Assessing trade-offs between food production, biodiversity and carbon sequestration for Eastern Europe

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Abstract

Understanding the trade-offs between food production, biodiversity, carbon sequestration and other ecosystem services has become increasingly important for local, national and global policy making. This paper contributes to this understanding 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 with ecosystem services data for 18 Central and Eastern European countries. Results show that opportunity costs of changes in biodiversity and carbon sequestration differ substantially between regions. Those areas having already relatively high levels of carbon sequestration have a comparative advantage in sequestering more carbon. Opportunity costs of biodiversity generally increase with increasing biodiversity up to a turning point after which they decrease. We argue that the method and resulting opportunity costs can lead to more integrated and rigorous policy support and dialogue.

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

JEL-codes: Q18, Q20, Q57, C14.

1. Introduction

Three of the major challenges the world faces today are to produce sufficient food to feed a growing world population, to halt global loss of (terrestrial) biodiversity and to halt the loss of carbon sequestration potential as one of the major mechanisms behind climate change. These three challenges are intrinsically interlinked. Transformation of natural habitat into agricultural land is a primary cause of biodiversity loss and of the loss of carbon sequestration potential. Loss of ecosystem services due to low biodiversity levels in areas of intensive agriculture may have detrimental effects on yield levels through negative feedback effects. Solutions to tackle these three challenges must integrate relations between food production, biodiversity conservation and carbon sequestration.

It is, however, difficult to determine how biodiversity and ecosystem goods and services – of which food production and carbon sequestration are examples – are connected and how they are affected by changes in land use (Turner et al., 2010; Barbier, 2011). In this paper, we analyze the ecological and economic effects of land use decisions from a supply side perspective. Important advances have been made in analyzing the effects of land use choices on ecosystems and the interactions and dependencies between the different ecosystem functions and services (Millennium Ecosystem Assessment, 2005). Recent integrated assessment models account for spatial heterogeneity and have expanded beyond singles services to consider various ecosystem services jointly produced by the ecosystem, including agricultural production, biodiversity and carbon sequestration.[1] The results of these models enable spatially explicit production possibility frontiers to be estimated, showing the bundles of ecosystem services that can be produced in a given area and the trade-offs between these ecosystem services when land use changes. In this way, they properly consider the ecological complexities and interactions of the integrated assessment models to analyze the trade-offs of land use changes at appropriate spatial scales (Polasky and Segerson, 2009; McShane et al., 2011).

In this paper we contribute to this literature by developing and applying a method to estimate the spatially explicit opportunity costs of several jointly produced ecosystem goods and services. 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 goods and services 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 (Diaz-Balteiro and Romero, 2008).

The results are used to explore the question where to preserve biodiversity or to stimulate agriculture. Is it cost-effective to separate reserves and intensive agriculture or it is possible to reach food production and biodiversity targets on a multifunctional landscape in which agriculture and nature protection coexists? The results provide relevant information on spatial differences of trade-offs between ecosystem services and on the areas that have a comparative advantage for supplying particular ecosystem services; information that is essential for

[1] See e.g. results from integrated assessment model IMAGE (Bouwman et al., 2006), global biodiversity model GLOBIO (Alkemade et al., 2009) and additional analyses for analyzing trends in ecosystem services (Schulp et al., 2012). Results of these models are used e.g. for OECD (2012), PBL (2012) and UNEP (2012).

supporting land use decisions. The results show that biodiversity, food production and carbon sequestration can all increase if land use choices account for the opportunity costs. By investing in biodiversity in areas with low opportunity costs for biodiversity and expanding agricultural production in areas in which opportunity costs are high, food production or biodiversity may decrease in particular areas, but overall levels may increase substantially in a cost-effective way.

Our method differs from previous studies in several ways. The extant literature on analyzing trade-offs between ecosystem services in monetary terms or deriving their 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 multinational scale. Previous studies that aim at higher spatial scales, such as Haines-Young et al. (2012) and Maes et al. (2012) do not make the step to estimate opportunity costs, and those that do, like Hussain et al. (2011), base estimates on debatable benefit transfer results. For analyses at higher spatial scales obtaining good quality, low resolution data of ecosystem services is still difficult. For that reason, we use spatially explicit data on agricultural revenues, cultural services, carbon sequestration and biodiversity for 18 Central and Eastern European countries that 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). These models give the state-of-the-art knowledge of the relations between land use, agricultural production, ecosystem services production and ecosystem processes. Results of these models have been used recently in the OECD Environmental Outlook (OECD, 2012) and UNEP GEO4 (UNEP, 2012). Whereas these models by themselves are not capable of producing information on trade-offs or effects of marginal land use changes, their results can be used in the semi-parametric method set up in this paper to recover from the data the production possibility frontier for ecosystem services. Using model data is the only feasible option for this type of analysis because of the lack of reliable observations of the relevant variables at higher spatial scales.

Related to our work are recent studies using bio-economic models to evaluate trade-offs between bundles of jointly produced ecosystem services (Polasky et al., 2008; Crossman and Bryan, 2009; Barraquand and Martinet, 2011; Keeler et al., 2012) and studies using GIS and heuristic routines combining ecological and economic concepts (Bateman, 2009; Bateman et al., 2011; White et al., 2012). Distinct from these studies, we use parametric and non-parametric frontier estimation techniques to explicitly estimate a multidimensional frontier. 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. 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.

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 ecosystem services due to land use changes. We consider a situation in which a social planner considers land use choices – where to grow crops, where to preserve biodiversity, where to keep a multifunctional landscape. These land use choices affect ecosystem services supply levels and the resulting trade-offs. To represent the bundle of ecosystem services supplied in a certain location, let conventional marketed outputs be denoted by vector y and non-marketed outputs by vector q. Together the bundle of outputs (y,q) covers the different land use dependent ecosystem goods and services that can be distinguished in a given area. Vector y includes agricultural revenues (provisioning services) and marketed cultural services (e.g. tourism). Vector 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.

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^*). 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:

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

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. Nevertheless, observing the output levels in a region, 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. In this paper, quasi-concavity of the transformation function will be tested for.

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. 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):

$$\Psi^{FDH}(x, y, z) = \left\{ (x, y, z) \in \mathbb{R}_+^{M+N+K} \mid y \le y_l, x \ge x_l, z_l \in [z-h, z+h] \exists l = 1,\dots,L \right\} \quad (3)$$

The FDH frontier is a stairway-shaped curve connecting the Pareto optimal observations. 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:

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

This function measures for each observation the distance of the output vector to the frontier. For the Pareto optimal observations $\lambda = 1$. For the other points $\lambda > 1$, where $(\lambda$-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):

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

for observation (x_i,y_i), with $S_Y\left(y \middle| x\right) = \Pr\left(y \le Y, x \ge X\right) / \Pr\left(x \ge X\right)$ 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 $\left\{\left(x_l, y_l, z_l\right) \middle| l = 1,\dots,L\right\}$ 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)

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

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:

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

for all l = 1,…,*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. In what follows, input variables are dropped from the notation. In the empirical analysis only one input variable is included, land, which is constant across all observations. Input variable *x* can easily be included in the framework, however. Let $\delta(y|z)$ be the Shephard output distance function, which is equal to the inverse of the Farrell-

Debrue distance function introduced in (4) and (6), $\delta(y|z) = \lambda^{-1}(y|z)$. Introduce a parametric distance function $\varphi(y,z;\theta)$, 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 $\delta(.)$:

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

Assume a translog function $\ln\varphi\left(y,z;\theta\right) = \alpha_0 + \beta'\ln y + \frac{1}{2}\ln y'\Gamma\ln y + \gamma'\ln z$. In which $\theta=(\alpha_0,\beta,\Gamma,\gamma)$ are the parameters of function φ, with α_0 a scalar, β and γ vectors of length M and K, respectively, and Γ a symmetric MxM-matrix, $\Gamma=\Gamma'$ (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 $\delta_i = \delta\left(y_i \mid z_i\right)$)

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

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

$$\ln y_{i1}^* = -\left(\alpha_0 + \beta_{-1}'\ln\tilde{y}_{i,-1} + \tfrac{1}{2}\ln\tilde{y}_{i,-1}'\Gamma_{22}\ln\tilde{y}_{i,-1} + \gamma'\ln z_i\right) \tag{9}$$

is estimated using OLS. Using the conditions on β and Γ as given above, distance function $\varphi(y,z;\theta)$ 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 (9) 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 $B(p)=\sup_y \{p'y \mid (y,z)\in\Psi\}$. As $y/\delta(y\mid z)$ is a feasible output vector, it has to hold that $B(p)\geq p'y/\delta(y\mid z)$ and so $\delta(y\mid z)=\max_p\left[p'y/B(p)\right]$. As a result, for each m = 1,…,M

$$\frac{\partial\delta(y\mid z)}{\partial y_m}=\frac{p_m}{B(p)} \tag{10}$$

If the market price is known for one of the outputs, e.g. for the first output, from (10) 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

$$p_m = p_1\cdot\frac{\partial\delta(y\mid z)/\partial y_m}{\partial\delta(y\mid z)/\partial y_1}=p_1\frac{y_1}{y_m}\left(\frac{\beta_m+\Gamma_m^{'}\ln y}{\beta_1+\Gamma_1^{'}\ln y}\right) \tag{11}$$

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 foregone agricultural revenues. 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.

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 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. As discussed in the introduction, data originate from the integrated assessment model IMAGE, global biodiversity model GLOBIO and additional ecosystem services models. Data are given on the level of grid cells of size 50x50 km^2 for the year 2000. Land use patterns per grid cell (share of each cell covered with agricultural land, grassland, shrub land, forest or cultivated land) were derived from the GLC2000 land use map (EC-JRC, 2003). This results in 1166 observations. The following output variables are included:

1. Agricultural revenues (provisioning services; in 2000 international $/km2).[2] 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. [3] 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. 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* (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.
3. 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

[2] A better measure would be to use agricultural returns, i.e. gross revenues minus costs. Due to the difficulty to estimate production costs in a reliable way, however, many studies use gross revenues instead.
[3] 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.

the biomass harvested. Data are based on the GLC2000 land use map and EURURALIS carbon model (see e.g. Schulp et al., 2008).

4. Cultural services: a composite index consisting of attractiveness for tourism and for hunting and gathering activities (Schulp et al., 2012). Tourist attractiveness is an index ranging from 0 (unattractive) to 1 (attractive) and depends on the 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.

The following conditional variables are included, which assure a peer-to-peer comparison between cells when deriving the frontier.

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.
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 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. 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).

Maps of the base data are shown in Figure 3 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. The relatively high correlations between some of the ecosystem services corresponds with similar observations by Raudssep-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. 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. Results

The two-stage approach discussed in Section 3 is used to analyze the following questions: What are the opportunity costs of marginal changes in biodiversity and carbon sequestration? Which regions have a comparative advantage for conservation or for agricultural development? Can land use changes lead to win-win situations or will there always be a trade-off between the different variables included? The results, discussed in detail below, can be summarized in the following 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 certain ecosystem service. For agricultural revenues, opportunity costs generally increase if production increases. In contrast, enhancing carbon

sequestration becomes cheaper in regions with higher sequestration levels, whereas, in general, promoting biodiversity becomes more expensive with higher levels biodiversity.

- By improving biodiversity in areas with low opportunity costs for biodiversity and improving agriculture in areas in which they are high, both biodiversity and agricultural revenues can be improved substantially. The same applies for carbon sequestration. Within an area there will be a trade-off between the different variables and areas will specialize more in either biodiversity or crop production; overall, for the entire study area, agricultural revenues, biodiversity as well as carbon sequestration may increase simultaneously.

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 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). 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 (10) 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 (11) 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 (9), is estimated. The coefficients of the Shephard output distance function and frontier function (9) 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 agricultural revenues showing that within each cell there is a trade-off between the different outputs.

The eigenvalues of the Hessian show that the distance function is not quasi-convex and so that the frontier is non-concave. This result is robust, as the same result is also obtained for different formulations of the model. As an illustration, Figure 1 plots the 3-dimensional frontier for the sub-region SE for agricultural revenues, biodiversity and carbon sequestration. As this plot is in fact an extrapolation, it should be interpreted with care, especially at the boundaries and the areas with only few observations. The plots show violations of the curvature assumptions.

Table 3: Parameter estimates of the translog distance function and frontier function (9) [(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*)	$-5.7x10^{-4}$	
$\Gamma_{14} = \Gamma_{41}$	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.

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. Arrow et al. (2003) argue that “a large body of empirical work has revealed that ecosystem interactions in many cases involve transformation possibilities among environmental goods and services that, together, constitute non-convex sets" (p.499). As a result, output specialization often 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.

5.2 Opportunity costs

The second result concentrates on opportunity costs. For each observation, opportunity cost ratios or marginal rates of transformation are estimated using (11) and the coefficients given in Table 3.[4] They reflect the gross benefits from agricultural production foregone due to a marginal increase in any of the other output variables. They are presented in Table 4 and Figure 2. For this analysis, we particularly focus on the opportunity costs for biodiversity and carbon sequestration.

[4] 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.

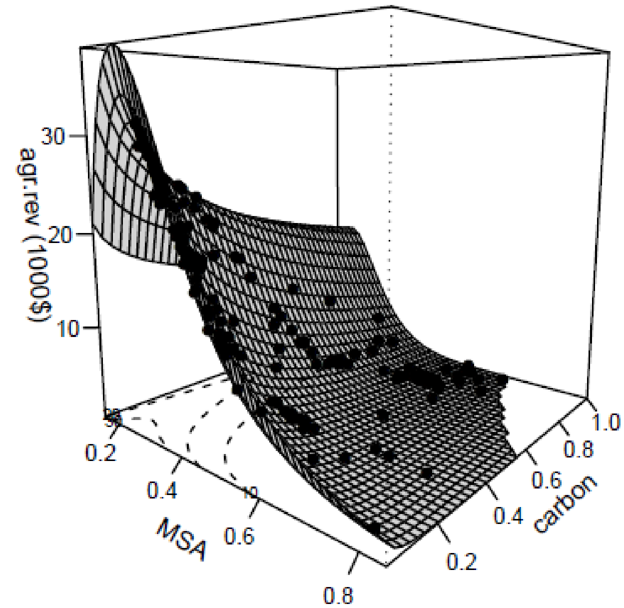


Figure 1: 3-dimensional plot of the relationship between agricultural revenues, biodiversity (MSA) and carbon sequestration for the sub-region SE – see also Figure 4.

Note: The plots for the other three sub-regions are similar. A contour plot is given on the x-y plane. The dots on the frontier are the observations projected on the frontier. To draw the plot, 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 (9).

Due to their particular interpretation in this analysis, it is difficult to directly compare the estimates 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 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: Median opportunity cost ratios and standard deviations per country

	MSA ($ per % MSA)		Carbon ($ per tonne carbon)	
	Median	St.Dev.	Median	St.Dev.
Total[1]	1,027	1,029	202	220
Belarus	1,008	597	125	55
Estonia	247	310	62	65
Latvia	667	482	139	89
Lithuania	702	254	211	78
Moldova	1,534	707	490	151
Ukraine	1,184	944	220	190
Czech	363	486	246	170
Hungary	1,308	543	342	232
Poland	792	615	224	183
Slovakia	521	831	110	97
Bosnia	1,904	2,344	284	134
Croatia	870	772	222	108
Macedonia	1,817	770	1,169	419
Serbia	1,206	833	374	151
Slovenia	111	312	66	67
Albania	1,208	1,046	315	260
Bulgaria	1,635	779	347	181
Romania	2,064	1,439	155	294

Notes: 1) Opportunity costs for MSA range between $1 and $8,846 with a mean of $1,276 per % MSA. Opportunity costs for carbon sequestration range between $1.4 and $1,986 with a mean of $263 per ton C.

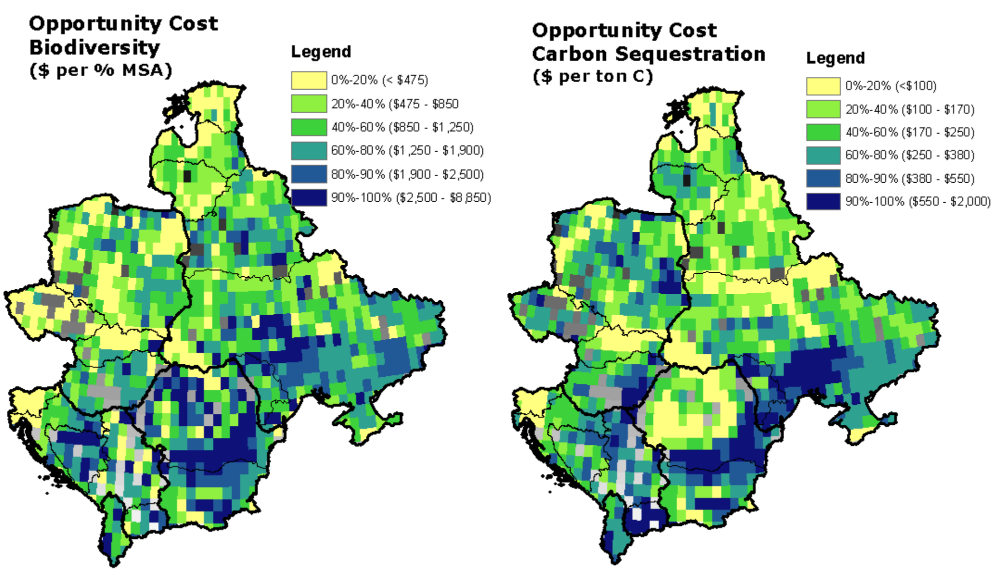


Figure 2: Maps of opportunity costs per cell for a) mean species abundance ($ per % MSA), b) cultural services ($ per % cultural services index) and c) carbon sequestration ($ per ton carbon).
Note: Classification of the cells is such that each color corresponds with 10% or 20% of the observations. Grey cells are non-monotonic observations or outliers.

As shown in Figure 2, opportunity costs differ substantially between the different regions but also within-country differences are large. The results show that carbon opportunity costs increase with decreasing levels of carbon, biodiversity and cultural services and with increasing levels of agricultural revenues. Moreover, 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 agricultural revenues 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 agricultural revenues. 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. Their opportunity costs increase with higher levels of agricultural revenues. Cells with high levels of agricultural production have a comparative disadvantage in providing further biodiversity. 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. It turns out that the relationship between agricultural revenues 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 agricultural revenues, carbon sequestration and land cover. The higher the levels of agricultural revenues, the higher the opportunity costs for biodiversity and the higher the rate at which these opportunity costs increase with rising biodiversity levels. However, after a certain threshold level, which is cell specific, opportunity costs start to decrease again.

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.

To conclude, the maps in Figure 2 show which regions have comparative advantages (low opportunity costs) or comparative disadvantages (high opportunity costs) in the provision of biodiversity or carbon sequestration. There is considerable variation within each country, but each country has regions in which increasing these outputs is more cost-effective. So, these maps are helpful for prioritizing conservation policies.

5.3 Effects of land use changes

As argued in the introduction, internationally objectives are set to improve biodiversity levels, to reduce greenhouse gas emissions and to produce enough food to feed a growing world population. In the analysis above, it has been argued that improving any of the output variables considered in this analysis can be reached cost-effectively if regions are targeted carefully. The question remains whether win-win situations can be attained in which land use changes in such a way that several of the output variables increase simultaneously. In this section, using the frontier function, it is analyzed which output changes are feasible and to what extent alternative land use choices affect the outputs generated. The following policy alternatives are considered:

1. Increase overall biodiversity levels by 5%
    a. Uniformly improve biodiversity by 5% in all cells.
    b. Improve biodiversity by 15% in the cells having below average opportunity costs for biodiversity and decrease biodiversity by 15% in the remaining cells.
    c. Improve biodiversity by 30% in the cells having opportunity costs lower than halve of the mean opportunity costs, improve biodiversity by 15% in the remaining cells having below average opportunity costs and allow for a decrease of biodiversity by 30% in the remaining cells.
2 Increase overall carbon sequestration by 5%
    a. Uniformly improve carbon sequestration by 5% in all cells.
    b. Improve carbon sequestration by 8% in cells having below average opportunity costs for carbon and decrease sequestration by 8% in the remaining cells.

To analyze the above questions, the levels of biodiversity or carbon sequestration are changed individually, after which the agricultural revenues are recalculated using the frontier function (9). Note that even though biodiversity and carbon sequestration are correlated, in the model they are treated as independent variables.

The results show that increasing biodiversity by 5% in all cells (case 1a) results in a reduction of total agricultural revenues by 13%. In all cells agricultural production falls, with a maximum of 42% and a minimum of 1%. A 5% improvement of aggregate biodiversity is also realized if biodiversity levels are increased by 15% in the cells having below average opportunity costs and decreased by the same amount in the remaining cells (case 1b). In that case,

Table 5: Effects of changes in biodiversity or carbon sequestration on agricultural revenues

	Case 1a		Case 1b		Case 1c		Case 2a		Case 2b	
	Agric. Rev.	Biodi-versity	Agric. Rev.	Biodi-versity	Agric. Rev.	Biodi-versity	Agric. Rev.	Biodi-versity	Agric. Rev.	Biodi-versity
CIS	-12%	5%	14%	6%	41%	6%	-2%	5%	-1%	5%
CE	-11%	5%	3%	8%	19%	12%	-2%	5%	0%	4%
YUG	-15%	5%	47%	2%	148%	1%	-2%	5%	1%	2%
SE	-15%	5%	48%	-2%	129%	-7%	-2%	5%	0%	5%
tot	-13%	5%	23%	5%	67%	5%	-2%	5%	0%	5%

agricultural production dramatically drops in some cells (over 80%; especially in the Carpathians and in Belarus) but it substantially improves in other cells (especially in the former Yugoslavian republics and central and south-east Romania) in such a way that total agricultural revenues increase with 23%. The cells for which agricultural production decreases are those having low levels of agricultural output and high levels of biodiversity in the base situation. In those cells, opportunity costs are low as well. The same aggregate level of biodiversity can also be attained in case 1c in which extra effort is put on improving biodiversity in the cells having the lowest opportunity costs. In that case, total agricultural revenues even increase by more than 60%. In the cases 1b and 1c, agricultural revenues increase in all sub-regions, but aggregate biodiversity in Romania (sub-region SE) decreases somewhat. The results show that specialization improves aggregate levels of biodiversity and agricultural production. Some regions, however, especially those already having low biodiversity levels, will become even more agriculture oriented and biodiversity poor than they were before.

The effect of improving carbon sequestration on agricultural revenues is smaller than the effect of improving biodiversity, but the pattern is the same. Case 2a, in which carbon sequestration is improved by 5% in all cells shows that aggregate agricultural revenues only decrease by 2%. If those regions are targeted in which opportunity costs are lowest (case 2b), aggregate production remains stable. In that case, in some regions agricultural production increases whereas it decreases in those regions in which land use changes are made to support carbon sequestration.

For both cases, the results show that by carefully targeting areas to protect biodiversity, to sequester carbon or improve agricultural production, substantial gains can be attained. Aggregate agricultural production can still improve and it not necessarily has to lead to a continued loss of biodiversity. It will result, however, in increased specialization in any of the functions, at the expense of multifunctional and diverse landscapes.

6. Discussion and Conclusion

The principal aim of this paper was to provide monetary estimates of opportunity costs of ecosystem services, to analyze the comparative advantages areas have for producing particular ecosystem services and to assess the possibility to attain win-win situations in which food production, biodiversity and carbon sequestration increase simultaneously. 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, which regions have a comparative advantage in producing particular ecosystem services and in which regions should biodiversity conservation or agricultural extension be targeted? 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. Generally, opportunity costs increase with higher levels of agricultural revenues. 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 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. Further analyzing the cost-effectiveness of improving biodiversity or carbon sequestration, shows that win-win changes can be attained if land use changes are carefully targeted in such a way that biodiversity or carbon sequestration are improved in areas with low opportunity costs and agricultural production is intensified in areas with higher opportunity costs,

The results for the Central and Eastern European case study show the advantages of the proposed method. Our frontier approach enables the opportunity costs of biodiversity and carbon sequestration to be assessed. These values show what is actually lost in terms of agricultural revenues 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.

References

Alkemade, R., M. v. Oorschot, L. Miles, C. Nellemann, M. Bakkenes and B. t. Brink (2009). "GLOBIO3: A Framework to Investigate Options for Reducing Global Terrestrial Biodiversity Loss." Ecosystems 12: 374-390.

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

Arrow, K. J., P. Dasgupta and K. G. Maler (2003). "Evaluating projects and assessing sustainable development in imperfect economies." Environmental & Resource Economics 26(4): 647-685.

Barbier, E. B. (2011). "Pricing Nature." Annual Review of Resource Economics 3(1): 337-353.

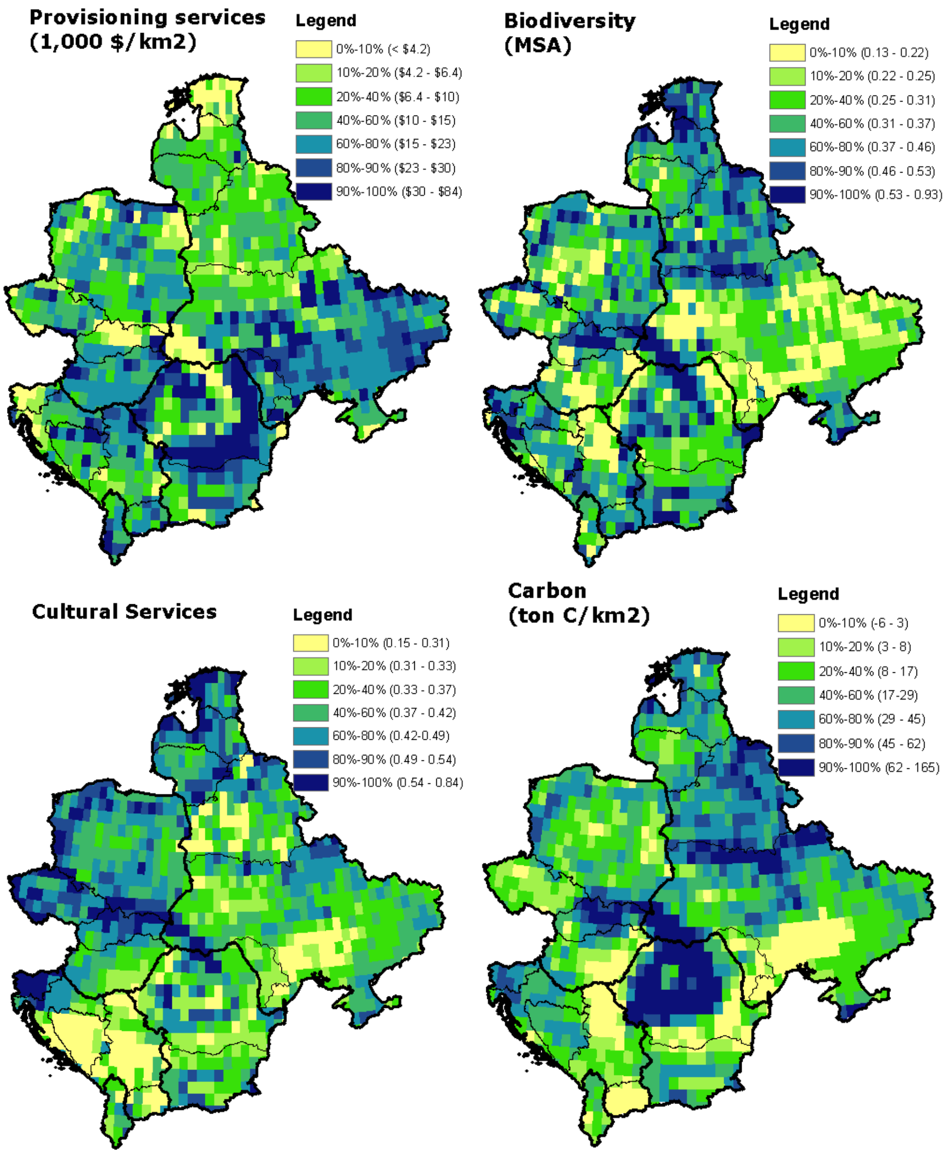
Barraquand, F. and V. Martinet (2011). "Biological conservation in dynamic agricultural landscapes: Effectiveness of public policies and trade-offs with agricultural production." Ecological Economics 70(5): 910-920.

Bateman, I. J. (2009). "Bringing the real world into economic analyses of land use value: Incorporating spatial complexity." Land Use Policy 26, Supplement 1(0): S30-S42.
Bateman, I. J., G. M. Mace, C. Fezzi, G. Atkinson and K. Turner (2011). "Economic Analysis for Ecosystem Service Assessments." Environmental & Resource Economics DOI 10.1007/s10640-010-9418-x.
Bellenger, M. J. and A. T. Herlihy (2010). "Performance-based environmental index weights: Are all metrics created equal?" Ecological Economics 69(5): 1043-1050.
Bosetti, V. and B. Buchner (2009). "Data Envelopment Analysis of different climate policy scenarios." Ecological Economics 68(5): 1340-1354.
Bostian, M. B. and A. T. Herlihy (2012). Shadow pricing wetland function. Paper presented at the EAERE 2012 Conference. Prague, Czech Republic.
Bouwman, A. F., T. Kram and K. Klein Goldewijk, Eds. (2006). Integrated modelling of global environmental change. An overview of IMAGE 2.4. Bilthoven, The Netherands, PBL - Netherlands Environmental Assessment Agency.
Brown, G., T. Patterson and N. Cain (2011). "The devil in the details: Non-convexities in ecosystem service provision." Resource and Energy Economics 33(2): 355-365.
Cazals, C., J. P. Florens and L. Simar (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.
Crossman, N. D. and B. A. Bryan (2009). "Identifying cost-effective hotspots for restoring natural capital and enhancing landscape multifunctionality." Ecological Economics 68(3): 654-668.
Daraio, C. and L. Simar (2005). "Introducing Environmental Variables in Nonparametric Frontier Models: a Probabilistic Approach." Journal of Productivity Analysis 24: 93-121.
Daraio, C. and L. Simar (2007a). Advanced robust and nonparametric methods in efficiency analysis: Methodology and applications. New York, Springer.
Daraio, C. and L. Simar (2007b). "Conditional nonparametric frontier models for convex and nonconvex technologies: a unifying approach." Journal of Productivity Analysis 28: 13-32.
Dasgupta, P. and K. G. Maler (2003). "The economics of non-convex ecosystems: Introduction." Environmental & Resource Economics 26(4): 499-525.
De Witte, K. and B. Geys (2011). "Evaluating efficient public good provision: Theory and evidence from a generalised conditional efficiency model for public libraries." Journal of Urban Economics 69: 319-327.
De Witte, K. and M. Kortelainen (2009). Blaming the exogenous environment? Conditional effciency estimation with continuous and discrete exogenous variables. MPRA Paper. Munich.
Diaz-Balteiro, L. and C. Romero (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.
Egoh, B. N., B. Reyers, J. Carwardine, M. Bode, P. J. O'Farrell, K. A. Wilson, H. P. Possingham, M. Rouget, W. de Lange, D. M. Richardson and R. M. Cowling (2010). "Safeguarding Biodiversity and Ecosystem Services in the Little Karoo, South Africa." Conservation Biology 24(4): 1021-1030.
Färe, R. and S. Grosskopf (2000). "Theory and Application of Directional Distance Functions." Journal of Productivity Analysis 13(2): 93-103.
Färe, R., S. Grosskopf, D. W. Noh and W. Weber (2005). "Characteristics of a polluting technology: theory and practice." Journal of Econometrics 126(2): 469-492.
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).
Ferraro, P. J. (2004). "Targeting conservation investments in heterogeneous landscapes: A distance-function approach and application to watershed management." American Journal of Agricultural Economics 86(4): 905-918.
Florens, J. P. and L. Simar (2005). "Parametric approximations of nonparametric frontiers." Journal of Econometrics 124(1): 91-116.
Haines-Young, R., M. Potschin and F. Kienast (2012). "Indicators of ecosystem service potential at European scales: Mapping marginal changes and trade-offs." Ecological Indicators 21(0): 39-53.

Hof, J., C. Flather, T. Baltic and R. King (2004). "Forest and rangeland ecosystem condition indicators: Identifying national areas of opportunity using data envelopment analysis." Forest Science 50(4): 473-494.
Hughes, G. and P. Hare (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 and L. Mathieu (2011). The Economics of Ecosystems and Biodiversity Quantitative Assessment - Final Report. Geneva, UNEP.
Keeler, B. L., S. Polasky, K. A. Brauman, K. A. Johnson, J. C. Finlay, A. O'Neill, K. Kovacs and B. Dalzell (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.
Lester, S. E., C. Costello, B. S. Halpern, S. D. Gaines, C. White and J. A. Barth (forthcoming 2012). "Evaluating tradeoffs among ecosystem services to inform marine spatial planning." Marine Policy.
MacLeod, M., D. Moran, V. Eory, R. M. Rees, A. Barnes, C. F. E. Topp, B. Ball, S. Hoad, E. Wall, A. McVittie, G. Pajot, R. Matthews, P. Smith and A. Moxey (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., P. P. Principe and E. R. Smith (2010). "A directional distance function approach to regional environmental-economic assessments." Ecological Economics 69(10): 1918-1925.
Maes, J., M. L. Paracchini, G. Zulian, M. B. Dunbar and R. Alkemade (2012). "Synergies and trade-offs between ecosystem service supply, biodiversity, and habitat conservation status in Europe." Biological Conservation 155(0): 1-12.
McShane, T. O., P. D. Hirsch, T. C. Trung, A. N. Songorwa, A. Kinzig, B. Monteferri, D. Mutekanga, H. V. Thang, J. L. Dammert, M. Pulgar-Vidal, M. Welch-Devine, J. Peter Brosius, P. Coppolillo and S. O'Connor (2011). "Hard choices: Making trade-offs between biodiversity conservation and human well-being." Biological Conservation 144(3): 966-972.
Millennium Ecosystem Assessment (2005). Ecosystems and Human Well-Being: Current State and Trends. Washington D.C., Island Press.
Montgomery, C. A., R. A. Pollak, K. Freemark and D. White (1999). "Pricing Biodiversity." Journal of Environmental Economics and Management 38(1): 1-19.
Naidoo, R., A. Balmford, P. J. Ferraro, S. Polasky, T. H. Ricketts and M. Rouget (2006). "Integrating economic costs into conservation planning." Trends in Ecology & Evolution 21(12): 681-687.
Nalle, D. J., C. A. Montgomery, J. L. Arthur, S. Polasky and N. H. Schumaker (2004). "Modeling joint production of wildlife and timber." Journal of Environmental Economics and Management 48(3): 997-1017.
O'Donnell, C. J. and T. J. Coelli (2005). "A Bayesian approach to imposing curvature on distance functions." Journal of Econometrics 126(2): 493-523.
OECD (2012). OECD Environmental Outlook to 2050. Paris, OECD.
PBL (2012). Roads from RIO+20. Den Haag, The Netherlands, PBL - Netherlands Environmental Assessment Agency.
Polasky, S., E. Nelson, J. Camm, B. Csuti, P. Fackler, E. Lonsdorf, C. Montgomery, D. White, J. Arthur, B. Garber-Yonts, R. Haight, J. Kagan, A. Starfield and C. Tobalske (2008). "Where to put things? Spatial land management to sustain biodiversity and economic returns." Biological Conservation 141(6): 1505-1524.
Polasky, S. and K. Segerson (2009). "Integrating Ecology and Economics in the Study of Ecosystem Services: Some Lessons Learned." Annual Review of Resource Economics 1: 409-434.
Raudsepp-Hearne, C., G. D. Peterson and E. M. Bennett (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.
Schulp, C. J. E., R. Alkemade, K. Klein Goldewijk and K. Petz (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., G. J. Nabuurs and P. H. Verburg (2008). "Future carbon sequestration in Europe - effects of land use change." Agriculture, Ecosystems and Environment 127: 251-264.

Tschirhart, J. (2011). "Biology as a Source of Non-convexities in Ecological Production Functions." Environmental and Resource Economics: 1-25.
Turner, R. K., S. Morse-Jones and B. Fisher (2010). Ecosystem valuation A sequential decision support system and quality assessment issues. Ecological Economics Reviews. 1185: 79-101.
UNEP (2012). GEO5: Environment for the future we want. Nairobi, Kenya, United Nations Environment Program.
White, C., B. S. Halpern and C. V. Kappel (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.

Appendix: Maps of base data



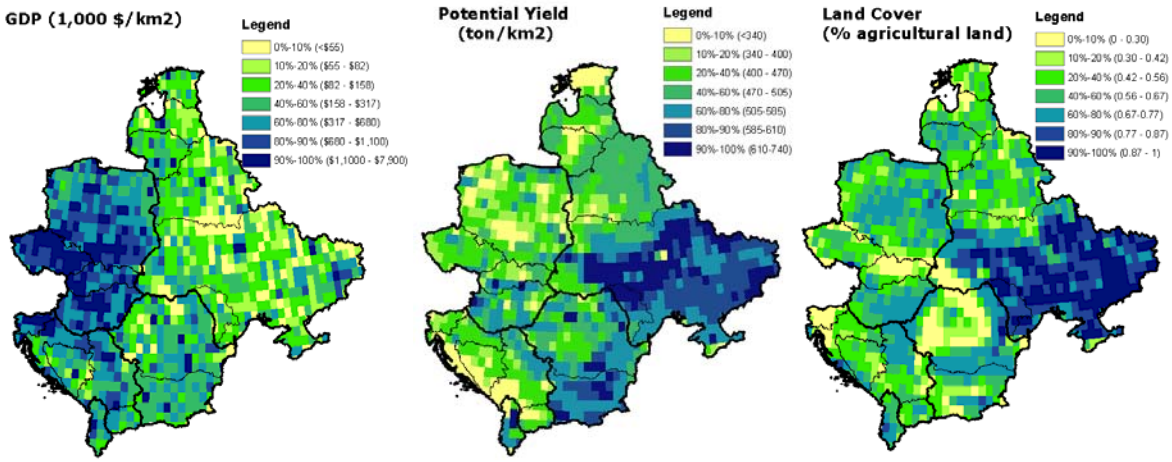


Figure 3: Maps of the base data: a) agricultural revenues in $ of agricultural output per km^2; b) biodiversity (MSA); c) cultural services; d) carbon sequestration (tonnes C/km^2); e) GDP ($/km^2); f) potential yield (ton/km^2/yr) and g) land cover (% agricultural land)).

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

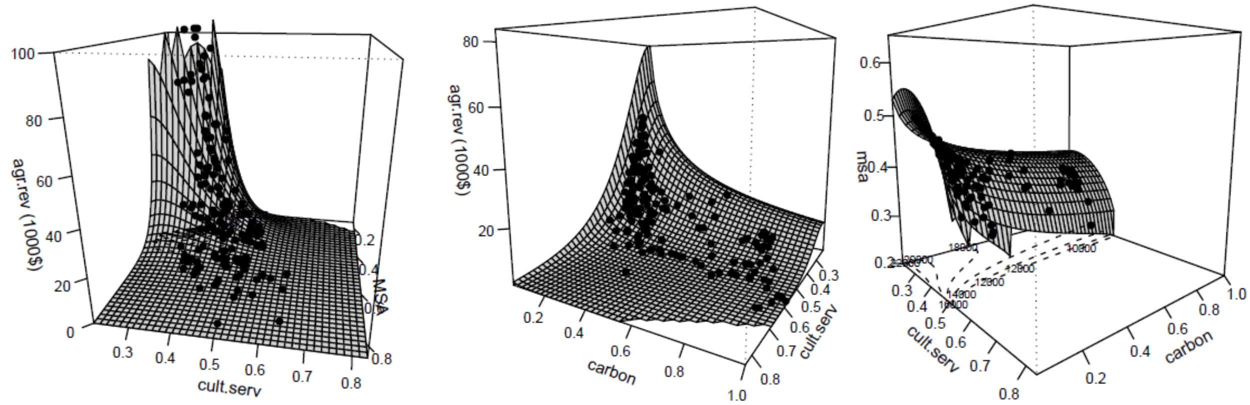


Figure 4: 3D-plots of the frontier for sub-region SE – see also Figure 1.