

To overcome the difficulties highlighted above we have in this study explored further the use of 'expert-' and 'literature-driven' models. The study builds on our earlier work (Kienast et al., 2009) which sought to establish the multiple links between land cover and use, and potential ecosystem service outputs in different geographical contexts across Europe. The specific set of potentials related to land characteristics referred to as 'landscape functions' in this earlier study; in this paper we follow a similar logic and regard these landscape functions as part of a wider set ecosystem functions associated with land and landscapes. Although our earlier work looked at the impacts of land cover and use change through scenarios, it did not consider the more immediate kind of problem that decision makers would face when monitoring incremental change against a base-line. This situation arises, for example, where we seek to understand the impact of transformations in land cover on the marginal or incremental values of services (see for example Bateman, 2011). It also arises in the context of land and ecosystem accounting.

In the current paper we focus mainly on the 'accounting problem', and in particular, the issue of *physical* accounting for land

¹ <http://www.naturalcapitalproject.org/InVEST.html>.

² <http://www.uvm.edu/giee/mimes/>.

³ Under conditions where change is non-linear and irreversible (i.e. non-marginal), other ethical and political considerations may be more important than economic ones in any assessment.

⁴ <http://www.uknea.org.uk>.

cover and use, and the potential output of ecosystem services. The context is provided by the *Land and Ecosystem Accounts for Europe 1990–2000* (EEA, 2006) which showed that while much more work needed to be done to develop these accounts for whole ecosystems, for those characteristics related to land cover at least, conceptual frameworks and data infrastructures are sufficiently well developed for land accounts to be made operational. This study and subsequent work by the EEA has taken the CORINE Land Cover for 2000 as the base-line, and use a 'change-only' approach to make estimates of past trends (since 1990, as recorded in the earlier CORINE mapping study for 1990), and more recent changes. The case of monitoring change since 2000 is particularly interesting, because the 2006 CORINE update involves the use of new sources of remotely sensed data (GlobCover 2006) with different spatial spectral characteristics. Although direct comparisons of the state of land cover at two times is possible, using rule-based or modelled functions, in this paper we explore whether the types of changes observed can be used as indicators of the changing outputs of ecosystem services as monitored against some base-line.

As in our earlier work (Kienast et al., 2009) we note that the task of using rule-based or expert-driven assessments is a challenging one. If these methods are to be used operationally it is clear that they require checking to determine whether they are capable of providing plausible yet parsimonious outputs given the current state of knowledge about the way ecosystem functions link to potential service output. That same proposition also applies here, and as in the initial study we seek to make independent tests of the mapping output in order to better understand how robust these methods are. In this way we hope to come close to mainstreaming ecosystem services in the way that Cowling et al. (2008) and others suggest.

3. Analysis

Our original study (Kienast et al., 2009) used binary links defined by a panel of five experts to express whether specific land cover and use types or other environmental properties have a supportive or neutral role in the potential to generate ecosystem services. The mapping of these relationships was made at a relatively coarse spatial resolution, namely NUTS-x regions of Europe, the mean size of which is around 8000 km²; the study area was the EU-25 plus Switzerland and Norway. The current work refines these methods in several ways with the aim of making the outputs consistent with the Land and Ecosystem Accounting (LEAC) framework that has been developed by the EEA; we highlight the major changes to the methodology compared to the 2009 study:

- (a) We use the same 1 km × 1 km scale accounting grid employed by LEAC and the three CORINE land cover maps for 1990, 2000 and 2006 that are now available. Thus the mapping and accounting outputs are at a much finer scale resolution. These data are used alongside a range of context variables describing associated land characteristics (Table 1). As reporting and mapping units we still use the NUTS-x regions.
- (b) Our approach uses the hierarchical classification of ecosystem services proposed by the EEA in their work on the revision of the SEEA 2003 (United Nations, 2003). This new hierarchical classification, known as CICES,⁵ seeks to provide consistent but more thematically balanced classification of ecosystem services than that used in the Millennium Ecosystem Assessment (MA, 2005) and The Economics of Ecosystems and Biodiversity⁶

(TEEB). In the classification the categories are designed to be more equal their conceptual scope. It has also been designed to allow an easier read-across between different studies by taking account of the way service definitions vary at different spatial and thematic scales. Four ecosystem services at the third 'service type' level in the CICES hierarchy were selected for analysis: "Crop-based production", from within the Nutrition class of the Provisioning theme; "Wildlife products", from the class of Biotic Materials, again in the Provisioning theme; "Habitat diversity", from the Lifecycle Maintenance and Habitat Protection class in the Regulating thematic group; and finally "Recreation", as an example of a service from the Experiential class under the Cultural theme.

Different ways of linking the landscape functions with independent parameters have been presented in the literature as a way of generating ecosystem service indicators. These include process-based links (Krönert et al., 2001; Haase et al., 2007) or look-up tables expressing to what degree land characteristics hinder or support a particular landscape function (Burkhard et al., 2009). Given our experience reported in Kienast et al. (2009), we decided to use the link table shown in Table 1. Binary links (0/1 look-up tables) are used for the land cover and use data (Table 2) to express whether a land characteristic has a supportive role (value 1) or a neutral role (value 0), for a given CICES ecosystem service. As the scoring system shows, context information relating to geographic location or other land characteristics is used to modify the lookup scores. The land characteristics were net primary production gathered via remotely sensed MOD17 data at 500 m resolution (Zhao et al., 2005), and bioclimatic and landscape properties such as whether the area for which a projection was being made was within mountainous terrain, or adjacent areas with a landscape and nature protection designation (see Table 1 for further details of data sources). The link table (Table 2) was generated with the aid of expert knowledge and the scientific literature. It took several iterations and rounds of discussion before the findings from the literature and the expert assessments were consistent and considered to be a credible framework for describing how specific land characteristics are associated with each service.

In addition to estimating the current potential of different areas to deliver services, a novel aspect of this study was the analysis of two components of future change: the assessment of the impact of marginal changes in service output resulting from recent historical land cover and use change (1990–2000); and, how projected changes up to 2030 might alter the capacity of land to deliver specific services.

The historic assessment of marginal changes was undertaken using the Land and Ecosystem Accounting database (LEAC) created by the EEA using successive CORINE Land Cover data. The analysis of these incremental changes was included in the study in order to examine whether recent trend data could add additional insights to spatial assessment techniques, particularly where change against some base-line status is of interest to decision makers. For the analysis, two time steps were examined: the changes identified by a comparison of CORINE Land Cover for 1990 and the 2000 base-line; and, a comparison between CORINE Land Cover 2000 and the recently published data derived from GlobCORINE that provides an insight into the state of land cover for 2006. Unfortunately GlobCORINE is not available for the UK and Switzerland and so these countries have been left out of this part of the analysis.

The accounting framework developed by the EEA (EEA, 2006) includes both a classification of land cover and use types and the processes by which one type is transformed into another. The classification of flows is hierarchical in structure, like the classification of land cover and use, and has been used to map where particular types change, like urban sprawl or agricultural

⁵ Common International Classification of Ecosystem Services, see Common International Classification of Ecosystem Services, see <http://unstats.un.org/unsd/envaccounting/ceea/meetings/JNCEEA-5-7-Bk1.pdf>.

⁶ <http://www.teebweb.org/>.

Table 1
Final set of independent data describing the basic land characteristics.

Dataset number and short description	Number of classes, resolution Class description	Source	Approx. year of reference
1. Corine land cover, level 2, including Norway and Switzerland	15 classes (level 2), 1 ha 1.1 urban fabric 1.2 industrial, commercial and transportation 1.3 mine, dump and construction sites 1.4 artificial non-agricultural vegetated areas 2.1 arable land 2.2 permanent crops 2.3 pastures 2.4 heterogeneous agricultural areas 3.1 forest 3.2 shrub and/or herbaceous vegetation associations 3.3 open spaces with little or no vegetation 4.1 inland wetlands 4.2 coastal wetlands 5.1 inland waters	EEA (2002) and Hazeu et al. (2007)	2000
2. Mountain terrain	2 classes, 0.2 km ² 0: no mountain terrain 1: mountain terrain: >1000 m or 500–1000 m and >5% slope	GTOPO30	
3. Nature protection zones	3 classes, 1 ha 0: not adjacent to protection zone 1: inside a 10 km × 10 km square with 7–25% protection zone 2: inside a 10 km × 10 km square with >25.1% protection zone	World Database of Protected Areas (“WDPA”) compiled by the WDPA Consortium, including UNEP-WCMC	1970–2000
4. Landscape protection zones	3 classes, 1 ha 0: not adjacent to protection zone 1: inside a 10 km × 10 km square with 6–20% protection zone 2: inside a 10 km × 10 km square with >20.1% protection zone	World Database of Protected Areas (“WDPA”) compiled by the WDPA Consortium, including UNEP-WCMC	1970–2000
5. Mean Actual Net Primary Production (aNPP)	3 classes, 1 km ² 1: 0.009–0.35 kg C/m ² /year 2: 0.351–0.68 kg C/m ² /year 3: >0.681 kg C/m ² /year	MODIS MOD17 product (Zhao et al., 2005)	2001–2005
6. Buffered coast, wetlands, large rivers	2 classes, 1 ha 0: outside 500 m buffer 1: inside 500 m buffer	EEA (2002) and Hazeu et al. (2007)	2000
7. Land accounts for Europe for (a) 1990–2000; (b) 2000–2006, and (c) 2000–2030 (Eururalis 2.0 scenarios)	LEAC change classes 1 km 1990–2000, 9 classes, 1 km ² LCF1 Urban land management LCF2 Urban residential sprawl LCF3 Sprawl of economic sites and infrastructures LCF4 Agriculture internal conversions LCF5 Conversion from forested and natural land to agriculture LCF6 Withdrawal of farming LCF7 Forests creation and management LCF8 Water bodies creation and management LCF9 Changes of Land Cover due to natural and multiple causes	LEAC, EEA (2006), Meijl et al. (2006), Verburg et al. (2006), Westhoek et al. (2006), Verburg et al. (2009).	1990–2000 2000–2006 2000–2030

conversion, is occurring. For the purposes of the present study, the different types of change have been classified using the system of binary links with those changes likely to further support or enhance a particular service being scored as ‘S’, and those that are more neutral in their effects being given the score ‘N’. In contrast to the overall mapping of potential for the marginal changes we did attempt to recognise changes that might damage or detract from service output; these changes were scored as ‘D’ (Table 2).

The futures component of the work was based on EURURALIS 2.0 land use scenarios for 2000–2030 (Meijl et al., 2006; Verburg et al., 2006; Westhoek et al., 2006; Verburg et al., 2009), which are based on the four IPCC SRES land use scenarios. The land cover types used in EURURALIS were cross referenced with the land cover classes used for the preparation of the EEA land cover accounts. For the analysis of the impacts of the different scenarios the land cover and use changes were assigned a supportive, neutral or degrading role, as for the analysis of recent historic patterns. Although other studies have examined how ecosystem services might change under different assumptions about the future (e.g. Carpenter et al., 2006; Nelson et al., 2009), the analysis presented here sought to break new ground by identifying how different geographical areas might be grouped according to their change trajectories for bundles of ecosystem services. Recent service mapping studies have tended to focus on individual service responses. A motivation

for our analysis is the belief that, if society is to make decisions about the implications of trade-offs between services, then a more integrated perspective on bundles of services is necessary.

Given the fact that the methods applied in this study were based on expert judgement and literature review, we felt it important to test the robustness of the outputs critically (see Kienast et al., 2009 for details). Thus the assessment made for each of the four target services were compared with other independently generated continent-wide assessments (maps of ecosystem services or environmental parameters/indicators). While we would argue that these other assessments tended to be more generalised than the outputs from this study, they are sufficiently useful for testing the plausibility of the assessment made here. The details of the independent data used for testing the outputs are discussed below.

4. Results

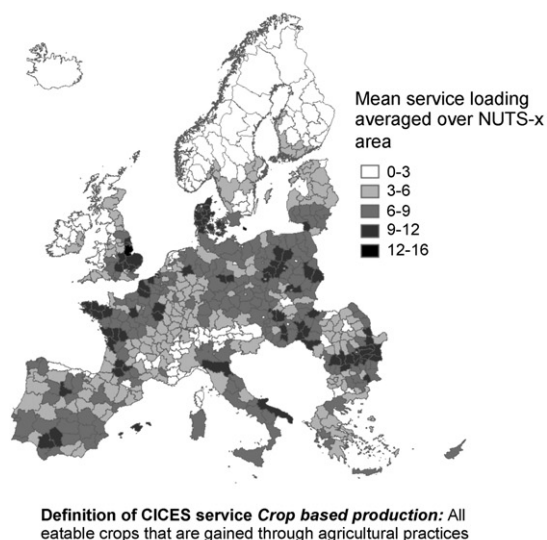
In Figs. 1–4 we present the results of the multi-criteria mapping for the target services; the scoring system used for each service and associated criteria are defined in Table 2. Although the analysis was made at the 1 km × 1 km resolution, the mapping has been aggregated to NUTS-x level so that the broad geographical patterns can be seen more easily. A full set of graphics is provided for “Crop-based production” (Figs. 1 and 2) to illustrate the nature of the output that

Table 2
Calculating the importance of a CICES ecosystem service for any given parcel of land (1 ha pixel) (see Table 1 for definition of datasets).

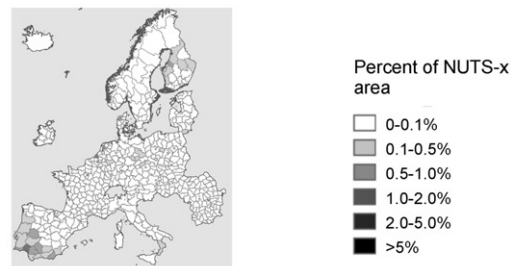
Dataset number, description	Ecosystem services				
	Food and beverage – example Crop-based Production	Biotic materials – example Wildlife products	Lifecycle maintenance and habitat protection – example Habitat diversity	Experiential – example Recreation	
<i>1 Corine land cover, level 2, including Norway and Switzerland (0: neutral; 1: supportive)</i>					
Class	Description				
1.1	Urban fabric	0	0	1	1
1.2	Industrial, commercial and transportation	0	0	0	0
1.3	Mine, dump and construction sites	0	0	0	0
1.4	Artificial non-agricultural vegetated areas	0	0	0	1
2.1	Arable land	1	0	0	0
2.2	Permanent crops	1	0	1	1
2.3	Pastures	0	0	1	1
2.4	Heterogeneous agricultural areas	1	0	0	1
3.1	Forest	0	1	1	1
3.2	Shrub and/or herbaceous vegetation associations	0	1	1	1
3.3	Open spaces with little or no vegetation	0	1	1	1
4.1	Inland wetlands	0	0	1	1
4.2	Coastal wetlands	0	1	1	1
5.1	Inland waters	0	1	1	1
	Weight for service loading (if pixel is "1", multiply by...)	*10	*10	*10	*10
<i>2 Mountain terrain (weight to be added to basic service loading of dataset 1)</i>					
0	No mountain terrain	0	0	0	0
1	Mountain terrain: >1000 m or 500–1000 m and >5% slope	-2	1	2	3
<i>3 Nature protection zones (weight to be added to basic service loading of dataset 1)^a</i>					
0	Not adjacent to protection zone	Not relevant	Not relevant	0	0
1	Inside a 10 km × 10 km square with 7–25% protection zone	Not relevant	Not relevant	2	1
2	Inside a 10 km × 10 km square with >25.1% protection zone	Not relevant	Not relevant	4	2
<i>4 Landscape protection zones (weight to be added to basic service loading of dataset 1)^a</i>					
0	Not adjacent to protection zone	Not relevant	0	0	0
1	Inside a 10 km × 10 km square with 6–20% protection zone	Not relevant	1	1	1
2	Inside a 10 km × 10 km square with >20.1% protection zone	Not relevant	2	2	2
<i>5 Mean Actual Net Primary Production (weight to be added to basic service loading of dataset 1)^a</i>					
1	0.009–0.35 kg C/m ² /year	2	2	Not relevant	Not relevant
2	0.351–0.68 kg C/m ² /year	4	4	Not relevant	Not relevant
3	>0.681 kg C/m ² /year	6	6	Not relevant	Not relevant
<i>6 Buffered coast, wetlands, large rivers (weight to be added to basic service loading of dataset 1)^a</i>					
0	Outside 500 m buffer	Not relevant	0	0	0
1	Inside 500 m buffer	Not relevant	1	2	3
16	Maximum service loading per pixel	20	20	20	20
<i>7 Land accounts for Europe 1990–2000; 2000–2030 (D: degradative; N: neutral; S: supportive) (area of D, N, S added over dominant land use type or biogeographic region)</i>					
LCF1	Urban land management	D	D	N	S
LCF2	Urban residential sprawl	D	D	D	S
LCF3	Sprawl of economic sites and infrastructures	D	D	D	D
LCF4	Agriculture internal conversions	N	N	N	N
LCF5	Conversion of forest and natural land to agriculture	S	D	D	D
LCF6	Withdrawal of farming	D	S	S	S
LCF7	Forests creation and management	D	S	S	S
LCF8	Water bodies creation and management	D	S	S	S
LCF9	Changes of cover by natural and multiple causes	N	N	N	N

^a Factors labelled 'Not relevant' were assumed to have no influence on that service and were omitted from the calculation.

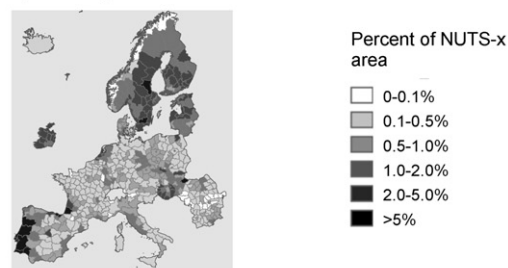
A1 Mean importance score for Crop-based production by NUTS-x region in 2000



A3.1. Change of supportive land-use 2000-2006² for Crop-based production

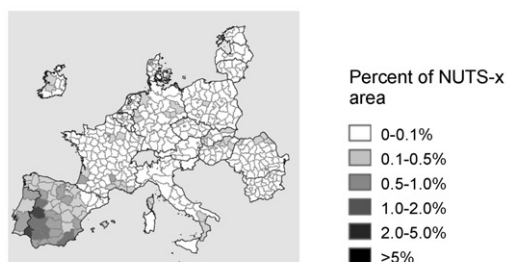


A3.2 Change of degradative land-use 2000-2006² for Crop-based production

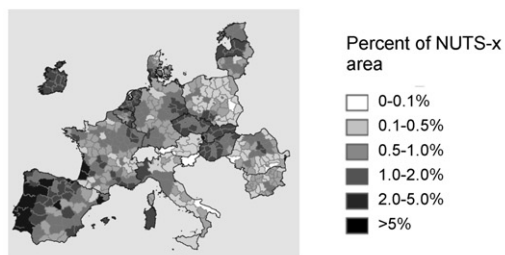


² for countries where GISAT 2000-2006 database applies

A2.1. Change of supportive land-use 1990-2000¹ for Crop-based production

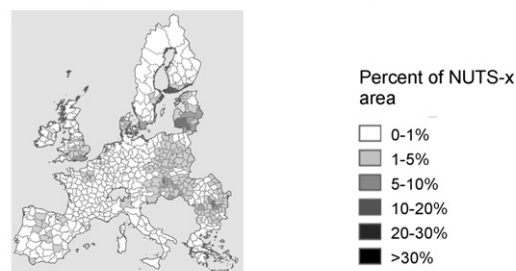


A2.2 Change of degradative land-use 1990-2000¹ for Crop-based production

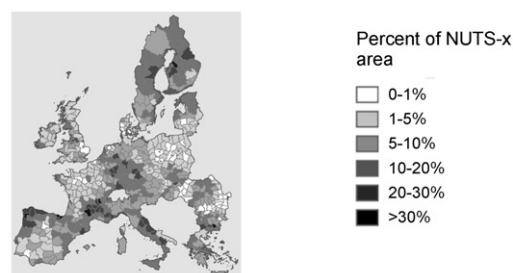


¹ for countries where LEAC database applies

A4.1. Expected change of supportive land-use 2000-2030 (scenario A1)³ for Crop-based production



A4.2 Expected change of degradative land-use 2000-2030 (scenario A1)³ for Crop-based production



³ for countries where EURURALIS 2.0 database applies

Fig. 1. Analysis of results for Crop-based production (part 1).

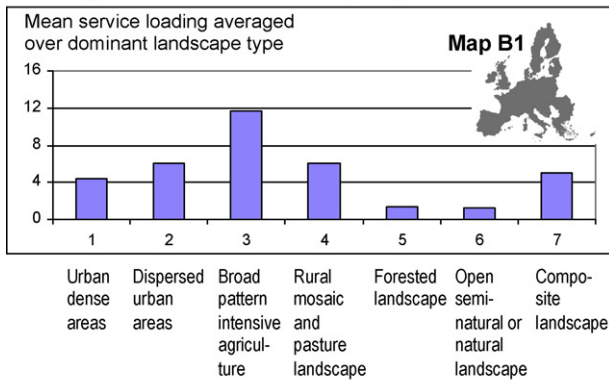
can be generated by the approach; for “Wildlife products”, “Habitat diversity” and “Recreation” we present a limited number of maps and diagrams (Figs. 3 and 4).

The analysis for “Crop-based production” (Figs. 1 and 2) maps all the areas that are important for food crops produced through commercial agriculture. The spatial analysis of the situation in year 2000 (Fig. 1, map A1) clearly highlights the expected hotspots of arable production such as the eastern part of the UK, northern France, parts of Belgium and the Netherlands, and Denmark, together with

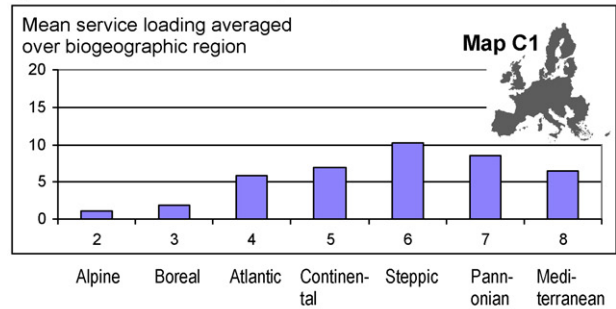
a broad sweep of land in the northern part of Germany and Poland. Areas with low potential scores are the mountains and the Nordic regions.

The analysis of the marginal changes between 1990 and 2000 in the potential to support crop-based services is particularly interesting. A comparison of the two maps (Fig. 1, A2.1 and A2.2) suggests that changes tending to reduce the potential for “Crop-based production” tend to be more widespread than those that would tend to enhance it. Although a comparison of the relative changes for

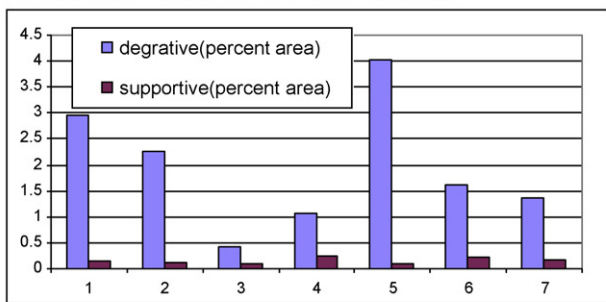
B1 Mean importance score for Crop-based Production by dominant landscape type in 2000 (for area shown on map B1)



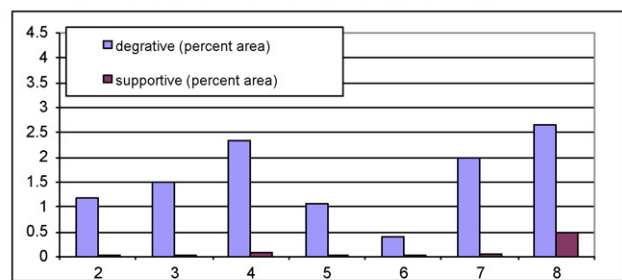
C1 Mean importance score for Crop-based Production by biogeographic region in 2000 (for area shown on map C1)



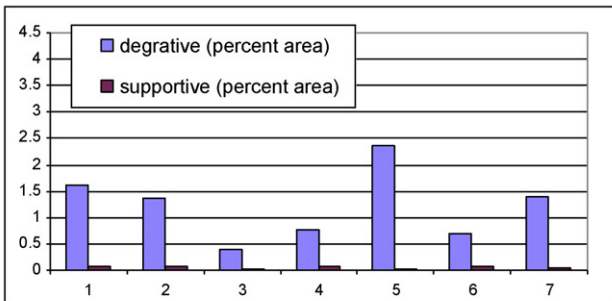
B2 Relative change in area (%) of supportive/degradative land-use 1990-2000¹ for Crop-based Production per dominant landscape type



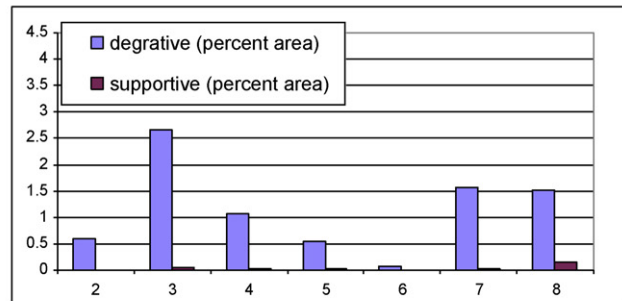
C2 Relative change in area (%) of supportive/degradative land-use 1990-2000¹ for Crop-based Production per biogeographic region



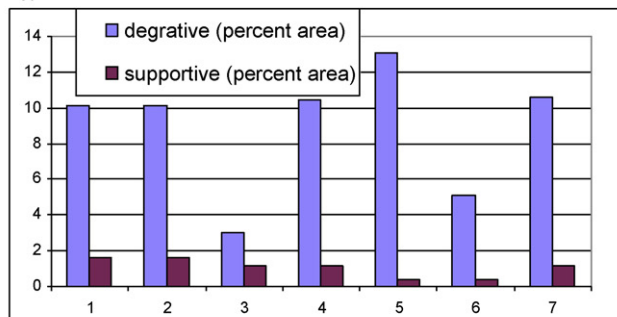
B3 Relative change in area (%) of supportive/degradative land-use 2000-2006² for Crop-based Production per dominant landscape type



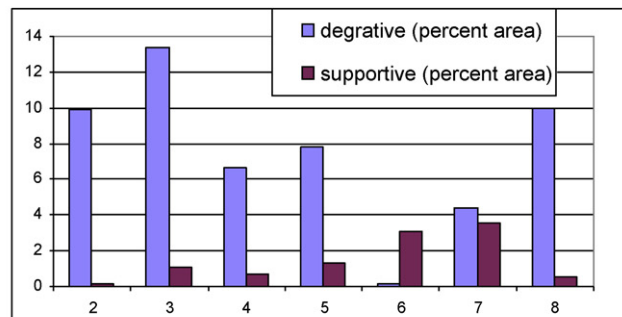
C3 Relative change in area (%) of supportive/degradative land-use 2000-2006² for Crop-based Production per biogeographic region



B4 Relative change in area (%) of expected supportive/degradative land-use (2000-2030)³ for Crop-based Production per dominant landscape type



C4 Relative change in area (%) of expected supportive/degradative land-use (2000-2030)³ for Crop-based Production per biogeographic region



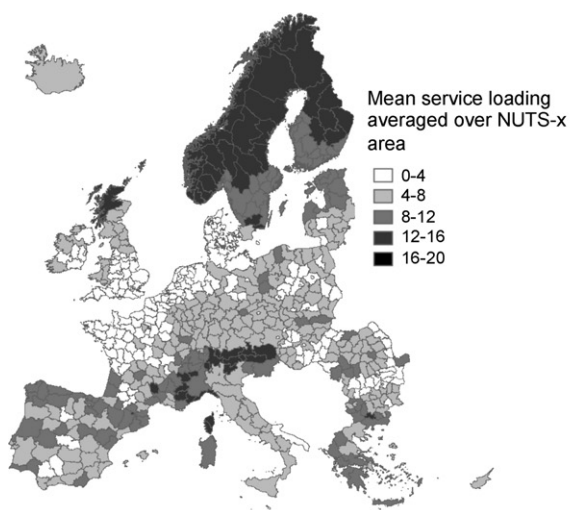
¹ for countries where LEAC database applies

² for countries where GISAT 2000-2006 database applies

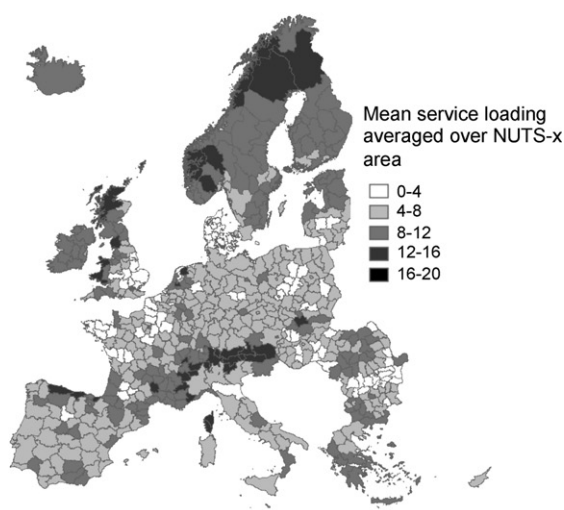
³ for countries where EURURALIS 2.0 database applies

Fig. 2. Analysis of results for Crop-based production (part 2).

A1w Mean importance score for Wildlife products by NUTS-x region in 2000



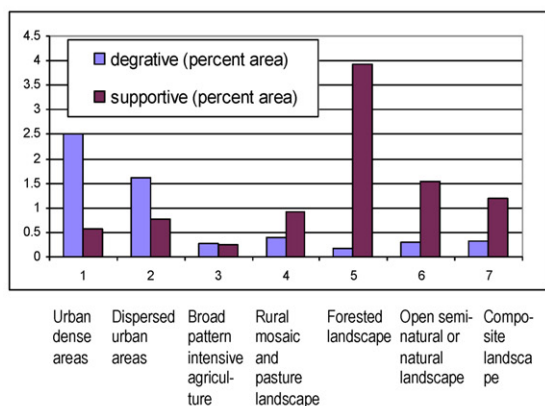
A1h Mean importance score for Habitat diversity by NUTS-x region in 2000



Definition of CICES service Wildlife products: All non-eatable raw products that are gained through non-agricultural practices or as a side-product of commercial and non-commercial forests, primarily in non-intensively used land or semi-natural and natural areas

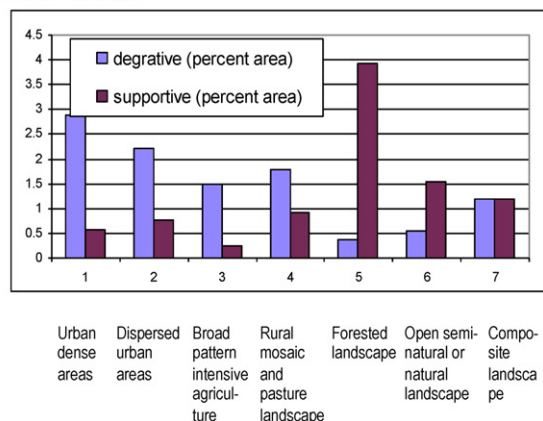
Definition of CICES service Habitat diversity: All areas that contribute to biological diversity and are thus assumed important for biodiversity enhancement

B2w Relative change in area (%) of supportive/degradative land-use 1990-2000¹ for Wildlife products per dominant landscape type



¹ for countries where LEAC database applies

B2h Relative change in area (%) of supportive/degradative land-use 1990-2000¹ for Habitat diversity per dominant landscape type



¹ for countries where LEAC database applies

Fig. 3. Analysis of results for Wildlife products and Habitat diversity.

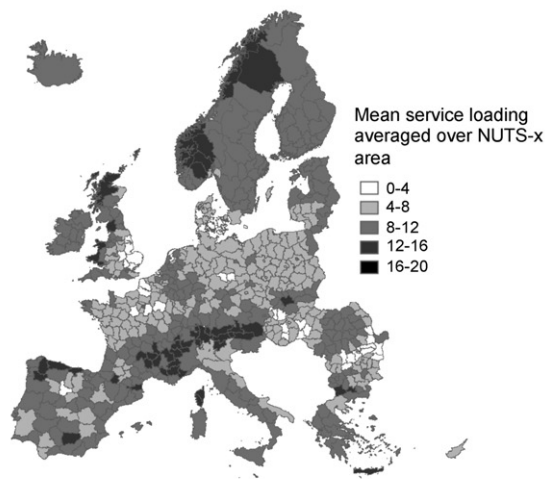
the two historic time periods (1990–2000; 2000–2006) is made more difficult because of the lack of data for some countries, it is clear that degradative changes continue to dominate. As is clearly visible in maps A3.1 and A3.2 (Fig. 1) Portugal and the western Mediterranean coast of Italy as well as the Nordic countries show the most marked transformations. The analysis of the EURUALIS scenarios suggests that these degradative trends may continue into the future, at least for the A1 ('Global Economy') scenario (Fig. 1, maps A4.1 and A4.2). Since the aim of this paper is a methodological one, we only present the results for the A1 scenario as a test of concept because the land cover and use changes under this narrative are potentially the most extreme. This scenario envisages rapid economic growth, global population peaking at around 9 billion in 2050, rapid uptake of new technologies and globalised societies.

Fig. 2 provides a comparison of potentials for "Crop-based production" across the dominant landscape types and major

biogeographical zones defined in the earlier accounting work of the EEA (EEA, 2006). The classification of dominant landscape was based on the most widespread land cover and use types in each 1 km × 1 km cell of the European grid. The biogeographical zones used in the EEA account study were based on those devised in support of the EU Habitats Directive (EEA, 2006). The loss of potential in "Crop-based Production" is particularly marked in forested landscapes (Fig. 2, diagrams B2–B4) supporting the hypothesis that changes in land management leading to land abandonment may be important in these areas. Other processes tending to lead to a reduced potential for "Crop-based production" include urban sprawl in those areas assigned to the Urban and Dispersed Urban landscape types defined by the EEA (Fig. 2, diagrams B2–B4).

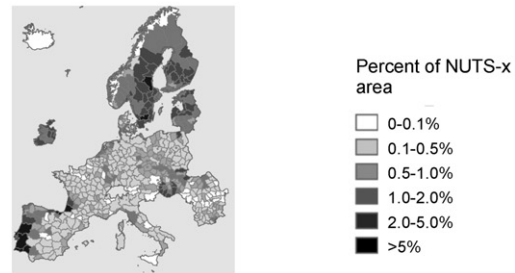
"Wildlife products" belongs to the service group Biotic Materials in the CICES system; it includes the provisioning of all non-edible raw material products that are gained through non-agricultural

A1 Mean importance score for Recreation by NUTS-x region in 2000

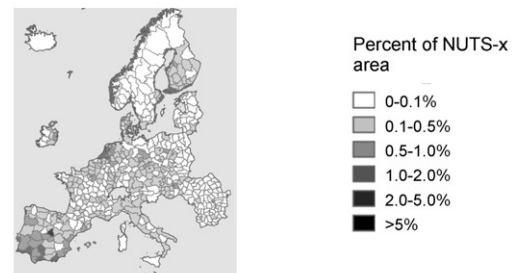


Definition of CICES service Recreation:
All areas that where landscape properties are favorable for recreation purposes

A3.1. Change of supportive land-use 2000-2006² for Recreation

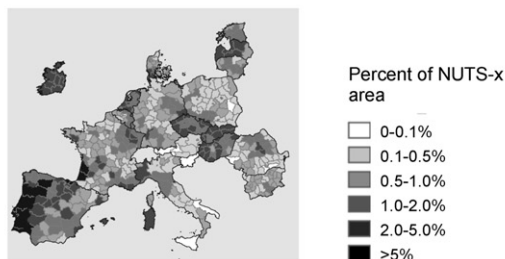


A3.2 Change of degradative land-use 2000-2006² for Recreation

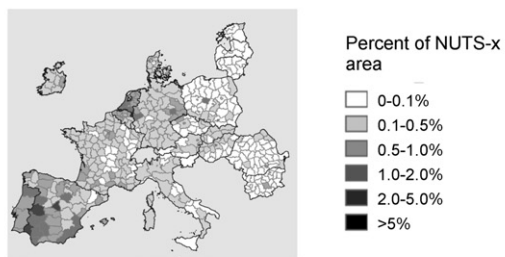


² for countries where GISAT 2000-2006 database applies

A2.1. Change of supportive land-use 1990-2000¹ for Recreation

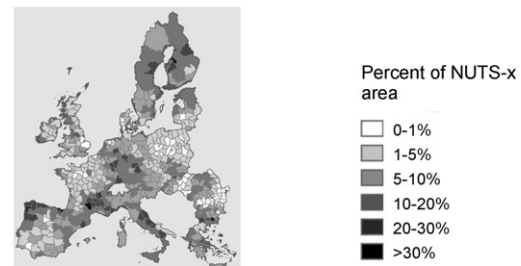


A2.2 Change of degradative land-use 1990-2000¹ for Recreation

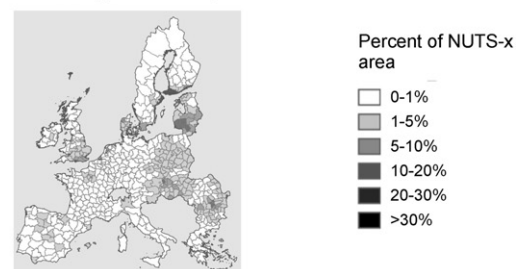


¹ for countries where LEAC database applies

A4.1. Expected change of supportive land-use 2000-2030 (scenario A1)³ for Recreation



A4.2 Expected change of degradative land-use 2000-2030 (scenario A1)³ for Recreation



³ for countries where EURURALIS 2.0 database applies

Fig. 4. Analysis of results for Recreation.

practices or which are produced as a by-product of commercial and non-commercial forests, primarily in non-intensively used land or semi-natural and natural areas. The map of service potential (Fig. 3, map A1w) shows high loadings for the Alpine, Boreal and Mediterranean zones, partly due to the more natural or less intensively used character of these landscapes and their associated land-uses. As might be anticipated given the changes in “Crop-based production”, those landscapes where recent changes have tended to enhance the provision of these services are those where land abandonment may be underway; forested and semi-natural and natural

landscapes show the most marked positive changes for this service, whereas those dominated by urban cover show the most negative scores. The changes in areas dominated by intensive agriculture are more balanced (Fig. 3, diagram B2w).

The analysis for the regulating service “Habitat diversity” seeks to identify all the areas with potential to support biodiversity (Fig. 3). The results suggest that the potential to provide this service is fairly evenly distributed across Europe except in areas with intensive agriculture. Urban habitats are assumed to contribute important habitats that are increasingly vanishing in Europe such

Table 3
Quantitative and qualitative map evaluation with independent spatial data.

Service	Independent map	Method of quantitative comparison		Qualitative comparison (visual comparison)
		Proportion of agreement incl. 95% error margins	Un-weighted Kappa	
Food and beverage – example Crop-based Production	Map of ecosystem service “farmer livelihood” (source: Metzger et al., 2006, 2008)	0.4713 (chance agreement: 0.25) upper 95% 0.52 lower 95% 0.42 (4 class comparison; class boundaries = quartiles)	0.30 (fair agreement) upper 95% 0.36 lower 95% 0.23 (4 class comparison; class boundaries = quartiles)	Fair
Biotic materials – example Wildlife products	(a) Map “Areas with relatively little influence from urbanisation, transport or intensive agriculture” (EEA, 1998) (b) Map “Ratio of forest and semi-natural areas to agriculture and urban areas” (EEA, 1999)	No quantitative comparison possible	No quantitative comparison possible	Fair
Lifecycle maintenance and habitat protection – example Habitat diversity	Map “number of species (bird, plant, trees, reptiles)” (Metzger et al., 2008)	Below chance agreement	Below chance agreement	Low
Experiential – example Recreation	Map of indicator “tourist accommodation, bed places” (source: http://epp.eurostat.ec.europa.eu)	0.69 (chance agreement: 0.61) upper 95% 0.74; lower 95% 0.65 (2 class comparison; class boundary = 75% percentile)	0.22 (fair agreement) upper 95% 0.32; lower 95% 0.11 (2 class comparison; class boundary = 75% percentile)	Fair

as bare plots, open gravel or extensively used transport corridors (Fig. 3, A1h). Aside from urban areas, Scandinavia, Scotland and mountain areas are clearly highlighted as important zones, where increased habitat diversity is likely to increase the potential to support biodiversity. Overall, the extent of areas where the potential to provide this ecosystem function is being degraded appear to balance out those where changes are tending to enhance it. However, the spatial distributions of these changes are very uneven, so that the balance of pressures may be changing (Fig. 3, diagram B2h); the analyses suggest that while the potential is increasing in forested, open semi-natural and composite landscapes, degradative processes appear to dominate in the other landscapes. Such pressures suggest a polarisation in the ability of ecosystems to provide this important ecosystem function across Europe.

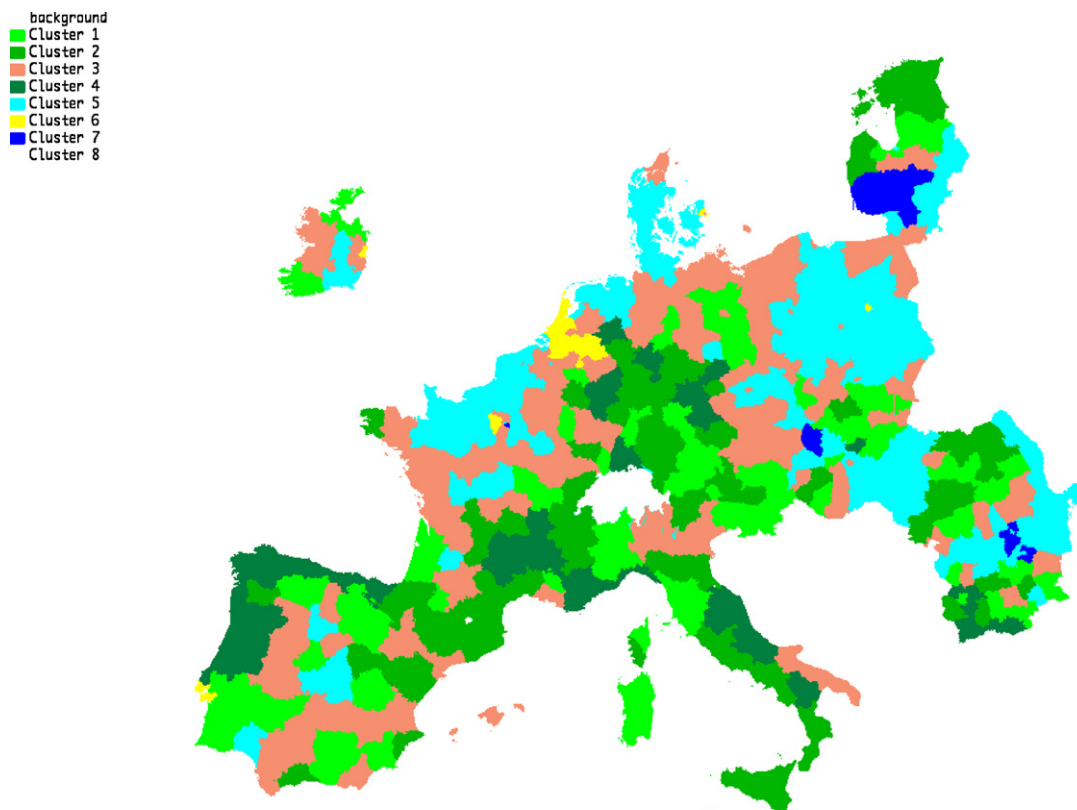
The final service considered here is “Recreation”, which belongs to the Intellectual and Experiential service group. It is broadly defined as all areas where landscape properties are favourable for active recreation purposes. The resulting analysis of potential shows a good match with the major European summer and winter destinations e.g. the Mediterranean and mountain areas (Fig. 4, map A1), although given the size of the NUTS-x units used for mapping the coastal areas of the Mediterranean do not stand out as might be expected. The observed changes in land cover and use suggest that the areas where the potential to supply this service is being degraded are relatively small in contrast to those where the potential is increasing. Many of the areas showing positive change are undergoing some form of re-wilding. As with the analysis of “Crop-based production”, a comparison of the relative changes for the two historic time periods is made more difficult because of the non-availability of data for some countries (see Fig. 4, maps A2.1, A2.2, A3.1, A3.2). However, it appears that the positive changes appear to dominate in both time periods in those areas where information exists, such as in the western part of the Iberian Peninsula and the Hungarian Plain. The analysis of the EURUALIS scenarios suggests that these positive tendencies may continue into the future (Fig. 4, maps A4.1 and A4.2).

Five independent sources have been used to check the output of the analysis of service potential presented above (Table 3). Comparisons have been made quantitatively by using proportional

agreement and the un-weighted Kappa metric (Hagen-Zanker, 2006 or Visser and de Nijs, 2006). The levels of agreement for the Kappa metric (low, fair, and good) follow the international standards (Landis and Koch, 1977). The qualitative evaluation, which was performed by visual comparison of maps, follows Pontius et al. (2008) and is based on a strict protocol. Maps were compared region by region by describing type and portion of agreement/disagreement on three qualitative levels (*low*: patches at different geographical locations, quantities not equal; *fair*: major patches match geographically, quantities match within a range of approx. 25%; *good*: major and minor patches match geographically, quantities match within a range of approx. 10%). “Crop-based production” showed fair agreement with the corresponding ecosystem service maps of Metzger et al. (2006, 2008). No statistically significant agreement was found for “Habitat diversity” when compared with the indicator map for number of species. Fair agreement was observed for “Recreation” when compared with tourist accommodation and bed-places. No qualitative or quantitative comparisons were possible for the provision of “Wildlife products”.

It is recognised that both landscapes and ecosystems are generally multi-functional in character; that is they can potentially deliver a range of benefits to people. The portfolio of services that are actually delivered depends on a range of external factors, including local biophysical conditions and the nature of the human use or management of the land. Rarely, however, can the output of benefits be optimised across all services. Instead, trade-offs often exist, with the option of increasing the output of particular types of service only being made possible by reducing the capacity to deliver others. While most other service mapping studies have considered changes in the output of single services, rarely have the analysis of such potential trade-offs been reported. A recent attempt is provided by Maes et al. (2011), however, who looked at the current spatial associations of services derived from applying the lookup of Burkhard et al. (2009) to patterns across Europe. The approach used in this study allows such an analysis of trade-off over time to be made.

The data on the mean service loading and marginal impacts of land cover and use change for the four target services across the two time periods (1990–2000 and 2000–2006) were aggregated to NUTS-x level, and used to define clusters or groupings of units with



Cluster Description

Cluster	Description
1	Nature-dominated areas where the potential to grow crops is strongly reduced since 1990 in favor of nature and recreation; moderate rewilding changes expected by 2030
2	Nature-dominated areas with little changes between 1990 and 2006; High nature / recreation potential by 2000; moderate rewilding changes expected by 2030
3	Stable mixed ecosystem service areas with medium potential for crop production and low potential for wildlife products, only little changes expected by 2030
4	Nature-recreation dominated areas, strong changes in crop potential between 1990 and 2006 in favor of rewilding, rewilding trend will continue strongly
5	Crop zones with relatively low nature / wilderness potential. Degrading crop potential 1990-2006 but positive perspectives by 2030. Nature capital will not be affected by increased production capacities
6	Highly populated areas with moderate crop production potential, high recreation and habitat potential. Recreation potential will increase at the expense of habitat, crop and wildlife services.
7	Intensive crop zones where crop potential decreased since 1990 but will strongly increase by 2030.
8	No data

Fig. 5. Mapping trade-off clusters based on trajectories of Crop-based production, Wildlife products, Habitat diversity and Recreation.

similar change trajectories. The results are shown in Table 4 and the clusters are mapped in Fig. 5.

In general there is a clear trade-off between the potential for the service “Habitat diversity” and “Crop-based production”. There are three clusters where the potential for “Habitat diversity” and “Recreation” have improved at the expense of “Crop-based production” (clusters 1, 2, and 4). Cluster 1 aggregates all areas where agricultural abandonment started early in the 1990s. The process is expected to continue in a moderate way by 2030. Cluster 2, on the other hand, shows little changes in the crop production potential between 1990 and 2006 and moderate re-wilding or extensification changes are expected by 2030. Finally, cluster 4 shows a strong reduction in the potential to grow crops between 1990 and 2006 in favour of extensification, a trend that appears to continue strongly under the A1 EURUALIS scenario.

There are two clusters that show a strong potential to grow crops (clusters 5 and 7). The NUTS-x regions belonging to cluster 5 have a relatively low score for “Habitat diversity” and “Wildlife

products”. There is a loss of potential for crop-based production in these regions between 1990 and 2006, but a more positive perspective projected for the period up to 2030. The natural capital of these regions, however seems to be least affected by increased production capacities. The NUTS-x areas in cluster 7 show only little changes in the dominant crop production mode (1990–2006) and will most likely maintain or even increase their potential to grow crops by 2030 according to the scenario. Their nature and recreation potential, however, is expected to decrease.

Among the seven clusters, only one (cluster 3) suggests that there are areas with a fairly stable mix of ecosystem services; it shows a medium potential for crop production, low potentials for “Wildlife products”, and only small changes are expected by 2030. Finally cluster 6 represents highly populated areas with moderate scores for “Crop-based production” and high potentials for “Recreation” and “Habitat diversity”. Due to a high projected demand in 2030, the potential for “Recreation” is likely increase at the expense of habitat, crop and wildlife services.

Table 4
Cluster definition for NUTS-x regions showing different patterns of trade-offs between four CICES ecosystem services through time, from 1990 to 2030.

Cluster	Relative area of cluster	Food and beverage – example Crop-based Production			Lifecycle maintenance and habitat protection – example Habitat diversity			Experiential – example Recreation			Biotic materials – example Wildlife products				
		Mean service loading ^a	Mean net balance of supporting minus degrading land use ^c	2000–2030	Mean service loading ^b	Mean net balance of supporting minus degrading land use ^c	2000–2030	Mean service loading ^b	Mean net balance of supporting minus degrading land use ^c	1990–2000	Mean service loading ^b	Mean net balance of supporting minus degrading land use ^c	1990–2000	2000–2030	
1	19.4	4.9	-22.9	-11.2	-79	7.3	7.6	8.7	20.4	9.3	79	6.8	18	8.3	64.3
2	19.4	4.6	-10.6	-6.4	-132.9	7.9	-0.2	9.4	8	4.2	132.9	7.4	5.5	3.1	108.5
3	26.9	6.5	-11.7	-5.9	-42.5	5.6	-3.9	6.9	8.5	3.4	42.5	4.5	6.3	2	15.9
4	9.3	4.4	-28.1	-19.9	-240.1	7.8	16.7	10	24.4	17.5	240.1	8	20.3	16.2	211.7
5	22	7.9	-10.4	-6.9	5.9	4.1	-2.1	5.2	7.5	4.3	-5.9	3	5.2	3.2	-18.5
6	1	5	-35.5	-18	-142.5	6.1	-30.7	8	8.9	3.5	142.5	2.3	-22	-8.6	-83.1
7	1.7	8.1	-12.2	-11.5	105.6	3.5	-1.2	5	11.6	10.2	-105.6	3.4	10.8	9.6	-107.8
No data	0.4	No data	No data	No data	No data	No data	No data	No data	No data	No data	No data	No data	No data	No data	No data

^a Mean of all pixels belonging to cluster (max. 16; min 0); see Table 2.

^b Mean of all pixels belonging to cluster (max. 20; min 0); see Table 2.

^c In per mille of total land area of all NUTS-x regions belonging to a cluster.

5. Discussion

The aim of this work has been to devise approaches for assessing changes in the capacity of ecosystems to deliver ecosystem services based on multi-criteria methods, that could be used as part of the fast track accounting approach being developed by the EEA. The results suggest that there is some potential in the methods described, with mapped outputs for four target services showing reasonable agreement with independent sources that describe similar types of ecosystem output. However, to ensure a successful extension and application of these methods there are several important issues that must be considered:

- (a) *Landscape dynamics and changes in service potential*: The assumption that land cover and use data are reasonable proxies (indicators) for estimating the potential of land to provide services, can be questioned, given their temporal dynamics and variability. Our analysis has showed, however, that at the coarse spatial level of NUTS-x regions (mean area 8000 km²), land use data are acceptable proxies for a mid-term assessment (10–20 yrs), if historic land-use change data are used or a sensitivity analysis with scenario-generated data is undertaken. At the moment we do not see any valid alternative that could replace the land use data.
- (b) *Linking ecosystem services and land characteristics*: A major purpose of the study was to refine and apply the methodology proposed by Kienast et al. (2009) to a set or bundle of ecosystem services identified in the CICES framework. We are, however well aware that given the rather general character of input data we are still very likely to estimate the potential of a pixel of land to provide ecosystem services (stock) rather than the realized service (flow) itself. Nevertheless, while absolute flows may not be measurable, the marginal changes in these flows might be estimated by the analysis of the types of land cover and use change (flow accounts for land). The analysis showed that at the selected spatial resolution (NUTS-x, dominant landscape types and biogeographic regions), land use change can be used as a proxy to assess past and projected degradative or supportive land use for some ecosystem services.
- (c) *Validating the continent-wide approach*: As highlighted here and in the scientific literature (Burkhard et al., 2012; Beck et al., 1997; Mayo and Spanos, 2004), the quantitative evaluation of ecosystem service estimates with independent data is difficult at the continental scale. Only a few of the independent data are continent-wide service assessments. Most are so-called indicator maps, representing only a small thematic proportion of a service. We recommend improving our evaluation by more sophisticated procedures as more independent ecosystem maps become available. This could involve techniques such as fuzzy-based Kappa (Visser and de Nijs, 2006) or budgeting spatially explicit components of shared information (Pontius et al., 2008).
- (d) *Trade-off analysis*: To our knowledge this is one of the first continental-wide analysis where land use trajectories over 40 yrs (1990–2030) are taken into account to estimate the trade-offs between the selected services. We are able to distinguish seven spatially explicit clusters with distinct evolutionary trajectories of ecosystem services. Such clusters and their dynamics could be used as the basis of an additional set of indicators for tracking the capacity of land to generate ecosystem services.

6. Conclusions

The aim of this paper has been to develop the logic for assessing the potential of large areas to deliver ecosystem services as first

reported in Kienast et al. (2009). This approach and the follow-up work have been designed to be:

- *Transparent and parsimonious*: in that the decision rules (see e.g. Gustavsson et al., 2006; Metzger et al., 2006) representing links between land characteristics and ecosystem service classes fit the available knowledge at continental scales;
- *Expert-driven*: to the extent that information from experts and from literature is implemented to supplement and extend empirical knowledge;
- *Temporally and spatially explicit*: in that the method is applicable to multiple time steps; and incorporates historical as well as projected land use change; and
- *Theoretically consistent*: in that the proposed rules are consistent with the currently accepted ecosystem goods and service concepts.

While the methods may stand alone as one way of assessing the potential of land to provide ecosystem services, they clearly have relevance in the context of land accounting and the attempt to construct a rapid or 'quick scan' assessment of the state of services at European scales. Thus they are not meant to replace more detailed process- or model-based assessments which may, for example, more explicitly take account of feedbacks. Instead, they aim to allow a rapid initial appraisal that could potentially flag up important trends as they occur, and so help decision makers prioritise intensive scrutiny using specialised scientific tools.

By explicitly linking the analysis to land characteristics and the processes of land cover and use change, the outputs from this study also represent an initial attempt to construct 'physical' accounts for services that so provide a first approximation to broad-scale ecosystem accounts. The methods can therefore be used to identify the state and trends of different spatial units and their capacity to provide ecosystem services, and thus give some insights into the state of our natural capital, at least from a terrestrial perspective. The importance of assessing the significance of incremental or marginal changes has been emphasised recently in the economic literature (Fisher et al., 2008; Bateman et al., 2010). Indeed the approach has explicitly been used as the basis of valuation of ecosystem services in the recent UK National Ecosystem Assessment, by making comparisons between scenario outcomes and current conditions. We suggest that indicators of change, such as the ones considered here, that include reference to these marginal changes may be useful in taking such work forward.

The link we have suggested to scenarios is a further area where these kinds of approach may be useful. The methods proposed here are sufficiently flexible to be applied to both historical cover and projected change. The approach can therefore form something of a bridge between the rapid or quick-scan assessments of current trends, and the kinds of exploratory, 'what-if' modelling that decision makers often use in the process of policy development and appraisal. Such work would require the more detailed analysis of the way different drivers of change are likely to impact on land cover and use.

Although our work suggests that plausible outputs can be obtained using the simple methods proposed, we recognise that the more sophisticated sets of criteria can be developed. Thus probabilistic methods could be applied in order to produce more spatially nuanced or refined mapped and tabular outputs; cross-scale effects should also be considered. More explicit analysis of the drivers of change could also be incorporated into the analysis. We see these possibilities as an important next step in the development of indicators of the capacity of land to generate ecosystem services. This study confirms, however, that indicators or proxies of the capacity of land to generate ecosystem services at broad spatial scales can be constructed in ways that can address current policy concerns.

Acknowledgements

We acknowledge the European Environment Agency for support in undertaking this analysis (Framework Contract: Support for the Development of Ecosystem Accounting, Contract Number EEA/BSS/07/007). The views expressed here are, however, our own. We also thank the anonymous reviewers for their swift and insightful comments.

References

- Bastian, O., Haase, D., Grunewald, K., 2012. Ecosystem properties, potentials and services – The EPPS conceptual framework and an urban application example. *Ecological Indicators* 21, 7–16.
- Bateman, I.J., Mace, G.M., Fezzi, C., Atkinson, G., Turner, K., 2010. Economic analysis for ecosystem service assessments. *Environmental and Resource Economics* 48 (2), 177–218.
- Bateman, I.J., 2011. Valuing changes in ecosystem services: scenario analyses. In: The UK National Ecosystem Assessment: Technical Report. UNEP-WCMC, Cambridge. Available from: <http://uknea.unep-wcmc.org/>.
- Beck, M.B., Ravetz, J.R., Mulkey, L.A., Barnwell, T.O., 1997. On the problem of model validation for predictive exposure assessments. *Stochastic Hydrology and Hydraulics* 11 (3), 229–254.
- Bindraban, P.S., Stoorvogel, J.J., Jansen, D.M., Vlaming, J., Groot, J.J.R., 2000. Land quality indicators for sustainable land management: proposed method for yield gap and soil nutrient balance. *Agriculture, Ecosystems & Environment* 81 (2), 103–112.
- Bolliger, J., Kienast, F., 2010. Landscape functions in a changing environment. *Landscape Online* 21, 1–5.
- Boyd, J., Banzhaf, S., 2007. What are ecosystem services? The need for standardized environmental accounting units. *Ecological Economics* 63 (2–3), 616–626.
- Burkhard, B., Kroll, F., Müller, F., Windhorst, W., 2009. Landscapes' capacities to provide ecosystem services – a concept for land-cover based assessments. *Landscape Online* 15, 1–22.
- Burkhard, B., Kroll, F., Nedkov, S., Müller, F., 2012. Mapping ecosystem service supply, demand, and budgets. *Ecological Indicators* 21, 17–29.
- Busch, M., La Notte, A., Laporte, V., Erhard, M., 2012. Potentials of quantitative and qualitative approaches to assessing ecosystem services. *Ecological Indicators* 21, 89–103.
- Carpenter, S.R., Bennett, E.M., Peterson, D.G., 2006. Scenarios for ecosystem services: an overview. *Ecology and Society* 11 (1), 29. Available from: <http://www.ecologyandsociety.org/vol11/iss1/art29/>.
- Chan, K.M.A., Shaw, M.R., Cameron, D.R., Underwood, E.C., Daily, G.C., 2006. Conservation planning for ecosystem services. *PLoS Biology* 4, 2138–2152.
- Comber, A.J., 2008. Land use or land cover? *Journal of Land Use Science* 3 (4), 199–202.
- Cowling, R.M., Ego, B., Knight, A.T., O'Farrell, P.J., Reyers, B., Rouget, M., Roux, D.J., Welz, A., Wilhelm-Rechman, A., 2008. An operational model for mainstreaming ecosystem services for implementation. *Proceedings of the National Academy of Sciences of the United States of America* 105 (28), 9483–9488.
- Daily, G.C., Matson, P.A., 2008. Ecosystem services: from theory to implementation. *Proceedings of the National Academy of Sciences of the United States of America* 105 (28), 9455–9456.
- De Groot, R.S., 2006. Function-analysis and valuation as a tool to assess land use conflicts in planning for sustainable, multi-functional landscapes. *Landscape and Urban Planning* 75, 175–186.
- De Groot, R.S., 2010. Integrating the ecological and economic dimensions in biodiversity and ecosystem service valuation. In: Kumar, P. (Ed.), *The Economics of Ecosystems and Biodiversity: Ecological and Economic Foundations*. Earthscan.
- Ego, B., Reyers, B., Rouget, M., Richardson, D.M., Le Maitre, D.C., van Jaarsveld, A.S., 2008. Mapping ecosystem services for planning and management. *Agriculture, Ecosystems & Environment* 127, 135–140.
- European Commission, 2011. Our life insurance, our natural capital: an EU biodiversity strategy to 2020. Communication from the Commission to the European Parliament, the Council, the Economic and Social Committee and the Committee of the Regions. Document COM(2011) 244 final, issued May 3, 2011, p. 16. Available from: <http://ec.europa.eu/environment/nature/biodiversity/comm2006/pdf/2020/1.EN.ACT.part1.v7%5b1%5d.pdf>.
- European Environmental Agency (EEA), 1998. State of the environment report No. 2. A Report on the Changes in the pan-European Environment as a Follow-up to 'Europe's Environment: The Dobbris Assessment' (1995) Requested by the Environment Ministers for the Whole of Europe to Prepare for the Fourth Ministerial Conference in Aarhus, Denmark, June 1998.
- European Environmental Agency (EEA), 1999. Environmental assessment report No. 2. Report on the State of the European Environment, Copenhagen, p. 446.
- European Environmental Agency (EEA), 2002. Corine Land Cover Update 2000: Technical Guidelines. EEA, Copenhagen.
- European Environmental Agency (EEA), 2006. Land accounts for Europe 1990–2000. Towards integrated land and ecosystem accounting. EEA Report 11/2006, Copenhagen, p. 107.
- Fisher, B., Turner, R.K., Zylstra, M., Brouwer, R., de Groot, R., Farber, S., Ferraro, P., Green, R., Hadley, D., Harlow, J., Jefferiss, P., Kirkby, C., Morling, P., Mowatt, S., Naidoo, R., Paavola, J., Strassburg, B., Yu, D., Balmford, A., 2008. Ecosystem ser-

- vices and economic theory: integration for policy-relevant research. *Ecological Applications* 18 (8), 2050–2067.
- Fisher, B., Turner, R.K., Morling, P., 2009. Defining and classifying ecosystem services for decision making. *Ecological Economics* 68 (3), 643–653.
- Fohrer, N., Haverkamp, S., Frede, H.G., 2005. Assessment of the effects of land use patterns on hydrologic landscape functions: development of sustainable land use concepts for low mountain range areas. *Hydrological Processes* 19 (3), 659–672.
- Gustavsson, M., Kolstrup, E., Seijmonsbergen, A.C., 2006. A new symbol-and-GIS based detailed geomorphological mapping system: renewal of a scientific discipline for understanding landscape development. *Geomorphology* 77 (1–2), 90–111.
- Haase, D., Walz, U., Neubert, M., Rosenberg, M., 2007. Changes to Central European Landscapes – analysing historical maps to approach current environmental issues, examples from Saxony, Central Germany. *Land Use Policy* 24, 248–263.
- Hagen-Zanker, A., 2006. Map comparison methods that simultaneously address overlap and structure. *Journal of Geographical Systems* 8 (2), 165–185.
- Hazeu, G., Mächer, S., Kienast, F., Zimmermann, N., 2007. Land cover maps for environmental modelling at multiple scales. Project Report EU Project ECOCHANGE FP6 2006 GOCE 036866.
- Haines-Young, R., 2009. Land use and biodiversity relationships. *Land Use Policy* 26 (1), S178–S186.
- Haines-Young, R., Potschin, M., 2010. The links between biodiversity, ecosystem services and human well-being. In: Raffaelli, D., Frid, C. (Eds.), *Ecosystem Ecology: A New Synthesis*. BES Ecological Reviews Series. Cambridge University Press, Cambridge.
- Haines-Young, R., Patterson, J., Potschin, M., 2011. Scenarios: development of storylines and analysis of outcomes. In: *The UK National Ecosystem Assessment: Technical Report*. UNEP-WCMC, Cambridge. Available from: <http://uknea.unep-wcmc.org/>.
- Jansen, L.J.M., Di Gregorio, A., 2002. Parametric land cover and land-use classification as tools for environmental change detection. *Agriculture, Ecosystems & Environment* 91, 89–100.
- Jax, K., 2005. Function and functioning in ecology: what does it mean? *Oikos* 111 (3), 641–648.
- Kienast, F., Bolliger, J., Potschin, M., de Groot, R.S., Verburg, P.H., Heller, I., Wascher, D., Haines-Young, R.H., 2009. Assessing landscape functions with broad-scale environmental data: insights gained from a prototype development for Europe. *Environmental Management* 44, 1099–1120.
- Krönert, R., Steinhart, U., Volk, M., 2001. *Landscape Balance and Landscape Assessment*. Springer, Heidelberg, New York, 304 pp.
- Landis, J.R., Koch, G.G., 1977. The measurement of observer agreement for categorical data. *Biometrics* 33, 159–174.
- Leibowitz, S.G., Loehle, C., Li, B.L., Preston, E.M., 2000. Modeling landscape functions and effects: a network approach. *Ecological Modelling* 132, 77–94.
- Lesta, M., Mauring, T., Mander, U., 2007. Estimation of landscape potential for construction of surface-flow wetlands for wastewater treatment in Estonia. *Environmental Management* 40, 303–313.
- Maes, J., Paracchini, M.L., Zulian, G., 2011. A European Assessment of the Provision of Ecosystem Services: Towards an Atlas of Ecosystem Services. Publications Office of the European Union, Luxembourg, doi:10.2788/63557, p. 81.
- Mayo, D.G., Spanos, A., 2004. Methodology in practice: statistical misspecification testing. *Philosophy of Science* 71 (5), 1007–1025.
- Meijl, H.V., van Rheenen, T., Tabeau, A., Eickhout, B., 2006. The impact of different policy environments on agricultural land use in Europe. *Agriculture, Ecosystems & Environment* 114, 21–38.
- Metzger, M., Schröter, D., Leemans, R., Cramer, W., 2008. A spatially explicit and quantitative vulnerability assessment of ecosystem service change in Europe. *Regional Environmental Change* 8, 91–107.
- Metzger, M.J., Rounsevell, M.D.A., Acosta-Michlik, L., Leemans, R., Schroter, D., 2006. The vulnerability of ecosystem services to land use change. *Agriculture, Ecosystems & Environment* 114, 69–85.
- Millennium Ecosystem Assessment, 2005. *Ecosystems and Human Well-Being: Synthesis*. Island Press, Washington, DC.
- Naidoo, R., Balmford, A., Costanza, R., Fisher, B., Green, R.E., Lehner, B., Malcolm, T.R., Ricketts, T.H., 2008. Global mapping of ecosystem services and conservation priorities. *Proceedings of the National Academy of Sciences of the United States of America* 105 (28), 9495–9500.
- Nedkov, S., Burkhard, B., 2012. Flood regulating ecosystem services – Mapping supply and demand, in the Etropole municipality, Bulgaria. *Ecological Indicators* 21, 67–79.
- Nelson, E., Mendoza, G., Regetz, J., Polasky, S., Tallis, H., Cameron, D.R., Chan, K.M.A., Daily, G.C., Goldstein, J., Kareiva, P.M., Lonsdorf, E., Naidoo, R., Ricketts, T.H., Shaw, R.M., 2009. Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. *Frontiers in Ecology and Environment* 7 (1), 4–11.
- Peterseil, J., Wrбка, T., Plutzer, C., Schmitzberger, I., Kiss, A., Szerencsits, E., Reiter, K., Schneider, W., Suppan, F., Beissmann, H., 2004. Evaluating the ecological sustainability of Austrian agricultural landscapes – the SINUS approach. *Land Use Policy* 21 (3), 307–320.
- Pontius, R.G., Thontteh, O., Chen, H., 2008. Components of information for multiple resolution comparison between maps that share a real variable. *Environmental and Ecological Statistics* 15 (2), 111–142.
- Potschin, M., Haines-Young, R.H., 2011. Ecosystem services: exploring a geographical perspective. *Progress in Physical Geography* 35 (5), 575–594.
- Tallis, H., Kareiva, P., Marvier, M., Chang, A., 2008. An ecosystem services framework to support both practical conservation and economic development. *Proceedings of the National Academy of Sciences of the United States of America* 105, 9457–9464.
- Seppelt, R., Fath, B., Burkhard, B., Fisher, J.L., Grêt-Regamey, A., Lautenbach, S., Pert, P., Hotes, S., Spangenberg, J., Verburg, P.H., van Oudenhoven, A.P.E., 2012. Form follows function? Proposing a blueprint for ecosystem service assessments based on reviews and case studies. *Ecological Indicators* 21, 145–154.
- Tallis, H., Polasky, D., 2009. Mapping and valuing ecosystem services as an approach for conservation and natural-resource management. *The year in ecology and conservation biology, 2009*. *Annals of the New York Academy of Science* 1162, 265–283.
- Tallis, H., Polasky, S., 2011. Assessing multiple ecosystem services: an integrated tool for the real world. In: Karvera, P., Tallis, H., Ricketts, T.H., Daily, G.C., Polasky, S. (Eds.), *Natural Capital. Theory and Practice of Mapping Ecosystem Services*. Oxford University Press, Oxford, pp. 34–52.
- ten Brink, P., 2011. *The Economics of Ecosystems and Biodiversity in National and International Policy Making*. Earthscan.
- United Nations, 2003. *Handbook of National Accounting: Integrated Environmental-Economic Accounting 2003*. United Nations, European Commission, International Monetary Fund, Organisation for Economic Co-operation and Development, World Bank. Series F, No. 61, Rev. 1 (ST/ESA/STAT/SER.F/61/Rev.1).
- Verburg, P.H., Veldkamp, A., Rounsevell, M.D.A., 2006. Scenario-based studies of future land use in Europe. *Agriculture, Ecosystems & Environment* 114 (1), 1–6.
- Verburg, P.H., Eickhout, B., van Meijl, H.A., 2008. A multi-scale, multi-model approach for analyzing the future dynamics of European land use. *Annals of Regional Science* 42, 57–77.
- Verburg, P.H., van Berkel, D.B., van Doorn, A.M., van Eupen, M., van den Heiligenberg, H.A.R.M., 2009. Trajectories of land use change in Europe: a model-based exploration of rural futures. *Landscape Ecology* 25 (2), 217–232.
- Visser, H., de Nijs, T., 2006. The map comparison kit. *Environmental Modelling and Software* 21 (3), 346–358.
- Westhoek, H.J., van den Berg, M., Bakkes, J.A., 2006. Scenario development to explore the future of Europe's rural areas. *Agriculture, Ecosystems & Environment* 114 (1), 7–20.
- Willemsen, L., Verburg, P.H., Hein, L., van Mensvoort, M.E.F., 2008. Spatial characterization of landscape functions. *Landscape and Urban Planning* 88 (1), 34–43.
- Wrбка, T., Erb, K.-H., Schulz, N.B., Peterseil, J., Hahn, C., Haberl, H., 2004. Linking pattern and process in cultural landscapes. An empirical study based on spatially explicit indicators. *Land Use Policy* 21, 289–306.
- Wu, J., Jenerette, G.D., David, J.L., 2003. Linking land-use change with ecosystem processes: a hierarchical patch dynamic model. In: Guhathakurta, S. (Ed.), *Integrated Land Use and Environmental Models*. Springer, Berlin, pp. 99–119.
- Zhao, M., Heinsch, F.A., Nemani, R.R., Running, S.W., 2005. Improvements of the MODIS terrestrial gross and net primary production global data set. *Remote Sensing of Environment* 95, 164–176.