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A. Ruijs
M. Kortelainen
A. Wossink
C.J.E. Schulp
R. Alkemade

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Manchester M13 9PL

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A. Ruijs[1], M. Kortelainen[2,3], A. Wossink[2], C.J.E. Schulp[1,4] and R. Alkemade[1]

1) PBL – Netherlands Environmental Assessment Agency, Bilthoven, The Netherlands

2) VATT – Government Institute for Economic Research, Finland.

3) Economics, School of Social Sciences, University of Manchester, UK.

4) IVM-VU, Amsterdam, The Netherlands

Correspondence:

Arjan Ruijs

PBL-Netherlands Environmental Assessment Agency

PO box 303

3720 AH Bilthoven

The Netherlands

e-mail: arjan.ruijs@pbl.nl

tel. +31 30 274 2167

fax. +31 30 274 4479

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Abstract

Understanding the economic value of nature and the services it provides to humans has become increasingly important for local, national and global policy. Consensus is still lacking on how to properly integrate the importance of biodiversity and ecosystem services into decision making processes regarding land use change. This paper seeks to contribute to the literature on ecosystem service assessment by developing and demonstrating a method by which the spatially explicit opportunity costs of several jointly produced ecosystem services can be estimated. The method is based on a two-stage frontier approach using parametric and non-parametric estimation techniques. The approach is implemented for provisioning services, biodiversity, cultural services and carbon sequestration with data for 18 Central and Eastern European countries. Results show that opportunity costs of changes in ecosystem services provision, given in terms of foregone benefits of provisioning services, differ substantially between regions. Those areas having already relatively high levels of carbon sequestration have a comparative advantage in sequestering carbon. Opportunity costs of biodiversity generally increase with increasing biodiversity up to a turning point after which they decrease again. We argue that the method and the resulting opportunity costs can lead to more integrated and rigorous policy support and dialogue.

Keywords: opportunity costs, ecosystem services, biodiversity, non-separability, nonparametric estimation, trade-offs, comparative advantage

1. Introduction

Understanding the economic value of nature and the services it provides to humans has become increasingly important for local, national and global policy.[1] In spite of the numerous recent studies on the topic, consensus is still lacking on how to properly integrate the importance of biodiversity and ecosystem goods and services into decision making processes regarding land use change (see e.g. Millennium Ecosystem Assessment, 2005; PBL, 2010; Secretariat of the Convention on Biological Diversity, 2010; TEEB, 2010; UK National Ecosystem Assessment, 2011). Problems arise as the majority of these goods and services have no formal market or are characteristically intangible. In addition it is often difficult to determine how biodiversity and ecosystem goods and service are connected and how this is affected by a change in land use (Turner et al., 2010; Barbier, 2011).

The ecological and economic effects of land use decisions can be analyzed from a demand and a supply side perspective. Most previous studies start from consumer sovereignty and revealed or stated preferences. Environmental valuation methods are used to judge the net impact on human wellbeing of changes in biodiversity and ecosystem services (see e.g. Naeem et al., 2009; TEEB, 2010). As is well documented, there are several caveats with the use of valuation methods in this context.[2] First, valuation analyses are limited in that they are not well suited to include interactions between ecosystem goods and services (Montgomery et al., 1999; Batabyal et al., 2003). If the resulting overall policy estimate is to be robust it is vital, however, to account for how their contributions overlap in the economic system (Carbone and Smith, 2010;

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[1] In the Millennium Ecosystem Assessment (2005) *ecosystem goods and services* are defined as the benefits humans obtain from ecosystems. Usually four categories of ecosystem services are distinguished (see also Daily, 1997): 1. provisioning services, e.g. food, wood, water and fiber, fuel;2. regulating services, e.g. climate regulation, flood regulation, water purification, disease regulation, pollination; 3. cultural services, e.g. aesthetic, spiritual, educational, recreational; and 4. supporting services, e.g. nutrient cycling, soil formation, primary production.

[2] Overviews and in-depth discussions of the methods to value ecosystem services were initiated in the USA by the National Academy of Sciences (National Research Council, 2005) and the US Environmental Protection Agency (US Environmental Protection Agency, 2009). Recently, the TEEB study (the economics of ecosystems and biodiversity) provided an overview of the state-of-the-art knowledge on valuation methods and marginal values of ecosystem services for ecosystems around the globe (TEEB, 2010). TEEB is based on an evaluation of the environmental valuation literature of the last decades. As a follow-up of this, several European governments are planning to perform or finalizing a TEEB-like study for their own country in order to get a better idea of the importance of their natural resources for their economy (see e.g. UK National Ecosystem Assessment, 2011).

Balmford et al., 2011). Second, the services and estimated economic values are location specific and thus it is difficult to generalize the analyses to higher spatial scales. Third, for unfamiliar goods and services preferences may be underdeveloped (Plott and Zeiler, 2005; Bateman et al., 2008) and/or unstable, for example, due to anchoring and framing effects (Ariely et al., 2003). These issues are particularly germane to cases involving regulating and supporting services which are often of concern to policy makers but have no direct consumer appeal due to their unfamiliarity (Johnston and Russell, 2011). Finally, the valuation methods risk to double count the benefits from several ecosystem services (Fisher et al., 2008). For these reasons estimates from valuation studies remain somewhat controversial to inform decisions on management and allocation of land resources.

The supply side perspective starts from land use as such and estimates the provision of ecosystem goods and services in biophysical terms. Integrated assessment studies show the effects of land use choices on ecosystems and the interactions and dependencies between the different ecosystem functions and services (Millennium Ecosystem Assessment, 2005). Important advances have been made in this area. Recent studies account for spatial heterogeneity and have expanded beyond singles services to consider various ecosystem services jointly produced by the ecosystem (see e.g. Raudsepp-Hearne et al., 2010; Maes et al., 2012). This enables the analysis of production possibility frontiers for ecosystem services. However, much remains to be done. Spatially explicit analysis of the trade-offs between ecosystem services in particular is crucial to be able to judge effects of land use changes in policy decisions (Bateman et al., 2011; Haines-Young et al., 2012). Hence, there is an obvious need for new methods which properly consider the ecological complexities and interactions and which show the inevitable trade-offs of land use changes at appropriate spatial scales (Polasky and Segerson, 2009; McShane et al., 2011).

Against this background we develop and demonstrate a method by which the spatially explicit opportunity costs of several jointly produced ecosystem goods and services can be estimated. This method, which draws and expands on the two-stage frontier approach proposed by Florens and Simar (2005), traces out production frontiers showing the combinations of ecosystem services and goods that can be generated. The suggested approach enables the assessment of marginal rates of transformation over a range of levels of these public goods. The resulting trade-offs or opportunity costs - the value of the foregone alternative – reflect the (economic) implication of biophysical and ecological changes. They derive their economic meaning from the scarcity of the underlying resources and the jointness in the generation of ecosystem goods and services. Thus the trade-offs reflect the underlying relationship between priced and non-priced ecosystem services and enable ecosystem service synergies to be covered without the risk of double counting. This supply side opportunity cost perspective is especially suited to situations where current preference setting is ill-informed or lacking and where future generations are under consideration when decisions need to be made. As is the situation with biodiversity and regulatory and supporting ecosystem services. The results provide relevant spatial information on trade-offs between ecosystem services and on the areas that have a comparative advantage for supplying particular ecosystem services; information that is essential for supporting land use decisions.

Our method differs from previous studies in several ways. The extant literature on analyzing trade-offs or deriving opportunity costs is limited (see e.g. Montgomery et al., 1999; Ferraro, 2004; Nalle et al., 2004; Naidoo et al., 2006; Polasky et al., 2008; Egoh et al., 2010; Macpherson et al., 2010; Bostian and Herlihy, 2012; Lester et al., forthcoming 2012). Most studies focus on limited geographical scales such as regions or catchments, whereas we focus on a much higher multinational or global scale. Previous studies that aim at higher spatial scales, such as Haines-Young et al. (2012) do not make the step to estimate opportunity costs. The approach most similar to our method to estimate production frontiers is that presented by Polasky et al. (2008). While these authors adopt a bio-economic model to derive a two-dimensional efficiency frontier, we use parametric and non-parametric frontier estimation techniques to estimate a multidimensional frontier. The multidimensional approach allows for the analysis of interdependencies between different services. Recent examples using frontier methods include Bosetti and Buchner (2009) for an evaluation of climate scenarios, Ferraro (2004) for an analysis of the allocation of conservation funds across a spatially heterogeneous landscape, Hof et al. (2004) and Macpherson et al. (2010) for an evaluation of environmental performance.[3] Moreover, where most studies adopt parametric or non-parametric approaches to estimate the frontier, we adopt a two-stage, combined non-parametric and parametric approach (Florens and Simar, 2005). The current paper is one of the first empirical applications of this method, which has as advantage over many of the other possible frontier approaches its flexibility with regard to assumptions on the convexity of the frontier and the distribution of the error term. In particular, as no convexity assumptions are made, we are able to test for non-convexities in the system. This is a feature of many ecosystems but an issue often ignored in economic studies (Chavas, 2009; Brown et al., 2011) even though acknowledged by Dasgupta and Maler (2003) to be a feature having important consequences for the functioning of the price mechanism. Assuming convexity relations where they do not exist may result in misinterpretations and false conclusions.

The information provided by the method developed in this paper is expected to be useful to guide decision makers in targeting and prioritizing the areas most or least suitable for conservation or agricultural development. Moreover, the results show to what extent ecosystem services are mutually exclusive or can be generated jointly at reasonable amounts. The results also highlight areas with steeply increasing opportunity costs — which might point at local turning points in ecosystem functioning. The method is illustrated for a case study of eighteen Central and Eastern European countries.

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[3] In addition, Cherchye (2001) and Cherchye et al. (2008) use frontier methods to derive aggregate indicators for human development and macro-economic performance. Similar methods are also applied for efficiency analysis in the regulated energy and water markets (see e.g. Thanassoulis, 2000; Zhou et al., 2008; De Witte and Dijkgraaf, 2010) or for eco-efficiency analysis of firms producing desirable and undesirable outputs (see e.g. Färe et al., 2007; Kortelainen and Kuosmanen, 2007).

The remainder of this paper is set up as follows. In the next section, the theoretical framework is discussed, while Section 3 presents our empirical approach. In section 4, the data used is discussed. The results of the analysis are presented in Section 5. Finally, Section 6 discusses the method and results and concludes.

2. Theoretical Framework

The focus in this study is with the trade-offs between biodiversity and ecosystem services due to land use changes. Let conventional marketed outputs be denoted by y and non-marketed outputs by q. Together these two outputs cover the different land use dependent ecosystem goods and services that can be distinguished in a given area. Variable y includes provisioning services and marketed cultural services (e.g. tourism). Variable q is defined as non-marketed cultural services and the regulating and supporting services which maintain benefits in the longer term (e.g. carbon sequestration). Also biodiversity is included in q as a proxy for several intermediate services necessary for maintaining services like nutrient cycling, water purification and pest control. The effect of land use choices on marketed and non-marketed outputs is dependent upon a number of factors exogenous to the decision makers, such as geographical location and soil type. These are covered by conditional variable z.

To properly evaluate the trade-offs between the different ecosystem services and biodiversity, production theory provides useful insights. It gives the integrated framework that is needed for an assessment of the relationship and the non-separability of the market activities, the non-priced ecosystem services and the underlying supporting services represented by y and q. Production theory states that providing multiple interacting services is ‘costly’ in terms that producing one service may take away scarce resources that could potentially be used for producing other services. These costs, given in terms of foregone production of other services, depend on the interactions and synergies between different system elements. For this, information is required on the production structure involved. Besides, the biological and physical processes in this structure should be consistent with insights that ecological systems may be characterized by non-linear and non-convex relationships (Swinton and Wossink, 2007; Chavas, 2009; Brown et al., 2011).

Within ecosystems, there is a wide range of possible combinations of ecosystem services; joint products y and q do not necessarily have to be produced in fixed proportions. Rather, this arises from land use choices made by humans – which crops to grow, cultivate large acreages or keep a landscape with scattered agricultural plots, deforest an area or keep it covered with trees? Moreover, due to differences in z, these choices will differ for different areas within a larger region. Feasible joint outputs depend to a large extent on biophysical characteristics affecting growth potentials. This feasible range can be derived from the transformation function $F(y,q|x,z)=0$. The transformation function describes how in a specific area outputs y and q are jointly produced using inputs x (including land) in a given environment described by the vector z. The slope of the transformation curve F_{q^*}/F_{y^*} at a given point (y^*,q^*) gives the marginal rate of transformation. This reflects the trade-offs or opportunity costs, the foregone output of y due to a marginal increase of q at (y^*,q^*).[4] It shows whether such a change entails high or low costs or whether coordinated management of a bundle of services in a certain area is better than specializing in one of them.

According to production theory, a producer selects the inputs that generate the largest net profits. If prices p_y and p_q were known, maximum profits would be reached at the point where the uni-profit line, $\pi = p_y y + p_q q$ impinges on the transformation function $F(\cdot)$. At that point, the marginal rate of transformation, the slope of the transformation function, equals the price ratio:

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[4] In the literature the terms opportunity costs and shadow prices are often used interchangeably. This may cause confusion, however. Where the definition of opportunity costs is clear (referring to the output foregone due to an increase in one of the other outputs) shadow prices may have different connotations. In micro-economics it often refers to the marginal utility of the output for society, whereas in constrained optimization it refers to the increase of the objective function due to a marginal relaxation of a constraint. In order to avoid confusion, we rather use opportunity costs instead of shadow prices when referring to trade-offs.

$$\frac{p_q}{p_y} = \frac{F_q}{F_y} \tag{1}$$

In the case of ecosystem services, the price of the marketed outputs p_y may be known. The price of the non-marketed outputs p_q is not known, however. Observing the output levels in a region, however, yields the implicit producer price ascribed to the non-marketed outputs at that point. At output level (y^*,q^*), this implicit producer price, given in monetary terms, will be equal to

$$p_q = p_y \frac{F_q\left(y^*,q^*\right)}{F_y\left(y^*,q^*\right)} \tag{2}$$

Note that due to market imperfections and public goods characteristics, it is unlikely that observed output levels (y^*,q^*) also reflect the social optimum. Thus land use change at the regional level by means of policy interventions will be called for. Given transformation function $F(\cdot)$ and the resulting opportunity costs, it can be evaluated which areas have *comparative advantages* in producing more of any of the outputs. An area *a* has a comparative advantage in producing output *q* over area *b* if it can produce this output at a lower opportunity cost. If for example the regional objective is to improve biodiversity, the areas with a low opportunity costs for biodiversity have a comparative advantage. Properly targeting the most suitable areas for particular land uses will result in considerable savings for society.

The transformation function is commonly assumed to be quasi-concave. For bio-economic interactions, however, it is now understood that feedback effects from natural systems, *q,* into social systems, *y,* may result in non-convexities that have not been fully appreciated (Brown et al., 2011). In terms of the model above this means that marginal products from reallocation of *x* may be positive but non-decreasing due to the indirect effects through the joint output *q*. For locations on the transformation function where this applies the gradient F_{yq} would be positive. In this paper, quasi-concavity of the transformation function will be tested for by checking whether the bordered Hessian, the matrix of second partial derivatives of the Lagrangian function of (1), is semi-definite.

3. Empirical Approach

3.1 Robust Conditional FDH Model

To derive opportunity costs, the transformation function has to be known. We adapt the two-stage procedure as set up by Florens and Simar (2005) to estimate the frontier of feasible combinations of q and y at the given level of z-variables based on spatially explicit data on ecosystem services. Where Florens and Simar (2005) focus on production frontier estimation with one output only, we consider ecosystem service production with several outputs (see also Daraio and Simar, 2007a, for the extension to multi-output case), Moreover, in contrast to other applications of the Florens-Simar method, we account for environmental variables in the first stage using the so-called conditional efficiency approach.

The proposed method is implemented as follows. In the first stage, based on a nonparametric frontier estimator, for each observation the distance to the frontier of observations is determined. For this, the output-oriented, robust, conditional Free Disposal Hull (FDH) method, as developed by Cazals et al. (2002) and Daraio and Simar (2005; 2007a; 2007b), is adopted (see also De Witte and Geys, 2011) which is an extension of the robust FDH-method employed by Florens and Simar (2005). FDH requires no prior assumptions about the convexity of the frontier in contrast to the popular Data Envelopment Analysis (DEA) method. This is a major advantage in our application because prior convexity assumptions in situations characterized by non-convex frontiers may lead to flawed conclusions. 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 (Cazals et al., 2002; De Witte and Kortelainen, 2009). Moreover, the *conditional* FDH approach assures that only observations having similar characteristics are compared with each other (Daraio and Simar, 2005). This extension is important for the type of data adopted in the current paper, as we assess opportunity costs for a large number of regions many of which have totally different characteristics with different output potentials. In our empirical analysis, we use the *output orientated* FDH model to reflect that authorities can only partially influence land use decisions (the inputs into the model) by policies aiming at changing the amount of biodiversity and ecosystem services (the outputs).

For the second stage, Florens and Simar (2005) propose to approximate the nonparametric frontier obtained in the first stage with a parametric function such that unique opportunity costs can be derived. For this, we adopt the translog function, especially for its flexibility. The advantage of this two-stage procedure 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, in contrast to standard parametric methods, this two-stage method can avoid critical homoskedasticity and distributional assumptions.

We now more formally discuss the suggested two-stage approach and the estimation methods. For the first stage, introduce for each region a vector of outputs $y = (y^1, \dots ,y^M)$ (which cover vectors y and q introduced above), inputs $x = (x^1,\dots,x^N)$ which cover the land use choices, and conditional variables $z = (z^1,\dots,z^K)$ which are beyond the control of the decision makers. The feasible output set is defined as

$$\Psi = \left\{ (x, y, z) \middle| x \text{ can produce } y \text{ given characteristics } z \right\}.$$

In empirical studies, Ψ should be estimated from a random sample of L observations $\left\{ (x_l, y_l, z_l) \middle| l = 1,\dots, L \right\}$. The Free Disposal Hull (FDH) estimator for the production possibility set Ψ is (with bandwidth parameter h):

$$\Psi^{FDH}\left(x,y,z\right)=\left\{\left(x,y,z\right)\in \mathbb{R}_{+}^{M+N+K} \mid y\leq y_{l},x\geq x_{l},z_{l}\in\left[z-h,z+h\right]\exists l=1,\ldots,L\right\} \qquad (3)$$

The FDH frontier is a stairway-shaped curve connecting the Pareto optimal observations. In Figure 1 line AB represents the FDH-frontier and region OAB the set of feasible outputs.

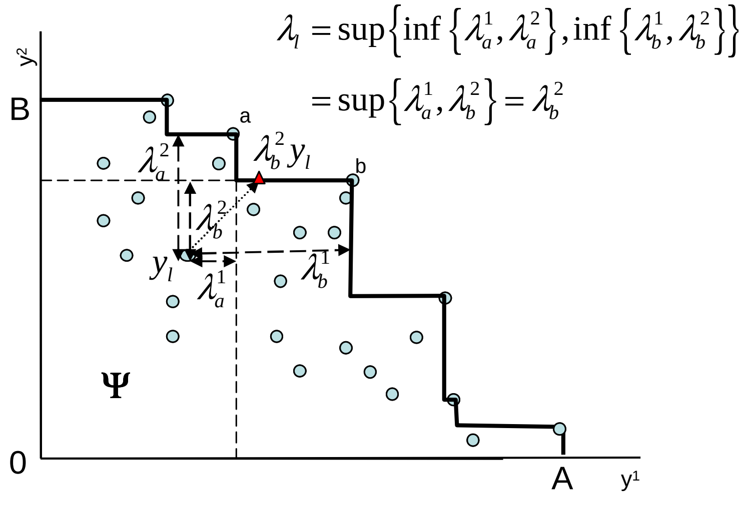

Figure 1: Representation of stage 1 showing the feasible output set Ψ and distance to the frontier of observation y_l which is equal to λ_l for an example with two outputs.

Note: Observation y_l can improve at most with λ^1_a before it reaches the frontier of the Pareto optimal observation *a* and with λ^2_b before it reaches the frontier of the Pareto optimal observation *b*. So, the observation can improve with $\max\left\{\lambda_a^1,\lambda_b^2\right\}$ before it reaches frontier AB. If $y_l=(y^1, y^2)$ improves with λ^2_b, the observation would move from y_l to $\lambda^2_b y_l$, which is located at the frontier.

Under the assumption of free disposability (see e.g. Färe and Grosskopf, 2000, for an explanation of the assumptions), for each observation (*x,y,z*), the Farrell-Debreu distance function can be defined as:

$$\lambda\left(x,y\mid z\right)=\sup\left\{\lambda\mid(x,\lambda y,z)\in\Psi\right\} \qquad (4)$$

This function measures for each observation the distance of the output vector to the frontier – see also Figure 1. For the Pareto optimal observations $\lambda = 1$. For the other points $\lambda > 1$, where $(\lambda\text{-}1)$*100% measures the percentage increase of each output necessary to reach the frontier.

For estimating (4), first consider a situation without conditional variables *z*. In that case, the estimator of (4), written in probabilistic format, is (Daraio and Simar, 2005; De Witte and Kortelainen, 2009):

$$\lambda(x_i, y_i) = \underset{\lambda_i}{Sup}\left\{\lambda_i \left| S_Y\left(\lambda_i y_i \left| x_i\right.\right) > 0\right.\right\} = \underset{\lambda_i}{Sup}\left\{\lambda_i \left| \frac{I\left(\lambda_i y_i \le y_l, x_i \ge x_l\right)}{I\left(x_i \ge x_l\right)} > 0, l = 1, ..., L\right.\right\} \tag{5}$$

for observation (x_i,y_i), with $S_Y\left(y\left|x\right.\right) = \Pr\left(y \le Y, x \ge X\right)/\Pr\left(x \ge X\right)$ the survivor function of Y and $I(\cdot)$ the indicator function. Secondly, to estimate the robust, order-*m* efficiency measure conditional on the *z*-variables, for each observation (x_i,y_i,z_i) a sample of size *m* is drawn with replacement from the original sample $\left\{\left(x_l, y_l, z_l\right)\left| l = 1, ..., L\right.\right\}$ repeatedly for a large number of times after which the expectation is taken. Cazals et al. (2002) showed that the conditional order-*m* efficiency score is (see also Daraio and Simar, 2005)

$$\lambda^m\left(x_i, y_i \left| z_i\right.\right) = \int_0^\infty \left[1 - \left(1 - S_Y\left(u y_i \left| x_i, z_i\right.\right)\right)^m\right] du \tag{6}$$

For estimating the conditional survivor function $S_Y(y|x,z)$ nonparametrically, 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). For this, different from what is given in (5), $S_Y(y|x,z)$ changes into:

$$S_Y\left(y_i \left| x_i, z_i\right.\right) = \frac{\sum_{l=1}^{L} I\left(y_i \le y_l, x_i \ge x_l\right) K_h\left(\left(z_i - z_l\right)/h\right)}{\sum_{l=1}^{L} I\left(x_i \ge x_l\right) K_h\left(\left(z_i - z_l\right)/h\right)} \tag{7}$$

for all $l = 1,\ldots,L$ and with $K_h(.)$ a Kernel function with bandwidth parameter *h*.

3.2 Estimation of Opportunity Costs

In the second stage, following Florens and Simar (2005) and Daraio and Simar (2007a), we approximate the nonparametric frontier function with a flexible parametric production function. As derived below, this frontier function directly follows from the distance function which gives the distance from each observation to the frontier. Let $\delta(x,y|z)$ be the Shephard output distance function, which is equal to the inverse of the Farrell-Debrue distance function introduced in (4) and (6), $\delta(x,y|z) = \lambda^{-1}(x,y|z)$. Introduce a parametric distance function $\varphi(x,y,z;\theta)$, which is homogenous of degree one in y, and with unknown parameters given by vector θ. The aim is to estimate the values of θ which give the best approximation of the multivariate output distance function $\delta(.)$:

$$\theta_0 = \arg\min_{\theta}\left[\sum_{i=1}^{L}\left(\delta\left(x_i, y_i \mid z_i\right) - \varphi\left(x_i, y_i, z_i; \theta\right)\right)^2\right]. \tag{8}$$

Assume a translog production function

$$\ln\varphi\left(y, z; \theta\right) = \alpha_0 + \beta'\ln y + \tfrac{1}{2}\ln y'\Gamma\ln y + \gamma'\ln z \tag{9}$$

where $\Gamma=\Gamma'$ is symmetric (see Daraio and Simar, 2007a). Due to homogeneity of degree one in y, it has to hold that $\beta' \cdot i_M = 1$ and $\Gamma \cdot i_M = 0$, 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 $\tau = \left(\tau_1 \;\; \tau_{-1}'\right) \in \mathbb{R}^M$, τ_{-1} an (M-1)-vector and Γ_{22} an (M-1)x(M-1)-matrix. Due to the homogeneity assumption it follows that $\beta_1 = 1 - \beta_{-1}' \cdot i_{M-1}$, $\tau_1 = -\tau_{-1}' \cdot i_{M-1}$ and $\tau_{-1} = -\Gamma_{22} \cdot i_{M-1}$. For the translog function, (8) equals (with $\delta_i = \delta\left(x_i, y_i \mid z_i\right)$)

$$
\begin{aligned}
\theta_0 &= \arg\min_{\theta}\left[\sum_{i=1}^{L}\left(\ln\delta_i - \left(\alpha_0 + \beta'\ln y_i + \tfrac{1}{2}\ln y_i^{'}\Gamma\ln y_i + \gamma'\ln z_i\right)\right)^2\right] \\
&= \arg\min_{\theta}\left[\sum_{i=1}^{L}\left(-\ln y_{i1}^{*} - \left(\alpha_0 + \beta_{-1}^{'}\ln\tilde{y}_{i,-1} + \tfrac{1}{2}\ln\tilde{y}_{i,-1}^{'}\Gamma_{22}\ln\tilde{y}_{i,-1} + \gamma'\ln z_i\right)\right)^2\right]
\end{aligned}
$$

with $y_{i1}^{*} = y_{i1} / \delta_i$ the values of y_{i1} projected on the frontier and $\tilde{y}_{i,-1} = y_{i,-1} / y_{i1} = y_{i,-1}^{*} / y_{i1}^{*}$.

In words, to estimate the best parametric approximation of the multivariate output distance function, the output values are projected on the output frontier using the distance values estimated in the first stage, after which the frontier function

$$
\ln y_{i1}^{*} = -\left(\alpha_0 + \beta_{-1}^{'}\ln\tilde{y}_{i,-1} + \tfrac{1}{2}\ln\tilde{y}_{i,-1}^{'}\Gamma_{22}\ln\tilde{y}_{i,-1} + \gamma'\ln z_i\right) \qquad (10)
$$

is estimated using OLS. Using the conditions on β and Γ as given above, distance function $\varphi(x,y,z;\theta)$ immediately follows. One of the major advantages of this approach is that no restrictive homoskedasticity or distributional assumptions have to be made for the error term in (8). A disadvantage, because of the first-stage estimation, is that in the second stage standard errors should be obtained using a computationally intensive bootstrapping procedure (see Florens and Simar, 2005).

As a final stage, opportunity costs or trade-offs between the output combinations are derived in physical and monetary terms. The slope of the frontier function (10) represents the marginal rate of transformation. This gives the opportunity costs, the output foregone due to an increase in one of the other outputs. This opportunity cost ratio can be derived using the duality relationship between the benefit function and the distance function (Färe and Grosskopf, 2000; Bellenger and Herlihy, 2010). For output price $p \in \mathbb{R}^M$, the benefit function is defined as $B(p) = \sup_{y}\{p'y \mid (x,y,z) \in \Psi\}$. As $y/\delta(x,y \mid z)$ is a feasible output vector, it has to hold that $B(p) \geq p'y/\delta(x,y \mid z)$ and so $\delta(x,y \mid z) = \max_{p}\left[p'y/B(p)\right]$. As a result, for each $m = 1,\ldots,M$

$$\frac{\partial\delta(x,y\,|\,z)}{\partial y_m}=\frac{p_m}{B(p)} \tag{11}$$

If the market price is known for one of the outputs, e.g. for the first output, from (11) opportunity costs for the other outputs can be derived in monetary terms. This reflects the slope of the production possibility frontier, i.e. the marginal rate of transformation. For the translog distance function, opportunity costs are

$$p_m=p_1\cdot\frac{\partial\delta(x,y\,|\,z)/\partial y_m}{\partial\delta(x,y\,|\,z)/\partial y_1}=p_1\frac{y_1}{y_m}\left(\frac{\beta_m+\Gamma_m^{'}\ln y}{\beta_1+\Gamma_1^{'}\ln y}\right) \tag{12}$$

with Γ_m the mth row of vector Γ.

For example if y_1 is defined as agricultural production and p_1 its market price, this second stage gives opportunity costs of the non-monetary outputs in terms of revenues from provisioning services foregone. These opportunity costs show in a positive (not a normative) way the trade-offs between monetary and non-monetary outputs. They serve as an input into the decision making process in which it has to be decided whether society is willing to make this trade-off. They, therefore, differ from the values estimated using environmental valuation methods. Different from valuation, they show the effects of a land use change in income equivalents without explicit reference to the (largely unknown) trade-offs households are willing to make for these changes.

3.3 Estimation of Morishima elasticities

The effects of marginal changes in the output variables on the opportunity costs can be investigated using the indirect Morishima Elasticity of Transformation (MET), which provides a measure for the curvature of the frontier (Blackorby and Russell, 1989; Mundra and Russell, 2010; Bostian and Herlihy, 2012). The MET is defined as the percentage change in the opportunity cost ratio due to a percentage change in the output ratio (Mundra and Russell, 2010; Färe et al., 2012). It is a so-called two-price, one output elasticity in which only one of the outputs in the output ratio changes (Frondel, 2011; Stern, 2011). As a result, the elasticity is asymmetric, depending on which output changes. The MET, for a change in output y_i is defined as:

$$MET_{ij} = \frac{\partial \ln\left(p_i / p_j\right)}{\partial \ln\left(y_j / y_i\right)} = \frac{\partial \ln\left(p_j\right)}{\partial \ln\left(y_i\right)} - \frac{\partial \ln\left(p_i\right)}{\partial \ln\left(y_i\right)}. \tag{13}$$

It follows that the change in the opportunity cost ratio depends on two quantity elasticities. A negative MET_{ij} implies that decreasing the quantity of *i* increases the opportunity cost of output *i* relative to that of output *j*, or the more negative MET_{ij}, the more costly it is to increase y_j. In that case, output *j* is a Morishima substitute to output *i*. Similarly, if $MET_{ij}>0$, output *j* is a dual Morishima complement to output *i*. For positive elasticities, it holds that the larger MET_{ij}, the less costly it is to increase y_i. Given (12),

$$M_{i1} = 1 - \frac{\tau_{ii}}{\beta_i + \Gamma_i \ln y} + \frac{\tau_{1i}}{\beta_1 + \Gamma_1 \ln y}$$

4. Data

The approach discussed above is illustrated for a case study of eighteen Central and Eastern European countries: Albania, Belarus, Bosnia and Herzegovina, Bulgaria, Croatia, Czech Republic, Estonia, Hungary, Latvia, Lithuania, Macedonia, Moldova, Poland, Romania, Serbia, Slovakia, Slovenia and Ukraine. In order to illustrate the method only a limited number of variables is included, but this can be easily extended in later applications. To differentiate ecosystem services between input and output variables, we distinguish between intermediary services (especially regulating and supporting services) and final services (especially provisioning and cultural services) where the intermediary services are the inputs or processes necessary for producing the goods and services providing human benefits (see e.g. Boyd and Banzhaf, 2007; Brown et al., 2007; de Groot et al., 2010; Haines-Young and Potschin, 2010). In the current analysis, land is the only input variable included. This is a limitation: it is not just intermediary ecosystem services but also human induced inputs that contribute to the provision of final ecosystem services. Because of a lack of data on land use intensity or other human inputs, it was decided to include land as the only input variable.

Data are given on the level of grid cells of size 50x50 km^2 for the year 2000. Land use patterns (percentage of area per grid cell) were derived from the GLC2000 land use map (EC-JRC, 2003). This results in 1166 observations. The following variables are included:

Output variables

1. Provisioning services: Agricultural revenues (in 2000 international $/km2). For each cell total revenues for the production of cereals, grass, maize, pulses, roots, tubers, and oil crops are calculated based on land use data from the GLC2000 map, the cropping pattern, cropping intensities and potential yields from the IMAGE modeling system (Bouwman et al., 2006), and prices from FAOstat.[5] Aggregate production per crop is based on FAO-data, which is allocated over the cells using the IMAGE modeling system and the GLC2000 map.
2. Cultural services: a composite index consisting of attractiveness for tourism and recreation and for hunting and gathering activities (Schulp et al., 2012). Tourist and recreation attractiveness is an index ranging from 0 (unattractive) to 1 (attractive) and depends on GDP per capita, percentage protected area, percentage urban and arable land, distance to coast and geographic relief. Potential for gathering and hunting is based on statistics from FAO and the European Forestry Institute.
3. Biodiversity: Biodiversity is treated as an output variable, serving as a proxy for the positive impact of several regulating and supporting services on ecosystems and as an indicator which is important in nature policies. We measure biodiversity as *mean species abundance*

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[5] 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.

(MSA), i.e. the current mean abundance of species compared to their abundance in an undisturbed, pristine environment as calculated by global biodiversity model GLOBIO (see Alkemade et al., 2009). MSA is contingent upon land cover, habitat, percentage of the cell covered with certain vegetation, land use intensity, and distance to roads and cities.

4. Carbon sequestration: Carbon sequestration is used as a proxy variable for climate regulation. It is measured as net biome productivity in tonnes C per km^2 which is calculated as net primary production of carbon minus soil respiration minus the carbon sequestered in the biomass harvested. For respiration and sequestration factors long-term averages are taken. Data are based on the GLC2000 land use map and EURURALIS carbon model (see e.g. Schulp et al., 2008).

Conditional variables

5. GDP PPP per km2 for the year 2000 (in international $/km2): GDP levels per grid cell are based on World Bank data on GDP per country, agricultural shares in GDP and on urban and rural population per cell from the IMAGE modeling system. National GDP is allocated over the grid cells by considering differences in income for the rural and urban population.
6. Share of agricultural 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 temperate zone cereals in tonnes/ha based on climate, soil and slope characteristics. Temperate zone cereals (wheat, rye, corn, barley, oats) are chosen as this is the main crop grown (around 60% of the cropland is covered with temperate zone cereals).
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 of the Commonwealth of Independent States (CIS) Belarus, Estonia, Latvia, Lithuania, Moldova and Ukraine, 2.

Central European countries (CE) Czech Republic, Hungary, Poland and Slovakia, 3. the republics that formerly constituted Yugoslavia (YUG) Bosnia, Croatia, Macedonia, Serbia and Slovenia, 4. the south-eastern European countries (SE) Albania, Bulgaria and Romania (see, Fenger, 2007).

For the analysis, model data are used because no field observations exist for the ecosystem services and biodiversity variables. The results from IMAGE and GLOBIO, however, give the state-of-the-art knowledge of the relationship between global land use decisions, a number of ecosystem and environmental indicators and biodiversity. They are regularly used in global integrated assessments (see e.g. Nelleman et al., 2009; Van Vuuren and Faber, 2009; TEEB, 2010; OECD, 2012; PBL, 2012; UNEP, 2012).

Maps of the base data are shown in the Appendix. Table 1 and Table 2 give some descriptive statistics. The large standard deviation for some variables reflect differences in population density (urban vs. rural areas) and differences between average country development levels. Signs of correlation coefficients are as expected and related to land cover and land use. Agricultural production is higher in cells with higher percentages of agricultural land. MSA, cultural services and carbon sequestration levels are higher on areas with less cultivated plots and therefore prevailing in cells with lower agricultural production. Levels of cultural services are higher in areas with higher MSA levels as these areas are more attractive for recreation and hunting. These are, generally, also the areas sequestering more carbon. The relatively high correlations between some of the ecosystem services corresponds with similar observations by Raussep-Hearne et al. (2010) for Canada.

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

		Prov. services US$/km^2	MSA	Cultural services	Carbon sequest. Tonnes C/km^2	GDP US$/km^2	Pot. Yield ton/ha	% agric. + grass- land
Total	**Mean**	15,674	0.359	0.408	29.15	491,823	481	0.59
	St.Dev.	(11,394)	(0.128)	(0.102)	(29.14)	(774,723)	(109)	(0.21)
	Min - Max	0 - 83,928	0.133 - 0.929	0.144 - 0.900	-10 - 165	0 - 7,881,679	139 - 738	0.00 - 1.00
CIS	**Mean**	14,419	0.358	0.402	31.96	209,673	516	0.66
	St.Dev.	(11,089)	(0.130)	(0.093)	(26.10)	(372,633)	(108)	(0.22)
CE	**Mean**	14,982	0.349	0.457	20.21	1,082,414	431	0.56
	St.Dev.	(9,090)	(0.114)	(0.086)	(17.15)	(1,132,678)	(78)	(0.16)
YUG	**Mean**	14,677	0.365	0.358	16.43	629,314	384	0.50
	St.Dev.	(11,249)	(0.121)	(0.140)	(14.70)	(669,721)	(114)	(0.17)
SE	**Mean**	21,393	0.372	0.387	43.16	352,835	516	0.51
	St.Dev.	(13,826)	(0.147)	(0.089)	(47.37)	(503,458)	(80)	(0.22)

Table 2: Correlation coefficients of the variables included in the analysis

	1. Provisioning services	2. Mean species abundance	3. Cultural services	4. Carbon sequestration	5. GDP	6. Potential yield	7. %agric. +grass land
1. PS	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. Results

The two-stage approach discussed in Section 3 is illustrated for the data discussed above. We are particularly interested in the following questions: What are the opportunity costs of marginal changes in the different output variables, to what extent do they depend on regional characteristics and what is their rate of change? This information will guide decision makers in targeting the regions having a comparative advantage for conservation or for agricultural development and for judging the effects of decisions on land use changes. It also shows sub-regions characterized by steep increases in opportunity costs which signals towards upcoming turning points in ecosystems functioning.

The estimation results, discussed in detail below, can be summarized in the following two main conclusions:

- The production possibility frontier, showing the Pareto optimal output combinations, is non-concave. This has implications for the interpretation of the opportunity costs.
- Regional differences in trade-offs are large. Each country has regions that have a comparative advantage for a particular ecosystem service. For provisioning services, opportunity costs generally increase if levels increase. In contrast, enhancing carbon sequestration levels becomes cheaper in regions with higher carbon sequestration levels, whereas, in general, promoting biodiversity is found to become more expensive with higher levels biodiversity.

5.1 Shape of production possibility frontier

We first test concavity of the production possibility frontier by checking for quasi-convexity of the output distance function. Few applied studies test for convexity of the distance function (O'Donnell and Coelli, 2005). Most studies estimating opportunity costs (or shadow prices) with frontier methods simply impose it by the particular choice of the functional form or by adding convexity constraints (see e.g. Färe et al., 2005; Bellenger and Herlihy, 2010; Bostian and Herlihy, 2012). In this way, the production possibility frontier is nicely concave, but one can wonder to what extent the resulting curvature and opportunity costs still reflect reality. Curvature violations have consequences for the interpretation of the opportunity costs because the duality assumption between the distance function and benefit function no longer holds. For an observation on a concave frontier, opportunity cost p_m in (11) is a benefit maximizing opportunity cost. For a downward sloping but convex frontier, however, this does not apply. In all cases, however, the opportunity cost ratio (12) still reflects the trade-off between y_m and y_1 in the neighborhood of the observation *y*.

To test for concavity of the production possibility frontier, first, function (10), is estimated. The coefficients of the Shephard output distance function (9) and frontier function (10) are listed in Table 3. The first-order derivatives of the distance function are positive. So, the distance to the frontier reduces if output levels increase. Similarly, at the frontier, higher levels of biodiversity, cultural services or carbon sequestration result in lower levels of provisioning services showing that there is a trade-off between the different outputs.

Table 3: Parameter estimates of translog functions (9) and (10) $^{(1)}$

Coeff.$^{(2)}$		**Estimate**	**95% Confidence Interval$^{(3)}$**
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$^{(4)}$	-0.040	(0.012-0.098)
γ_{43}	Sub-region = YUG$^{(4)}$	0.133	(0.032-0.133) *
γ_{44}	Sub-region = SE$^{(4)}$	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	

Notes: (1) As in many empirical applications, each of the continuous variables is divided by their respective sample means such that each has a mean equal to one. Moreover, non-monotonous observations are removed from the sample. (2) $\beta_1 = 1 - \beta_2 - \beta_3 - \beta_4$, $\Gamma_{1i} + \Gamma_{2i} + \Gamma_{3i} + \Gamma_{4i} = 0$ for all i=1,2,3,4; (3) Variables marked with a * are significant at the 95% level. Confidence intervals are based on bootstrapping procedure with 200 runs. (4) 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.

The eigenvalues of the Hessian show that the distance function is not quasi-convex and so that the frontier is non-concave. All observations have both positive and negative eigenvalues implying that the frontier has a saddle point. This result is robust, as the same result is also obtained for different formulations of the model. As an illustration, Figure 2 plots the frontier in 3-dimensional plots for the sub-region SE, each time fixing one of the output variables at its mean value. As these plots in fact are extrapolations, they should be interpreted with care, especially at the boundaries and the areas with only few observations. These plots show violations of the curvature assumptions especially for carbon.

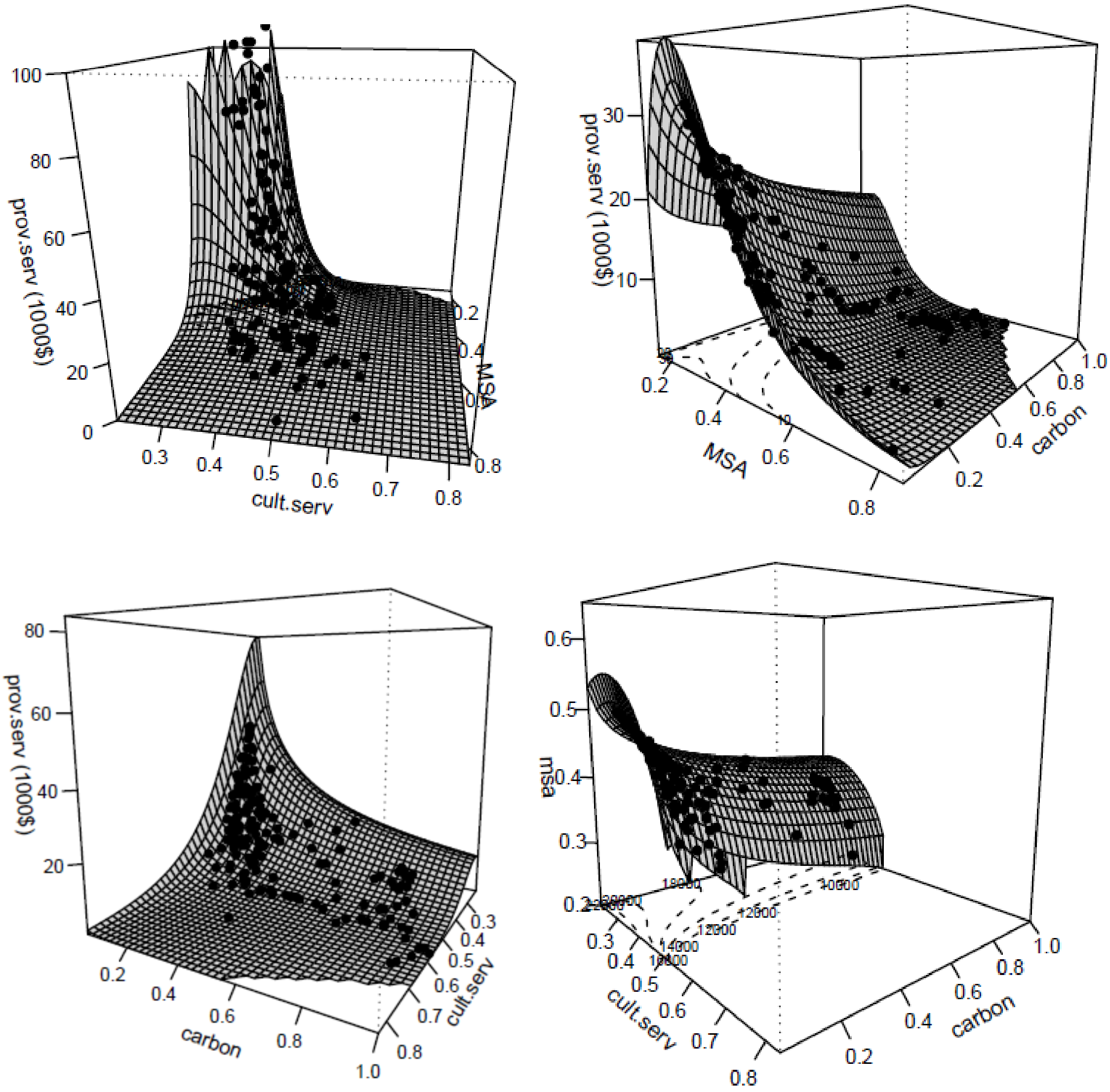

Figure 2: 3-dimensional plots of the frontier for the sub-region SE.

Note: The plots for the other three sub-regions are similar. Contour plots are given on the x-y plane. 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. For the range of values given by the x-y coordinates, the corresponding level of z-values is determined using (10).

The violation of the convexity axiom is inconvenient from an economic point of view. Dasgupta and Maler (2003), Brown et al. (2011) and Tschirhart (2011), however, argue that such non-concavities apply more often. In fact, Dasgupta and Mahler (2003) argue that "the word “convexity” is ubiquitous in economics, but absent from ecology. ... We now know that the price system can be an efficient allocation mechanism if transformation possibilities among goods and services … constitute a convex set. However, in non-convex environments, we still do not have a clear understanding of the mechanisms by which resources are allocated …. So we economists continue to rely on the convexity assumption, always hoping that it is not an embarrassing simplification. ... Ecologists have no comparable need to explore the structure of convex sets [as they don’t optimize anything]. They are interested in identifying pathways by which the constituents of ecosystem interact with one another and with the external environment. A large body of empirical work has revealed that those pathways in many cases involve transformation possibilities among environmental goods and services that, together, constitute non-convex sets" (p.499)". For example, a meta-analysis of data on land use change and carbon sequestration indicate that conversion from grassland to financially more attractive cropland results in a significant decline of soil carbon stocks and vice versa (Guo and Gifford, 2002). Similarly, Boscolo and Vincent (2003) conclude that in forestry the joint production of timber and non-timber products may have a non-convex relationship. As a result, output specialization gives higher net returns than output diversification. These non-convexities may reflect spillovers, positive externalities, species interactions and feedback effects from natural systems into social systems.

Another interesting observation from Figure 2 is that for sufficiently low levels of biodiversity it seems to be possible to improve biodiversity and provisioning services at the same time. We have to be careful drawing strong conclusions here because it is based on only a few observations. Such a complementary relationship, however, is plausible as for biodiverse poor areas increasing biodiversity also supports biotic processes beneficial to agricultural production.

The lower agricultural production due to a loss in agricultural land can be compensated by the positive externalities of higher biodiversity. Once biodiversity exceeds a certain level, the positive externalities cannot compensate for the loss in land. Likewise too high levels of biodiversity may also harm agricultural productivity (Swinton and Wossink, 2007).

5.2 Opportunity costs and Morishima elasticities

The second result concentrates on opportunity costs. For each observation, opportunity cost ratios or marginal rates of transformation are estimated using (12) and the coefficients given in Table 3.[6] They reflect the gross benefits from provisioning services foregone due to a marginal increase in any of the other output variables. They are presented in Table 4, Table 5, and Figure 3.[7] The effects of small output changes on the level of the opportunity costs are exemplified using the Morishima Elasticity of Transformation (MET) which provides a measure for the curvature of the frontier– see (13). Estimates of the MET are given in Table 6.

Due to their particular interpretation in this analysis, it is difficult to directly compare the estimates for cultural services and for mean species abundance with other studies. The estimates for carbon sequestration however can be compared with those in other studies. Antle et al. (2003) construct a marginal opportunity cost curve for sequestering carbon for five cropping system in the US. The resulting marginal costs range from \$20 to \$100 per ton carbon. Similarly, MacLeod et al. (2010) estimate for the UK a carbon abatement cost curve for agricultural emissions from crops and soils. They argue that 11.5% of emissions from agriculture can be abated at a marginal abatement cost of £168 ≈ \$261 per ton carbon sequestered. At higher levels, marginal abatement costs increase fast. In the latter studies, the opportunity cost

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[6] Note that the opportunity costs are estimated for each observation. For the observations not on the frontier, these values reflect the trade-offs for the situation in which they moved towards the frontier. The distance to the frontier depends among other things on the variables not included in the analysis which partly explain why data points for a cell are not at the frontier.

[7] The results are stable. A Monte Carlo simulation has been carried out in which the model is solved 100 times, with provisioning services in each cell drawn from a normal distribution with mean equal to the original value and standard deviation equal to 10% of this mean value. Results show that the distribution of the resulting distance measures (4) has a standard deviation equal to only 1.6% of the mean distance value. The distribution of the resulting opportunity costs has, for each variable, a standard deviation of 11% of the mean opportunity cost.

estimates are given in terms of net farm benefits and not in terms of gross benefits, as in our case. For transforming gross to net benefits, the social profit rate should be used, which compares the value added of an economic activity to the value of gross output of this activity at world prices. Hughes and Hare (1994) provide an estimate of the average medium run social profitability of agriculture for Eastern Europe of 7.25%. Using this rate, our average opportunity cost of $263 of gross benefits lost due to an extra ton of carbon sequestered implies a loss of net benefits of $19 per ton of carbon sequestered, which is quite low compared to the above studies.

Table 4: Mean opportunity cost ratios

	Mean	Median	Standard deviation	Min	Max
MSA ($ per % MSA index)	1,276	1,027	1,029	1	8,846
Cult. Serv. ($ per % cult.serv. index)	1,865	1,368	1,706	0.3	12,587
Carbon ($ per ton C)	263	202	220	1.4	1,986

Table 5: Median opportunity cost ratios and standard deviations per country

	MSA ($ per % MSA)		Cultural services ($ per % cult.serv index)		Carbon ($ per tonne carbon)	
	Median	St.Dev.	Median	St.Dev.	Median	St.Dev.
Total	1,027	1,029	1,368	1,706	202	220
Belarus	1,008	597	666	503	125	55
Estonia	247	310	233	339	62	65
Latvia	667	482	699	587	139	89
Lithuania	702	254	1,234	510	211	78
Moldova	1,534	707	2,633	1,664	490	151
Ukraine	1,184	944	2,323	1,878	220	190
Czech	363	486	1,103	1,149	246	170
Hungary	1,308	543	2,286	1,741	342	232
Poland	792	615	1,329	1,110	224	183
Slovakia	521	831	560	1,364	110	97
Bosnia	1,904	2,344	694	2,031	284	134
Croatia	870	772	934	1,414	222	108
Macedonia	1,817	770	1,412	1,243	1,169	419
Serbia	1,206	833	1,159	2,043	374	151
Slovenia	111	312	314	539	66	67
Albania	1,208	1,046	1,577	1,204	315	260
Bulgaria	1,635	779	1,947	960	347	181
Romania	2,064	1,439	2,464	2,440	155	294

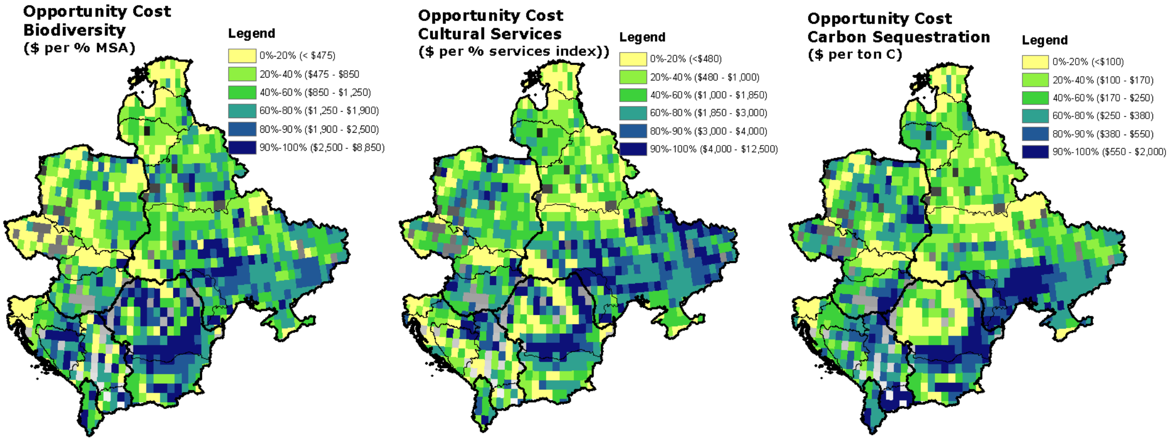

Figure 3: Maps of opportunity costs per cell for a) mean species abundance ($ per % MSA), b) cultural services ($ per % cultural services index) and c) carbon sequestration ($ per tonne carbon).

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.

As shown in Figure 3, opportunity costs differ substantially between the different regions but also within-country differences are large. Further analysis of the opportunity cost function shows that carbon opportunity costs increase with decreasing levels of carbon, biodiversity and cultural services and with increasing levels of provisioning services. Moreover, as shown by the positive but decreasing Morishima elasticities, it becomes cheaper to increase carbon sequestration when sequestration levels increase, but at a decreasing rate. For low carbon levels, an extra unit of carbon stored will result in more provisioning services foregone than when carbon levels are higher. If sequestration is higher, a larger part of the cell will be covered with forest and sequestering marginally more will not demand a huge sacrifice in terms of provisioning services. These cells have a comparative advantage in sequestering more carbon. They are in a situation with increasing returns to scale and thus it is more cost-effective to sequester more carbon in areas already having high levels of carbon sequestration and thus less interesting for agricultural production. It also implies that it may be cost-effective to have a certain level of specialization per cell, with those cells having a comparative advantage in sequestration focusing on carbon instead of attempting to improve all services simultaneously.

A more mixed picture emerges for the opportunity costs for biodiversity and cultural services. For both these outputs, opportunity costs increase with higher levels of provisioning services. Cells already having high levels of agricultural production have a comparative disadvantage in providing further biodiversity and cultural services. The relationship between biodiversity levels and their opportunity costs, however, is more complex. Regions having a comparative advantage in providing biodiversity may be either biodiversity-poor or biodiversity-rich regions. The same applies for cultural services. Moreover, the Morishima elasticities are negative for most observations. It turns out that the relationship between provisioning services and biodiversity is concave at first but becomes convex after a certain threshold level of biodiversity. The level at which this point is observed depends on cell characteristics including levels of provisioning services, carbon sequestration and land cover. The higher the levels of provisioning services, the higher the opportunity costs for biodiversity and the higher the rate at which these opportunity costs increase with rising biodiversity levels (very negative Morishima elasticities). However, after a certain threshold level, which is cell specific, opportunity costs start to decrease again. For cultural services, more or less the same pattern is observed as for biodiversity.

On a related point, cells having higher carbon sequestration levels have a comparative advantage in improving sequestration, whereas for biodiversity it are the cells with high and with low levels that have a comparative advantage. Increased carbon sequestration can be achieved at low (opportunity) costs, but at the same time increasing biodiversity is more difficult/costly. To increase sequestration, transforming agricultural land to extensive grassland or monoculture forests may be sufficient. Improving biodiversity demands more effort. Only once a certain biodiversity level is realized (which is already rather high), further increasing biodiversity becomes easier/less costly. In those cases external pressures are low and ecosystem processes are complementary instead of competitive.

Table 6: Morishima elasticity estimates

	mean species abundance			cultural services			Carbon		
	mean	**median**	**st.dev.**	**mean**	**median**	**st.dev.**	**mean**	**median**	**st.dev.**
Total	-6.50	-0.80	144.73	-2.12	-0.40	24.09	0.87	0.87	0.04
Belarus	-0.15	-0.11	0.36	-4.60	-1.24	21.58	0.84	0.84	0.02
Estonia	-0.67	-0.52	0.74	-1.09	-0.65	1.91	0.83	0.83	0.02
Latvia	-0.89	-0.56	1.35	-19.12	-0.57	116.76	0.84	0.85	0.04
Lithuania	-2.50	-1.42	4.37	-0.31	-0.21	0.38	0.87	0.87	0.01
Moldova	-1.68	-1.84	0.87	-0.34	-0.11	0.60	0.90	0.90	0.02
Ukraine	-2.15	-1.27	4.00	-0.89	-0.21	7.45	0.87	0.88	0.05
Czech	-5.37	-2.48	11.38	-0.16	-0.09	0.26	0.88	0.88	0.02
Hungary	-9.15	-1.21	30.23	-0.37	-0.25	0.44	0.89	0.89	0.03
Poland	-2.69	-1.22	5.93	-0.42	-0.26	0.53	0.87	0.87	0.03
Slovakia	-183.80	-0.81	931.40	-0.68	-0.44	0.82	0.84	0.84	0.04
Bosnia	-0.04	0.17	0.60	-5.03	-3.75	4.28	0.87	0.87	0.02
Croatia	-2.02	-0.67	5.02	-1.46	-0.50	3.37	0.87	0.86	0.03
Macedonia	-0.56	-0.23	0.74	-2.18	-1.28	2.37	0.91	0.90	0.01
Serbia	-4.92	-0.34	11.59	-1.66	-0.85	1.96	0.89	0.88	0.04
Slovenia	-3.12	-2.14	2.86	-0.25	-0.12	0.50	0.83	0.83	0.02
Albania	-1.67	-0.70	1.97	-0.90	-0.49	1.54	0.88	0.88	0.02
Bulgaria	-0.61	-0.37	0.88	-0.94	-0.78	0.74	0.87	0.88	0.02
Romania	-1.04	-0.58	2.34	-2.84	-0.51	10.00	0.86	0.87	0.07

To conclude, the maps in Figure 3 show which regions have comparative advantages (low opportunity costs) or comparative disadvantages (high opportunity costs) in the provision of biodiversity, cultural services or carbon sequestration. There is considerable variation within each country, but each country has regions in which increasing these outputs is more cost-effective. Moreover, especially in regions characterized by high negative Morishima elasticities, further increasing the ouputs becomes more expensive very fast. Thus if these regions already have high opportunity costs, these regions are least suitable for further land use changes. More interesting from a cost-effectiveness perspective are regions with lower opportunity costs and low negative or positive Morishima elasticities. Moreover, regions characterized by high and steeply increasing costs may be nearing turning points in ecosystem functioning. So, these maps are helpful for prioritizing conservation policies.

6. Discussion and Conclusion

The principal aim of this paper was to present a method that is capable of providing monetary estimates of opportunity costs of ecosystem services, their dependence on regional characteristics and the comparative advantages areas have for producing particular ecosystem services. The method is an extension of the two-stage procedure proposed by Florens and Simar (2005) for estimating production possibility frontiers from which opportunity costs can be derived. Advantages of the proposed frontier approach are that no assumptions have to be made on the convexity of the frontier and the distribution of the error term and that the approach is flexible in the inputs, outputs and conditional variables included.

The approach presented here adds to the growing literature on integrated assessments and on mapping the effects of changes in ecosystem services; what are the opportunity costs of changes in ecosystem services, to what extent do they differ per region and which regions have a comparative advantage in producing particular ecosystem services? These insights are helpful for prioritizing the regions requiring attention and for searching cost-effective policies. The paper also adds to the growing literature the empirical insight that trade-offs depend distinctly on the spatial variation in biophysical interactions between biodiversity and ecosystem services.

Based on spatially explicit data on agricultural revenues, biodiversity (mean species abundance), cultural services and carbon sequestration, a number of relevant policy insights can obtained. First, the production possibility frontier is non-concave. While an inconvenient result from an economic viewpoint this will be of no surprise to ecologists. Moreover, opportunity cost information shows that trade-offs differ substantially between regions. On average, higher income countries have lower opportunity costs than poorer countries in our sample. Within-country variation, however, is large. This finding emphasizes that caution must be taken when using benefit transfer methods to generalize results from environmental valuation studies. Generally, opportunity costs increase with higher levels of provisioning services. For most regions, they also increase with higher levels of biodiversity and cultural services, but they decrease with higher carbon levels. Therefore, regions having high carbon sequestration levels have a comparative advantage in further increasing sequestration. This implies that for carbon sequestration there are economies of scale. On the other hand, regions having a comparative advantage in improving biodiversity or cultural services are those with low and those with high levels of biodiversity or cultural services. Up to a certain level of biodiversity, there are economies of scope but at higher levels specializing becomes more cost-effective.

The results from the Central and Eastern European case study shows the advantages of the proposed method. Our frontier approach enables the opportunity costs of biodiversity and other difficult-to-value ecosystem services to be assessed. These values show what is actually lost in terms of agricultural production if biodiversity changes and in that way help shaping land use decisions. There are several possible extensions to the application of the method. First, the number and type of policy implications offered by future analyses will benefit from fewer restrictions on data availability. Obviously, if more outputs are modeled, more trade-offs can be analyzed. Moreover, a richer analysis will follow if also other inputs are included. This refers both to regulating and supporting ecosystem services affecting the provisioning and cultural services (pollination, erosion prevention, water infiltration and natural pest management) and also to inputs reflecting land use intensity (like fertilizer and pesticide use and labor and capital input). By including both types of inputs, trade-offs between natural and modern inputs, or between more and less intensive agriculture can be analyzed in more detail. To enable such an analysis, reliable spatial data on land use intensity needs to be become available first. Secondly, with pooled cross-section and annual data, changes in the shape and position of the frontier can be assessed. Positions of the frontier may change due to technical changes or changes in climate. Moreover, due to differences in economic development patterns, evolution of country frontiers may follow different patterns. In addition, the position of each region on the frontier may change over time. Evaluation of the intertemporal changes of the frontier and the position on the frontier provides relevant information on the dynamic effects of land use choices on the opportunity cost of ecosystem services .

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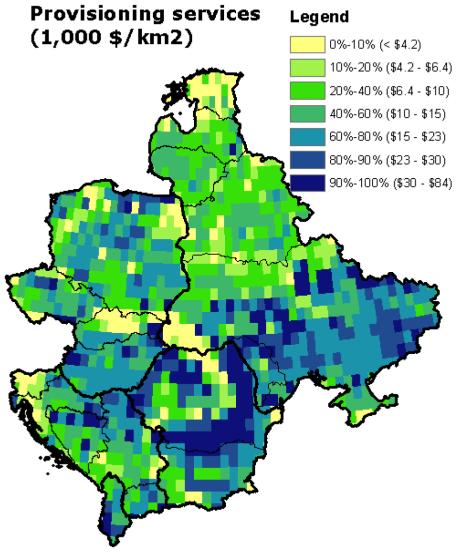
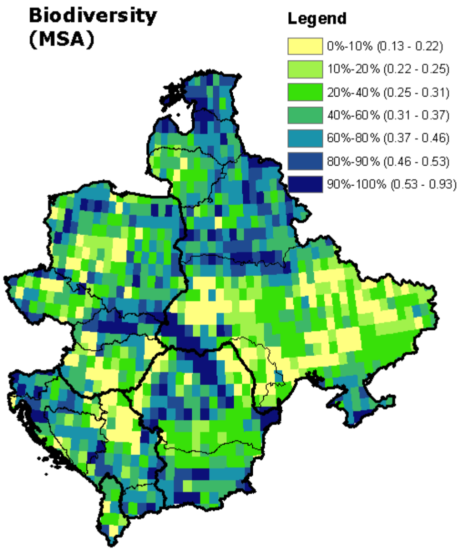
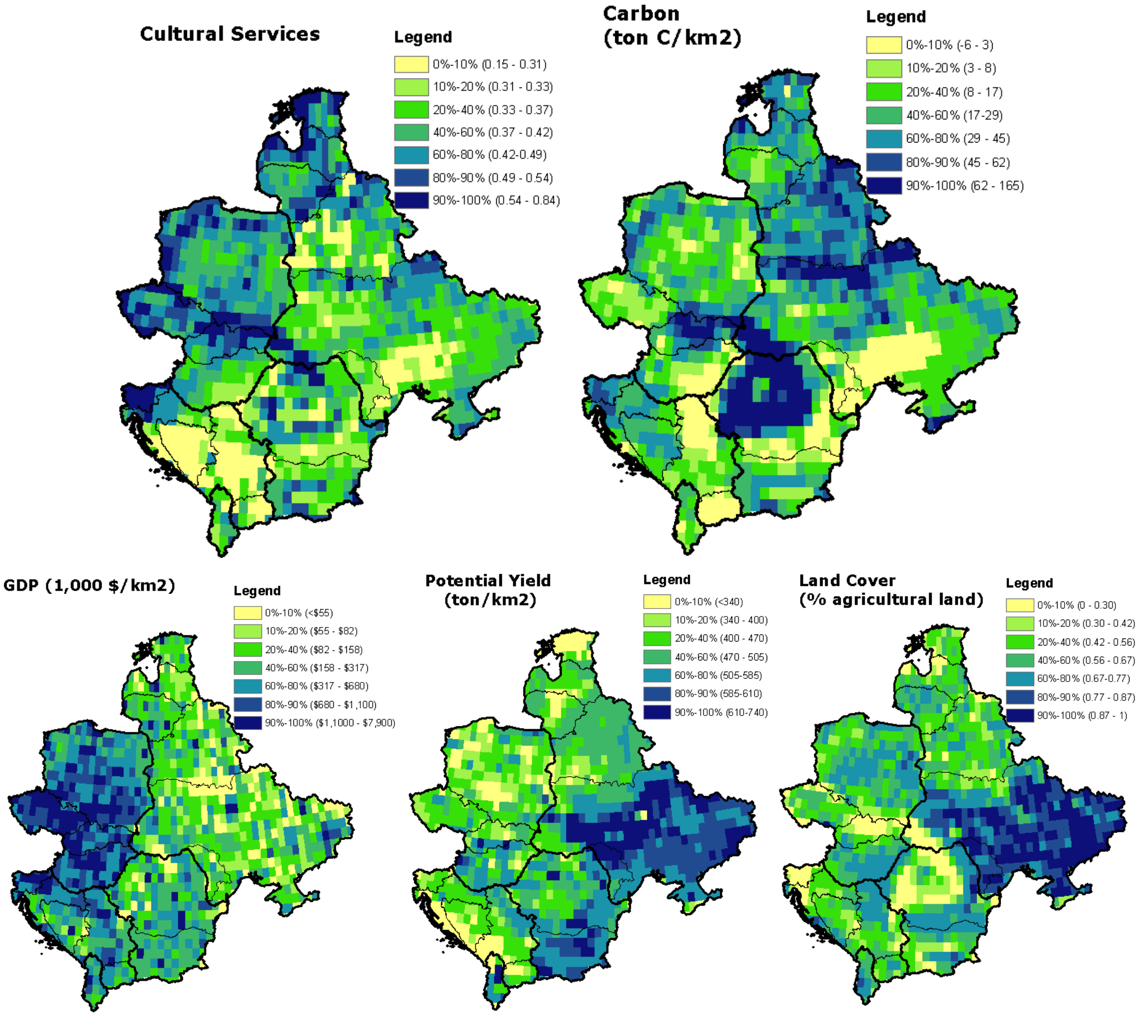
Appendix: Maps of base data







Map 1: Maps of the base data: a) provisioning services in $ of agricultural output per km^2; b) biodiversity (MSA); c) cultural services; d) carbon sequestration (tonnes C/km^2); e) GDP ($/km^2); f) potential yield (ton/km^2/yr) and g) land cover (% agricultural land)).

Note: Classification of the cells is such that each color corresponds with 10% or 20% of the observations.