Ecosystem Services 4 (2013) 82–94

Contents lists available at SciVerse ScienceDirect

Ecosystem Services

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

Trade-off analysis of ecosystem services in Eastern Europe

A. Ruijs [a,*], A. Wossink [b], M. Kortelainen [c], R. Alkemade [a], C.J.E. Schulp [d]

[a] PBL—Netherlands Environmental Assessment Agency, PO Box 303, 3720 AH, Bilthoven, The Netherlands
[b] Economics, School of Social Sciences, University of Manchester, Manchester M13 9PL, UK
[c] VATT, PO Box 1279, FI-00101, Helsinki, Finland
[d] VU-IVM Institute for Environmental Studies, De Boelelaan 1087, 1081 HV, Amsterdam, The Netherlands

ARTICLE INFO

Article history:
Received 20 May 2012
Received in revised form
21 January 2013
Accepted 1 April 2013
Available online 14 May 2013

Keywords:
Production possibility frontiers
Biodiversity
Carbon sequestration
Opportunity costs
Non-parametric estimation

ABSTRACT

In this paper we assess trade-offs between ecosystem services in a spatially explicit manner. From a supply side perspective, we estimate opportunity costs, which reflect in monetary terms the trade-offs between ecosystem services due to a marginal land use change. These are based on estimation of the frontier function, which gives the feasible bundles of ecosystem services that can be generated. For this, a two-stage semi-parametric method is applied and spatial data are used on agricultural revenues, cultural services, carbon sequestration and biodiversity for 18 Central and Eastern European countries. Based on the estimates, we assess which regions are most suitable for expanding any of the ecosystem services. Where opportunity costs are low, a further expansion of any of the ecosystem services is cost-effective. If areas are targeted carefully, joint improvement of several ecosystem services can be reached. If carbon sequestration levels are to be increased, it is best to focus on areas already having high sequestration levels because opportunity costs of carbon sequestration decrease with increasing sequestration levels. For biodiversity and cultural services the pattern is less clear as low opportunity cost were found both in areas rich and poor with these services.

1. Introduction

Since the Millennium Ecosystem Assessment (2005), ecosystem services research has gained momentum. Step by step, it is becoming better understood how ecosystem functions and services are interrelated and which factors affect the provision of ecosystem services (see e.g. Daily et al., 2009; Haines-Young and Potschin, 2010; Isbell et al., 2011; UK National Ecosystem Assessment, 2011). Gaining insight into where particular services are weak or strong is important for making land use decisions (Daily et al., 2009). At different levels of decision making. maps can be generated, quickly and transparently showing the bundles of ecosystem services that can jointly be supplied (e.g Daily and Matson, 2008; Naidoo et al., 2008; Haines-Young, 2009; Maes et al., 2012; Martínez-Harms and Balvanera, 2012; Schulp et al., 2012).

Similarly, awareness of the importance of maintaining ecosystem services for human welfare has increased (see e.g. US Environmental Protection Agency, 2009; Haines-Young and Potschin, 2010; TEEB, 2010; Bateman et al., 2011). The economics of the natural environment is receiving increased attention owing to numerous initiatives including the TEEB studies (TEEB, 2010), assessments in the UK (UK National Ecosystem Assessment, 2011) and USA (National Research Council, 2005; US Environmental Protection Agency, 2009) and attempts by the WAVES Partnership and the UN SEEA to integrate the value of ecosystem services into national accounts and economic growth plans.[1] As a result of these initiatives, the notion that natural resources have economic value is increasingly finding its way into policy analyses and government decision processes.[2]

However, at national, regional or global scales, much remains still unknown about the trade-offs between ecosystem services resulting from land use changes. Trade-offs are location specific. Thus it is pertinent to understand where changes in land use can improve total food production, biodiversity levels, climate change mitigation, etc. in the most cost-effective way. To answer such questions, trade-offs need to be known in monetary terms.

* Corresponding author. Tel.: +31 30 2742167; fax: +31 30 2744479.
E-mail addresses: Arjan.Ruijs@pbl.nl (A. Ruijs), Mika.Kortelainen@vatt.fi (M. Kortelainen), Rob.Alkemade@pbl.nl (R. Alkemade), Nynke.Schulp@vu.nl (C.J.E. Schulp), Ada.Wossink@manchester.ac.uk (A. Wossink).

2212-0416/$ - see front matter © 2013 Elsevier B.V. All rights reserved.
http://dx.doi.org/10.1016/j.ecoser.2013.04.002

[1] See www.wavespartnership.org and unstats.un.org/unsd/envaccounting.

[2] An example is the White Paper published by the government of the UK in 2012 entitled 'The Natural Choice: securing the value of nature', in which it is stressed that the economic and social benefits of the environment should be properly valued. Moreover, the EU Biodiversity Strategy specifically calls on the member states to assess the status of ecosystem services in their territory and to assess their economic value. Furthermore, the WAVES Global Partnership has brought together a broad coalition of organizations to mainstream ecosystem services valuation in national accounts and development planning and the UN SEEA proposes to add ecosystem services accounts to their system of environmental economic accounts.

The objective of this paper is to present a method to estimate the trade-offs between the different ecosystem services due to a land use change. The tradeoffs are to be spatially explicit and in monetary terms. With these results, the method aims to answer two questions of practical policy relevance: first, which regions should decision makers target to achieve national and international biodiversity objectives in a cost-effective way? and second, is it better to jointly generate ecosystem services in a region or to specialize in one of them?

The novelty of this paper is derived from the way in which the reported approach combines a supply side perspective for ecosystem services with a non-parametric method to estimate transformation functions and opportunity costs and a unique data set. Most directly related to our work are recent studies at the micro level (e.g. Macpherson et al., 2010; Bostian and Herlihy, 2012; Sauer and Wossink, 2013). We are not aware of studies that use a non-parametric methods to estimate transformation functions for the analysis of ecosystem services and apply this to analyse supply at sub-national to global levels.

By following a supply side approach, we assess the expected change in ecosystem services supply due to a land use change. In contrast to most existing supply side analyses, that quantify trade-offs of land use changes in biophysical terms (see e.g. Millennium Ecosystem Assessment, 2005; Maes et al., 2012), we quantify these trade-offs in monetary terms. Studies that evaluate the monetary value of ecosystem services, commonly follow a demand side approach (see TEEB, 2010 for a review of the literature) and in that way evaluate how people appraise the changes. Ideally, a combined supply–demand side approach should be adopted in which it is shown how supply changes due to a land use change and how people value these changes. This makes it possible to evaluate whether the changes are welfare improving. However, demand side valuation analyses are less reliable for studies at higher spatial scales.[3] For that reason, we refrain from demand side valuation techniques and approach cost-effectiveness of land use changes from a supply side. We assess trade-offs between ecosystem services that are jointly produced in a given area in monetary terms, i.e. we estimate opportunity costs of land use changes, with which it can be evaluated whether land use changes are cost-effective (see Diaz-Balteiro and Romero, 2008). Such analyses at higher spatial scales are rare.

We derive opportunity costs from transformation functions, which summarize the feasible bundles of ecosystem services generated in a region (see e.g. Smith et al., 2012). The estimated transformation functions are then used to show the effects of the land use choices available to authorities—where to develop agriculture, where to preserve biodiversity, where to keep a multifunctional landscape. Trade-offs are contingent on the curvature of the frontier function at each point. The transformation functions are estimated empirically using a two-stage, semi-parametric, distance function approach. Hof et al. (2004), Bellenger and Herlihy (2010) and Macpherson et al. (2010) also adopt non-parametric or semi-parametric estimation techniques (though different from the approach adopted by us) to select the areas that jointly produce multiple environmental outputs in the most efficient way. However, whereas these existing studies focus on efficiency, we extend the approach by explicitly considering the opportunity costs of land use changes as a basis for selecting the areas most appropriate for particular land uses. This extra dimension results in trade-off information in monetary terms which is essential but often missing in supply side analyses.

For our application we use spatial data on agricultural revenues, cultural services, carbon sequestration and biodiversity for 18 Central and Eastern European countries on the level of grid cells of a size of 0.5×0.5 degree. The data originate from the integrated assessment model IMAGE (Bouwman et al., 2006), biodiversity model GLOBIO (Alkemade et al., 2009) and additional ecosystem services models (Schulp et al., 2012).[4] These models give the state-of-the-art knowledge of the interrelations between land use, agricultural production and ecosystem functioning and results are used extensively in e.g. OECD Environmental Outlook (OECD, 2012), UNEP GEO4 (UNEP, 2012) and several other global assessments of environmental change (see e.g. Van Vuuren and Faber, 2009; Brink et al., 2010; PBL, 2012). These models, however, do not directly yield information on trade-offs or effects of a marginal land use change. Yet, their results can be used in the semi-parametric method as set up in this paper, to recover from the data the transformation function and derive the opportunity costs of a marginal land use change. Using model data is the only feasible option for our analysis because of the lack of reliable observations of the relevant variables at higher spatial scales.

Our work differs from other recent studies at the aggegated level, for example those that use bio-economic models. Examples are the InVEST model (see e.g. Daily and Matson, 2008; Polasky et al., 2008; Nelson et al., 2009; Keeler et al., 2012), the bio-economic models used to derive cost-effective ecological restoration of the Murray Darling basin in south-east Australia (Crossman and Bryan, 2009; Bryan, 2010; Bryan et al., 2011) and bio-economic models by e.g. Hauer et al. (2010) and Barraquand and Martinet (2011). Other examples of spatially explicit trade-off analyses for ecosystem services use GIS or heuristic routines to combine ecological and economic concepts (Bateman, 2009; Bateman et al., 2011; White et al., 2012) or do not estimate trade-offs in monetary terms (Naidoo et al., 2008; Raudsepp-Hearne et al., 2010; Maes et al., 2012). Most of these analyses, however, are less suitable for doing trade-off analysis at sub-national to global scales. Hussain et al. (2011) also base their analysis on IMAGE and GLOBIO data (Brink et al., 2010). They however employ benefit transfer methods to evaluate values of changes in ecosystem services provision. Such methods remain controversial especially when applied at high spatial scales.

The remainder of this paper is set up as follows. In Section 2 we briefly discuss the economic model. The data used are discussed in Section 3. In Section 4, we present estimation results. Finally, Section 5 ends with a discussion and conclusions.

2. Theoretical and empirical model

This paper focuses on the trade-offs between ecosystem services due to land use changes. We consider a situation in which a social planner makes land use choices—where to grow crops, where to preserve biodiversity, where to keep a multifunctional landscape. These choices have a multitude of effects on the

[3] Revealed or stated preference studies yield informative results about people's preferences especially for analyses at low spatial scales, such as local or regional public project assessments. For analyses at higher spatial scales, however, demand side valuation approaches are less reliable. See e.g. the discussion of Costanza et al. (1997) and the reply in Costanza et al. (1998). Moreover, the use of benefit transfer methods which is used more and more for analyses at higher spatial scales (see e.g. Hussain et al., 2011) often gives unacceptably large transfer errors (Brouwer et al., 2012), despite of recent methodological improvements (Ghermandi et al., 2010; Brander and Koetse, 2011).

[4] IMAGE (Integrated Model to Assess the Global Environment) simulates the environmental consequences of human activities worldwide. It represents interactions between society, the biosphere and the climate system to explore the long-term dynamics of global change as the result of interacting demographic, technological, economic, social, cultural and political factors. GLOBIO (Global Biodiversity model) is used in the assessment of policy options for reducing global biodiversity loss and is based on the GLC2000 land use map.

ecosystem services provided at different locations. To derive the trade-offs we consider transformation functions of biodiversity and ecosystem services. A transformation function represents the output producible from a given input base and existing conditions, which also represents the feasible production set. Thus these functions show production relationships, interactions between ecosystem services and effects of spatial differences in biotic and abiotic characteristics. In this section, we first discuss the theory behind transformation curves and how they are related to opportunity costs. Second, it is discussed how these curves and opportunity costs can be estimated empirically.

To show how the transformation functions are derived, introduce two vectors of ecosystem services, y_m a vector of marketed services and y_n a vector of non-marketed services. These vectors together cover the bundle of ecosystem services that are generated in a given location. The services distinguished provide direct human benefits or serve as a proxy for longer term benefits. Vector y_m includes provisioning services and marketed cultural services (e.g. tourism). Vector y_n represents non-marketed cultural services and regulating and supporting services which maintain benefits in the longer term. This includes carbon sequestration and biodiversity. Several of the non-marketed services y_n are common pool resources. They are non-excludable and offer rival benefits. Non-excludability means that there is access to their use at zero marginal costs for the user. As a result, price signals do not reveal the true value and supply and demand may not be welfare optimizing (Romstad, 2008). The way land use choices affect marketed and non-marketed outputs is dependent upon a number of factors exogenous to the decision makers, like geographical location, soil type and regional income (which depends on population density and economic structure). These are covered by the vector of conditional variables z.

The way in which the marketed and non-marketed ecosystem services $y=(y_m,y_n)$ are jointly produced in a specific location with a given environment z can be described using the transformation function $F(y|z)$—see Fig. 1 for a simplified example with y_m and y_n as scalars. In each location, the bundle of services (y_m, y_n) can be produced in variable proportions depending on land use and input choices related to which crops to grow, which acreages to conserve, which areas to develop, etc. The transformation function, also called production possibility frontier, gives the combinations of y_m and y_n that can maximally be produced in a given location. The slope of the transformation function at a certain point reflects the change of y_n due to a small change in y_m. This is called the opportunity cost of y_n, or the trade-off between y_n and y_m. Transformation functions may differ by location and therefore also trade-offs between y_m and y_n differ spatially.

Whereas more information becomes available on how different factors affect the generation of individual ecosystem services, there still is little empirical information on transformation functions

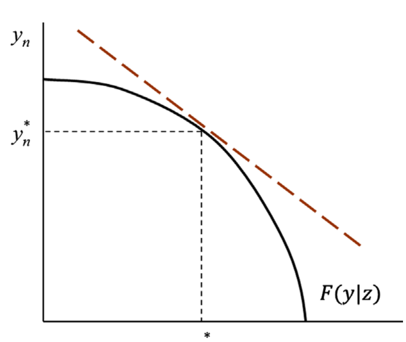

Fig. 1. Representation of a transformation function for a situation with two outputs.

showing which bundles can efficiently be generated in a certain geographical location. By comparing for a large number of locations spatially explicit information on the bundles of ecosystem services generated and by properly reckoning for differences in feasible outputs because of different environmental, social and historical characteristics, the transformation function can be uncovered from the data. Depending on the position of a certain location at the transformation function (is a large share of the land used for agriculture with low biodiversity levels or is most land natural habitat with low agricultural production?), the trade-offs between the ecosystem services generated at that location can be derived. To estimate this frontier of feasible ecosystem services bundles, we adopt the two-stage semi-parametric estimation approach as proposed by Florens and Simar (2005) and Daraio and Simar (2007a). The approach is explained in detail in Ruijs et al. (2012).

In the first stage, we non-parametrically estimate the efficient frontier and the distance of each observation to the frontier using the output oriented, robust conditional Free Disposal Hull (FDH) method (see also Cazals et al., 2002; Daraio and Simar, 2005, 2007b; De Witte and Kortelainen, 2009; De Witte and Marques, 2010 and the appendix to this paper). The Free Disposal Hull is a nonparametric frontier estimator proposed by Deprins et al. (1984). FDH is a more flexible (or less restrictive) frontier model than the more well-known Data Envelopment Analysis (DEA), because in contrast to DEA it does not require the production possibility set to be convex, but only freely disposable. The output oriented FDH method evaluates for which observations no other observation exists that has equal or higher output levels for all elements of the output vector. The frontier shaped by these observations represents the Pareto-optimal bundles of ecosystem services levels that can be generated in a certain location. For the other observed bundles of ecosystem services the distance to the frontier is measured, which represents the efficiency improvement the region could theoratically reach. The advantage of the robust, conditional FDH method is that, different from parametric and DEA methods, it requires no prior assumptions about the convexity of the production possibility set. As ecosystem services provision is likely characterized by non-convexities (Chavas, 2009; Brown et al., 2011) *a priori* convexity assumptions may lead to misleading policy recommendations. In addition, in comparison to traditional FDH, the robust (or order-m) FDH approach is much less sensitive to noise and outliers, since it allows some observations to be outside of the frontier. Moreover, the conditional FDH approach assures that only observations having similar characteristics are compared with each other (see Daraio and Simar, 2005, 2007b). So, rich and poor regions or areas with arid and those with humid climates are not compared when estimating efficient points (or observations).

In the second stage, we parametrically approximate the non-parametric frontier obtained in the first stage with a flexible translog frontier function. In that way, a smooth curve is estimated approximating the stairway-shaped frontier derived in the first stage and for each location unique opportunity costs can be determined. The advantage of first deriving the frontier non-parametrically and then parametrically approximating this frontier is that not the shape of the center of a cloud of observations is estimated, but the shape of the observations near the frontier (Florens and Simar, 2005). Moreover, the advantage of a translog functional form is that no convexity or restrictive parametrics assumptions have to be made. In the appendix we show how the frontier function can be deduced from the distance function, which gives for each observation (y,z) its distance, δ, to the frontier. Although the presentation there is quite technical, the basic idea is relatively simple: the parametric approximation of the multi-variate output distance function can be implemented by projecting the output values on the output efficient frontier using the distance values estimated in the first stage, and then estimating with the ordinary least squares (OLS) method the following function

$$\ln y_{i1}^{*} = -\left(\alpha_0 + \beta'_{-1} \ln \tilde{y}_{i,-1} + \frac{1}{2} \ln \tilde{y}'_{i,-1} \Gamma_{22} \ln \tilde{y}_{i,-1} + \gamma' \ln z_i\right), \quad (1)$$

where $y_{i1}^{*} = y_{i1}/\delta_i = y_{i1}\lambda_i$ are the values of y_{i1} projected on the output efficient frontier and $\tilde{y}_{i,-1} = y_{i,-1}/y_{i1} = y_{i,-1}^{*}/y_{i1}^{*}$, β and γ are coefficient vectors and Γ is the coefficient matrix.

According to Daraio and Simar (2007a), one of the major advantages of this approach is that no restrictive homoskedasticity or distributional assumptions have to be made on the error term ε in the distance function.

Using the estimated frontier function, trade-offs are determined for each observation by evaluating the slope of the frontier at that particular point projected on the frontier. In order to evaluate cost-effectiveness of possible scenarios of land use changes, we need to translate this marginal trade-off into opportunity costs in monetary terms. If the market price is known for one of the ecosystem services, e.g. for the first output, p_1, it can be shown for the translog frontier function that the opportunity costs, p_k, for the kth element of vector y are (see Appendix A):

$$p_k = p_1 \frac{y_1}{y_k}\left(\frac{\beta_k + \Gamma'_k \ln y}{\beta_1 + \Gamma'_1 \ln y}\right) \quad (2)$$

with Γ_k the kth row of matrix Γ and β_k the kth element of vector β. If y_1 represents agricultural production and $y_1 p_1$ agricultural revenues, the opportunity costs reflect the foregone agricultural revenues due to a marginal increase in one of the ecosystem services.

3. Data

We estimate transformation functions and opportunity costs for a case study of 18 Central and Eastern European countries—see Fig. 2. For the non-parametric analysis, we have to make a clear distinction between input and output variables. The subdivision of ecosystem services into supporting, regulating, cultural and provisioning services, even though a helpful categorization for ecological analyses, is inappropriate for economic analyses (Wallace, 2007; Fisher et al., 2009). For our purpose, the subdivision in final or output services (especially provisioning and cultural services) and intermediary or input services (especially the supporting services and some regulating services) is more useful. This corresponds closely with the ecosystem services definitions given by Boyd and Banzhaf (2007), Haines-Young and Potschin (2010) and de Groot et al. (2010).

Fig. 2. Map of the 18 Central and Eastern European countries and four sub-regions considered.

Data are obtained for the year 2000 from the GLC2000 land cover map, the IMAGE integrated assessment model (Bouwman et al., 2006), biodiversity model GLOBIO (Alkemade et al., 2009), new estimates of ecosystem services from Schulp et al. (2012) (see also EC-JRC, 2003) and data from the World Bank World Development Indicators and FAOstat. In total, data are generated for 1166 grid cells of size 50×50 km^2. As explained in the introduction, we employ model data because field observations are unavailable for the ecosystem services and biodiversity variables at higher spatial scales. No data set on ecosystem services of comparable quality exist yet for analyses at high spatial scales covering multiple countries.

The following output variables are included:

1. Agricultural revenues (provisioning services): for each cell total agricultural revenues (in 2000 international \$/km^2) are calculated based on land use data from the GLC2000 map, the cropping pattern from IMAGE, yield data and prices per crop per country for the year 2000 from FAOSTAT.[5]
2. Cultural services: a composite index is set up consisting of attractiveness for tourism and recreation and attractiveness for hunting and gathering activities. Tourist and recreation attractiveness depend on percentage protected area, percentage urban and arable land, distance to coast and geographic relief. Attractiveness for hunting and gathering activities depends on the regional potential for gathering wild foods, fruits, and mushrooms, catching fish and hunting game and is based on statistics from FAO and the European Forestry Institute.
3. Biodiversity: mean species abundance (MSA) as determined by GLOBIO (Alkemade et al., 2009) is included as proxy for the positive impact of several regulating and supporting services on ecosystems and as an indicator which is important in nature policies. MSA is an index indicating the level of disturbance compared to the maximally possible level for the particular habitat. Note that MSA does not completely cover the complex biodiversity concept; e.g. it does not properly describe changes in threatened or red list species. Other indicators are not available yet at the required level of detail, however.
4. Carbon sequestration: net biome productivity in tons C per km^2 is included as a proxy for climate regulation. Net biome productivity equals long term averages of net primary production of carbon minus soil respiration minus the carbon stored in the biomass harvested. Estimates are based on the GLC2000 land cover map and the EURURALIS carbon model (see e.g. Schulp et al., 2008).

For the current analysis, we include land as the only input variable. Including more inputs is left for future research. The regulating service 'climate regulation' is treated as an output variable. Better functioning climate regulation, i.e. more carbon sequestration, is assumed to reflect long term stability of output potentials. Finally, we included conditional variables to assure a peer-to-peer comparison between cells when deriving the production possibility frontier. The variables included affect the position of the frontier. We include.

[5] A better measure would be to use agricultural returns, which reflects gross revenues minus costs. It is, however, difficult to obtain reliable information about production costs, which is why many studies use agricultural gross revenues. This also affects the interpretation of the opportunity costs. Opportunity costs reflect agricultural gross revenues foregone and not agricultural income foregone, which would better show the economic consequences of land use changes. In Section 4, the social profit rate is used to transform gross revenues into net revenues such that they can be compared with results from other studies.

Table 1
Averages and standard deviations of the variables included for the different sub-regions (standard deviations are given in brackets).

	Agricult. revenues US$/km^2	MSA	Cultural services	Carbon sequest. T C/km^2	GDP US$/km^2	Pot. Yield t/ha	% agric.+grass- land
Total							
Mean	15,674	0.36	0.41	29.15	491,823	481	0.59
St. Dev.	(11,394)	(0.13)	(0.10)	(29.14)	(774,723)	(109)	(0.21)
CIS							
Mean	14,419	0.36	0.40	31.96	209,673	516	0.66
St. Dev.	(11,089)	(0.13)	(0.09)	(26.10)	(372,633)	(108)	(0.22)
CE							
Mean	14,982	0.35	0.46	20.21	1082,414	431	0.56
St. Dev.	(9,090)	(0.11)	(0.09)	(17.15)	(1132,678)	(78)	(0.16)
YUG							
Mean	14,677	0.37	0.36	16.43	629,314	384	0.50
St. Dev.	(11,249)	(0.12)	(0.14)	(14.70)	(669,721)	(114)	(0.17)
SE							
Mean	21,393	0.37	0.39	43.16	352,835	516	0.51
St. Dev.	(13,826)	(0.15)	(0.09)	(47.37)	(503,458)	(80)	(0.22)

Table 2
Correlation between the variables included.

	1. Agricultural revenues	**2. Mean species abundance**	**3. Cultural services**	**4. Carbon sequestration**	**5. GDP**	**6. Potential yield**	**7. %agric. +grass land**
1. AR	1	−0.55	−0.39	−0.37	0.02	0.49	0.55
2. MSA	−0.55	1	0.50	0.55	−0.12	−0.33	−0.78
3. CS	−0.39	0.50	1	0.44	0.16	−0.21	−0.60
4. CAR	−0.37	0.55	0.44	1	−0.15	−0.12	−0.62
5. GDP	0.02	−0.12	0.16	−0.15	1	−0.12	−0.11
6. YLD	0.49	−0.33	−0.21	−0.12	−0.12	1	0.47
7. Agri	0.55	−0.78	−0.60	−0.62	−0.11	0.47	1

5. GDP PPP per km^2 for the year 2000 (in international \$/km^2) based on World Bank data on GDP per country which is allocated over the grid cells by considering differences between agricultural and non-agricultural income and rural and urban population in order to properly distinguish between rural and urban cells.
6. Share of arable and grassland: share of each cell used for production of agricultural crops and for grazing. In the analysis, a distinction is made between arable land, grassland, forests, shrub and herbaceous land and artificial surface.
7. Potential yield: potential yield of the main crop in Central and Eastern Europe (temperate cereals) in t/km^2 is included as conditional variable to assure that only regions with similar bio-physiological characteristics are compared with each other. Potential yields depend on climate, soil and slope characteristics.
8. Sub-region typology: categorical variable reflecting differences in historical, political and social development patterns which may affect the technical possibilities available to the regions. Four sub-regions are considered: (1) member countries to the Commonwealth of Independent States (CIS), (2) Central European countries (CE), (3) the former Yugoslavian republics (YUG), and (4) the south-eastern European countries (SE) (see, Fenger, 2007).

4. Results

4.1. Descriptive statistics

The descriptive statistics given in Table 1 show that the levels of the different variables are highly variable within countries and between sub-regions. This reflects differences in population density and differences between average country development levels. Signs of correlation coefficients shown in Table 2 are as expected and are related to land cover and land use. Agricultural production is higher in cells with higher percentages of agricultural land. Generally, more forested areas have higher MSA levels, have higher levels of carbon sequestration and are more attractive for cultural services like recreation and hunting. The pattern of correlations between the ecosystem services is consistent with similar observations by Raudsepp-Hearne et al. (2010) for Canada. These correlations indicate that the joint generation of ecosystem services is non-separable. It is noted that the observed level of correlation coefficients does not affect estimates of the opportunity costs.

4.2. Production possibility frontier

Before discussing the opportunity costs, we first discuss some elements of the frontier function to give insight in the shape of the production possibility frontier. Parameter estimates of function $\ln\delta = \alpha_0 + \beta' \ln y + \frac{1}{2} \ln y'\Gamma \ln y + \gamma \ln z$ are given in Table 3. A first result is that these coefficients are such that the derivative of the frontier function, $\partial\delta/\partial y_i$, is positive for all observations (or that $\partial\lambda/\partial y_i$ is negative for all observation).[6] This implies that the distance to the frontier reduces if output levels increase. Second, the coefficients are such that the first order derivatives of the frontier function $y_1 = f(y_{-1}, z, \delta)$ for the different output variables are negative. This shows that at the frontier, higher levels of biodiversity, cultural services or carbon sequestration result in lower levels of agricultural revenues, implying that there are trade-offs between the different outputs.

Second, quasi-convexity of the frontier function is investigated. If the objective is to investigate which bundles of ecosystem services can be generated, it is not relevant whether the frontier is concave or convex. In economic analyses, however, transformation functions are

[6] Note that $\partial\delta/\partial y = \delta/y \cdot \partial\ln(\delta)/\partial\ln(y)$.

Table 3
Parameter estimates of the translog frontier function.

Coeff.[a]		Estimate	95% Confidence interval[b]
α_0	Intercept	−0.211	(−0.252–−0.144)*
β_2	Ln(*msa*)	0.374	(0.308–0.401)*
β_3	Ln(*cult.serv.*)	0.458	(0.371–0.496)*
β_4	Ln(*carbon*)	0.060	(0.035–0.089)*
Γ_{22}	½Ln(*msa*)Ln(*msa*)	0.622	(0.513–0.867)*
$\Gamma_{23}=\Gamma_{32}$	½Ln(*msa*)Ln(*cult.serv*)	−0.677	(−0.910–−0.554)*
$\Gamma_{24}=\Gamma_{42}$	½Ln(*msa*)Ln(*carbon*)	0.045	(−0.032–0.093)
Γ_{33}	½Ln(*cult.serv*) Ln(*cult.serv*)	0.701	(0.549–1.034)*
$\Gamma_{34}=\Gamma_{43}$	½Ln(*cult.serv*)Ln(*carbon*)	−0.024	(−0.111–0.028)
Γ_{44}	½Ln(*carbon*) Ln(*carbon*)	−0.002	(−0.017–0.065)
γ_1	*GDP*	0.018	(−0.003–0.006)
γ_2	*Potential yield*	−0.159	(−0.098–−0.020)
γ_3	*Cover*	0.295	(0.121–0.200)
γ_{42}	Sub-region=CE[c]	−0.040	(0.012–0.098)
γ_{43}	Sub-region=YUG[c]	0.133	(0.032–0.133)*
γ_{44}	Sub-region=SE[c]	0.036	(−0.007–0.104)
β_1	Ln(*prov.serv*)	0.108	
Γ_{11}	½Ln(*prov.serv*)Ln(*prov.serv*)	0.008	
$\Gamma_{12}=\Gamma_{21}$	½Ln(*prov.serv*)Ln(*msa*)	0.010	
$\Gamma_{13}=\Gamma_{31}$	½Ln(*prov.serv*)Ln(*cult.serv*)	-5.7×10^{-4}	
$\Gamma_{14}=\Gamma_{41}$	½Ln(*prov.serv*)Ln(*carbon*)	−0.019	

[a] $\beta_1=1-\beta_2-\beta_3-\beta_4$, $\Gamma_{1i}+\Gamma_{2i}+\Gamma_{3i}+\Gamma_{4i}=0$ for all $i=1,2,3,4$. Before the model is estimated, first the continuous variables are standardized by dividing them by their respective sample means and the non-monotonous observations are removed from the sample.

[b] Variables marked with a * are significant at the 95% level. Confidence intervals are based on bootstrapping procedure with 200 runs.

[c] The conditional variable sub-region is modeled as three dummy variables for the sub-regions CE, YUG an SE, where they have the value 1 if the respective cell is part of the sub-region considered and 0 otherwise.

usually assumed to be quasi-concave. A quasi-concave frontier function is required to reach an efficient allocation of the goods and services from a welfare optimizing social planning perspective (Dasgupta and Maler, 2003). In case of a convex frontier, welfare optimization might lead to specialization in one of the resources. Moreover, if quasi-convexity is falsely assumed, bundles of services thought to be welfare maximizing may turn out to be welfare minimizing or local optima. In many economic studies, quasi-concavity of the transformation curve is simply assumed or imposed without testing for it (O'Donnell and Coelli, 2005). If policy analyses are based on false assumptions, misleading recommendations may be given.

Using the parameter values given above, the shape of the transformation function is analyzed by inspection of the eigenvalues of the bordered Hessian matrix—see Ruijs et al. (2012). The bordered Hessian gives the second-order partial derivatives of the frontier function. For the curve to be quasi-concave, the eigenvalues must be positive semi-definite. Evaluation of the eigenvalues for our data demonstrates that all observations have both positive and negative eigenvalues, implying that the frontier has a saddle point. So it follows that the frontier is not quasi-concave. If the frontier functions are used in economic analyses to evaluate whether certain bundles are welfare optimizing, this should be considered carefully.

Finally, the shapes of the frontier functions are shown in 3-dimensional plots, each time fixing the fourth output variable and the conditional variables at the mean value—see Fig. 3. These plots give an indication of how the curvature of the frontier function depends on the position at the frontier. As it is not possible to visualize shapes in more than three dimensions and as these figures are in fact extrapolations, they have to be interpreted with care for individual observations, especially at the boundaries and the areas with only few observations. The shapes, however, confirm the non-concavity of the frontier function. Especially for the relation between agricultural revenues and carbon sequestration, output specialization seems to be cost-effective with some cells focusing more on agricultural revenues and others more on carbon sequestration. For the relation between agricultural revenues and biodiversity, the picture is more mixed but shows that to a certain extent it is cost-effective if agricultural production, biodiversity and cultural services are jointly generated in one cell.

4.3. Opportunity costs

In this section, we estimate opportunity cost ratios for a situation in which all observations reached the frontier. The opportunity cost ratios reflect the marginal rates of transformation at the frontier. They estimate the gross agricultural revenues foregone due to a change in any of the other output variables (see also footnote 5).

Some statistics of the opportunity costs are presented in Tables 4 and 5. For carbon, the average opportunity cost of $263 per ton of carbon sequestered, corresponds with net agricultural benefits foregone of $19 per ton of carbon sequestred if a social profit rate of 7.25% is adopted (Hughes and Hare, 1994). This corresponds with the lowest estimates from the ranges estimated by MacLeod et al. (2010) and Antle et al. (2003). It is difficult to directly interprete the results for mean species abundance and the cultural services indicator as no other studies have used these indicators for evaluating trade-offs.

As argued above, opportunity costs reflect foregone gross agricultural revenues due to a marginal change in the respective ecosystem service. It is noted that they do not reflect preferences for a particular land use change but indicate the agricultural losses if land use changes such that biodiversity, cultural services or carbon sequestration increase marginally. For example, high opportunity costs for biodiversity indicate that a marginally higher level of biodiversity will entail a substantial loss in agricultural benefits. If the social benefits of having a higher level of biodiversity exceed these costs, it may still be worthwhile to invest in biodiversity. Similarly, in areas with low opportunity costs for cultural services, investments in these services may engender only a low loss of agricultural benefits. Such investments may still not be socially beneficial if people attach low values to additional services provision. Even though the opportunity costs do not provide information about the social benefits of land use changes, they do provide interesting information. The information on the order of magnitude of foregone gains or losses of agricultural benefits is seldomly available but essential for making decisions on land use changes and can only be obtained from trade-off analyses.

Figs. 4–6 show for biodiversity, cultural services and carbon sequestration the levels of these variables and their opportunity costs for the different cells (see Appendix A for the derivations). The figures clearly demonstrate that the within-country variation of the opportunity costs is substantial. Neighboring cells may have totally different opportunity costs, depending on the scarcity values in each cell and cell characteristics like land use patterns, agro-climatical and soil characteristics (represented by potential yield per cell) and population pressure.

For carbon sequestration, the pattern observed in Fig. 6 confirms the shape of the frontier given in Fig. 3. Generally opportunity costs for carbon sequestration are high in areas with low levels of carbon sequestration and vice versa. Moreover, they decrease at a decreasing rate when sequestration levels increase. These results are plausible. Generally, in a cell having a low level of carbon sequestration, the majority of the land is used for crop production and agricultural production is high. A marginal increase of carbon sequestration can only be realized by transforming agricultural land to grassland or forest, resulting in a relatively large loss of agricultural benefits. Cells having relatively lower levels of agricultural production, generally have higher

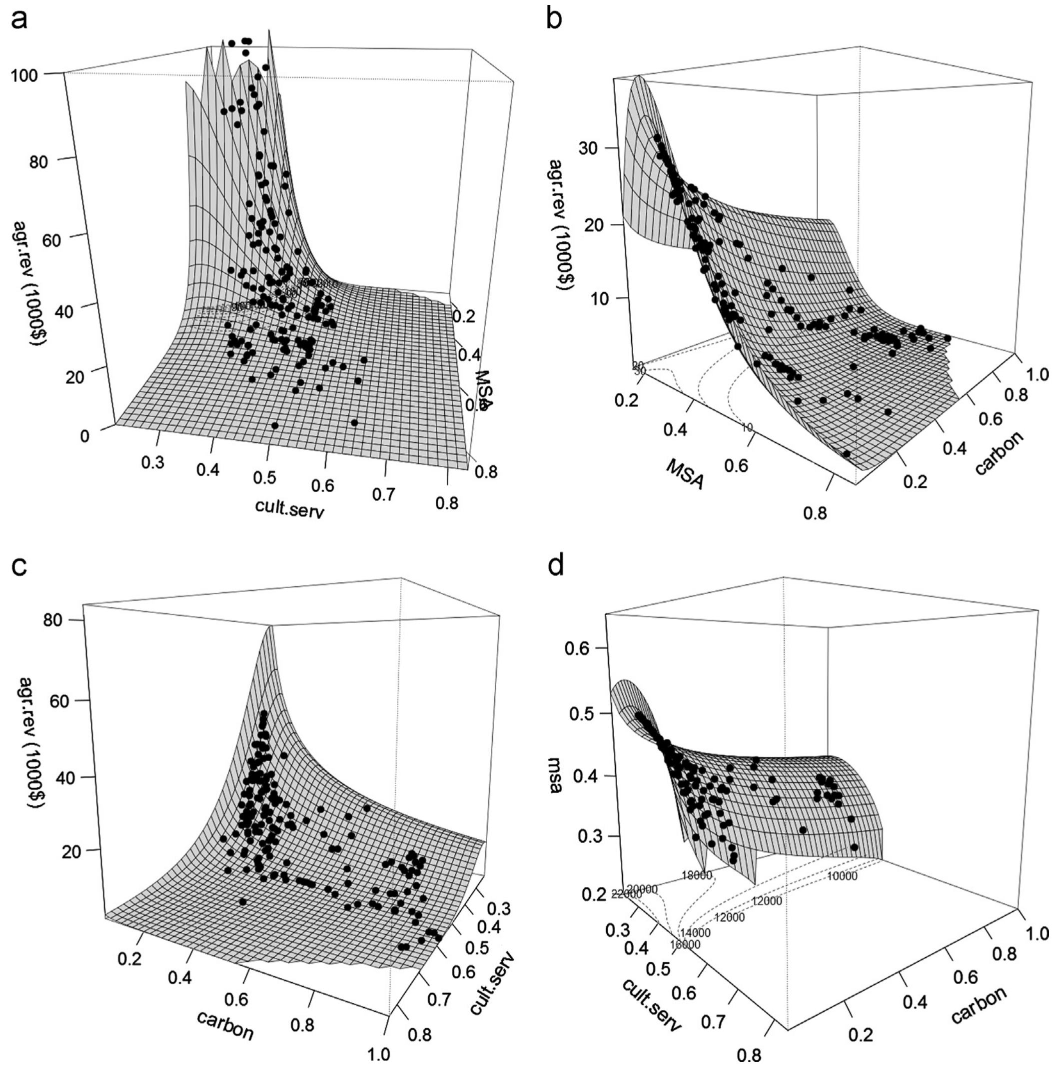

Fig. 3. 3-dimensional plots of the frontier for the sub-region SE. Note: The plots for the other three sub-regions are similar. The dots on the frontier are the observations on which the regression is based projected on the frontier. To draw the different plots, the output not given in the plot and the conditional variables are fixed at their mean values.

Table 4
Statistics of the opportunity costs for the ecosystem services distinguished.

	Mean	Median	Standard deviation	Min	Max
MSA ($ per % MSA index)	1276	1027	1029	1	8846
Cultural services ($ per % cult.serv index)	1865	1368	1706	0.3	12587
Wild fish ($ per kg)	415	305	380	0.1	2802
Wild fruit ($ per kg)	76	56	70	0.01	513
Wild game ($ per kg)	8857	6499	8102	1.5	59777
Wild mushrooms ($ per kg)	81	59	74	0.01	543
Tourism ($ per tourism point)	4998	3668	4572	0.8	33736
Carbon ($ per ton C)	263	202	220	1.4	1,986

percentages of grassland or forest or are cells in which potential yields are lower. In these cells, carbon sequestration can be improved without the need to sacrifice productive agricultural land. As a result opportunity costs are lower in these cells.

Figs. 4 and 5 indicate that this pattern is different for biodiversity and for cultural services. Generally, opportunity costs increase at a decreasing rate if levels of biodiversity or cultural services increase. After a certain threshold level, opportunity costs

Table 5
Median opportunity cost ratios and standard deviations by country.

	MSA ($ per % MSA)		Cultural services ($ per % cult.serv index)		Carbon ($ per tonne carbon)	
	Median	St. Dev.	Median	St.Dev.	Median	St. Dev.
Total	1027	1029	1368	1706	202	220
Belarus	1008	597	666	503	125	55
Estonia	247	310	233	339	62	65
Latvia	667	482	699	587	139	89
Lithuania	702	254	1234	510	211	78
Moldova	1534	707	2633	1664	490	151
Ukraine	1184	944	2323	1878	220	190
Czech	363	486	1103	1149	246	170
Hungary	1308	543	2286	1741	342	232
Poland	792	615	1329	1110	224	183
Slovakia	521	831	560	1364	110	97
Bosnia	1904	2344	694	2031	284	134
Croatia	870	772	934	1414	222	108
Macedonia	1817	770	1412	1243	1169	419
Serbia	1206	833	1159	2043	374	151
Slovenia	111	312	314	539	66	67
Albania	1208	1046	1577	1204	315	260
Bulgaria	1635	779	1947	960	347	181
Romania	2064	1439	2464	2440	155	294

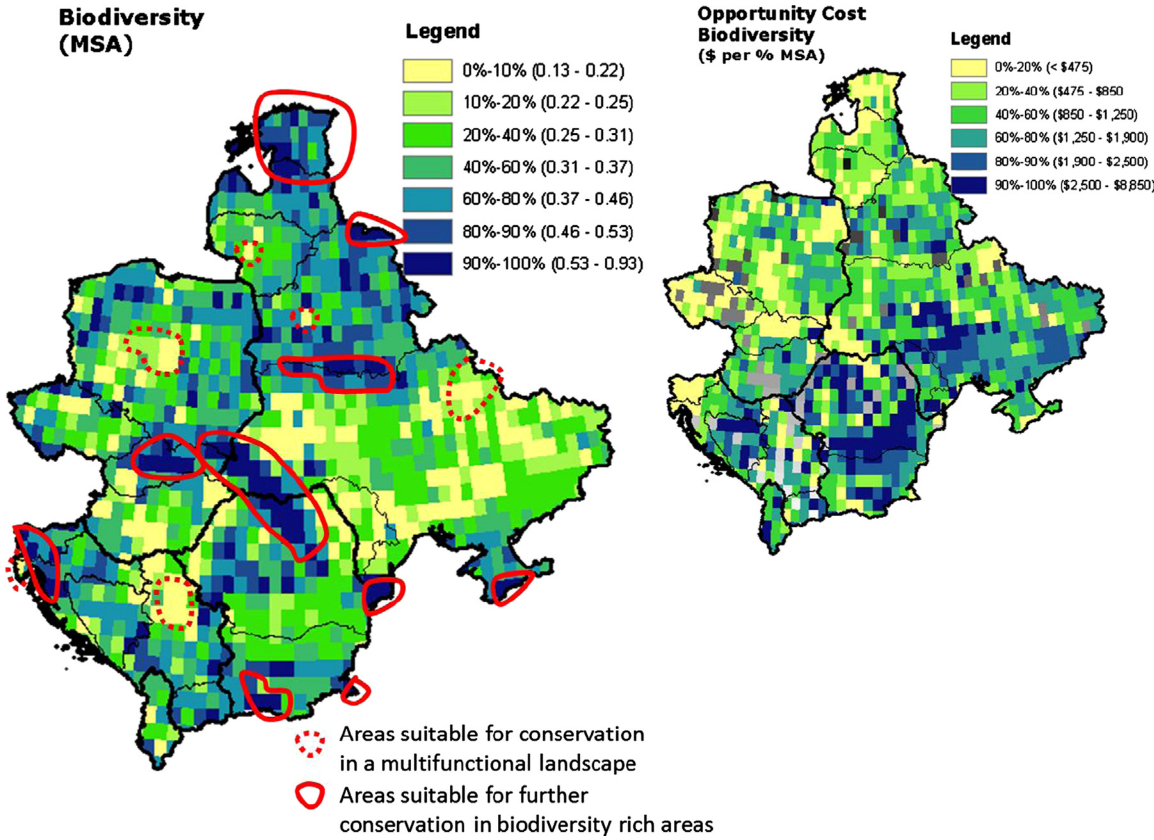

Fig. 4. Maps of levels (MSA) and opportunity costs ($ per % MSA) for biodiversity. Areas suitable for increasing biodiversity have low levels of opportunity costs. *Note:* Classification of the cells is such that each color corresponds with 10% or 20% of the observations. Grey cells are non-monotonic observations or outliers.

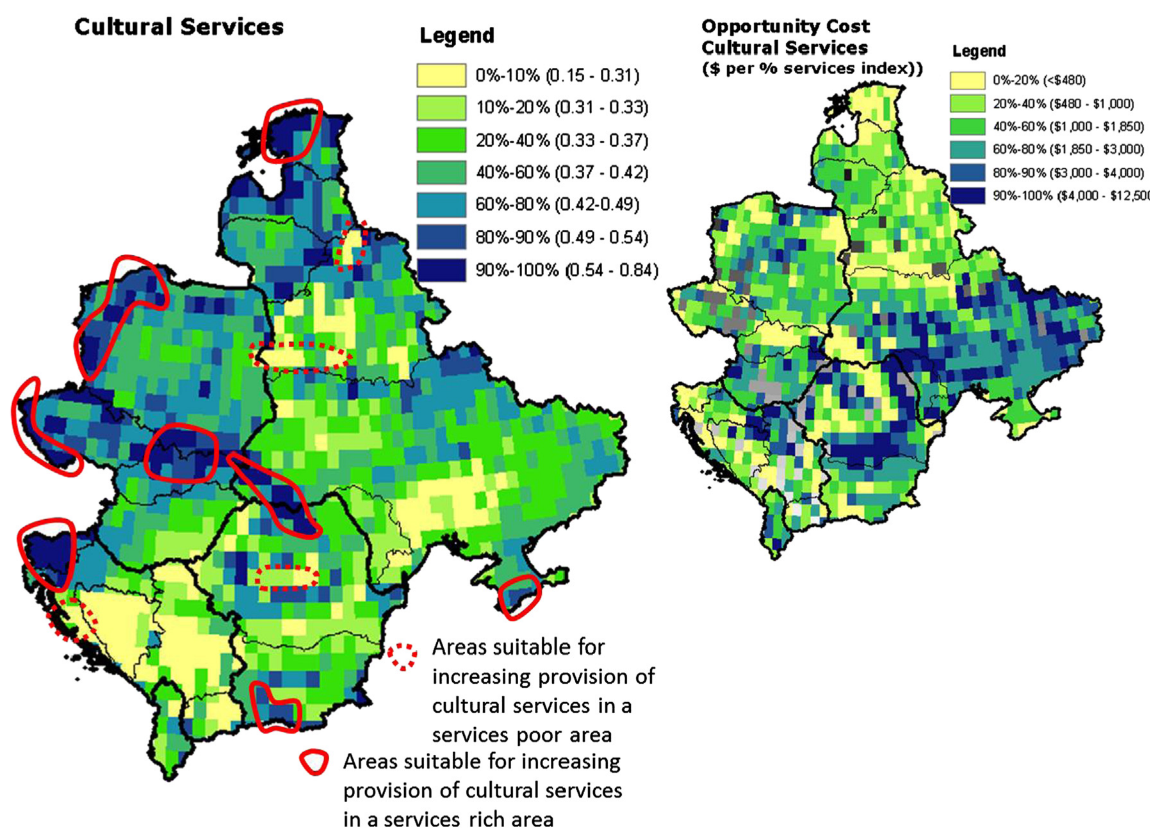

Fig. 5. Maps of levels and opportunity costs ($ per % cult.serv.) for cultural services. Areas suitable for increasing cultural services are the areas with low opportunity costs. *Note:* Classification of the cells is such that each color corresponds with 10% or 20% of the observations. Grey cells are non-monotonic observations or outliers.

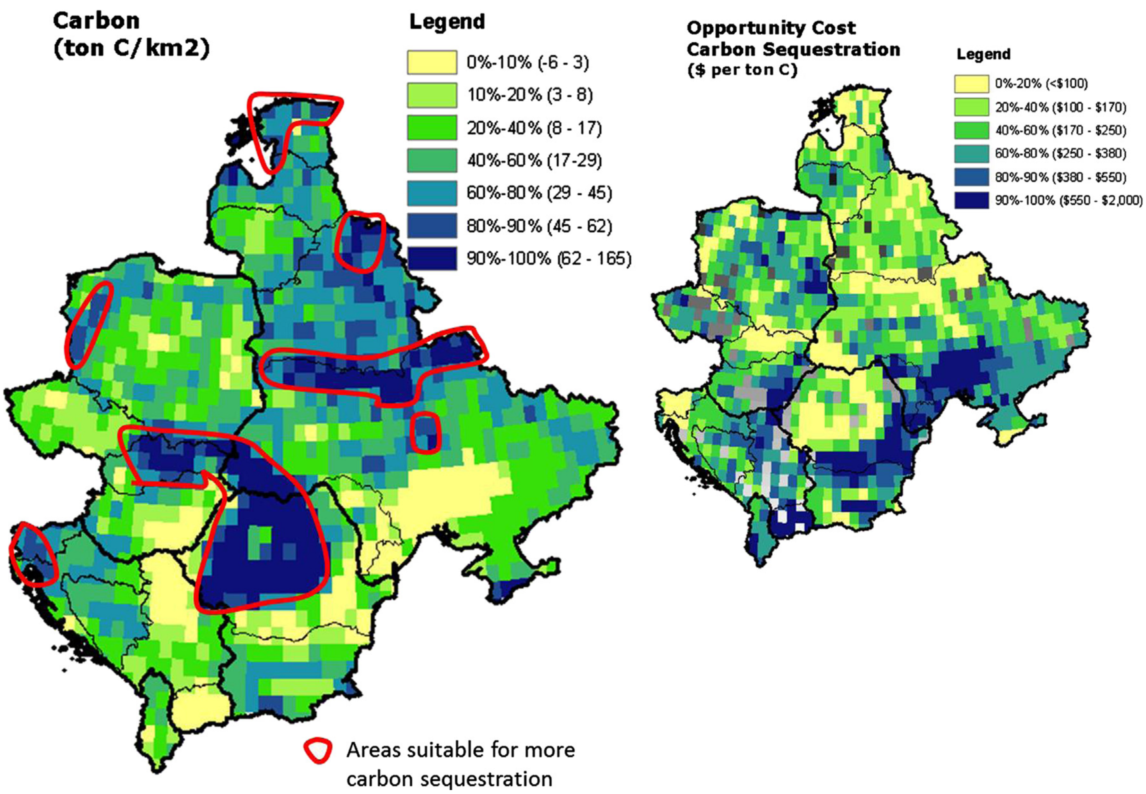

Fig. 6. Maps of levels (ton C/km^2) and opportunity costs ($ per tonne C) for carbon sequestration. Areas suitable for more carbon sequestration have low levels of opportunity costs for sequestration.
Note: Classification of the cells is such that each color corresponds with 10% or 20% of the observations. Grey cells are non-monotonic observations or outliers.

may start to decrease; this threshold is not the same for each cell type. The implication is that improving biodiversity can be relatively cheap in biodiversity poor but also in biodiversity rich areas. This is a plausible result. In biodiversity poor areas, small land use changes can positively affect biodiversity. In biodiversity rich areas, due to species interactions and network effects, small land use changes result in a more than linear growth in biodiversity due to which opportunity costs of biodiversity improvements are relatively low.

Figs. 4–6 also show in which areas expanding biodiversity, cultural services or carbon sequestration may be most suitable. Generally, an area is more suitable for expansion of one of the output variables if the corresponding opportunity costs are low. The figures show that the cells being most suitable for improvement of biodiversity or cultural services may have high or low output levels. Not all cells with high or low levels of biodiversity or cultural services are suitable for expansion, however. Opportunity costs may also be high for such cells, depending on the cell characteristics. In a similar way, it may be possible to indicate in which regions biodiversity or cultural services should not be expanded. Cells with high opportunity costs are particularly unsuitable. For carbon sequestration, only cells already having high sequestration levels are suitable. Hardly any cell has low sequestration levels and low opportunity costs. Based on the shadow price estimates, we can also indicate in which areas expanding agricultural production is most suitable—see Fig. 7. If opportunity costs are high, marginally reducing biodiversity, cultural services or carbon sequestration may result in a large gain of agricultural production. The figure shows that this is especially the case in areas already having high levels of agricultural revenues.

Mapping the areas suitable for improving biodiversity, cultural services, carbon sequestration or agricultural production all in one figure, shows in which areas expanding any of the ecosystem services entails low opportunity costs—see Fig. 8. Many of the areas suitable for improving biodiversity, cultural services or carbon sequestration overlap. The areas suitable for agricultural expansion are located in different regions. This is of no surprise. Biodiversity, cultural services and carbon sequestration are positively correlated. Expanding forested area is, under certain circumstances, beneficial for all three services. Expanding agricultural production requires in our model setting an expansion of the area of agricultural land as land use intensity is not included in the analysis. This leads to lower levels of the other three ecosystem services considered. This figure shows that by carefully selecting the areas in which nature conservation is stimulated, a win–win situation can be attained in which several ecosystem services benefit at the same time. This does not happen automatically as not all areas suitable for biodiversity improvement lead to higher levels of carbon sequestration or cultural services.

5. Discussion and conclusions

In this paper we assessed trade-offs between ecosystem services in a spatially explicit manner. The estimated opportunity costs represent the trade-offs between on the one hand biodiversity, cultural services or carbon sequestration and on the other hand agricultural revenues (provisioning services) resulting from a marginal change in land use. These opportunity costs are based on estimates of the transformation function. This function shows the feasible bundles of ecosystem services that can be generated in a region depending on a number of regional and agro-climatical characteristics. For this, a two-stage semi-parametric robust, conditional FDH method is set up and spatial data are used on agricultural revenues, cultural services, carbon sequestration and biodiversity for 18 Central and Eastern European countries.

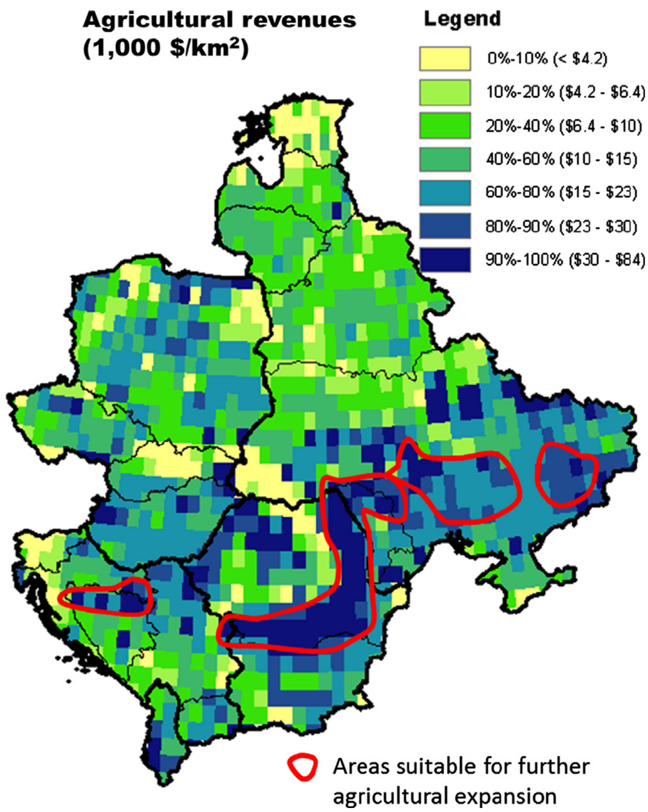

Fig. 7. Map of agricultural production (1000$/km^2). Areas suitable for agricultural expansion have high opportunity costs for one or more of the other ecosystem services. Note: Classification of the cells is such that each color corresponds with 10% or 20% of the observations. Grey cells are non-monotonic observations or outliers.

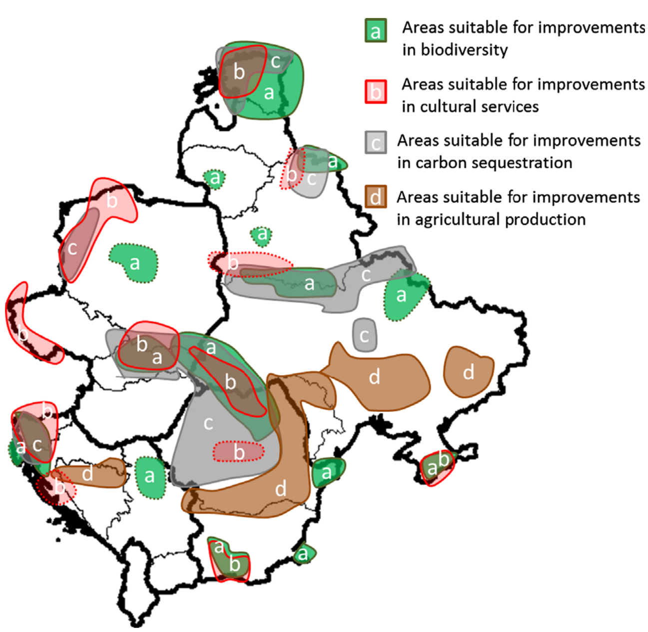

Fig. 8. Map of areas suitable for expansion of biodiversity, cultural services, carbon sequestration or agricultural production.

The objectives of this paper were twofold. The first aim was to analyze trade-offs between ecosystem services in a spatially explicit manner in order to assess in which regions ecosystem services can be increased cost-effectively. The second aim was to assess whether it is better to jointly generate ecosystem services or to specialize in one of them.

As for the first objective, we indicated which areas are most suitable for expanding provision of each of the ecosystem services. If areas are targeted carefully, joint improvement of biodiversity, cultural services and carbon sequestration can be reached. Moreover, in certain areas expansion of agricultural production only leads to a marginal loss of biodiversity or one of the other ecosystem services. Generally, it is more cost-effective to target regions with low opportunity costs. For increasing carbon sequestration, targeting areas already having high sequestration levels is cost-effective. Opportunity costs decrease at a decreasing rate when carbon sequestration levels increase. For biodiversity and cultural services this pattern is less clear. We also found that in general opportunity costs increase if biodiversity levels increase, even though at a decreasing rate. In some biodiversity rich areas opportunity costs start to decrease again if biodiversity levels cross a certain threshold level. It is noted that these opportunity costs indicate the gross agricultural benefits foregone due to a marginal increase of biodiversity, cultural services or carbon sequestration. They do not indicate whether society is willing to pay for these foregone benefits. Nevertheless, these results do provide valuable information on effects of land use changes which is relevant for making land use decisions.

As for the second objective, we conclude that the transformation function is not quasi-concave. As a result, the assumption often made in economic analysis that the cost of producing an additional unit of a certain good or service gradually increases, does not apply for all cases. Especially for the relation between gross agricultural revenues (provisioning services) and carbon sequestration, specialization in one of the ecosystem services seems to be cost-effective. The relationship between agricultural revenues and biodiversity or cultural services is more complex. In most areas combining bundles of ecosystem services is cost-effective. But especially if biodiversity levels are high, focusing on biodiversity conservation, instead of combining agricultural production and biodiversity, becomes cost-effective.

The approach presented here provides interesting insights in the interactions between ecosystem services if land use changes are to be proposed. Showing the trade-offs between ecosystem services helps to understand the linkages within ecosystems. An important observation is that these linkages may not follow a concave relationship, an assumption made in several related studies. If concavity of the frontier is imposed in a situation where this is in fact non-concave, flawed inference will follow and thus the conclusions on the suitability for multifunctional land use or specialization will be incorrect. In addition, the shape of the frontier depends on the variables included. Polasky et al. (2008) estimate a concave frontier for the relation between agricultural income and number of species. A rough estimate of MSA can be obtained by translating the acreages of their land use categories to MSA values and then drawing the frontier, instead of the number of species. It turns out that in that case the shape of their frontier would be different and be much more comparable to the shape found in our study. Moreover, results also depend on the services included. Sensitivity analysis showed that deleting one of the ecosystem services did not change much the relationship between the remaining variables. Similarly, adding another ecosystem services is not expected to alter the relationships between the services currently analyzed. The interactions with the newly added service, however, may show unexpected new insights. Adding more variables will also make the analysis more complex. Furthermore, this analysis only included one input variable, land allocation, and adopted a grid size of 50×50 km. A grid size of 50×50 km is a large scale for ecosystem services analyses because it misses local heterogeneity which is important for some ecosystem services. Currently it is not yet feasible to obtain data at a lower resolution, but this will be possible in the near future when IMAGE will be based on smaller grid cells. Note, however, that the grid data in fact are aggregates of sub-grid information on shares of each grid cell covered with a particular land use type and the ecosystem services generated by each of these land use types. Including more input variables may also become possible in the near future. Data on regulating services like pollination, erosion protection and pest control are already available, but they should be complemented with data on human inputs or land use intensity in order to be able to properly show the trade-off between using human or natural inputs. This is left for future research. Finally, the lessons learned on the interactions between the different ecosystem services may also complement and improve demand side environmental valuation studies searching for human preferences for land use changes. They may complement these studies as the results show which combinations of ecosystem services actually are feasible and what trade-offs of proposed land use changes will be. It may also improve valuation studies. In these studies respondents are asked to evaluate a proposed change without them actually knowing what the effects will be. In that case, it can be wondered whether values expressed truly reflect preferences. The trade-off information from our analysis provides insights about these effects which may in the end shape and sharpen preferences. Therefore, the relevance of the method and results presented here goes beyond its direct results but also feeds into other, related fields of research.

Acknowledgments

We would like to thank the reviewer for the valuable comments which have improved the quality of our paper. We would further like to thank the participants of the ESP-conference in Wageningen (Sept 2011) and the TEEB-conference in Leipzig (April 2012) and appreciate and acknowledge the feedback on earlier versions of this paper from Herman Vollebergh, Aart de Zeeuw, Sido Mylius and staff members from PBL—Netherlands Environmental Assessment Agency and the School of Social Sciences from the University of Manchester.

Appendix A. Two-stage semi-parametric estimation approach

In this appendix, the two-stage semi-parametric estimation approach is briefly discussed. Full details on the method are given in Ruijs et al. (2012). In the first stage, the frontier of efficient observations is estimated non-parametrically using the output oriented, robust conditional FDH method (Cazals et al., 2002; Daraio and Simar, 2005). Introduce for each observation a vector of outputs y and a vector of conditional variables z which are beyond the control of the decision makers. For the current analysis no input variables are considered. The output vector contains the ecosystem services produced in a certain region. The vector of conditional variables contains geographical location, soil type and regional income. The feasible output set Ψ is defined as all possible combinations of y and z such that the ecosystem services y can be produced given characteristics z. Considering an empirical situation with L observations, the Free Disposal Hull (FDH) estimator for the production possibility set Ψ^{FDH} is (with bandwidth parameter h)

$$\Psi^{FDH}(y,z)=\{(y,z)|y\le y_l, z\in[z_l-h, z_l+h]\exists\ l=1,\ldots,L\} \tag{A1}$$

As opposed to Fig. 1, the FDH frontier is a stairway-shaped curve connecting the efficient observations. For each observation (y,z), the Farrell–Debrue measure of output-oriented efficiency can be defined as:

$$\lambda(y|z)=\sup\{\lambda|(\lambda y,z)\in\Psi\} \tag{A2}$$

This function measures for each observation the distance of the output vector to the frontier, where λ100% measures the percentage output increase necessary to reach the frontier. Daraio and Simar (2005) showed that for a situation without conditional variables the estimator for the efficiency score λ can be written in probabilistic format as follows:

$$\lambda(x_i,y_i)=\operatorname*{Sup}_{\lambda_i}\{\lambda_i|S_Y(\lambda_i y_i)>0\} \tag{A3}$$

for observation y_i, and with $S_Y(y)=\Pr(y\le Y)$ the survivor function. Cazals et al. (2002) and Florens and Simar (2005) showed that for empirically estimating the conditional robust efficiency score (A2), a sample of size m should be drawn with replacement from the sample of observations repeatedly to calculate (A3), after which the expectation is taken (see also Daraio and Simar, 2005). With $S_Y(y|z)$ the conditional survivor function, the estimator can be written as

$$\lambda^m(y_i|z_i)=\int_0^\infty[1-(1-S_Y(uy_i|z_i))^m]du \tag{A4}$$

For estimating the conditional survivor function $S_Y\,(y|z)$ non-parametrically, smoothing techniques are needed such that in the reference samples of size m observations with comparable z-values have a higher probability of being chosen (see Daraio and Simar, 2005; De Witte and Kortelainen, 2009).

In the second stage, the non-parametric frontier obtained in the first stage is approximated parametrically with a flexible translog functional form. Following Daraio and Simar (2007b), the parametric frontier function can be derived from the Shephard output distance function that gives the distance of an observation from the frontier as a function of the output and conditional variables. The Shephard output distance measure, $\delta(y|z)$, is an alternative indicator for the distance from each observation to the frontier which is equal to the inverse of the Farrell–Debrue distance measure introduced in (A2), $\delta(y|z)=\lambda^{-1}(y|z)$. The aim is to estimate from the distance measures δ and variables y and z the translog distance function $\varphi(y,z;\theta)$, where the vector θ represent the unknown parameters of the translog distance function. The parameters θ that can be estimated by solving the following optimization problem:

$$\theta_0=\arg\min_\theta\left[\sum_{i=1}^{L}(\ln\delta(y_i|z_i)-\ln\varphi(y_i,z_i;\theta))^2\right] \tag{A5}$$

However, to use (A5) in practice we have to first write it in terms of observed (or estimated) variables. To this end, introduce the translog distance function $\ln\varphi(y,z;\theta)=\alpha_0+\beta'\ln y+\frac{1}{2}\ln y'\Gamma\ln y+\gamma'\ln z$, where matrix $\Gamma=\Gamma'$ is symmetric (see Daraio and Simar, 2007a), with $\theta=(\alpha_0,\beta,\Gamma)$ the parameters of the distance function; α_0 is a scalar and β a vector. Due to homogeneity of degree one in y, it has to hold that $\beta'\times i_M=1$ and $\Gamma\times i_M=0$, with i_M the identity vector of size M. Define β_{-1} the $(M-1)$-vector of coefficients not containing β_1 and

$$\Gamma=\begin{bmatrix}\tau_1 & \tau'_{-1}\\ \tau_{-1} & \Gamma_{22}\end{bmatrix}$$

with $\tau=(\tau_1\quad\tau'_{-1})\in\mathbb{R}^M$, τ_{-1} an $(M\text{-}1)$-vector and Γ_{22} an $(M-1)\times(M-1)$-matrix. Due to the homogeneity assumption $\beta_1=1-\beta'_{-1}\times i_{M-1}$, $\tau_1=-\tau'_{-1}\times i_{M-1}$ and $\tau_{-1}=-\Gamma_{22}\times i_{M-1}$. With these notations, we can now estimate the translog distance function with OLS by solving the following problem (A5)

$$\theta_0=\arg\min_\theta\left[\sum_{i=1}^{L}(\ln\delta_i-(\alpha_0+\beta'\ln y_i+\frac{1}{2}\ln y'_i\Gamma\ln y_i+\gamma'\ln z_i))^2\right]$$

$$= \arg\min_{\theta} \left[\sum_{i=1}^{L} \left(-\ln y_{i1}^{*} - \left(\alpha_0 + \sum_{j=2}^{M} \beta_j \ln \tilde{y}_{ij} + \frac{1}{2} \sum_{j=2}^{M} \sum_{l=2}^{M} \ln \tilde{y}_{ij} \Gamma_{jl} \ln \tilde{y}_{il} + \gamma' \ln z_i \right) \right)^2 \right]$$

with $y_{i1}^{*} = y_{i1}/\delta_i = y_{i1}\lambda_i$ the values of y_{i1} projected on the output efficient frontier and $\tilde{y}_{i,-1} = y_{i,-1}/y_{i1} = y_{i,-1}^{*}/y_{i1}^{*}$. In words, estimate the best parametric approximation of the multivariate output distance function is the same as projecting the output values on the output efficient frontier using the distance values estimated in the first stage, and then estimating with ordinary least squares (OLS) the frontier function

$$\ln y_{i1}^{*} = -\left(\alpha_0 + \beta'_{-1} \ln \tilde{y}_{i,-1} + \frac{1}{2} \ln \tilde{y}'_{i,-1} \Gamma_{22} \ln \tilde{y}_{i,-1} + \gamma' \ln z_i \right) \quad \text{(A6)}$$

Based on the frontier function (A6), opportunity costs or trade-offs between the different variables can be derived. For this we use the duality relationship between the benefit function and the distance function (Färe and Grosskopf, 2000; Bellenger and Herlihy, 2010). For vector of prices p, the benefit function is defined as $B(p) = \sup_y \{p'y | (x, y, z) \in \Psi\}$. As $y/\delta(x, y|z)$ is a feasible output vector, it has to hold that $B(p) \geq p'y/\delta(x, y|z)$ and so $\delta(x, y|z) = \max_p [p'y/B(p)]$. As a result, for each output m

$$\frac{\partial \delta(x, y|z)}{\partial y_m} = \frac{p_m}{B(p)} \quad \text{(A7)}$$

It follows that if the market price is known for one of the outputs, e.g. for the first output, opportunity costs for the other outputs are equal to

$$p_k = p_1 \times \frac{\partial \delta(y|z)/\partial y_k}{\partial \delta(y|z)/\partial y_1} \quad \text{(A8)}$$

References

Alkemade, R., Oorschot, M.V., Miles, L., Nellemann, C., Bakkenes, M., Brink, B.T., 2009. GLOBIO3: a framework to investigate options for reducing global terrestrial biodiversity loss. Ecosystems 12, 374–390.

Antle, J., Capalbo, S., Mooney, S., Elliott, E., Paustian, K., 2003. Spatial heterogeneity, contract design, and the efficiency of carbon sequestration policies for agriculture. Journal of Environmental Economics and Management 46 (2), 231–250.

Barraquand, F., Martinet, V., 2011. Biological conservation in dynamic agricultural landscapes: effectiveness of public policies and trade-offs with agricultural production. Ecological Economics 70 (5), 910–920.

Bateman, I.J., 2009. Bringing the real world into economic analyses of land use value: incorporating spatial complexity. Land Use Policy 26 (Suppl. 1(0)), S30–S42.

Bateman, I.J., Mace, G.M., Fezzi, C., Atkinson, G., Turner, K., 2011. Economic analysis for ecosystem service assessments. Environmental & Resource Economics , http://dx.doi.org/10.1007/s10640-010-9418-x.

Bellenger, M.J., Herlihy, A.T., 2010. Performance-based environmental index weights: are all metrics created equal?". Ecological Economics 69 (5), 1043–1050.

Bostian, M.B., A.T. Herlihy (2012). Shadow Pricing Wetland Function. Paper Presented at the EAERE 2012 Conference. Prague, Czech Republic.

Bouwman, A.F., Kram, T., Goldewijk, K.Klein (Eds.), 2006. PBL—Netherlands Environmental Assessment Agency, Bilthoven, The Netherands.

Boyd, J., Banzhaf, S., 2007. What are ecosystem services? The need for standardized environmental accounting units. Ecological Economics 63 (2–3), 616–626.

Brander, L.M., Koetse, M.J., 2011. The value of urban open space: meta-analyses of contingent valuation and hedonic pricing results. Journal of Environmental Management 92 (10), 2763–2773.

Brink, B.T., Esch, S.V.D., Kram, T., Oorschot, M.V., Alkemade, R., Ahrens, R., Bakkenes, M., Bakkes, J., Berg, M.V.D., Christensen, V., Janse, J., Jeuken, M., Lucas, P., Manders, T., Meijl, H.V., Stehfest, E., Tabeau, A., Vuuren, D.V., Wilting, H., 2010. Rethinking Global Biodiversity Strategies: Exploring Structural Changes in Production and Consumption to Reduce Biodiversity Loss. PBL—Netherlands Environmental Assessment Agency, Bilthoven.

Brouwer, R., L. Brander, A. Wagtendonk (2012). Economic valuation of regulating services provided by wetlands in agricultural landscapes: a meta-analysis. In: TEEB-Conference. Leipzig.

Brown, G., Patterson, T., Cain, N., 2011. The devil in the details: non-convexities in ecosystem service provision. Resource and Energy Economics 33 (2), 355–365.

Bryan, B.A., 2010. Development and application of a model for robust, cost-effective investment in natural capital and ecosystem services. Biological Conservation 143 (7), 1737–1750.

Bryan, B.A., King, D., Ward, J.R., 2011. Modelling and mapping agricultural opportunity costs to guide landscape planning for natural resource management. Ecological Indicators 11 (1), 199–208.

Cazals, C., Florens, J.P., Simar, L., 2002. Nonparametric frontier estimation: a robust aproach. Journal of Econometrics 106, 1–25.

Chavas, J.P., 2009. On the productive value of biodiversity. Environmental & Resource Economics 42 (1), 109–131.

Costanza, R., d'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R.V., Paruelo, J., Raskin, R.G., Sutton, P., van den Belt, M., 1997. The value of the world's ecosystem services and natural capital. Nature 387, 253–260.

Costanza, R., d'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R.V., Paruelo, J., Raskin, R.G., Sutton, P., van den Belt, M., 1998. The value of ecosystem services: putting the issues in perspective. Ecological Economics 25 (1), 67–72.

Crossman, N.D., Bryan, B.A., 2009. Identifying cost-effective hotspots for restoring natural capital and enhancing landscape multifunctionality. Ecological Economics 68 (3), 654–668.

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.

Daily, G.C., Polasky, S., Goldstein, J., Kareiva, P.M., Mooney, H.A., Pejchar, L., Ricketts, T.H., Salzman, J., Shallenberger, R., 2009. Ecosystem services in decision making: time to deliver. Frontiers in Ecology and the Environment 7 (1), 21–28.

Daraio, C., Simar, L., 2005. Introducing environmental variables in nonparametric frontier models: a probabilistic approach. Journal of Productivity Analysis 24, 93–121.

Daraio, C., Simar, L., 2007a. Advanced Robust and Nonparametric Methods in Efficiency Analysis: Methodology and Applications. Springer, New York.

Daraio, C., Simar, L., 2007b. Conditional nonparametric frontier models for convex and nonconvex technologies: a unifying approach. Journal of Productivity Analysis 28, 13–32.

Dasgupta, P., Maler, K.G., 2003. The economics of non-convex ecosystems: introduction. Environmental & Resource Economics 26 (4), 499–525.

Deprins, D., Simar, L., Tulkens, H., 1984. Measuring labor inefficiency in post offices. In: Marchand, M., Pestiau, P., Tulkens, H. (Eds.), The Performance of Public Enterprises: Concepts and Measurements, Amsterdam, North-Holland, pp. 243–267.

de Groot, R.S., Alkemade, R., Braat, L., Hein, L., Willemen, L., 2010. Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making. Ecological Complexity 7 (3), 260–272.

De Witte, K., M. Kortelainen (2009). Blaming the Exogenous Environment? Conditional Effciency Estimation with Continuous and Discrete Exogenous Variables. MPRA Paper. Munich.

De Witte, K., Marques, R.C., 2010. Influential observations in frontier models, a robust non-oriented approach to the water sector. Annals of Operations Research 181, 377–392.

Diaz-Balteiro, L., Romero, C., 2008. valuation of environmental goods: a shadow value perspective. Ecological Economics 64, 517–520.

EC-JRC (2003). Global Land Cover 2000 Database, European Commission, Joint Research Centre.

Färe, R., Grosskopf, S., 2000. Theory and application of directional distance functions. Journal of Productivity Analysis 13 (2), 93–103.

Fenger, H.J.M., 2007. Welfare regimes in Central and Eastern Europe: incorporating post-communist countries in a welfare regime typology. Contemporary Issues and Ideas in Social Sciences 3, 2.

Fisher, B., Turner, R.K., Morling, P., 2009. Defining and classifying ecosystem services for decision making. Ecological Economics 68 (3), 643–653.

Florens, J.P., Simar, L., 2005. Parametric approximations of nonparametric frontiers. Journal of Econometrics 124 (1), 91–116.

Ghermandi, A., van den Bergh, J., Brander, L.M., de Groot, H.L.F., Nunes, P., 2010. Values of natural and human-made wetlands: a meta-analysis. Water Resources Research, 46.

Haines-Young, R., 2009. Land use and biodiversity relationships. Land Use Policy 26 (Suppl. 1(0)), S178–S186.

Haines-Young, R., Potschin, M., 2010. The links between biodiversity, ecosystem services and human well-being. In: Rafaelli, D.G. (Ed.), Ecosystem Ecology: A New Synthesis. Cambridge University Press, Cambridge.

Hauer, G., Cumming, S., Schmiegelow, F., Adamowicz, W., Weber, M., Jagodzinski, R., 2010. Tradeoffs between forestry resource and conservation values under alternate policy regimes: a spatial analysis of the western Canadian boreal plains. Ecological Modelling 221 (21), 2590–2603.

Hof, J., Flather, C., Baltic, T., King, R., 2004. Forest and rangeland ecosystem condition indicators: identifying national areas of opportunity using data envelopment analysis. Forest Science 50 (4), 473–494.

Hughes, G., Hare, P., 1994. The international competitiveness of industries in Bulgaria, Czechoslovakia, Hungary and Poland. Oxford Economic Papers 46, 200–221.

Hussain, S.S., L. Brander, A. McVittie, O. Vardakoulias, A. Wagtendonk, P. Verburg, R. Tinch, A. Fofana, C. Baulcomb, L. Mathieu (2011). The Economics of Ecosystems and Biodiversity Quantitative Assessment—Final Report. Geneva, UNEP.

Isbell, F., Calcagno, V., Hector, A., Connolly, J., Harpole, W.S., Reich, P.B., Scherer-Lorenzen, M., Schmid, B., Tilman, D., van Ruijven, J., Weigelt, A., Wilsey, B.J., Zavaleta, E.S., Loreau, M., 2011. High plant diversity is needed to maintain ecosystem services. Nature 477 (7363) 199-U196.

Keeler, B.L., Polasky, S., Brauman, K.A., Johnson, K.A., Finlay, J.C., O'Neill, A., Kovacs, K., Dalzell, B., 2012. Linking water quality and well-being for improved assessment and valuation of ecosystem services. Proceedings of the National Academy of Sciences of the United States of America 109 (45), 18619–18624.
MacLeod, M., Moran, D., Eory, V., Rees, R.M., Barnes, A., Topp, C.F.E., Ball, B., Hoad, S., Wall, E., McVittie, A., Pajot, G., Matthews, R., Smith, P., Moxey, A., 2010. Developing greenhouse gas marginal abatement cost curves for agricultural emissions from crops and soils in the UK. Agricultural Systems 103 (4), 198–209.
Macpherson, A.J., Principe, P.P., Smith, E.R., 2010. A directional distance function approach to regional environmental-economic assessments. Ecological Economics 69 (10), 1918–1925.
Maes, J., Paracchini, M.L., Zulian, G., Dunbar, M.B., Alkemade, R., 2012. Synergies and trade-offs between ecosystem service supply, biodiversity, and habitat conservation status in Europe. Biological Conservation 155 (0), 1–12.
Martínez-Harms, M.J., Balvanera, P., 2012. Methods for mapping ecosystem service supply: a review. International Journal of Biodiversity Science, Ecosystem Services & Management 8 (1–2), 17–25.
Millennium Ecosystem Assessment, 2005. Ecosystems and Human Well-Being: Current State and Trends. Island Press, Washington D.C..
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.
National Research Council, 2005. Valuing Ecosystem Services: Toward Better Environmental Decision-Making. Natl. Acad. Press, Washington, DC.
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, M.R., 2009. Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. Frontiers in Ecology and the Environment 7 (1), 4–11.
O'Donnell, C.J., Coelli, T.J., 2005. A Bayesian approach to imposing curvature on distance functions. Journal of Econometrics 126 (2), 493–523.
OECD, 2012. OECD Environmental Outlook to 2050. OECD, Paris.
PBL, 2012. Roads from RIO+20. Den Haag. PBL—Netherlands Environmental Assessment Agency, The Netherlands.
Polasky, S., Nelson, E., Camm, J., Csuti, B., Fackler, P., Lonsdorf, E., Montgomery, C., White, D., Arthur, J., Garber-Yonts, B., Haight, R., Kagan, J., Starfield, A., Tobalske, C., 2008. Where to put things? Spatial land management to sustain biodiversity and economic returns. Biological Conservation 141 (6), 1505–1524.
Raudsepp-Hearne, C., Peterson, G.D., Bennett, E.M., 2010. Ecosystem service bundles for analyzing tradeoffs in diverse landscapes. Proceedings of the National Academy of Sciences of the United States of America 107 (11), 5242–5247.
Romstad, E., 2008. The informational role of prices. European Review of Agricultural Economics 35 (3), 263–280.
Ruijs, A., M. Kortelainen, A. Wossink, C.J.E. Schulp, and R. Alkemade (2012). Performance Differences and Trade-Offs in the Provision of Ecosystem Goods and Services. PBL Working Papers 5. The Hague, PBL Netherlands Environmental Assessment Agency.
Sauer, J., Wossink, G.A.A., 2013. Marketed outputs and non-marketed ecosystem services: the evaluation of marginal costs. European Review of Agricultural Economics , http://dx.doi.org/10.1093/erae/jbs040.
Schulp, C.J.E., Alkemade, R., Klein Goldewijk, K., Petz, K., 2012. Mapping ecosystem functions and services in Eastern Europe using global-scale data sets. International Journal of Biodiversity Science, Ecosystem Services & Management 8, 156–168.
Schulp, C.J.E., Nabuurs, G.J., Verburg, P.H., 2008. Future carbon sequestration in Europe—effects of land use change. Agriculture, Ecosystems and Environment 127, 251–264.
Smith, F.P., Gorddard, R., House, A.P.N., McIntyre, S., Prober, S.M., 2012. Biodiversity and agriculture: production frontiers as a framework for exploring trade-offs and evaluating policy. Environmental Science & Policy 23, 85–94.
TEEB, 2010. The Economics of Ecosystems and Biodiversity: Ecological and Economic Foundations. Eartscan Publishers, London.
UK National Ecosystem Assessment (2011). The UK National Ecosystem Assessment: Synthesis of the Key Findings, UNEP-WCMC, Cambridge.
UNEP (2012). GEO5: Environment for the Future we Want. Nairobi, Kenya, United Nations Environment Program.
US Environmental Protection Agency (2009). Valuing the Protection of Ecological Systems and Services. Washington, DC, EPA Sci. Advis. Board Rep.
Van Vuuren, D., Faber, A. (Eds.), 2009. Growing within Limits. A Report to the Global Assembly 2009 of the Club of Rome. PBL—Netherlands Environmental Assessment Agency, Bilthoven.
Wallace, K.J., 2007. Classification of ecosystem services: problems and solutions. Biological Conservation 139 (3–4), 235–246.
White, C., Halpern, B.S., Kappel, C.V., 2012. Ecosystem service tradeoff analysis reveals the value of marine spatial planning for multiple ocean uses. Proceedings of the National Academy of Sciences of the United States of America 109 (12), 4696–4701.