Economics of Biodiversity and Productivity in Intensive Agricultural Systems

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Omer[1], A., Pascual[2*], U., Russell[1], N.

[1] *School of Economic Studies, University of Manchester, UK*
[2*] *Corresponding author; Department of Land Economy, University of Cambridge, UK*

ABSTRACT

This paper explores the economic effects of biodiversity loss on marketable agricultural output in the context of intensive agricultural systems, where due to specialisation and ecological simplification such systems require an increasing level of artificial capital inputs. A theoretical bioeconomic model is used to derive a hypothesis regarding the effect of the state of biodiversity on the optimal supply of crop output both in the longer run and in the transitional path towards the steady state equilibrium. The hypothesised positive relationship between biodiversity stock and optimal levels of crop output is empirically tested using a stochastic production frontier approach that controls for any potential inefficiency deviation from the crop output frontier. The empirical approach is based on data from a panel of UK specialised cereal farms for the period 1989-2000. The result supports the theoretical hypothesis. This is reflected by the empirical finding that, once the relevant set of labour and capital inputs are controlled for, increases in biodiversity can lead to a continual outward shift in the output frontier (although at a decreasing rate). This may indicate that agricultural transition towards biodiversity conservation is consistent with the increase in the supply of crop output in already biodiversity poor modern agricultural landscapes.

KEYWORDS: Agrobiodiversity, technical change, agricultural transition, sustainable intensive agriculture

1. Introduction

The emphasis in agricultural practice in industrialised countries is on creating the optimum environment for a single target species (the ‘crop’). This is pursued by adjusting the environment so that growing conditions for the target species are optimised while those for competing species (e.g. ‘weeds’ and ‘pests’) are deliberately worsened. This view of the agro-ecosystem as involving managed competitive relationships between species has dominated modern agricultural practice implying the simplification of ecosystem structures (Jackson et al. 2005; *forthcoming*; Pascual and Perrings. *forthcoming*; Perrings et al. 2006). The result is that modern intensive agriculture has largely ignored symbiotic interactions and resource use complementarities between species.

More recently, agricultural intensification is being seen as a mechanism for reducing *ex situ* impacts on non-agricultural habitats in order to conserve wild biodiversity at the landscape level (Green. et al. 2005). At the same time, it is being proposed that ecosystem sustainability is related to the maintenance of specific ecosystem functions rather than species per se, thus pointing towards the role of functional diversity (Altieri, 1999). This implies that sustainability is less related to the diversity of biological species than to preserving particular species that support the necessary ecosystem functions (Tilman et al. 1996; 2001). Obviously, the economic implications are necessarily profound, and this is something that it is beginning to permeate into agricultural and natural resource economics (Pascual and Perrings, forthcoming).

Thus, the role of biodiversity in agro-ecosystems is disputed and further research is being called for (Jackson et al., 2005). For instance additional species might reduce agricultural productivity of the main crop through competition (for nutrients, light etc.), or alternatively might increase output by supporting landscape-level ecosystem functions that help to enhance productivity, e.g. through pollination, soil nutrient enhancement, integrated pest control, rotational effects, etc. (Tscharntke et al. 2005). Although the time scales of these effects may differ, thus creating a complex picture of the effect of biodiversity on crop output, this points to a balance being struck between direct competition between different species, and the support provided by non-crop species for the growing crop through agro-ecosystem functions.

While agroecologists analyse the potential balance/trade-off often in farm-based field experiments (e.g., Tscharntke et al. 2005), economists often focus on the actual use-value provided by agriculture through agricultural markets, i.e., the ‘realized’ supply of biodiversity and crop output. For instance, some authors have analyzed the contribution of crop diversity to the actual (realized) productivity and variability of a main crop output and farm income (Smale et al. 1998; Di Falco and

Perrings 2003, 2005). However, these studies are mostly based on non-intensive agricultural systems where biodiversity is relatively high.

But biodiversity-related loss of ecosystem services matters both on- and off-farm, and may matter more in biodiversity-poor managed or heavily impacted systems than in biodiversity-rich 'wild' or lightly impacted systems. This has significant implications for biodiversity conservation strategies in agro-ecosystems that require policies and incentives to allocate resources (such as investing in on-farm biodiversity) to generate flows of services valued by society (Perrings et al. 2006).

Here we draw from this debate and contribute to it by investigating, from an economics perspective, the effects of biodiversity conservation on productivity in the context of an intensive agricultural landscape. Such systems have been, by contrast to more complex agroecological ones, explicitly simplified in ecological terms by substituting the natural system's internal regulatory processes with high levels of chemical and mechanical inputs and continued human intervention (Jackson et al. 2005; *forthcoming*). We specifically address the dynamics of this relationship using a bio-economic model that describes the effect of 'associated' on-farm biodiversity' (Altieri, 1999), on the marketable supply of crop output. The theoretical results on the optimal dynamic relationship between crop output, changes in productivity and biodiversity are empirically tested using economic and ecological data from a panel of specialised cereal producers in the UK, where there is evidence that on-farm biodiversity is declining over recent decades (Winter, 2000; Stoate et al. 2001).

Notwithstanding the existing studies that assess the productivity of the UK agricultural sector (Thirtle and Bottomley, 1992; Wilson et al. 2001), there have been few attempts to assess such productivity accounting for environmental externalities. Notable exceptions are those by Barnes (2002) and Thirtle and Holding (2003), which mostly focus on pesticide pollution. The empirical model used in this paper instead focuses on productivity and changes in biodiversity.

The rest of the paper is organised as follows: The next section presents the set-up of the theoretical model, which focuses on the effects of biodiversity on crop output. Then, section 3 describes the data used in the empirical analysis and section 4 presents the econometric results, based on a stochastic production frontier approach. The final section recapitulates the main findings and draws out the main implications for the sustainable conservation of biodiversity in intensive agricultural systems.

2. A model of biodiversity change in intensive agriculture

The model assumes that economic decisions such as the optimal allocation of agricultural inputs, for a given area of farm-land are motivated both by levels of crop output and by the agro-ecosystem's environmental quality, reflected by the state of on-site biodiversity. Further, it is assumed that the decision maker's objective is the maximisation of the discounted present value of utility flows derived from both outputs.[1] The stylised direct utility function is specified as $U=U(y_t, b_t)$ where y_t represents the flow of 'marketable' agricultural output at time *t*, and b_t stands for biodiversity loss, also a flow variable. This loss is attributable to intensive use of artificial inputs, x_t, which negatively impacts on utility, i.e. $U_y > 0, U_{yy} < 0,$ and $U_b < 0, U_{bb} < 0,$ for a strictly concave and linearly separable utility function. This setup reflects a subset of economic decisions that would principally affect land use activities, and the welfare that these activities generate. The problem is to find the inter-temporal optimal levels of utility yielding services (flows) based on (i) marketable agricultural supply and (ii) physical depreciation of biodiversity.[2]

Following recent studies (e.g., see Tscharntke et al. 2005), the crop production function is assumed to be positively affected by the stock of biodiversity, z_t, alongside the conventional agricultural input set x_t. In addition, the 'state of the art' of agricultural technology is captured by a_t, as an exogenous shifter of the production possibility frontier, thus representing neutral technical progress. Normalising the unit price of crop output, the value production function is represented by $f(x_t, z_t, a_t)$, assumed to exhibit well behaved properties, i.e. $f_i > 0, f_{ii} < 0$ for $i=x_t$, z_t and a_t, We further assume that the stock of z_t can be increased by conservation investment, c_t. In this sense, the farmer is assumed to choose the optimal transitional time paths of y_t and x_t, that allow maximisation of inter-temporal utility levels. This also implies that the farmer is accounting for the evolution of the stock of biodiversity in the agro-ecosystem, and allocating the total proceeds from agricultural production, $f(x_t, z_t, a_t)$, to: (i) some 'marketable (value) output' y_t, and (ii) some expenditure (investment) for biodiversity conservation.[3] This implies that,

$$c_t = f(x_t, z_t, a_t) - y_t \tag{1}$$

By focusing on the functional diversity of species, the effect of a change in z_t, on the marginal product of x_t, is likely to be different at each level or sublevel of z_t. For example, an increase in insect or micro-organism diversity would increase the marginal product of fertiliser since it enhances soil productivity ($f_{xz} \geq 0$). Alternatively, an increase in natural vegetation diversity would decrease the marginal product of fertiliser as it increases the competition against the cultivated crops ($f_{xz} \leq 0$).

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[1] The model is set up in a general fashion to accommodate a 'decision maker' at different levels, including an individual farmer, a social group or a national or regional authority. The choice on the level of the decision maker is however tied to the notion of a 'given area of land'. In the case of individual farmers, the scale of land-use is the plot or farm, whereas for a social decision maker the agricultural landscape scale is am or appropriate focus. Due to data availability, subsequent empirical analysis to test the results of the model is based on the individual farm level. In this case, following the standard tradition in agricultural household models, it is assumed that our hypothetical rational and forward looking farmer follows an optimal (utility maximizing) pathway.

[2] While z_t refers to the 'level' (*stock*) of biodiversity in time *t*, b_t refers to biodiversity 'loss' or physical depreciation (a *flow* variable). Furthermore, while b_t is an argument in the direct utility function, it depends on the level of application of artificial inputs and the 'current' level of biodiversity through a 'biodiversity impact function'.

[3] This general formulation allows conservation investment to be interpreted as 'forgone output', either as direct current investment in conservation activities or as reduced output arising from adopting environmentally enhancing production practices.

Similar examples could be stated for other components of biodiversity. For simplification, the production function *f(•)* is assumed linearly separable in all its arguments.

Similarly, the biodiversity impact (or loss) function which results in disutility, is expressed by $b_t=b(x_t,z_t)$.[4] The impact function captures the idea of density dependent biodiversity, i.e. a greater stock of biodiversity makes a positive contribution to ecological integrity, implying a lower level of biodiversity loss. This relationship stems from biodiversity's role in enhancing the ability of the agro-ecosystem to tolerate and overcome the potential adverse effects of agricultural land use activities (Altieri, 1999). Additionally, it is assumed that at the margin, biodiversity loss increases (decreases) at an increasing (decreasing) rate due to increases in input intensification (biodiversity stock), i.e. $b_x > 0, b_{xx} > 0$, $b_z < 0, b_{zz} > 0$ and for simplicity $b_t=b(x_t,z_t)$ is assumed to be linearly separable in x_t and z_t.

To maximize utility, the farmer has to choose the optimal levels of the control variables y_t and x_t, at each point in time, subject to the evolution of z_t in the agro-ecosystem. This evolution reflects biodiversity stock, conservation investments, c_t, and artificial input use, x_t, that proxies the level of intensification. This can be expressed as:

$$\dot{z} = g(z_t, c_t, x_t) \tag{2a}$$

The evolution of biodiversity is captured by equation (2a) which can be interpreted as an extended logistic function to allow a closed form solution:

$$\dot{z} = \alpha_1 z_t (1 - z_t / k) + \alpha_2 c_t - \alpha_3 x_t \tag{2b}$$

where α_1, α_2, α_3 and k are all constant parameters. The natural rate of growth of the biodiversity stock is given by $\alpha_1 > 0$. The parameter k portrays a stylized idea of the maximum potential diversity that can be sustained in the ecological system.[5] According to equation (2b), z_t is positively density dependent and it also increases with investment in conservation, α_2 being the rate of induced growth. The parameter α_2 also can be interpreted as the marginal degradation in z_t caused by increase in y_t, i.e., the opportunity cost of c_t. The biodiversity stock is also assumed to be negatively affected by input intensification, reflected by the parameter α_3. It is worth noting that whilst biodiversity is considered to be natural capital, it is assumed that no depletion in biodiversity occurs as a result of its supporting role in the production process. On intensified agricultural systems which are biodiversity poor relative to its potential maximum, the term z_t/k can be interpreted as negligible, and thus (2b) can be simplified through further approximation as:

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[4] See footnote 2.

[5] The parameter k is linked to the idea of 'carrying capacity' in standard species population growth models. In a biodiversity context, it portrays a rather more abstract concept and thus we do not attempt to measure it. It only states what would be the maximum potential level of diversity which could be sustained naturally. Clearly in highly intensive agricultural systems, the term z_t/k (relative biodiversity stock) is likely to be negligible.

$$\dot{z} = \alpha_1 z_t + \alpha_2 c_t - \alpha_3 x_t \tag{2c}$$

The optimisation problem is expressed, for a positive utility discount rate ($\rho > 0$) as:

$$\max_{y,x,c} W(y_t, b_t) = \int_{t=0}^{\infty} e^{-\rho t} U(y_t, b_t) dt \tag{3}$$

subject to (i) the environmental conservation investment function (c.f. equation 1), (ii) the evolution of z_t, (c.f. equation 2a), (iii) the impact function *b(.)*, (iv) the initial condition $z(0) = z_0$ and (v) the non-negativity constraints $x \geq 0$ and $b \geq 0$. This yields the current-value Hamiltonian:

$$\widetilde{H} = U(y_t, b_t) + \varphi(\alpha_1 z + \alpha_2 f(.) - \alpha_2 y_t - \alpha_3 x_t) \tag{4}$$

where φ is the current shadow value of biodiversity. The properties of the optimal trajectories for the state and control variables can be deduced after applying the Maximum Principle, and a subset of these properties are illustrated by a phase diagram in the (z_t, y_t) space (Figure 1). The diagram depicts the joint evolution of $\dot{z} = g(z_t, y_t)$ and $\dot{y} = h(z_t, y_t)$ as a saddle-path towards the steady state (long run) equilibrium with two convergent isosectors (labelled I and III). In the context of the current analyses attention is focused on low-biodiversity intensive agro-ecosystems notionally represented by points within isosector I.

[FIGURE 1]

In this context the effect on optimal crop output supply of a change in the stock of biodiversity z_t, can be investigated from both a static and a dynamic comparative analyses perspective. It can be shown that the optimal supply of marketable output can increase (albeit at a declining rate) along the transition path to the long run equilibrium of output and biodiversity stock when the latter increases in the transition towards the steady state.[6]

The issue being addressed here is whether increasing productivity is consistent with biodiversity conservation in agricultural landscapes and hence whether policies to promote sustainable agriculture should focus on promoting low-yield extensive production practices or seek to develop high yielding sustainable technologies with their attendant advantages for food security and land use. The positive relationship emerging from the theoretical analysis would support the latter approach. In the remainder of this paper we set out an empirical test of this relationship, using a frontier production function to represent the production surface in the theoretical model and constructing a data set that allows estimates of the relevant parameters.

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[6] It can also be shown that the supply of crop output can be increased by investing in improving the state of biodiversity-neutral agricultural technology. These two theoretical propositions and the details of the optimal solution can be obtained from the authors upon request. The properties of the optimal adjustment pathway and an analysis of the impact on agricultural output of biodiversity are provided in Appendix A.

3. The Data

The production data used in this study come from a panel of approximately 230 cereal producers from the East of England, for the period 1989-2000, yielding a total sample size of 2,778 observations in an unbalanced panel. The data are from the UK's annual Farm Business Survey (FBS) undertaken by the Department of Environment, Food and Rural Affairs of the UK (Defra, 2002). Further, the UK Countryside Surveys (CS2000) undertaken in 1978, 1990 and 1998 have been used to construct a farm level biodiversity index (Haines-Young et al. 2000).

The data set includes information on cereal output, level of input application, participation in and payments from agri-environmental schemes, and socioeconomic characteristics of the farm households. In addition, a variable measuring on-farm functional biodiversity is constructed. The per-hectare variables used in the econometric model are: crop enterprise output (marketed)**,** hired and imputed family labour, use of machinery, fertilisers and pesticides, and the biodiversity index (BI). All the variables, except for BI, are derived from value measures deflated by the relevant Agricultural Price Index (base year 1989). Summary descriptive statistics for these variables appear in Table 1.

[TABLE 1]

The data allow the estimation of stochastic production frontier (SPF) models that provide an explicit representation of the production surface underlying the theoretical analysis, where it is assumed that farmers are optimally adjusting their production processes so that they are operating along the production frontier.

Frontier models can be traced back to Farrell's (1957) and Shephard's seminal theoretical work. Aigner, et al*,* (1977) and Meeusen and van den Broeck (1977) introduced contemporary empirical approaches to dealing with stochastic production frontier models. Since then a large and increasing body of literature that focuses on estimating agricultural production frontiers is being generated. Here we estimate separate stochastic frontier production functions, of the type proposed by Battese and Coelli (1995). While one weakness of the SPF approach, relative to nonparametric approaches such as data envelopment analysis (DEA) (e.g., Lansink and Reinhard, 2004), is the potential for misspecification of the functional form of the production technology, its strength is that it allows for white noise or random fluctuations representing influences outside the control of farmers. Since in agriculture, even in intensive systems, random fluctuations are important, we favour the use of the SPF approach relative to DEA.

The key dynamic relationship between agricultural activity and biodiversity is based on measures of species diversity from the UK Countryside Survey (Haines-Young et al. 2000, 2003) and indices of input use and conservation activity on panel farms from the UK Farm Business Survey

(Defra, 2002). Because data from the Countryside Survey is not fully integrated with data from the Farm Business Survey (due to issues related to confidentiality of farm business records), this relationship cannot be directly estimated in conjunction with the production frontier. The adopted approach instead is to initially estimate parameters of this relationship for the panel as a whole, and then apply these parameter values to the farm level data set to generate a farm level biodiversity index (BI) for all farms over the period 1989-2000. The BI is constructed following a three-step process.

The first step is to construct an aggregate biodiversity index for the whole area studied, based on measures of plant species richness from individual survey plots. This information corresponds to the so-called Environmental Zone 1 (EZ1) in the UK Countryside Survey that covers major parts of the eastern lowland counties of England and overlaps closely with the area spanned by the panel of farms in this study. The environmental zone thus reflects an aggregation of land classes chosen to reflect major environmental variation. Hence, the biodiversity index exploits information disaggregated by eight 'aggregate vegetation classes' (AVC)[7] and ten so-called 'broad habitat' (BH) types within EZ1.

The idea is to take into account biodiversity of agricultural landscapes that include non-cropped areas such as field margins, hedgerows and other semi-natural habitats embedded in the cropping area. This is consistent with a number of ecological studies (e.g. Altieri, 1999; Tscharntke et al. 2005) that emphasise the role of landscape level biodiversity (associated and functional) affecting the ecological functioning of arable agro-ecosystems. The aggregation approach used to construct the biodiversity index is described by Wenum et al. (1999). The index (representing the variable z in the theoretical model) is given by:

$$z = \sum_{j} \sum_{i} a_j n_{ij} s_{ij} \qquad (5)$$

where, s_{ij} is the mean plant species richness on a given plot located in aggregate vegetation class (AVC) i within broad habitat (BH) type j; n_{ij} stands for the measure of *AVC-i* dominance in BH type-*j*, i.e. the number of AVC-*i* plots in BH type-*j* relative to the total number of plots of all AVCs in BH-*j*; lastly a_j is a scalar associated with *BH-j* dominance, i.e. the relative area of BH_j within Environmental Zone 1. Using this approach an aggregated index is derived for five periods. Besides the 1978, 1990 and 1998 periods for which the data from the major ecological surveys are available, two additional observations, for 1997 and 1999, have been constructed from the national estimates on each AVC published as part of CS2000 results adjusted for EZ1. The data for 1978 is not presented by BH, so the BH breakdown from 1990 is used as a proxy for 1978 by merging the two data sets at plot level and then using only those plots for 1978 which are repeated in 1990 to construct the 1978 value of the index.

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[7] The eight Aggregate Vegetation Classes in the Countryside Survey are, Crops/Weeds, Tall Grass/Herbs, Fertile Grassland, Infertile Grassland, Lowland Woodland, Upland Woodland, Moorland Grass, and Heath/Bog. Not all of these are represented in EZ1.

In the second step, the evolution of the biodiversity index at the aggregate level is calibrated as a non-linear discrete-time aggregate version of equation (2b). This calls for calibrating the influence on biodiversity of a measure of biodiversity conservation, c_t, and a biodiversity-degrading input intensification measure, x_t. The corresponding values for indices of input use intensity and biodiversity conservation are derived from the Farm Business Survey sample.

$$z_{t+1} - z_t = \alpha_1 \ln z_t + \alpha_2 c_t - \alpha_3 \ln x_t, \qquad \text{(2b')}$$

where c_t is proxied by categorical-dummy variable (1/0) representing whether the farmer is a beneficiary of the introduction of agri-environmental schemes following the EU's Common Agricultural Policy reform in 1992, and x_t is average per hectare pesticide use on the sample farms.[8]

The calibrated parameter values (standard deviations in brackets) are: $\alpha_1=0.32$ *(0.18)*, $\alpha_2=0.31$ *(0.41)*, $\alpha_3=2.24$ *(0.88)*. The last step involves using the parameterisation of the state equation at EZ1 level, in an iterative process, to estimate the value of z_t for each farm in the panel, given the existing farm-level observations for c_t, x_t, and a farm-level starting value for z_0.

4. The Empirical Model

In order to test the key proposition from the theoretical model, a reduced form dynamic parametric frontier model is used and fitted to the data from the panel of cereal farmers from the East of England during 1989-2000. The stochastic production frontier (SPF) approach allows estimating both the output production frontier that represents best practice among farmers (as assumed in the theoretical model) and the possibility of real deviations from the frontier attributed to the effects of variation in the sampled farmers' level of technical efficiency (TE). After technical inefficiency is controlled for, it is possible to qualify the key relationships derived from the theoretical model along the production frontier as it evolves over time. It should be noted that the frontier provides a closer approximation to the 'optimal path' than a more traditional econometric specification which does not allow for technical inefficiency. Hence, the data on marketed crop output is used to estimate the output optimal path which is reduced to an estimable function $y(x_t, z_t, a_t)$.

The model fitted to the twelve years, t=1, 2,…,T, and farm-specific data, i, takes the following form:

$$y_{it} = \beta_0 + \sum_k \beta_k p_{kit} + v_{it} - u_{it} \qquad (6)$$

where:

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[8] See Winter (2000) and Kleijn and Sutherland (2003) for a review of effectiveness of the different the types of agri-environmental schemes applied in arable farming in the UK and EU, respectively, since 1992.

y_{it}: natural log of crop marketed output of farm i at time t (x £100 per hectare/Agricultural Price Index);

p_1: natural log of BI (biodiversity index);

p_2: natural log of fertiliser input value (x £100 per ha/API);

p_3: natural log of labour input value (x £100 per ha/API);

p_4: natural log of machinery input value (x £100 per ha/API);

p_5: natural log of pesticide input value (x £100 per ha/API);

p_6: year of observation where $p_6 = 1, 2,\ldots,12$.

Assuming that v_{it}s are independently and identically $N(0,\sigma_v^2)$ distributed random errors independent of the non-negative random error term, u_{it}, associated with technical inefficiency in production, β_k stands for the parameter vector to be estimated. In order to assume that farms cluster around the efficiency frontier and that their frequency decreases with rising inefficiency levels, the distribution of the residuals needs to be negatively skewed. In other words, it should have the 'longer tail' on the low efficiency side. Hence, the first step in the estimation procedure is to check the sign of the third moment and the skewness of the OLS residuals associated with the sample data (Waldman, 1982). The third moment of the OLS residuals for the models is -0.003. The negative sign suggests that the residuals of the sample data possess the correct pattern for the implementation of the MLE procedure, hence the justification of the model specification.[9] Results for three general Cobb-Douglas SFP models, based on different specifications for the error term u_{it}, are presented, (c.f. equations 7a-7c).

Several versions of each of these three models are estimated using FRONTIER4 (Coelli, 1996), such as the trans-log model, although this is not presented here because the statistical significance of the estimated parameters are masked by significant multicolinearity between the interaction terms. We thus present the results of a standard generalised Cobb-Douglas functional form as the approximate representation (to the true yet unknown) agricultural technology.

Model 1 is a time-varying inefficiency model, in which the inefficiency effect is defined as (Battese and Coelli, 1992):

$$u_{it} = \{\exp[-\eta(t-T)]\}u_i \qquad (7a)$$

where η is an unknown parameter to be estimated, and $u_i = 1,2,\ldots,N$, are independent and identically distributed non-negative random variables obtained by the truncation, at zero, of a $N(\mu,\sigma_u^2)$ distribution. The relative technical efficiency between farms is obtained by scaling it by a factor that increases (if η is positive) or decreases (if η is negative) deterministically over time. This specification

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[9] Since, in the stochastic regression frontier, the $v_{it}s$ are symmetrically distributed, the third moment is also the third sample moment of the u_i. Consequently, if it is negative, it implies that the OLS residuals are negatively skewed, and suggests the presence of technical inefficiency. The computed value of Coelli's (1995) standard normal skewness statistic (M3T) based on the third moment of the OLS residuals is -14.06 (p = 0.000, for H_0: M3T = 0).

assumes that the ranking of farms is unchanged over time and the inefficiency evolves identically for all of them.

Model 2 corresponds to a neutral inefficiency effects model, the inefficiency effects being defined as (Battese and Coelli, 1995):

$$u_{it} = \delta_0 + \sum_j \delta_j q_{jit} + w_{it} \tag{7b}$$

where the w_{it} is an unobservable non-negative random variable assumed independent and identically distributed, obtained by the truncation at zero of a $N(0,\sigma_u^2)$ distribution. The δ_j coefficients are associated with the effects of the following inefficiency effects covariates:

q_1 : Natural log of farmer's age (years);

q_2 : Natural log of the amount of environmental payment (subsidies) obtained by the household;

q_3 : Dummy variable, 1 if the farm participates in any agri-environmental scheme introduced in 1992, 0 otherwise;

q_4 : Proportion of hired to total labour applied in the farm;

q_5 : Dummy variable, 1 if use of hired labour hours, 0 otherwise;

q_6 : Year of observation, $t=1,2...,12$.

Model 3 is a non-neutral inefficiency model (Battese and Broca, 1997):

$$u_{it} = \delta_0 + \sum_j \delta_j q_{jit} + \sum_j \sum_k \delta_{jk} p_{kit} q_{jit} + w_{it} \tag{7c}$$

This model is an extended version of model 2, with interactions between farm-specific variables and the input variables in the stochastic frontier. This approach is similar to the approach by Pascual (2005) to test the bidirectional effect of soil fertility (also an environmental input) with potential simultaneous effects on frontier output and TE.

Table 2 shows the results of various hypothesis tests regarding the specification of the three models.

[TABLE 2]

Given the specification of model 1, the null hypothesis that deviations from the frontier are insignificant (technical inefficiency is absent), i.e. H_0: $\gamma = 0$, is strongly rejected by the data, parameter γ being defined as $\gamma \equiv \sigma_u^2/(\sigma_v^2 + \sigma_u^2)$. The hypotheses of a time stationary frontier (no technical change), H_0: $\beta_6 = 0$, and deviations (technical inefficiency effects) as time invariant, H_0: $\eta=0$, are also rejected by the data at any meaningful significance level. In addition, the half-normal distribution is not an inadequate representation of the distribution of the technical inefficiency effects, i.e. H_0: $\mu = 0$ cannot be rejected at the 5% significance level. Under model 2, the null hypotheses that inefficiency is absent, H_0: $\gamma=\delta_0=\delta_j=\delta_{jk}=0$, and that there is no technical change, H_0:$\beta_6=0$, are both rejected as well. Additionally, the hypothesis that the neutral specification of the model outperforms model 3 (H_0:

$\delta_{jk}=0$) is also rejected. Similarly, the null for no year interaction with the explanatory variables in the inefficiency sub-model, H_0: $\delta_{6k}=0$ can be rejected. Therefore, the results presented in Table 3 correspond to models 1 and 3.

[TABLE 3]

Battese and Broca (1997) derive the elasticity of crop output with respect to k^{th} input variable (c.f. appendix B). The elasticity of mean output with respect to the k^{th} input variable has two components: (i) the elasticity of frontier output with respect to the k^{th} input, given by the estimated β_k parameters and (ii) the elasticity of TE with respect to the k^{th} input.[10] The mean output, frontier and efficiency elasticities for each of the variable inputs, averaged throughout the 1989-2000 period, are presented in Table 4.

[TABLE 4]

Change in crop supply can be investigated by obtaining estimates of the time derivative of the realized mean crop output. From Table 3 it can be seen that the estimated time coefficient is significantly different from zero, and points towards technical progress regarding frontier crop output of about 5% per annum. Additionally, the rate of productivity growth over the period can be decomposed into (i) technical change (or technical progress) in the frontier and (ii) TE change (Battese et al. 2000). This is depicted in Figure 2 for the whole period under scrutiny. According to the data, there has been technical progress in frontier output reaching 5.7% in 2000. This result is consistent with the results by Thirtle and Holding (2003) for cereal farms in the UK who find that that the frontier of efficient performance has moved forward indicating that some farms are exhibiting high productivity growth, attributed to technology improvement (at over 5% per year in the UK) (Thirtle and Holding, 2003). It can also be noted from the analysis that the rate of technical change along the frontier is positive and it has been non-declining (about 3.7% per year). In fact, the change in technical efficiency has been positive throughout the period, and may reflect the turnover of farms during the 1990s' where smaller less efficient farms have been leaving the sector and larger, more efficient ones are increasing in scale

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[10] Here we assume that a farm's input decisions are not influenced by its productivity. Otherwise, estimating the stochastic production frontier regression without considering this kind of endogeneity would yield inconsistent coefficient estimates. The size of this inconsistency may depend on the slope parameters δ_i and γ as well as the variances of the error terms. Hence, the model assumes that the choice of inputs used to maximize marketable output is only subject to 'human error' which in turn is uncorrelated with the error specification in the stochastic frontier model.

(data for 1997-2002 show average cereals area per farm increasing by around 14% from 48 to 55 hectares) (Defra, 2003).

[FIGURE 2]

It can be observed that for the whole period, biodiversity is positively affecting mean output levels even though greater levels of biodiversity appear to have negatively affected TE in the sector (Table 4). This has also occurred with the application of fertilisers and more dramatically with the use of farm labour. Regarding the latter, the negative effect on efficiency seems to outweigh the positive effect on the frontier, implying an excessive use of labour in cereal farming. By contrast, the use of pesticides shows a relatively large mean output elasticity due to its positive effect both on the frontier and on TE. This impact on mean output is consistent with a range of other studies that suggest underutilization of pesticides in crop production (e.g. see Lansink and Silva, 2004) while the increase in technical efficiency may point towards the role of pesticides in reducing output variability by controlling damaging pest outbreaks.

[FIGURE 3]

What is of more interest here is the effect of the evolution of the stock of biodiversity proxied by BI (z_t), on the levels of 'frontier output' as this is more directly associated with optimal marketable crop output as described in the theoretical model (variable y_t). The results as depicted in Figure 3 are consistent with the hypothesis from the theoretical model, that there is a positive, although declining, effect. The frontier elasticities with respect to BI are positive and have tended to decrease at a rate of 0.06% per annum. It also appears that the effect of the stock of biodiversity on TE has been different before and after 1996. While there is initially a negative elasticity of TE, after 1996 the elasticity becomes positive reaching 0.15 in 2000. The net effect of biodiversity through the impacts on both frontier output and TE indicates that while until 1993 (the year after broad environmental payments were introduced in the farming sector) higher levels of biodiversity were associated with declining mean yields (average elasticity of -0.1), after the incorporation of the environmental payments for biodiversity conservation the impact on mean output has reversed with an elasticity in 2000 of 0.26. These results suggest that biodiversity conservation schemes have not undermined the productive performance of the cereal sector.

5. Conclusions

A distinguishing characteristic of modern agricultural landscapes is the increasing size and homogeneity of crop monocultures. The concerns for the potentially negative environmental effects of monocultures are well established, although relatively less attention is being paid to the economic effects of biodiversity loss. While ecologists mostly agree that increased intensification is a driver of biodiversity loss, the feedback effects on productivity are less well understood. On the one hand increasing the number of species on a farm may reduce productivity levels of the main crop in the short run through greater competition for abiotic and biotic resources. On the other hand, biodiversity, by providing ecological services (e.g. through pollination, soil nutrient enhancement, and integrated pest control) may increase agricultural output in the longer run (Jackson et al.; *forthcoming*).

This paper has explored one key link between conservation of biodiversity and crop output in the context of a specialised intensive farming system. Departing from an agroecological perspective, a behavioural model is used to set out the hypothesis that biodiversity can support increased marketable output in the longer run, by outward shifts in the production frontier. The key factors behind this theoretical result are; 1) an agricultural technology in which there is a positive relationship between biodiversity and agricultural productivity; and 2) decision maker preferences that reflect this positive relationship and generate resource allocation decisions that support it.

The empirical analysis to test this hypothesis is based on an output distance function approach using data from cereal farms in England for the period 1989-2000. The econometric analysis cannot reject our hypothesis. This has important implications for the design of agri-environmental policy as it suggests that the introduction of biodiversity conservation policies can represent a win-win scenario. That is, this study supports the claim that biodiversity in agricultural landscapes can be enhanced without negatively affecting agricultural productivity in already very intensified agricultural systems if the correct incentives are put in place. In a way these empirical results complement the findings of McInerney et al (2000) that additional conservation investment induced by the agri-environmental policy system can generate additional efficiency benefits for farmers and society at large through supporting and enhancing agricultural multifunctionality.

These results would support the view that increasing productivity is consistent with biodiversity conservation and that policies to promote sustainable agriculture should focus on developing high yielding sustainable technologies with their attendant advantages for food security and land use. This is not inconsistent with the current structure of the reformed schemes of the CAP in that biodiversity conservation is encouraged, at least in principle, at all levels.[11] What remains however is a range of uncertainties about how the implementation of these schemes might be fine

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[11] The cross compliance provisions under the Single Farm Payment ensure adherence to principles of good agricultural practice that support conservation while basic and more advanced conservation practices are specifically encouraged under the Entry Level and Higher Level Stewardship schemes, respectively.

tuned to deliver a truly multifunctional agro-ecosystem that is following a trajectory of sustainable intensification.

This is an area of promising research that clearly needs to expand in interdisciplinary scope in order to provide greater focus especially on refining the integration between ecologically meaningful biodiversity information and economically consistent data at both the farm and the landscape scale. Landscape analysis is also important as the interactions between farmers and off-farm species should be taken into account in both theoretical and empirical research in the expanding topic of agrobiodiversity economics.

References:

Aigner, D., Lovell, C.A.K., and Schmidt, P., 1977. ‘Formulation and estimation of stochastic frontier production function models’. Journal of Econometrics, 6, 21 –37.

Altieri, P., 1999. ‘The ecological role of biodiversity in agroecosystems’. Agriculture, Ecosystems and Environment, 74, 19–31.

Barnes, A., (2002) ‘Publicly-funded UK agricultural R&D and “social” total factor productivity’ Agricultural Economics, 27, 65-74.

Battese, G.E., Heshmati, A., and Hjalmarsson, L., 2000. ‘Efficiency of labour use in the Swedish banking industry: a stochastic frontier approach’. Empirical Economics, 25, 623-640.

Battese, G.E., and Broca S., 1997. ‘Functional forms of stochastic frontier production functions and models for technical inefficiency effects: a comparative study for wheat farmers in Pakistan’. Journal of Productivity Analysis, 8, 395-414.

Battese, G.E., and Coelli, T.J., 1992. ‘Frontier production functions, technical efficiency and panel data: with application to paddy farmers in India’. Journal of Productivity Analysis, 3, 153-169.

Battese, G.E., and Coelli, T.J., 1995. ‘A model for technical inefficiency effects in a stochastic frontier production function for panel data’. Empirical Economics 20, 325-332.

Coelli, T.J., 1995. ‘Estimators and hypothesis tests for a stochastic frontier function: A Monte Carlo analysis’. Journal of Productivity Analysis, 6(3):247-268

Coelli, T.J., 1996. ‘A guide to FRONTIER v.4.1: A computer program for stochastic frontier production and cost function estimation’. Working Paper, vol. 7/96. Centre for Efficiency and Productivity Analysis. Department of Econometrics, University of New England, Australia.

Defra, 2002. ‘Farm Incomes in the United Kingdom 2000/01’. London: Department for Environment, Food and Rural Affairs.

Defra, 2003. ‘Agriculture in the United Kingdom 2002’, HMSO. London: Department for Environment, Food and Rural Affairs.

Di Falco, S., and Perrings, C., 2005. 'Crop biodiversity, risk management and the implications of agricultural assistance'. Ecological Economics, 55(4), 459-466.

Di Falco, S., and Perrings, C., 2003. 'Crop Genetic Diversity, Productivity and Stability of Agroecosystems: A Theoretical and Empirical Investigation'. Scottish Journal of Political Economy, 50(2), 207-216.

Farrell, M., 1957. 'The measurement of productivity efficiency'. Journal of the Royal Statistical Society, CXX (3), 253– 290.

Green, R.E., Cornell, S.J., Scharlemann, P.W., and Balmford, A., 2005. 'Farming and the fate of wild nature'. Science, 307, 550-555.

Haines-Young, R.H., Barr, C.J., Firbank, L.G., Furse, M., Howard, D.C., McGowan, G., Petit, S., Smart, S.M., and Watkins, J.W., 2003. 'Changing landscapes, habitats and vegetation diversity across Great Britain'. Journal of Environmental Management, 67(3), 267-281.

Haines-Young, R.H., Barr, C.J., Black, H.I.J., Briggs, D.J., Bunce, R.G.H., Clarke, R.T., Cooper, A., Dawson, F.H., Firbank, L.G., Fuller, R.M., Furse, M.T., Gillespie, M.K., Hill, R., Hornung, M., Howard, D.C., McCann, T., Morecroft, M.D., Petit, S., Sier, A.R.J., Smart, S.M., Smith, G.M., Stott, A.P., Stuart, R.C., and Watkins, J.W., 2000. 'Accounting for nature: assessing habitats in the UK countryside'. Department of the Environment Transport and the Regions, London.

Jackson, L., Pascual, U., and Hodgkin, T., (forthcoming). 'Utilizing and conserving agrobiodiversity in agricultural landscapes'. Agriculture, Ecosystems and the Environment.

Jackson, L., Bawa, K., Pascual, U., and Perrings, C., 2005. 'Agrobiodiversity: A new science agenda for biodiversity in support of sustainable agroecosystems'. DIVERSITAS report. N. 4. 40 pp. Paris.

Kleijn, D., and Sutherland, W.J., 2003. 'How effective are European agri-environment schemes in conserving and promoting biodiversity?' J. Applied Ecology, 40, 947–969.

Kodde, D., and Palm, A., 1986. 'Wald criteria for jointly testing equality and inequality restriction'. Econometrica, 54 (5), 1243– 1248.

Lansink, A.O., and Reinhard, S., 2004. 'Investigating technical efficiency and potential technological change in Dutch pig farming'. Agricultural Systems, 79(3), 353-367.

McInerney, J., Barr, D., MacQueen, G., and Turner, M., 2000. 'What's the Damage? A Study of Farm Level Costs of Managing and Maintaining the Countryside'. Special Studies in Agricultural Economics No. 51, Agricultural Economics Unit, University of Exeter.

Meeusen, W., and van den Broeck, J., 1977. 'Efficiency estimation from Cobb-Douglas production functions and composed error'. International Economic Review, 81, 435-444.

Omer, A. A. L., 2004. ‘An Economic Analysis of Interactions between Agriculture and Biodiversity’, Thesis submitted for the degree of Doctor of Philosophy in the Faculty of Social Sciences and Law, University of Manchester.

Pascual, U., 2005. ‘Land use intensification potential in slash-and-burn farming through improvements in technical efficiency’. Ecological Economics, 52(4), 497-511.

Pascual, U., and Perrings, C., (forthcoming). ‘The economics of in-situ biodiversity conservation in agricultural landscapes’. Agriculture, Ecosystems and the Environment.

Pascual, U., Russell, N.P., and Omer, A.A. 2003. ‘Does Loss of Biodiversity Compromise Productivity in Intensive Agriculture?’ Discussion Paper N. 0322, School of Economic Studies, University of Manchester.

Perrings, C., Jackson, L., Bawa, K., Brussaard, L., Brush, S., Gavin, T., Papa, R., Pascual, U., de Ruiter, P., 2006. ‘Biodiversity in agricultural landscapes: Saving natural capital without losing interest’, Conservation Biology, 20(2), 263-264.

Shephard, R., 1970. ‘Theory of Costs and Production Functions’. Princeton University Press, Princeton.

Smale, M., Hartell, J., Heisey, P.W., Senauer, B., 1998. ‘The contribution of genetic resources and diversity to wheat production in the Punjab of Pakistan’. American Journal of Agricultural Economics, 80, 482-493.

Stoate, C., Boatman, N.D., Borralho, R.J., Rio Carvalho, C., de Snoo, G.R., and Eden, P., 2001. ‘Ecological impacts of arable intensification in Europe’, Journal of Environmental Management, 63, 337–365.

Thirtle, C. and Holding, J., (2003). ‘Productivity of UK agriculture: causes and constraints’. Published final report on DEFRA project ER0001/3. London: Department for Environment, Food and Rural Affairs.

Thirtle, C., and Bottomley, P., (1992), ‘Total factor productivity in UK agriculture, 1967-90’, Journal of Agricultural Economics, 43(3), 381-400.

Tilman, D., Wedin, D., and Knops, J., 1996. ‘Productivity and sustainability influenced by biodiversity in grasslands ecosystems’. Nature, 379, 718–720.

Tilman, D., Reich, P., Knops, J., Wedin, D., Mielke, T., and Lehman, C., 2001. ‘Diversity and productivity in a long-term grassland experiment’. Science, 294, 843-845.

Tscharntke, T., Klein, A.M., Kruess, A., Steffan-Dewenter, I., and Thies, C., 2005. ‘Landscape perspectives on agricultural intensification and biodiversity ecosystem service management’. Ecology letters, 8(8), 857-874.

Waldman, D., 1982.. ‘A stationary point for the stochastic frontier likelihood.’ Journal of Econometrics, 18, 275-279.

Wenum, J., Buys J., and Wossink, A., 1999. ‘Nature Quality Indicators in Agriculture’, in Brouwer, F. and B. Crabtree, ed., Environmental Indicators and Agricultural Policy. Wallingford: CABI Publishers.

Wilson, P., Hadley, D., and Asby, C., 2001, ‘The influence of management characteristics on the technical efficiency of wheat farmers in eastern England’. Agricultural Economics, 24, 329-338.

Winter, M., 2000. ‘Strong policy or weak policy? The environmental impact of the 1992 reforms to the CAP arable regime in Great Britain’, Journal of Rural Studies, 16, 47-59

Appendix

Appendix A: The theoretical model

The Maximum Principle

From the Hamiltonian (c.f. equation 4), eliminating the time subscript for convenience, the first order conditions for an optimal interior solution are:

$$U_y - \alpha_2 \varphi = 0 \tag{A1a}$$

$$U_b b_x + \varphi(\alpha_2 f_x - \alpha_3) = 0 \tag{A1b}$$

$$\dot{\varphi} = -U_b b_z - \varphi(\alpha_2 + \alpha_2 f_z - \rho) \tag{A1c}$$

$$\dot{z} = \alpha_1 z + \alpha_2 [f(.) - y] - \alpha_3 x \tag{A1d}$$

Condition (A1a) establishes that the current shadow value of biodiversity (φ) is positive, while (A1b) states that x should be allocated such that the marginal utility and disutility of artificial input use are balanced. For an interior solution, the bracketed term $(\alpha_2 f_x - \alpha_3)$ is positive as φ is positive and the first term is unambiguously negative. Equation (A1c) is the standard non-arbitrage condition which dictates that for an optimal solution, no gain in utility can be achieved by reallocating natural capital in the form of biodiversity from one period to another. This occurs when the current marginal return to z equals its marginal cost.

It can be shown that the current value Hamiltonian is maximised and that the solution of the first order conditions leads to a steady state solution marked as ($\bar{z}, \bar{y}, \bar{x}, \bar{\varphi}$) and that this is reachable from the initial state condition $z(0) = z_0$. Thus, there is an implicit terminal state $\lim_{t \to \infty} z_t = \bar{z}(\phi)$ where ϕ is a vector of exogenous parameters (see Pascual et al. 2003, for further details).

Steady state and optimal paths for y_t and z_t:

The following differential equation for y_t is derived from the basic solution:

$$\dot{y} = -\frac{U_y}{U_{yy}} \left[\alpha_1 - \rho + \alpha_2 f_Z - (\alpha_2 f_X - \alpha_3) \frac{b_z}{b_x} \right] \tag{A2}$$

which together with the evolution of biodiversity (A1d) describes the dynamic system of equations in (z_t, y_t) space: $\dot{y} = h(z_t, y_t)$ and $\dot{z} = g(z_t, y_t)$. Since at equilibrium, the steady state occurs when $\dot{z} = \dot{y} = 0$. The isoclines for z_t and y_t are respectively given by:

$$g(z, y) \equiv \alpha_1 z + \alpha_2 f(.) - \alpha_2 y - \alpha_3 x = 0 \tag{A3a}$$

$$h(z, y) \equiv \alpha_1 - \rho + \alpha_2 f_z - (\alpha_2 f_x - \alpha_3) \frac{b_z}{b_x} = 0 \tag{A3b}$$

Qualitatively, the two demarcation curves ($\dot{z} = 0$ and $\dot{y}=0$) divide the phase space into four regions, with a different mix of time derivatives for y_t and z_t (c.f. Figure 1).

The steady state values of y_t and z_t are derived from the linearised system of equations (A3a-A3b). The Jacobian matrix, J_S, evaluated at the steady state $(\bar{z}, \bar{y})$ is:

$$J_S = \begin{bmatrix} \underset{(+)}{g_z} & \underset{(-)}{g_y} \\ \underset{(-)}{h_z} & \underset{(+)}{h_y} \end{bmatrix} \equiv$$

$$\begin{bmatrix} \alpha_1 + \alpha_2 f_z + (\alpha_2 f_x - \alpha_3)x & -\alpha_2 + (\alpha_2 f_x - \alpha_3)x \\ \alpha_2 f_{zz} - \left[\dfrac{(\alpha_2 f_x - \alpha_3) b_x b_{zz} + \alpha_2 b_x b_z f_{xx} x_z - (\alpha_2 f_x - \alpha_3) b_z b_{xx}}{b_x^2} \right] & -\dfrac{[\alpha_2 b_x b_z f_{xx} - (\alpha_2 f_x - \alpha_3) b_z b_{xx}] x_y}{b_x^2} \end{bmatrix} < 0 \tag{A4}$$

the sign of the J_s is negative provided that the slope of the $\dot{y} = h(z_t, y_t)=0$ isocline is steeper than that of the $\dot{z} = g(z_t, y_t)=0$ isocline. The steady state equilibrium for biodiversity and output is given, respectively, by (see):

$$\bar{z} = -\tfrac{1}{|J_S|} \left[h_y (\alpha_1 z + \alpha_2 f(\cdot) - \alpha_2 y - \alpha_3 x) + g_Y \frac{U_y}{U_{yy}} \left(\alpha_1 - \rho + \alpha_2 f_z - (\alpha_2 f_z - \alpha_3) \frac{b_z}{b_x} \right) \right] \tag{A5a}$$

$$\bar{y} = -\tfrac{1}{|J_S|} \left[-h_Z (\alpha_1 z + \alpha_2 f(\cdot) - \alpha_2 y - \alpha_3 x) - g_Z \frac{U_y}{U_{yy}} \left(\alpha_1 - \rho + \alpha_2 f_z - (\alpha_2 f_x - \alpha_3) \frac{b_z}{b_x} \right) \right] \tag{A5b}$$

The comparative dynamic analysis:

The time paths of z_t and y_t are identified as:

$$\begin{bmatrix} \dot{z} \\ \dot{y} \end{bmatrix} = \begin{bmatrix} \dot{z}_z & \dot{z}_y \\ \dot{y}_z & \dot{y}_y \end{bmatrix} \begin{bmatrix} z-\bar{z} \\ y-\bar{y} \end{bmatrix} = J_d \begin{bmatrix} z-\bar{z} \\ y-\bar{y} \end{bmatrix} \tag{A6a}$$

where:

$$\dot{z}_z = \frac{\partial \dot{z}}{\partial z} > 0, \quad \dot{z}_y = \frac{\partial \dot{z}}{\partial y} < 0, \quad \dot{y}_z = \frac{\partial \dot{y}}{\partial z} < 0, \quad \dot{y}_y = \frac{\partial \dot{y}}{\partial y} > 0$$

The general solution is in turn given as:

$$\begin{bmatrix} z_t \\ y_t \end{bmatrix} = \begin{bmatrix} \bar{z} \\ \bar{y} \end{bmatrix} + \begin{bmatrix} z_v \\ y_v \end{bmatrix} \tag{A6b}$$

where $\begin{bmatrix} \bar{z} \\ \bar{y} \end{bmatrix}$ represents the steady state equilibrium, and $\begin{bmatrix} z_v \\ y_v \end{bmatrix}$ represents the complementary functions based on the reduced equations of the system

$$\begin{bmatrix} \dot{z} \\ \dot{y} \end{bmatrix} - \begin{bmatrix} \dot{z}_z & \dot{z}_y \\ \dot{y}_z & \dot{y}_y \end{bmatrix} \begin{bmatrix} \bar{z} \\ \bar{y} \end{bmatrix} = \begin{bmatrix} 0 \\ 0 \end{bmatrix} \tag{A6c}$$

For a dynamically stable equilibrium, $\begin{bmatrix} z_v \\ y_v \end{bmatrix} \to 0$ as $t \to \infty$. The optimal transitional paths to the steady state is given by:

$$\begin{bmatrix} z(t; z_0, \phi) \\ y(t; z_0, \phi) \end{bmatrix} = \begin{bmatrix} \bar{z} \\ \bar{y} \end{bmatrix} + v_1 \begin{bmatrix} 1 \\ k \end{bmatrix} e^{r_1 t} + v_2 \begin{bmatrix} 1 \\ k \end{bmatrix} e^{r_2 t} \tag{A6d}$$

for $v_1 = z(0) - \bar{z}$ when $v_2 = 0$, $k_1 = \dfrac{-\dot{y}_z - (r_1 - \dot{z}_z)}{r_1 - \dot{y}_y + \dot{z}_y} < 0$, and r_1 and r_2 being characteristic roots that for a saddle equilibrium need to be of opposite signs, e.g. $r_1 < 0$ and $r_2 > 0$. The definite solution of the dynamic system of the model, is given as:

$$\begin{bmatrix} z(t; z_0, \phi) \\ y(t; z_0, \phi) \end{bmatrix} = \begin{bmatrix} \bar{z} \\ \bar{y} \end{bmatrix} + \left[z_0 - \bar{z} \right] \begin{bmatrix} 1 \\ k_1 \end{bmatrix} e^{r_1 t} \tag{A7}$$

This information allows the analysis of the comparative dynamics, i.e. the effect of z_t on levels of y_t. along its optimal transitional time path. It provides the main hypothesis that *the effect of biodiversity conservation is to increase marketable output along the optimal transitional path towards the new steady state equilibrium (at a declining rate).*

$$\left. \frac{dy(t; z_0, \phi)}{d\bar{z}} \right|_{z_0 = \bar{z}} = -k_1 e^{r_1 t} > 0 \tag{A8}$$

Note that in the long run, for r_1<0, the term $k_1 e^{r_1 t}$ approaches zero as time goes to infinity.

Appendix B. The empirical model:

Under model 3, the elasticity of crop output with respect to k^{th} input variable can be calculated as

$$\frac{\partial \ln E(y_{it})}{\partial p_k} = \frac{\partial \beta p}{\partial p_k} - C_{it}\left(\frac{\partial \mu_{it}}{\partial p_k}\right) \tag{B1a}$$

where

$$\mu_{it} = \delta_0 + \sum_j \delta_j q_{jit} + \sum_j \sum_k \delta_{jk} p_{kit} q_{jit} \tag{B1b}$$

$$C_{it} = 1 - \frac{1}{\sigma}\left\{\frac{\phi(\frac{\mu_{it}}{\sigma} - \sigma)}{\varphi(\frac{\mu_{it}}{\sigma} - \sigma)} - \frac{\phi(\frac{\mu_{it}}{\sigma})}{\varphi(\frac{\mu_{it}}{\sigma})}\right\}_{it} \tag{B1c}$$

and ϕ and φ represent the density and distribution functions of the standard normal random variable, respectively.

It follows from Battese and Broca (1997) that the elasticity of *frontier output* with respect to the k^{th} input, $\frac{\partial \beta p}{\partial p_k}$, is different from the elasticity of *TE* with respect to the k^{th} input: $-C_{it}\left(\frac{\partial \mu_{it}}{\partial p_k}\right)$.

Lastly, the decomposition of the rate of change of mean crop output with respect to time is given by equation (B2):

$$\frac{\partial \ln E(y)}{\partial t} = \frac{\partial \beta p}{\partial t} - C\left(\frac{\partial \mu}{\partial t}\right) \tag{B2}$$

where the first and second terms in the right-hand-side of the equation represents the impact of exogenous technical change and the change in TE levels, respectively.

Figure 1. Saddle point equilibrium in the biodiversity–marketable output (z_t,y_t) phase space

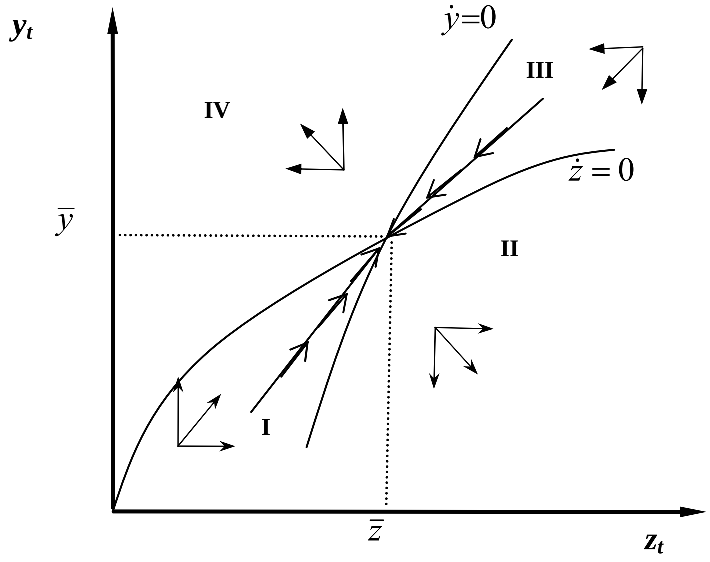

Figure 2: Technical change and productivity growth (1989-2000)

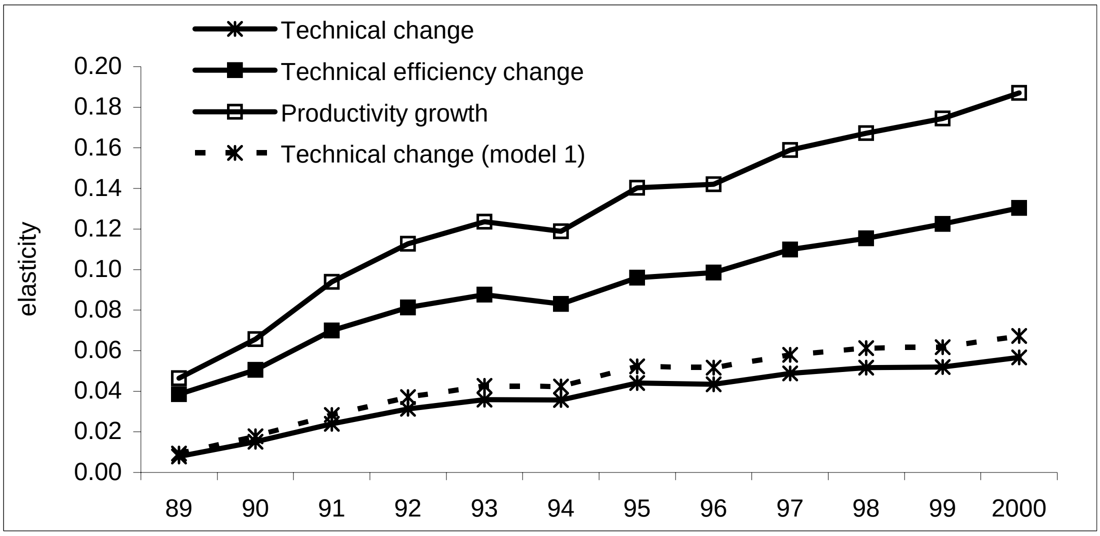

Figure 3. Change in elasticity of output with respect to Biodiversity (1989-2000)

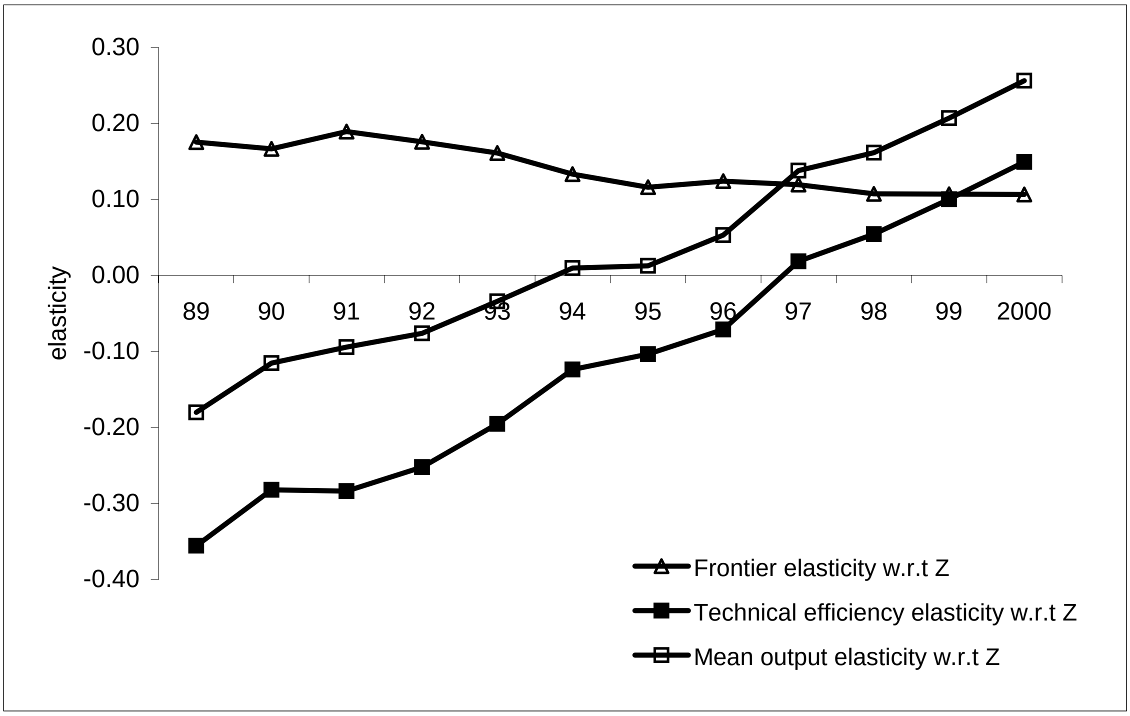

Table 1. Summary statistics for variables in the stochastic frontier models for cereal farmers in the East of England

Variable	Mean	St. Dev	Minimum	Maximum
Crop output (£/ha/API)	874.85	194.49	261.55	5141.61
Biodiversity index (BI)	13.63	1.04	9.99	16.22
Fertiliser application (£/ha/API)	87.55	32.78	0.68	571.90
Labour application (£/ha/API)	163.87	92.56	3.34	1093.45
Machinery application (£/ha/API)	208.98	93.51	12.55	1382.01
Pesticide application (£/ha/API)	91.41	27.57	1.99	345.62
Farm area (ha)	178.58	137.21	7.89	1008.18
Farmer's age (years)	50.91	10.52	27	79
Environmental Payments (£/ha/API)	2.77	11.00	0	93.63
Share of hired labour from total labour (0-1)	0.44	0.25	0	1

A total of 2788 observations were obtained in an unbalanced panel of approximately 230 different specialist cereal farms over the period 1989-2000. API: Agricultural Price Index for the relevant inputs (or output) and year.

Table 2. Generalized Likelihood-Ratio Tests for SPF models for Cereal Farmers in the East of England (1989-2000)

Null Hypothesis	Log likelihood	LR statistic	CV* (5%)
Model 1	1604.34		
H_0: $\gamma = 0$	1007.31	1194.07	7.05
$H_0 : \beta_6 = 0$	1311.12	586.44	3.84
$H_0 = \eta = 0$	1586.76	35.17	3.84
$H_0 = \mu = 0$	1602.66	3.36	3.84
Model 2	1261.79		
H_0: $\gamma = \delta_0 = \delta_j = 0$	1007.31	508.97	16.27
$H_0 = \beta_1 = 0$	1257.26	9.07	3.84
$H_0 = \beta_6 = 0$	1084.93	353.72	3.84
$H_0 : \delta_1 = \ldots = \delta_6 = 0$	1159.36	204.87	12.59
Model 3	1361.13		
H_0: $\gamma = \delta_0 = \delta_j = \delta_{jk} = 0$	1007.31	707.65	55.19
$H_0 = \beta_1 = 0, \delta_{1j} = 0, j = 1,....,6$	1352.69	16.87	14.07
$H_0 : \beta_6 = 0, \delta_{6j} = 0, j = 1,....,6$	1177.02	368.23	14.07
$H_0 : \delta_{jk} = 0, k, j = 1,.....6$	1261.79	198.67	43.77
$H_0 : \delta_{6k} = \delta_{k6} = 0, k = 1,...,6$	1318.76	84.73	11.07
$H_0 : \delta_{6j} = \delta_{6j} = 0, j = 1,...,6$	1313.58	95.09	11.07
$H_0 : \delta_{3k} = \delta_{4k} = 0, k = 1,...,6$	1341.35	39.56	19.92

*Critical Values are also obtained from Kodde and Palm (1986). LR: Likelihood Ratio.

Table 3. MLE parameter estimates of the generalized Cobb-Douglas SPF models 1 and 3

		Model 1		Model 3	
		Coefficient	t-ratio	Coefficient	t-ratio
Constant	β_0	1.81	23.49	1.69	12.33
p1: biodiversity	β_1	0.07	2.58	0.13	2.58
p2: fertilizer	β_2	0.04	5.17	0.05	4.03
p3: labour	β_3	0.02	2.91	0.01	2.91
p4: machinery	β_4	0.08	8.56	0.05	4.16
p5: pesticides	β_5	0.14	14.47	0.14	11.63
p6: time	β_6	0.05	35.91	0.04	31.67
Inefficiency model					
constant	δ_0			-0.60	-3.62
q1: age	δ_1			-0.05	-2.47
q2: environmental pay	δ_2			0.10	3.50
q3: d1	δ_3			-0.68	-0.73
q4: hired labour	δ_4			0.38	0.42
q5: d2	δ_5			0.71	0.77
q6: time	δ_6			0.29	2.16
p1.q1	δ_{11}			0.02	2.78
p1.q2	δ_{12}			-0.04	-3.50
p1.q3	δ_{13}			0.42	1.18
p1.q4	δ_{14}			-0.04	-0.11
p1.q5	δ_{15}			-0.24	-0.70
p1.q6	δ_{16}			-0.08	-1.66
p2.q1	δ_{21}			0.01	4.74
p2.q2	δ_{22}			-0.01	-2.83
p2.q3	δ_{23}			0.75	5.16
p2.q4	δ_{24}			0.22	2.41
p2.q5	δ_{25}			-0.20	-2.62
p2.q6	δ_{26}			-0.04	-6.27
p3.q1	δ_{31}			0.00	3.09
p3.q2	δ_{32}			0.00	1.81
p3.q3	δ_{33}			-0.19	-2.43
p3.q4	δ_{34}			-0.19	-3.33
p3.q5	δ_{35}			-0.05	-1.29
p3.q6	δ_{36}			0.02	4.02
p4.q1	δ_{41}			0.00	1.29
p4.q2	δ_{42}			-0.01	-2.93
p4.q3	δ_{43}			0.11	0.92
p4.q4	δ_{44}			-0.46	-5.14
p4.q5	δ_{45}			0.24	3.76
p4.q6	δ_{46}			0.00	-0.50
p5.q1	δ_{51}			0.01	5.45

p5.q2	δ_{52}			0.00	0.79
p5.q3	δ_{53}			0.10	0.92
p5.q4	δ_{54}			-0.05	-0.58
p5.q5	δ_{55}			-0.38	-5.81
p5.q6	δ_{56}			-0.05	-6.74
p6.q1	δ_{61}			0.00	1.63
p6.q2	δ_{62}			0.00	2.10
p6.q3	δ_{63}			-0.02	-1.59
p6.q4	δ_{64}			-0.05	-5.30
p6.q5	δ_{65}			-0.06	-4.86
p6.q6	δ_{66}			-0.01	-13.34
Variance Parameters					
σ^2		0.05	11.10	0.08	17.05
γ		0.73	28.35	0.86	63.98
η		0.04	6.00		
Log-likelihood		1586.76		1361.13	

Note: *d1*: Dummy variable for participation in agri-environmental schemes (1 if participating, 0 otherwise); *d2* dummy variable for hired labour (1, if positive expenditures on hired labour, 0 otherwise)

Table 4. Average crop output elasticities with respect to all the inputs in model 3 (1989-2000)

Variable	Frontier output elasticity	Technical efficiency elasticity	Mean output elasticity
Biodiversity	0.13	-0.10	0.04
Fertiliser	0.05	-0.02	0.03
Labour	0.01	-0.05	-0.03
Machinery	0.05	0.00	0.05
Pesticides	0.14	0.14	0.28
Time	0.04	0.09	0.13