Spatially explicit integrated modeling and economic valuation of climate change induced land use change and its indirect effects[1]

Ian Bateman[2], Amy Binner[2], Emma Coombes[2], Brett Day[2], Silvia Ferrini[2,3], Carlo Fezzi[2], Michael Hutchins[4], Andrew Lovett[2] and Paulette Posen[5].

Abstract:

Arguably the greatest challenge to contemporary research is to capture the inter-relatedness and complexity of the real world environment within models so at to better inform decision makers of the accurate and complete consequences of differing options. The paper presents an integrated model of the consequence of climate change upon land use and the secondary and subsequent effects arising subsequently. The model predicts the shift in land use which climate change is likely to induce and the impacts upon farm gross margins arising from this. However, both the direct driver of climate change and the induced shift in land use patterns will cause secondary effects upon the water environment for which agriculture is the major source of diffuse pollution. We model the consequent impact of changes in such pollution upon water ecology showing that these will be spatially specific. These impacts are likely to cause further knock-on effects upon the recreational benefits of water environments and these are assessed using a spatially explicit revealed preference database. Taken together this analysis permits a holistic examination of a much wider range of effects and net value consequences arising from climate change impacts upon land use.

Keywords:

Integrated models; land use; agriculture; climate change; water quality; recreation; economics; spatial analysis;

---

[1] This work was funded by the SEER project (ESRC Funder Ref: RES-060-25-0063).

[2] CSERGE, School of Environmental Sciences, University of East Anglia, Norwich, NR4 7TJ, UK. Contact i.bateman@uea.ac.uk. Ian Bateman is also Adjunct Professor in the Department of Agricultural and Resource Economics at the University of Western Australia, Perth and the Department of Economics, Waikato University.

[3] Department of Economics, University of Siena, Piazza S.Francesco,7 - 53100 Siena, Italy.

[4] Centre for Ecology and Hydrology, Crowmarsh Gifford, Wallingford, OX10 8BB, UK.

[5] Centre for Environment, Fisheries and Aquaculture Science, Pakefield Road, Lowestoft, Suffolk, NR33 0HT, UK.

1. Introduction

Recent years have seen a rapid growth in the development of integrated analyses bringing together the natural and social sciences in an attempt to meet the complex challenge of environmental degradation. Nowhere is this growth more evident than in the field of environmental economics where genuine efforts are being made to reverse more than a century of disciplinary separateness and work with the natural sciences to incorporate the complexities of the natural world within economic analyses. Examples can be drawn from around the world and across a variety of empirical contexts, however, perhaps the most central of foci have been the interactions of land use and water and the challenge of climate change (e.g. Ackerman et al., 2009; Avila-Foucat et al., 2009; Brouwer and De Blois, 2008; dale et al., 2011; Milne et al., 2009; van Ittersum et al., 2008; Wei et al., 2009). However, even here it is fair to say that models are still some way from being fully integrated and secondary impacts are rarely explored. This is unfortunate as the full chain of primary, secondary and further effects have to be incorporated if we are to undertake full cost-benefit analyses of impacts and hence guide policy response. It is within this context that the present study attempts to offer a contribution.

The study presented in this paper attempts to provide integrated models of the consequence of climate change upon arguably the most responsive of all the environments upon which it impacts; land use. Changes in land use will have major direct market and nonmarket impacts in terms of a shift in the productivity of land and hence the optimal mix of crop type and livestock intensity. This of course directly influences the income levels of farm enterprises and we model changes in farm gross margin as a measure of this. However, that change in land use will also have major secondary effects upon other environments, of which the above literature suggests that the water environment will be the most impacted. This impact will most obviously occur because changes in land use will result in a shift in the level and type of diffuse water pollution emanating from agriculture (which is the major source of diffuse pollutants, see Heathwaite et al., 2005). These effects will be further compounded by the direct effect of climate change on the water environment and to some extent either mitigated or elevated by the mixing of different waterways to generate changes in say water nutrient concentrations. A third round effect arises from such shifts in nutrient patterns in that these play a major role in determining the ecological quality of rivers as measured by indicators such as levels of chlorophyll. This effects the macrofauna and flora of rivers which leads to a fourth round effect in terms of the consequent impacts upon the (generally nonmarket) recreational benefits of river and lake environments.

Figure 1 summarizes the web of interlocking effects which forms the focus for our integrated modeling exercise. Ultimately such an analysis allows us to compare the economic costs and benefits of climate change impacts on land use to diverse groups ranging from farmers to recreational walkers. As the figure indicates, this all develops within a pre-existing policy framework which is further conditioned by historical, current and hence expected market forces and the baseline and expected level of natural environment quality. As such this analysis is of direct relevance to the rapidly developing literature on ecosystem services

(Balmford et al. 2002; Barbier, 2007; Bateman, 2009a; Bateman et al., 2011; Bockstael et al. 2000; Boyd and Banzhaf, 2007; CBD, 2006; Chapin, *et al*., 2000; Fisher et al., 2008; Fisher and Turner, 2008; GEF, 1998; Groot de, *et al*. 2002; Howarth and Farber, 2002; Koziell, 2001; Loreau et al., 2006; MA 2005; Mace et al., 2009; Maler et al. 2008; TEEB, 2009; Turner et al., 2010; Wallace, 2007).

Figure 1: Foci of research; Integrated modeling of the impacts of climate change upon land use and consequent secondary and subsequent effects

INSERT FIGURE 1 ABOUT HERE

The remainder of the paper is organized as follows. In the next section we present the spatially sensitive land use model which underpins our initial analysis, reviewing data sources and testing the validity of the model for predicting land use change as a result of any combination of policy, market or environmental drivers at a highly disaggregated spatial scale yet for the entire area of England and Wales. Section 3 applies this model to examine the impact which a medium term climate change scenario is likely to have upon land use and consequent farm incomes. The land use change predictions then form the basis of our analysis of consequences for the water environment considered in Section 4. This culminates in the estimation of a spatially explicit, transferable model for predicting the ecological impacts for the water environment of changes in agricultural land use within a world of altering climates. These predictions are then refined for use within a particular region for which we hold data on recreational demand and associated value. Section 5 focuses upon this area and develops a revealed preference model which encompasses the changes in the ecological quality of rivers derived previously. Application of this model provides estimates of the impact which our climate change scenario is predicted to have upon recreational values. A further analysis considers the recreational value of full implementation of a likely policy response as laid out under the EU Water Framework Directive. Section 6 summarizes and concludes.

2. Land use modeling

This section illustrates the agricultural land use model specification, the data used for its estimation and provides a summary of the main results. Due to space limitations, we limit this section to a brief overview with a more detailed illustration of the land use modeling approach being given in Fezzi and Bateman (2011).

2.1 Specification

Following and Chambers and Just (1989) we specify the farm profit function as:

$$(1)\quad \pi(\mathbf{p},\mathbf{w},\mathbf{z},l_1,\ldots,l_h) = \max\{\mathbf{p'}\,\mathbf{y} - \mathbf{w'}\,\mathbf{r} : \mathbf{y} \in Y(\mathbf{r},\mathbf{z},l_1,\ldots,l_h)\},$$

where $\mathbf{y}$ is the vector of m outputs, with $\mathbf{r}$ the vector of n inputs, $\mathbf{p}$ the vector of strictly positive output prices, $\mathbf{w}$ the vector of strictly positive input prices, $\mathbf{l}$ the vector of h land use allocations, L the total land available and $\mathbf{z}$ the vector of k other fixed factors (which may include physical and environmental characteristics, policy incentives and constraints, etc.). The farm profit maximization problem can be expressed, without any loss of generality, in terms of profit maximization per unit of land. Indicating with $\mathbf{s}$ the h land use shares corresponding to the land use allocations l, and with $\pi^L(.)$ the profits per unit of land, the optimal land use allocation problem can be written as:

$$(2)\quad \pi^L(\mathbf{p},\mathbf{w},\mathbf{z},L) = \max_{s_1,\ldots,s_h}\{\pi^L(\mathbf{p},\mathbf{w},\mathbf{z},L,s_1,\ldots,s_h) : \textstyle\sum_{i=1}^{h} s_h = 1\}\,.$$

Since the profit per area function is positively linearly homogenous and strictly convex in input and output prices, using the Hotelling's lemma one can derive the output supply (y^L) and input demand (r^L) per area (hereafter we will refer to these quantities as input and output *intensities*) as:

$$(3.a)\quad y_i^{\,L}(\mathbf{p},\mathbf{w},\mathbf{z},L) = \frac{\partial \pi^L(\mathbf{p},\mathbf{w},\mathbf{z},L)}{\partial p_i} = \frac{\pi^L(\mathbf{p},\mathbf{w},\mathbf{z},L,\bar{s}_1,\ldots,\bar{s}_h)}{\partial p_i}\text{, and}$$

$$(3.b)\quad r_j^{\,L}(\mathbf{p},\mathbf{w},\mathbf{z},L) = \frac{\partial \pi^L(\mathbf{p},\mathbf{w},\mathbf{z},L)}{\partial w_j} = \frac{\pi^L(\mathbf{p},\mathbf{w},\mathbf{z},L,\bar{s}_1,\ldots,\bar{s}_h)}{\partial w_j}\,,$$

where the superscript on s indicates the optimal shares, i.e. the shares that satisfy (3). The equations describing the optimal land allocations can be derived by recognizing that land is allocated to the different uses in order to equalize their marginal rent or shadow price. In terms of optimal land use shares this can be written as:

$$(4)\quad \frac{\partial \pi^L(\mathbf{p},\mathbf{w},\mathbf{z},L,\bar{s}_1,\ldots,\bar{s}_h)}{\partial s_i} = 0\text{, for } i = 1,\ldots,h.$$

When these equations are linear in the optimal land allocations, including the constraint that the sum of the shares needs to be equal to one leads to a linear system of h equations in h unknowns which can be solved to obtain the optimal land allocation as a function of $\mathbf{p}$, $\mathbf{w}$, $\mathbf{z}$ and L. For more details see Fezzi and Bateman (2011).

We specify the empirical profit function per hectare as a Normalized Quadratic (NQ) function. Defining with w_n the numeraire good, indicating with $\mathbf{x} = (\mathbf{p}/w_n, \mathbf{w}/w_n)$ the vector of normalized input and output (netput) prices and with $\mathbf{z}^*=(\mathbf{z}, L)$ the vector of fixed factors including policy and environmental drivers and also the total land available L, the NQ profit function can be written as:

$$(5)\ \bar{\pi}^L = \alpha_0 + \sum_{i=1}^{m+n-1} \alpha_i x_i + \frac{1}{2} \sum_{i=1}^{m+n-1} \sum_{j=1}^{m+n-1} \alpha_{ij} x_i x_j + \sum_{i=1}^{h-1} \beta_i s_i + \frac{1}{2} \sum_{i=1}^{h-1} \sum_{j=1}^{h-1} \beta_{ij} s_i s_j + \sum_{i=1}^{k+1} \gamma_i z_i^* +$$

$$+ \frac{1}{2} \sum_{i=1}^{k+1} \sum_{j=1}^{k+1} \gamma_{ij} z_i^* z_j^* + \sum_{i=1}^{m+n-1} \sum_{j=1}^{h-1} \delta_{ij} x_i s_j + \sum_{i=1}^{m+n-1} \sum_{j=1}^{k+1} \phi_{ij} x_i z_j^* + \sum_{i=1}^{h-1} \sum_{j=1}^{k+1} \varphi_{ij} s_i z_j^* ,$$

where $\bar{\pi}^L = \pi / w_n$ is the normalized profit per unit of land. This profit function is linearly homogeneous by construction, and symmetry can be ensured by imposing $\alpha_{ij} = \alpha_{ji}$, $\beta_{ij} = \beta_{ji}$ and $\gamma_{ij} = \gamma_{ji}$. Only h–1 land use shares appear in the profit function since the last one can be computed by difference and it is therefore redundant. Input and output intensities can be derived as in (3.a) and (3.b), whereas the optimal land use shares can be derived by solving the system (4) which contains h–1 equations with the land additivity constraint $\sum_{j=1}^{h} s_j = 1$. The resulting equations are linear function of the output prices, input prices, and fixed factors.

2.2. Estimation

Since micro-data on land use are typically censored (farms are very unlikely to comprise some element of all possible land uses) assuming normal disturbances and implementing ML leads to inconsistent estimates of the land use shares and input and output intensities equations (Amemiya, 1973). We address this issue by specifying a Tobit system of equations (Tobin, 1958) and following Pudney (1989), who suggests treating one of the shares as a residual category, defined by the identity:

$$(6)\ s_h = 1 - \sum_{j=1}^{h-1} s_j ,$$

and estimating the remaining h – 1 equations as a joint system. When the number of equations is higher than three the ML estimation of a Tobit system requires the evaluation of multiple Gaussian integrals which is computationally extremely intensive. In this paper we follow the practical and computationally feasible solution proposed by Yen et al. (2003), who suggest approximating the multivariate Tobit with a sequence of bivariate models, deriving a consistent Quasi Maximum Likelihood (QML) estimator (detailed in Fezzi and Bateman, 2011). We also account for possible heteroskedasticity in the error term allowing the standard errors to vary across observations as a function of a vector of exogenous variables. This QML estimator is consistent, allows the estimation of cross-equation correlations and the imposition of cross-equation restrictions.

2.3. Data sources

In order to correctly assess the financial, policy and environmental drivers of land use change, this analysis employs a unique database, which integrates multiple sources of information dating back to the late 1960s. The resulting data, collected on a 2km^2 grid square (400ha) basis, cover the entirety of England and Wales and encompass, for the past 40 years:

(a) land use shares and livestock number, (b) environmental and climatic determinants, (c) input and output prices, (d) policy and other drivers. However, we do not include yield and profits data, since the necessary information is simply not available at the disaggregated level required by this analysis. Data on agricultural land use hectares and livestock numbers, derived from the June Agricultural Census (JAC) on a $2km^2$ (400 ha) grid square resolution are available on-line from EDINA (www.edina.ac.uk), which aggregates information collected by the Department of Environment, Food and Rural Affairs (DEFRA) and the Welsh Assembly. These data cover the entirety of England and Wales for seventeen, unevenly spaced, years between 1969 and 2006 (in years 2005 and 2006 only Welsh data is available). This yields roughly 38,000 grid-square records each year. Regarding livestock numbers, we distinguish between dairy cows, beef cows and sheep. Concerning agricultural land use types, we explicitly model cereals (including wheat, barley, oats, etc.), oilseed rape, root crops (potatoes and sugar beet), temporary grassland (grass being sown every 3 to 5 years and typically part of an arable crop rotation), permanent grassland (grassland maintained perpetually without reseeding) and rough grazing. These six land use types together cover more than 88% of the total agricultural land within the country. We include the remaining 12% in an “other” land category encompassing horticulture, other arable crops, woodland on the farm, set-aside, bare, fallow and all other land (ponds, paths, etc.). Descriptive statistics for the agricultural land use types and livestock numbers are reported in Table 1 for three illustrative years and for the total dataset.

Table 1: Descriptive statistics, land uses (ha) and livestock numbers (head) per 2km^2 grid square

INSERT TABLE 1 ABOUT HERE

For each 2km^2 grid square we consider a detailed specification of the environmental determinants influencing farmers' decision making. For each grid square, we extract, from the National Soil Resources Institute LandIS database: average annual rainfall (denoted *aar*), autumn machinery working days (*mwd,* a measure of the suitability of the soil for arable cultivation), mean potential evapotranspiration (*pt,* indicating the amount of water that, if available, can be evaporated and transpired), median duration of field capacity (*fc*, reflecting water abundance in the soil), total number of degree days in the growing season (*dd*, from April to September) and mean elevation (*alt*). We also include the share of agricultural land with slope higher than 6 degrees (*smore6*) derived via GIS analysis of the Ordnance Survey, Digital Terrain Model. We also include in the model policy determinants, such as the share of each grid square designated as National Park, Nitrate Vulnerable Zone (NVZ) and Environmentally Sensitive Area (ESA). Further spatial control variables such as the distance to the closest sugar beet factory (to capture transportation costs) and the share of urban area are also included. Finally, we include input prices on a national level, whereas output prices are at a regional level using the agricultural output regional price statistics extracted from the UK Farm Business Survey for years 1982-2000.

2.4 Results

We implement the QML approach to estimate two censored Tobit systems: the 3 livestock intensity (dairy cows, beef cows, sheep) equation system; and the 6 land use shares (cereal, oilseed rape, root crops, temporary grassland, permanent grassland, rough grazing) system. Table 2 reports the final parameter estimates of the land use share equations. The sign and magnitude of the coefficients are consistent with our expectations and the model fit is satisfactory. Focusing on the economic determinants, in the upper part of the table, the own output price effects are always positive and the cross-price effects negative. Considering the environmental determinants of land use, reported in the lower part of the Table, favorable conditions for crop growth (e.g. more machinery working days, flatter land, etc.) increase the share of arable land, in particular of root crops. However, effects are highly non-linear. The coefficients of the livestock equations are not reported here to preserve space, but the results are in line with those of the land use ones (details are in Fezzi and Bateman, 2011).

Table 2: Land use share equations parameter estimates

INSERT TABLE 2 ABOUT HERE

3. From climate change to agricultural land use change

We simulate the land use changes arising from a "naive" climate change scenario obtained by holding all land use determinants (prices, policy, urbanization, etc.) constant[6] and increasing daily average temperature by 1°C. Although official UK climate change predictions estimate that a 1°C increase in mean daily temperature will occur by about 2030 (UKCP, 2009), the simple scenario used in the present paper is purely for illustrative purposes to show how our methodology operates. It does not conform to full UKCP scenarios in that we ignore monthly variation in the rate of climate change. Furthermore it simplifies the impact of climate upon land use in that we hold constant all those variables which are linked to temperature and precipitation (mwd, field capacity, evapotranspiration etc.). Ongoing work relaxes these assumptions. However, despite these caveats, the methodology developed in the present paper and illustrated through our simple climate change simulation, provides all the necessary flexibility required to embrace those more detailed changes. As such we feel that this example provides a useful illustration of that methodology.

Table 3 presents results obtained from our illustrative climate change scenario of a 1°C increase in mean daily temperature. The first two columns of the table list the various crops and livestock activities embraced by our land use model and their respective farm gross margin (FGM) in £/ha or £/head as appropriate. The third column provides the estimated intensity in terms of area or head for each of these activities under the present climate. The fourth column reveals our estimates for our climate change scenario while the final column reports the change induced under the latter scenario.

Table 3: Land uses and livestock numbers changes and FGM/ha as predicted by our land use model

INSERT TABLE 3 ABOUT HERE

Considering arable production, our climate change scenario induces a shift out of cereals and root crops and into more temperature tolerant crops such as oilseed rape. There is an apparent increase in permanent grassland however we have some reservations about the estimated size of this effect which may have been inflated by a decision to not directly model the 'other' land category leaving it as a residual from which permanent grassland may have overly drawn (we discuss this is more detail in Fezzi and Bateman, 2011, and will address this in ongoing work by directly modeling the 'other' land category).

One of the advantages of our modeling framework is that it is highly spatially sensitive. Focusing upon arable production, the left hand panel of Figure 2 uses the example of cereals to illustrate the highly spatially heterogeneous nature of changes to cereals area. This is increasing in the Northern parts of the country, where the warmer temperature will be beneficial to yield, and decreasing in the South where is substituted by other activities.

---

[6] All such determinants are fixed at 2004 levels as this is the last year that data for the entire study area of England and Wales are available at a consistent level.

Figure 2: Change in cereals and in FGM under “naive” climate change

INSERT FIGURE 2 ABOUT HERE

We can translate the predicted changes in land use into variations in measures of farm income. For example we can use the commonly adopted measure of Farm Gross Margin (FGM) which is defined as the difference between revenues from agricultural activities and associated variable costs. Again for illustrative purposes this can be achieved by simply using the average FGM for each activity calculated from the 2004 Farm Business Survey (as reported in Table 3) and apply this to the land use and livestock data in each $2km^2$ grid cell.[7] Results are illustrated in the right hand panel of Figure 2 and show that, under these hypotheses, climate change will be overall beneficial for UK farming incomes (a result which is in line with expectations given that, at present, temperatures are typically below the optimal level for plant growth). However, it will have locally negative impacts, mainly in the North-East part of the country, where there will be a decrease in high-revenue root crops.

4. From land use change to water quality impact

Changes in land use result in changes in nutrients available for leaching to water bodies and, consequently, in concentrations of these nutrients in rivers. In order to evaluate the impact of these changes in terms of water quality (the biological status of rivers) an understanding of the ecological response induced by various alterations to land use is needed. In this section we model the relationship between chlorophyll concentrations and land use using panel (repeated observation) data, split into winter and summer observations, provided by the Centre for Ecology and Hydrology (source: Davies and Neal, 2007) on the concentration of chlorophyll at individual monitoring points in rivers across England and Wales. Chlorophyll concentration, as a measure of the rate of algal production in a water body, can identify risk of eutrophication of aquatic ecosystems and is commonly used as an indicator of water quality. Observed concentrations of chlorophyll are partly affected by the characteristics of the surrounding area, including land use, through the impact that different land uses have on the levels of nutrients in the soil. Consequently, chlorophyll-*a* is a useful indicator of river water quality that can be used in assessing the biological impact of policy changes affecting land.

The explanatory variables can be typified as catchment characteristics (area and land use allocations), climatic, and hydrological variables. Land use affects many of the physical and chemical properties of rivers, such as the quantity of suspended sediment, levels of dissolved oxygen and concentrations of nutrients such as nitrate and orthophosphate. Therefore, we

---

[7] Note that FGM does not necessarily reflect profits because they do not include fixed costs. So, for example, in the UK this leads to a situation where dairy farms typically have higher FGM per hectare than arable farms but lower total profits (see, among others, Fezzi et. al 2008). Note also that these calculations implicitly assume that prices remain constant. This could be relaxed through reference to a number of sources, although predictions from OECD & FAO (2007) are that prices will stabilise and then decline somewhat between now and later this decade.

expect land use variables to be very important in determining chlorophyll concentration and overall river ecological quality. Among the climatic variables, we consider temperature, solar radiation and standard average annual rainfall levels. Lower temperatures are expected to be associated with lower concentrations of chlorophyll as lower thermal energy inhibits algal production. Radiation levels are also an important contributing factor reflecting the intensity of light, which is required for algal production. Hydrological variables include suspended sediment, representing the presence of particulate matter in the water, and the base flow index, relating to the speed and volume of river flow. A higher base flow index typically suggests a lower residence time in the river channel environment and is generally associated with lower observed concentrations of chlorophyll, as faster flow rates inhibit algal production and dilute nutrients.

4.1 Relating land use to ecological impact: GIS based methodology

In our prior work we relate land use and changes therein through spatially sensitive patterns of nutrient leaching, taking account of in-stream mixing processes to estimate the nutrient concentrations which in part determine ecological effect (Fezzi et al., 2008 & 2010; Hutchins et al., 2009). A problem with this somewhat convoluted process is that it is prone to error propagation arising from the multiple linkages inherent in such analyses. Therefore in the present study we adopt a more parsimonious approach by directly modeling the relationship between land use and its ecological impact on the water environment through the commonly adopted measure of chlorophyll-*a* concentrations. These allow us to assess the overall impact of particular changes in land use rather than relying purely on nutrient models. The land use variables are constructed using the same land use data as is used to construct the land use model discussed in Section 2 above.

Land use and chlorophyll modeling were performed for river basins contributing runoff and leached nutrients to water quality at 83 Environment Agency monitoring points along river networks throughout England and Wales, thus enabling modeled data to be evaluated against observed chlorophyll concentrations. The spatial extent of each contributing river basin was derived from the Ordnance Survey Land-Form PANORAMA DTM (www.edina.ac.uk) using ArcMap v9.2 (www.esri.com). Sets of land use variables representing livestock values and total areas of land under various agricultural and non-agricultural uses were compiled both at the basin level and at 5, 10 and 20 km radius buffer zones around each monitoring point. This facilitated the testing of the impact of land use on water quality at a range of distances from the monitoring points in question. Land use data were derived from the June Agricultural Census (www.edina.ac.uk) and MAGIC Agricultural Land Classification (www.magic.gov.uk) data sets.

Assigning land use values at the basin level involved a relatively straightforward proportional interpolation of 2 km grid resolution data to the extent of the basin boundaries. However, assigning land use to the buffer zones around monitoring points proved to be more complicated, since the proximity of some monitoring points meant that one or more of their respective buffer zones overlapped. It was, therefore, necessary to ensure that only land use corresponding to each discrete basin/buffer combination was assigned to its respective monitoring point. An example of such an occurrence is shown in Figure 3. The blue and green regions on the map represent areas of two adjacent river basins that each fall within a 20 km radius of their respective monitoring points (indicated by triangular symbols). The circular features in graded shades of grey represent 5, 10 and 20 km buffer zones around these and neighboring monitoring points. Only land falling within the area of each buffer zone within each individual basin was assigned to its respective monitoring point, thus avoiding double counting of land use values.

Figure 3: Example of the spatial relationships between two adjacent river basins and buffer zones surrounding water quality monitoring points at their respective outlets.

INSERT FIGURE 3 ABOUT HERE

4.2. The River Water Quality Model

Based on the observation that chlorophyll-*a* concentration (µg/l) is a function of land use, climatic and hydrological variables, we examined models of the form:

(7)

where $\boldsymbol{x}$ is a vector of land use variables, $\boldsymbol{y}$ is a vector of climatic variables, $\boldsymbol{z}$ is vector of hydrological variables, u is a residual component, θ is a vector of parameters to be estimated, i indicates the monitoring point and t indicates whether the observation relates to summer or winter.

Several functional forms were reviewed in our analysis, beginning with a simple linear model and progressing through a variety of functional forms[8] allowing for non-linear and interaction effects. The most theoretically plausible model that explained the largest proportion of the variation in chlorophyll-*a* levels was;

(8)

where is a vector of shares of different land uses, is the number dairy cows and beef cows per hectare of land, is the total area in the basin, is the average atmospheric temperature, is the base flow index, is a residual error term specific to the monitoring point i (random effect) and is a residual term. Both and are assumed to be normally distributed.

---

[8] Log-log, log-square and log-square root forms.

The parameters α, and are estimated via Generalized Least Squares (GLS, see Green, 2002) and the parameter standard errors obtained using the White (1980) sandwich correction. Table 4 presents the estimation results for the best fitting model. Considering the land use variables, the urban and non-agricultural land shares were combined into a single category while root crops were separated from other arable land as they have a disproportionate impact on water quality due to the high levels of nutrient fertilizers used in their production. The share of rough grazing provides the baseline for comparison. The total area of the catchment was tested but found to be an insignificant variable and was dropped from the model. Considering the livestock variables, the intensity of dairy cows (number of cows per hectare) is included separately in the model since dairy farms make a more intensive use of land and have higher nutrient inputs than other livestock farms (e.g. sheep) whose effect on water quality is captured by the grassland variables.

Table 4: Random effects (GLS) estimates of chlorophyll-*a* concentration (µg/l)

INSERT TABLE 4 ABOUT HERE

With regard to land use shares the results of Table 4 are clearly consistent with prior expectations. Relative to rough grazing the share of root crops has the largest positive association with chlorophyll-*a* concentration. The results suggest that the share of temporary grassland has the largest negative association compared to the rough grazing baseline. The coefficients relating to the share of urban and non-agricultural and the share of arable land are not statistically significant although the signs are consistent with our expectations that a greater share of arable land is associated with a rise in chlorophyll-*a* concentrations. The number of dairy cows per hectare is both statistically significant and of the anticipated sign, suggesting that dairy farms are characterized by more intense land use management practices than other livestock farms.

The remaining climatic and hydrological variables, temperature and base flow index, are both significant. The positive coefficient on temperature is consistent with the expectation that higher temperatures stimulate algal production, raising the concentration of chlorophyll-*a*, as temperature enters the equation in log form the coefficient represents an elasticity, it is greater than one indicating that chlorophyll concentration is elastic with respect to temperature. The negative coefficient on the base flow index is also consistent with expectations as a greater base flow index represents a faster flowing river in which nutrients are flushed through more quickly and there is less time for algal production to be stimulated.

The results reported here relate to basin wide land use variables. As discussed above alternative variables measuring the land use at 5, 10 and 20 km buffer levels were constructed and used in the regression analysis. The results illustrated the importance of including all of the relevant area in the calculation of land use variables. Regressions based on land use data from a region that did not represent the entire basin resulted in greater standard errors and a reduction in the overall explanatory power of the model. This is consistent with Baker (2003) who found that points near the edge of a watershed or basin are often more influential than those closer to the water quality monitoring point, which suggests that using basin level data is likely to be necessary for understanding the impact of land use.

4.3. Predicting the ecological impact of climate change: A case study

As noted, while both the land use and ecological quality model draw upon datasets which cover large areas, the data for both is obtained a high level of spatial accuracy. This means that derived models encompass a wide degree of data variability and should therefore be generally transferable following standard out-of-sample validation tests (the methodology for which is described in Bateman et al., 2002a, 2003, with successful transfer validation tests for the present analysis being reported in Fezzi and Bateman, 2011). As the model predictor variables are typically held for the entire coverage of the country, both land use and ecological quality estimates can be obtained for any decision-relevant area. As a case study demonstration we consider an area for which we also hold revealed preference data for the recreational value of the water environment; namely the catchment of the River Aire in Yorkshire as illustrated in Figure 4. This river basin covers 86,000 ha and is chosen as an interesting test catchment because of high diversity both in terms of land use, the water environment and socioeconomics. The western half of the catchment is sparsely populated, upland areas dominated by rough grazing and pastoral agriculture. However, the remainder of the catchment includes mixed and arable farming but is progressively dominated by high density urban zones, the latter including the large conurbations of Bradford and Leeds. While these urban areas are obviously unavailable for agriculture, nevertheless they have to be incorporated within our analysis as their location will be a major determinant of the recreational values generated by any change in water quality.

Figure 4: Land use in the River Aire catchment.

INSERT FIGURE 4 ABOUT HERE

Whereas data on the dependent variables of the land use model are available for a regular grid covering the entirety of England and Wales, values for the predictors in the water quality ecological impact model are only available for an irregular network of river monitoring points administered by the UK Environment Agency (EA). The EA maintain three water quality monitoring points on the River Aire and so these are used as points to transfer our ecological impact model to estimate likely changes in chlorophyll- *a* concentration, and hence water quality, arising from a 1 degree rise in temperature under climate change. The presence of these monitoring points allows for analysis at the basin and sub-basin scale, and their locations (one upstream of any large conurbation, one in central Leeds and one at the basin outlet; indicated as points A, B and C, respectively on Figure 5) permits differentiation between the urban and non-urban signatures. The monitoring points are sited according to the physical characteristics of the basin, taking account of hydrological response units (HRUs) corresponding to areas of land that drain to discrete river stretches. Aggregations of these HRUs can be thought of as sub-basins, with monitoring points located at their outlets, so that water quality at these points can be considered to be representative of the quality along the river stretch between each monitoring point and its next upstream neighbor. Additionally, water at point B will include inputs from Basin A and water at point C will include inputs from Basins A and B.

Figure 5: Three sub-basins of the River Aire corresponding to monitoring points A, B and C.

INSERT FIGURE 5 ABOUT HERE

Prior to presenting the results obtained from the transfer exercise it is worthwhile considering their use and interpretation. In economic value terms, while the change in land use had a direct impact upon farm incomes, the impact of changes in river ecology is more indirect, mainly occurring through effects upon the recreational values of rivers. In the subsequent section we describe the estimation of a revealed preference (travel cost) model of the recreational value of rivers embracing, amongst other determinants of that value, the ecological quality of rivers. A key out of the ecological quality analysis is the estimated level of Chlorophyll-*a* which in turn serves as a predictive input into our recreational value model. As a useful simplification we will relate the μg/l measure of Chlorophyll-*a* to the four point 'water quality ladder' (WQL) scale proposed by Hime et al., (2009) which seeks to identify broad classes of quality which might be perceived as distinct by recreational visitors[9]. The Hime et al., WQL denotes each of the four levels of water quality by a color, with blue being the highest quality level, followed by green, yellow and then the lowest quality water being denoted by the color red. Table 5 provides ecological descriptions and μg/l measures of Chlorophyll-*a* related to the Hime et al., WQL levels.

Table 5: Water quality classifications

INSERT TABLE 5 ABOUT HERE

Our ecological quality model was then applied to predict water quality under the present and future climate scenario. Such estimations require information on the likely change in the level of predictor variables under such a scenario. This information was gathered through personal communications with staff at the Centre for Ecology and Hydrology, Oxford. This suggested that a 1°C rise in air temperature may cause a greater than proportional increase in water temperature. However, in our naive climate change scenario rainfall is assumed to remain fixed at the annual average level (although this allows for the development of wetter winters and drier summers). Similarly the base flow index remains fixed and the levels of suspended sediment are assumed to increase by 10 per cent.

Table 6 details predictions from our transferable ecological quality model under the present and future climate scenario. Comparison of Chlorophyll-*a* measures shows that at all three monitoring points we predict a decline in ecological quality arising from both the direct effect

---

[9] This approach is similar to that proposed by UKTAG (2008). The Hime et al., paper includes a conversion table allowing comparison between the two scales.

of temperature increases within the water and the indirect effect of alterations in the level of nutrient leaching arising from the concurrent shift in land use. In relative terms this decline will be most marked at the upper levels of water quality and it is here where we see a shift down from blue to green quality on the WQL scale. In absolute terms the increase in Chlorophyll-*a* measures is greatest at lower levels of water quality. However, these are less marked in relative terms and do not breech the boundaries of respective WQL classes.

Table 6: Predicted reductions in water quality as a consequence of climate change.

INSERT TABLE 6 ABOUT HERE

The output of our ecological quality analysis forms an input to our assessment of the responsiveness of recreational values to the impacts induced by climate change. The key issue here is how the changes predicted in Table 6 will impinge upon those sites which are available for recreational access. Figure 6 indicates the recreational access sites for the study area (the definition of those sites being described in the following section) and their current ecological water quality described using the WQL color scale.

Figure 6: Sampling area and the quality of recreational access sites

INSERT FIGURE 6 ABOUT HERE

Analysis of Figure 6 is interesting. At first glance water quality looks generally good. However, the location of the medium and poor quality sites is predominantly in downstream areas and these coincide with the high density urban locations where most potential visitors live. Therefore, once recreational access is taken into consideration it would appear that there is considerable scope for improvement in this situation. Table 7 integrates the findings of our ecological analysis with the location of recreational access sites to reveal the impact of climate change upon those sites. As can be seen, the climate change scenario results in a substantial decline in the highest quality sites (which reduce by more than one-quarter) while the number of medium and poor quality sites increases markedly. In the following section we assess the loss of recreational value induced by this change.

Table 7: Percentage of all recreational access sites within the study area classified by ecological quality under current situation and climate change

INSERT TABLE 7 ABOUT HERE

5. Recreation value impacts

5.1 Sample survey and GIS data generation

In order to estimate the recreational impact of the changes in ecological quality indicated above a large sample survey of households was undertaken. In order to capture the spatial sensitivity of values to location and hence incorporate phenomena such as the distance decay of values away from an improvement site (Bateman et al., 2006), a large survey area was defined spanning a 70km diameter centered on the River Aire embracing its catchment and surrounding areas. In so doing we sought to capture likely substitution effects generated by competing resources (as well as the spatial complementarities noted in previous studies; see for example Jones et al., 2002). A sampling frame designed to capture spatial variation was designed and a household survey implemented.

The survey questionnaire was explicitly designed to capture large quantities of spatially explicit data from respondents through a highly accessible custom built computer aided personal interview (CAPI) system intended to avoid high cognitive load upon respondents. As part of the survey interview, respondents were shown an interactive map on a high resolution computer screen. This map showed the respondent's home location and all of the rivers around their home within an area of approximately 80km^2. Respondents then indicated on the interactive map[10] the river locations they visit for recreation and the frequency of their visits to each site[11]. In order to model the demand for water recreation we collect information regarding the total number of outdoor trips in the last 12 months, total frequencies to water bodies and detailed information about the rivers sites.

Once the interview was completed visit site locations were later matched to a real world recreational site using the Geographical Information Systems (ArcGIS 9.2, ESRI) software. River recreational sites, i.e. where it is possible to access the river for recreation by either walking or driving, were identified in the GIS from Ordnance Survey MasterMap data using a four-stage methodology. Firstly, stretches of river which are accessible to the public (defined as those river stretches which have either a public footpath or minor road within 50 metres) where identified. Secondly, these publicly accessible river stretches were assigned access points by identifying where the footpath or road first joined onto or met these accessible river stretches. Thirdly, some access points were extremely close together (within 150 metres of each other) and had similar environmental characteristics, and therefore these access points were grouped together to form a single recreational site. Finally, the locations of each of the recreational sites were verified using Ordnance Survey 1:50,000 maps and aerial photographs. In total, 531 recreational sites were identified along the study rivers, which span approximately 230km in length. GIS routines were employed to calculate distance data from each household in the survey sample to each of the recreational sites, included those not visited. This allows the analyst to examine the influence which substitute availability has upon the choice of recreation site visits. The GIS was also used to incorporate

---

[10] Typically the interviewer guided a mouse pointer to the location of the site indicated by the respondent and clicked this to record that location. Respondents were allowed to alter this location if they felt it was incorrect.

[11] Respondents' also indicated their own assessment of the water quality at each site although this is not used in the present analysis.

further information such as the population density of the household local area (a measure of whether the respondent lives in an urban area)[12].

The home location of each respondent was identified from Ordnance Survey Address Point data using their postcode. The distance by road, and travel time by car, from each respondent's home to all of the 531 recreational sites was calculated in the GIS. Lastly, information on the environmental characteristics of the recreational sites was identified in the GIS using Ordnance Survey MasterMap and Centre for Ecology and Hydrology (CEH) Land Cover Map of Great Britain datasets. These provided details of the predominant land use around each of the recreational sites, which were grouped into five broad categories including woodland, farmland, grassland, heath, and urban or other built land use. The current water quality at each of the recreational sites was calculated from Environment Agency long-term water quality monitoring data and categorized to the four-point given in Hime et al., (2009) as ranging from "Good" (blue color as discussed previously) through to "Poor" (red color). The location of these various sites have been illustrated previously in Figure 6.

Returning to consider our survey sample, after removing some 2% of respondents due to missing address information or other item non-response, in total some 1782 face-to-face, at-home household interviews were completed. Sample characteristics suggested a reasonable degree of representativeness had been achieved with 44% of respondents being male, average household size of 2.6, an average net income of £21,317 per annum (s.d.£11,700); 26% of respondents in full time employment, 13% part-time employed, 33% retired and 7% self-employed.

5.2 Modeling repeated recreational choices

The desirability of unifying physical environmental, locational and socioeconomic data has been repeated highlighted in valuation guidelines (Bateman et al., 2002b; Champ et al., 2003; Grafton et al., 2008) yet applications remain the exception rather than the rule (e.g. Barbier 1997; Sanciro and Wilen 2001; Smith and Wilen 2003; Bateman, 2009b; Egan et al., 2009; Jeon and Herriges, 2010) and in simulating land use choices (e.g.). However, the joining of physical data and economics effects is not a common practice in environmental valuation policies. This is perhaps unusual given that values demonstrably vary across space and indeed this is the basis of the travel cost random utility model (RUM).

The RUM provides the standard approach for analyzing recreational behavior and in the simplest multi-site model the only relevant information required is the site choice made by recreationalists. However, a change in natural resource quality will affect not only the choice of sites but also the visitation frequency. For this reason Phaneuf et al., (2000) proposed the general corner solution Kuhn-Tucker (KT) model as an improved RUM incorporate within

---

[12] GIS techniques also provide an ideal medium for linkage to a variety of other databases such as the UK Census, landcover data, etc., yielding a variety of variables (some of which are still to be analysed in ongoing work).

the same structural demand system the alternative choice demand and the frequencies of choice. This model represents a more realistic modeling of recreation choices. However, Bockstael and McConnell 2007 (p. 102) identify a number of limitations to KT-RUM models including:

1) complex implementation,
2) difficulties in straightforward interpretation of results,
3) limitations to the number of alternative choices (substitutes) which can be incorporated.

Therefore, in our case study, where more than 500 river access points have been identified the general corner solution model does not seem a viable solution.

Alternative models for to incorporate participation and site choice are discussed in Parson et al (1999) who demonstrate that the alternative RUM approaches proposed by Morey et al (1993) and Hausmann et al (1995) produce very similar results. Therefore, we specified a simplified version of the Morey et al. (1993) approach as described below. Although there are numerous discrete formulations for modeling site choice, we adopt a Conditional Logit Model (CLM) with alternative specific constant based on McFadden (1974) as a widely accepted option.

The basis of the economic analyses of recreational choices is to reveal how individuals trade money for improvements in natural resources quality. In the present application the utility associated with visiting recreational options area is specified as function of access costs, water quality levels and other site characteristics. The main problem in analyzing the recreation behavior is the travel cost calculation. Travel cost might be defined as a function of out-of-pocket costs (such as car fuels, etc.) plus the opportunity cost of time which is given by a proportion of respondent's wage. While different strategies have been proposed in the literature (e.g. Smith et al., 1983; Ward, 1984; Hynes et al., 2009), the travel cost calculation is still an unresolved question in recreation modeling (Randall, 1994; Common, 1999). Hynes et al., (2009) compared different methods of travel cost calculation and following one of his approaches we derive the travel cost as out-of pocket expenditure (at a rate of £0.25 per km travelled round trip) plus the opportunity costs of time calculated as a percentage of respondent's wage. Given the available information about household net income and household members, we derived the travel cost by considering the adults' wage value as a proportion of the family income for each adult.

Given the interest in understanding the benefits due to water quality changes for rivers flowing through an highly populated area, we assume that every day in the year might in theory provide a choice occasion (T). Therefore T is fixed at 365 and we observed the frequencies of choices to the 531 river access points, to other rivers in the sampling area, canals, lakes and other outdoor activities. Finally, we identify for each respondent the number of times they decide not to take outdoor trips. In this framework the individual i makes daily choices across the J options available (where $j=0,1, 2,\ldots, 535$ where $j=0$ is the option not to recreate, j = 1 to 531 are river access sites in the study area, and $j=532,\ldots,535$ are other rivers, canals, lakes and other outdoor trips). The individual chooses the option with the highest utility in each occasion. From the researcher's perspective we define this utility as the random function:

$$U_{ijt} = v_{ijt} + \varepsilon_{ijt} = f(X_j, Z_{ij}, W_i, \theta) + \varepsilon_{ijt} \tag{9}$$

where X_j includes sites characteristics that are constant across choice occasions, Z_{ij} includes individual characteristics such as travel costs that change across sites, W_i contains individual characteristics constant across sites and choice occasion, θ is the parameter vector, and ε_{ijt} is a random component that is unobservable to the analyst. This random utility model posits that, given J possible recreation options and the possibility of an opt-out (not to recreate) option, respondent i in period t will choose location j if the utility of j is higher than that of the other options as well as the choice of not to recreate in period t. Specifying a linear in parameter utility function, Equation (9) can be rewritten for each choice occasion as:

$$U_{ij=1,..,531} = \alpha + f(X_j, Z_{ij}, W_i, \eta) + \varepsilon_{ij} \text{ for river sites and} \tag{10}$$

$$U_{ij>531} = \alpha_j + \varepsilon_{ij} \text{ for the other options}$$

where $\theta=(\alpha, \alpha_j, \eta)$. In this structure the utility of visiting sites other than the river sites in our study area is captured by the alternative specific constant variables (α_j) and for identification, the utility of not recreating is fixed to zero. Note, however, that the utility of not recreating at the river site in our study area also captures the utility of leisure and recreation opportunities outside of outdoor trips. Furthermore, the individual specific variables (W_i) can be included in the model only as interactions with the alternative specific constant or sites variables.

Morey et al., (1993) formulate the recreational behavior model as nested choice; where in the first place individuals decide whether to engage in recreation or not and if so then they subsequently decide which site to visit in each choice occasion. In this format the error term is usually assumed to be distributed as a Generalized Extreme Value random term. However, a drawback here is that the likelihood function is not globally concave. If instead we assume that the random error term is identically and independently distributed as an Extreme Value type I then the model becomes is globally concave. With a dataset containing more than 900,000 observations (mainly as a result of capturing the spatial complexity of distances to all sites and substitutes from all households) then, for illustrative purposes we adopt the latter model due to its less intensive computational demands and accept that this may suffer from IIA problems. Alternative techniques such as the Mixed Multinomial Logit (Train 2003) approach would avoid IIA issues but, given the size of dataset concerned this would require customized 'smart' computation procedures or additional computing power. As such this is held back for future consideration.

In conclusion using the utility functions in Equation (10) the probability of respondent *i* undertaking a trip to site *j* is the standard conditional logit model with alternative specific constant variables, written as

$$\mathrm{P}\left[U_{ik} > U_{ij}\right] = \frac{e^{\alpha_k + \beta' x_k + \eta z_{ik}}}{\sum_j e^{\alpha_j + \beta' x_j + \eta z_{ij}}} \tag{11}$$

where β is the parameters of a vector of river characteristics and η is the travel cost parameter. The parameters are estimated via Maximum Likelihood.

The variables included in the linear utility function are described in Table 8. Note that the good (blue) quality level has been used as baseline and is not explicitly included in the model.

Table 8: Summary of variables

INSERT TABLE 8 ABOUT HERE

Using the variables in Table 8 and other socio-economic variables (e.g. number of children, etc.) interacted with the alternative specific constant, we specify and estimate models to analyze the impact of water quality change upon recreation behavior. Results from this analysis are reported in Table 9.

Table 9: Estimated coefficients from travel cost model

INSERT TABLE 9 ABOUT HERE

All the variables in Table 9 are highly statistically significant and accord with prior expectations. The “travel cost” variable is significant and negatively signed as expected. Similarly, the water quality variables are also significant and have expected negative signs indicating reductions from the baseline ‘good’ (blue) water quality. The positive sign on the “urban” parameter means that utility increases if the river site is in urban area suggesting that increasing natural resources in highly populated area might have a greater impact upon welfare than in rural areas, possibly because of the lack of alternative everyday source of environmental quality within cities. However, it is also possible that the positive sign of urban can be explained by considering correlations with other facilities available at river sites (e.g. car park, playgrounds, etc.) and the possibility to complement river recreation experiences with other sources of outdoor recreation (e.g. shopping). The GIS framework of our study is conducive to extensions of this analysis to address these possibilities. Finally, all alternative specific constants present a negative sign demonstrating the common sense finding that, over the year, respondents typically choose not to spend their time in other activities than river recreation.

In order to derive welfare measures we follow the method proposed by Small and Rosen (1982) and Hanemann (1999). The characteristics of river sites can change at a single site or at all sites. In both cases, using the linear utility function in Equation (11), we assume that the current quality levels defined by matrix X changes to X^*. Given the typically low budget shares of recreational activities, we can assume constant marginal utility of income and obtain willingness to pay (WTP) as per Equation (12):

$$WTP = \eta^{-1}\left[\ln\left(\sum_{j=1}^{J} e^{\alpha_k+\beta' x^*_k-\eta z_{ik}}\right) - \ln\left(\sum_{j=1}^{J} e^{\alpha_k+\beta' x_k-\eta z_{ik}}\right)\right] \qquad (12)$$

In special circumstances, where a change in a site attribute (e.g. poor quality - *pq*) is the same across sites and we can assume that the marginal utility of income remains constant over available options and choice occasions, the marginal WTP for that attribute can written as:

$$WTP_{pq} = \frac{\beta_{pq}}{\eta}. \qquad (13)$$

5.3. Estimating individual level values for changes in the ecological quality of rivers.

The ecological model detailed in Section 4 of this paper indicated that under our climate change scenario the direct effect of higher temperature combined with the indirect impact of induced changes in land use and consequent alterations in diffuse pollution would result in a decrease in water quality throughout the case study area of the River Aire. However, the same analysis suggested that the major impact of this change is likely to be a reduction in the number of high quality (blue) sites and a consequent increase in the number of medium quality (green) sites.

To estimate the consequences of the climate change scenario we take the changes in site quality predicted from our ecological model, as detailed in Table 7, and applying these to the parameter estimates given in our travel cost model, detailed in Table 9 (i.e. X^*= water quality levels estimated under the climate change scenario). Results from this calculation indicate that the average disutility expressed as compensation per year (i.e. negative WTP) is equal to £10.44 per person[13].

The losses likely to occur under climate change will of course be mitigated to a smaller or greater extent by the degree of policy intervention undertaken. Indeed the reductions in river water quality suggested by our climate change analysis stand in stark contrast to the policy targets set out for the EU under its Water Framework Directive (WFD) (European

---

[13] This assumes that there is no asymmetry between WTP and willingness to accept (WTA) compensation. There is a considerable body of research suggesting that this may not be the case and that per unit of provision WTP < WTA (see Horowitz and McConnell, 2002, for a review of this issue). However, recent work suggests that the large asymmetries claimed for nonmarket goods may in part reflect design problems (Bateman et al., 2009).

Commission, 2000). This requires member states of the European Union (EU) to avoid any reduction in water quality and instead act to improve biodiversity in aquatic ecosystems and achieve “good ecological and chemical status” for all water bodies by 2015 (*ibid.*). Setting aside the technical difficulties and costs involved in such an undertaking we can briefly extend our analysis of recreational benefits to assess the value of attaining such a goal. Starting from the present day and envisioning a shift directly to a situation where all recreation sites are improved to the highest (blue) quality (i.e. X^* = all rivers of good quality) we obtain an estimate and annual benefit equal to £17.89 per person[14].

Of course for decision purposes we need to aggregate these various individual estimates up to a population total level and we conclude this section with a demonstration of the issues involved in such an exercise.

5.4. Estimating aggregate level values for changes in the ecological quality of rivers.

In order to aggregate our values across the study area we require distance calculations from all possible recreation sites to all households (not just those sampled in our survey) in the case study area. We also require socioeconomic characteristics for all households. Our GIS based methodology allows us to perform these calculations with only a minimal degree of simplification; in this case working with UK Census Super Output Areas (SOAs) rather than individual households (although this could be achieved given necessary computing power).

The aggregation process considers two scenarios:

(i) A movement from the current baseline to the climate change scenario (with shifts in site quality as described in Table 7);

(ii) A movement from the current baseline to the WFD scenario (where all sites attain the highest (blue) quality).

These scenarios determine the level of dependent variables to be used in our travel cost model. This model is then applied to each SOA, taking into account its distance to each of the recreational sites, their quality, the socioeconomic characteristics of the population of that SOA and the number of households it contains.

Figure 8 illustrates the distribution of per person (left hand panel) and SOA aggregate (right hand panel) values for the climate change scenario. Note that the site colors illustrated show the baseline situation. Climate change will cause a decline in quality in the western area of the catchment (currently generally at the highest (blue) level of water quality). Aggregating

---

[14] In theory we could combine the two individual level estimates to obtain a value for a path in which losses from climate change occur after which policy initiatives raise all sites to the highest quality level. However, such a static analysis makes a number of assumptions, including that individuals are not subject to endowment effects (Kahneman et al., 1990). Empirical tests suggest this is unlikely to be the case in practice (Bateman et al., 1997).

across the entire case study area yields an estimate of the total loss induced by the climate change scenario of approximately £26million p.a. The distribution of these benefits is as one might expect, being concentrated in the western area of the catchment. Although populations are relatively low here, as noted this is the principle location where water quality losses will occur. The eastern area of the catchment does not suffer such appreciable losses, therefore, despite the higher population in that area, its aggregate values are relatively low.

Figure 7: The distribution of per person (left hand panel) and SOA aggregate (right hand panel) value changes for the climate change scenario.

INSERT FIGURE 7 ABOUT HERE

Figure 8 illustrates the distribution of per person (left hand panel) and SOA aggregate (right hand panel) values for the WFD scenario. Note again that the site colors illustrated show the baseline situation. Implementation of the WFD will cause an increase in quality in the eastern area of the catchment (currently generally at medium (green) or lower levels of water quality). Aggregating across the entire case study area yields an estimate of the total loss induced by the climate change scenario of approximately £65million p.a. The distribution of these benefits is again as one might expect, being very low in the west (where quality is and remains high) and instead being concentrated in the eastern area of the catchment. Given the high populations living here it is unsurprising that the total value of this scheme exceeds in absolute terms that under the climate change scenario.

Figure 8: The distribution of per person (left hand panel) and SOA aggregate (right hand panel) value changes for the WFD scenario.

INSERT FIGURE 8 ABOUT HERE

6. Conclusions.

The paper presents a unified series of models examining the direct secondary and further effects of a given driver upon natural resource based systems. The specific case study concerns the impact of climate change upon land use and water quality. We model the effects upon land use and its consequent impacts on farm incomes and ecological water quality. From there we consider the impact of changes in water ecology upon river recreation values.

Results show that, climate change is likely to generate highly spatially variable impacts upon both land use and consequent farm incomes. In some areas of the UK it will generate income gains while other areas will experience losses. This pattern directly reflects the diverse and highly heterogeneous nature of UK agriculture.

While our case study on the water quality impacts of climate change focuses down on a particular catchment. Here we see temperature increases leading to a general decline in the ecological quality of rivers. Translating this to recreational sites we apply a large sample revealed preference survey utilizing a novel spatially sensitive methodology to estimate the value losses associated with this change. These are contrasted with the value gains likely to arise from implementation of the WFD. We conclude our study with a aggregation exercise which incorporates the distance decay inherent in recreational values.

Taken together our recreational value results suggest that the benefits of implementing the WFD may be substantial, although we have not compared these against the costs of intervention. If full benefit cost analysis suggests that only limited implementation is justified then our methodology is well suited for the targeting of funds. The results presented here suggest that the most efficient target for WFD implementation is within highly polluted urban areas. While this may seem an obvious finding it contradicts the approach set out in the EU WFD documentation which make no distinction between locations in terms of the populace affected and benefits generated.

These recreational values have to be set against not only the costs of any policy intervention but also the impact on other areas of the economy. Our previous work has shown that WFD policies are likely to impose substantial costs upon the farming community (Fezzi et al, 2008). Our present analysis indicates that at least in the case study area addressed in our study of recreation values, that climate change may impose further financial strains upon an already beleaguered sector. We offer the integrated modeling methodology demonstrated throughout this paper as a tool to address the holistic effects of multiple environmental, policy and market influences acting simultaneously. While complex, we believe that such methodologies are vital to address the complexities of the real world and bring them within the remit of economic analysis.

References

Ackerman, F., DeCanio, S.J., Howarth, R.B. and Sheeran, S. (2009) Limitations of integrated assessment models of climate change, *Climatic Change*, Volume 95, Numbers 3-4: 297-315, DOI 10.1007/s10584-009-9570-x

Amemiya, Takeshi (1973) Regression analysis when the dependent variable is truncated normal, *Econometrica*, vol. 41, pp. 997–1016.

Avila-Foucat, V.S., Perrings, C. and Raffaelli, D. (2009) An ecological-economic model for catchment management: The case of Tonameca, Oaxaca, Mexico, *Ecological Economics*, Volume 68, Issues 8-9,Pages 2224-2231, DOI: 10.1016/j.ecolecon.2009.01.010.

Balmford, A., Bruner, A., Cooper, P., Constanza, R., Farber, S., Green, R.E., Jenkins, M., Jefferis, P., Jessamy, V., Madden, J., Munro, K., Myers, N., Naeem, S., Paavola, J., Rayment, M., Rosendo, S., Roughgarden, J., Trumper, K. and Turner, R.K. (2002) Economic Reasons for Conserving Wild Nature, *Science*, 297: 950 – 953, DOI: 10.1126/science.1073947.

Barbier B., 1997. Induced innovation and land degradation: Results from a bioeconomic model of a village in West Africa. *Agricultural Economics* 19: 15-25.

Barbier, Edward B., 2007. Valuing ecosystem services as productive inputs. *Economic Policy* 22:177–229.

Bateman, I.J. (2009a) Economic Analysis for Ecosystem Assessments: Application to the UK National Ecosystem Assessment (NEA), *CSERGE Working Paper EDM 09-12,* Centre for Social and Economic Research on the Global Environment, University of East Anglia.

Bateman, I.J. (2009b) Bringing the real world into economic analyses of land use value: Incorporating spatial complexity, *Land Use Policy*, 26S: S30–S42, doi:10.1016/j.landusepol.2009.09.010

Bateman, I.J., Carson, R.T., Day, B., Hanemann, W.M., Hanley, N., Hett, T., Jones-Lee, M., Loomes, G., Mourato, S., Özdemiroğlu, E., Pearce, D.W., Sugden, R. and Swanson, J. (2002b) *Economic Valuation with Stated Preference Techniques: A Manual*, Edward Elgar Publishing, Cheltenham.

Bateman, I.J., Day, B.H., Georgiou, S. and Lake, I. (2006) The aggregation of environmental benefit values: Welfare measures, distance decay and total WTP. Ecological Economics, 60(2): 450-460, doi:10.1016/j.ecolecon.2006.04.003.

Bateman, I.J., Day, B.H., Jones, A. P. and Jude, S. (2009) Reducing gains/loss asymmetry: A virtual reality choice experiment (VRCE) valuing land use change, *Journal of Environmental Economics and Management*, 58: 106-118, doi:10.1016/j.jeem.2008.05.003

Bateman, I.J., Jones, A.P., Lovett, A.A., Lake, I. and Day B.H. (2002) Applying geographical information systems (GIS) to environmental and resource economics, *Environmental and Resource Economics*, 22(1-2): 219-269. DOI: 10.1023/A:1015575214292

Bateman, I.J., Lovett, A.A. and Brainard, J.S. (2003) *Applied Environmental Economics: a GIS Approach to Cost-Benefit Analysis,* Cambridge University Press, Cambridge.

Bateman, I.J., Mace, G.M., Fezzi, C., Atkinson, G. and Turner, R.K. (2011) Economic analysis for ecosystem service assessments, *Environmental and Resource Economics*, 48(2): 177-218, DOI 10.1007/s10640-010-9418-x.

Bateman, I.J., Munro, A., Rhodes, B., Starmer, C. and Sugden, R. (1997) A test of the theory of reference-dependent preferences, *Quarterly Journal of Economics,* 112(2): 479-505.

Bockstael, N.E., Freeman, A.M. III, Kopp, R.J., Portney, P.R. and Smith, V.K. (2000) On measuring economic values for nature, Environmental Science and Technology, 34 (8): 1384–1389, DOI: 10.1021/es990673l

Bockstael N.E. and McConnell K.E. (2007). *Environmental Resource Valuation with Revealed Preferences. A Theoretical Guide to Empirical Models*. Springer, the Netherlands.

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

Brouwer, R. and De Blois, C. (2008) Integrated modelling of risk and uncertainty underlying the cost and effectiveness of water quality measures, *Environmental Modelling & Software*, Volume 23, Issue 7: 922-937

CBD (Convention on Biological Diversity) (2006) *Global Biodiversity Outlook 2*. Secretariat of the Convention on Biological Diversity. Montreal.

Champ, Patty A., Boyle, Kevin and Brown, Tom C. (eds.) (2003) *A Primer on Non-market Valuation*, The Economics of Non-Market Goods and Services: Volume 3, Kluwer Academic Press, Dordrecht. ISBN: 978-0-7923-6498-6.

Chapin, F. Stuart III, Erika S. Zavaleta, Valerie T. Eviner, Rosamond L. Naylor, Peter M. Vitousek, Heather L. Reynolds, David U. Hooper, Sandra Lavorel, Osvaldo E. Sala, Sarah E. Hobbie, Michelle C. Mack & Sandra Díaz (2000) Consequences of changing biodiversity, *Nature*, 405, 234-242 (11 May 2000) doi:10.1038/35012241

Chambers R.G., Just R.E. (1989) Estimating multioutput technologies, *American Journal of Agricultural Economics*, vol. 71, pp. 980-995.

Common, M., Bull, T. and Stoeckl, N. (1999) The Travel Cost Method: an Empirical Investigation of Randall's Difficulty, The Australian Journal of Agricultural and Resource Economics, 43(4): 457-477, DOI: 10.1111/1467-8489.00090

Dale, V.H., Efroymson, R.A. and Kline, K.L. (2011) The land use–climate change–energy nexus, *Landscape Ecology*, 26:755–773, DOI 10.1007/s10980-011-9606-2

Egan, K.J., Herriges, J.A., Kling, C.L. and Downing, J.A. (2009) Recreation Demand Using Physical Measures of Water Quality, *American Journal of Agricultural Economics*, Vol. 91, 1, pp. 106-123.

Fezzi, C., Hutchins, M., Rigby, D., Bateman, I.J., Posen, P. and Hadley, D., (2010) Integrated Assessment of Water Framework Directive Nitrate Reduction Measures, *Agricultural Economics*, 41: 123–134.

Fezzi C., Rigby D., Bateman I.J., Hadley D., Posen P. (2008) Estimating the range of economic impacts on farms of nutrient leaching reduction policies, *Agricultural Economics*, vol. 39, pp. 197-205.

Fezzi, C. and Bateman, I.J. (2011) Structural agricultural land use modeling for spatial agro-environmental policy analysis, *American Journal of Agricultural Economics*, 93 (4): 1168-1188, doi: 10.1093/ajae/aar037

Fisher, B., Turner, R.K., Zylstra, M., Brouwer, R., de Groot, R., Farber, S., Ferraro, P., Green, R., Hadley, D., Harlow, J., Jefferiss, P., Kirkby, C., Morling, P., Mowatt, S., Naidoo, R., Paavola, J., Strassburg, B., Yu, D. and Balmford, A. (2008) Ecosystem services and economic theory: integration for policy-relevant research. *Ecological Applications,* 18(8): 2050–2067.

Fisher, B. and Turner, R.K. (2008) Ecosystem services: classification for valuation. *Biol. Cons.* 141: 1167–1169.

GEF (Global Environment Facility) (1998) *Valuing the Global Environment: Actions and Investments for a 21st Century*. GEF, Washington, DC, USA.

Grafton, R. Quentin, Wiktor Adamowicz, Diane Dupont, Harry Nelson, Robert J. Hill, Steven Renzetti (2008) The Economics of the Environment and Natural Resources, Wiley.

Groot, de R.S. *et al*. 2002. A typology for the classification, description and valuation of ecosystem functions, goods and services. *Ecological Economics*, 41: 393–408.

Hanemann W.M., 1999. Welfare analysis with discrete choice models. In Herriges J. and Kling C. (eds) *Valuing Recreation and the Environment*. Cheltenham, UK and Northamption, MA: Edward Elgar.

Heathwaite, A.L., Quinn, P.F. and Hewett, C.J.M. (2005) Modelling and managing critical source areas of diffuse pollution from agricultural land using flow connectivity simulation, *Journal of Hydrology*, Volume 304, Issues 1-4: Pages 446-461, doi:10.1016/j.jhydrol.2004.07.043

Hime, S., Bateman, I., Posen, P., Hutchins, M. (2009). A transferable water quality ladder for conveying use and ecological information within public surveys. *CSERGE working paper EDM-2009-01*. http://www.uea.ac.uk/env/cserge/pub/wp/edm/edm_2009_01.htm

Horowitz, J. K., McConnell, K. E. (2002) A Review of WTA / WTP Studies, *Journal of Environmental Economics and Management*, 44: 426-447.

Howarth, R.B. & S. Farber. 2002. Accounting for the value of ecosystem services. *Ecol. Econ*. 41: 421–429.

Hutchins, M., Fezzi, C., Bateman, I.J., Posen, P. and Deflandre-Vlandas, A. (2009) Cost-Effective Mitigation of Diffuse Pollution: Setting Criteria for River Basin Management at Multiple Locations, *Environmental Management*, 44(2): 256-267. DOI 10.1007/s00267-009-9306-8.

Jeon, Y. and Herriges, J.A. (2010) Convergent Validity of Contingent Behavior Responses in Models of Recreation Demand, *Environmental and Resource Economics*, 45(2): 223 – 250, DOI 10.1007/s10640-009-9313-5.

Jones, A.P., Bateman, I.J. and Wright, J., (2002) Estimating arrival numbers and values for informal recreational use of British woodlands, report to the Forestry Commission, Edinburgh, published at http://www.forestry.gov.uk.

Kahneman, Daniel, Jack L. Knetsch and Richard H. Thaler (1990). Experimental tests of the endowment effect and the Coase theorem. Journal of Political Economy 98: 1325-1348.

Koziell, I. 2001, *Diversity not Adversity: Sustaining Livelihoods with Biodiversity*. International Institute for Environment and Development and Department for International Development (DFID). England.

Loreau, Michel, Alfred Oteng-Yeboah, M. T. K. Arroyo, D. Babin, R. Barbault, M. Donoghue, M. Gadgil, C. Häuser, C. Heip, A. Larigauderie, K. Ma, G. Mace, H. A. Mooney, C. Perrings, P. Raven, J. Sarukhan, P. Schei, R. J. Scholes & R. T. Watson, (2006) Diversity without representation, *Nature*, 442, 245-246: doi:10.1038/442245a.

MA (Millennium Ecosystem Assessment) (2005) *Millennium Ecosystem Assessment, Ecosystems and Human Wellbeing*. Synthesis Island Press.Washington, DC.

Mace, G., Bateman, I.J., Albon, S., Balmford, A., Church, A. and Winn, J. (2009) *Conceptual framework and methodology*, Report to the UK National Ecosystem Assessment, December 2009.

Mäler, K-G, Aniyar, S. and Jansson, Å., (2008) Accounting for ecosystem services as a way to understand the requirements for sustainable development, *PNAS*, 105(28): 9501–9506, www.pnas.org/cgi/doi/10.1073/pnas.0708856105.

Milne, E., Aspinall, R.J. and Veldkamp, T.A. (2009) Integrated modelling of natural and social systems in land change science, *Landscape Ecology*, Volume 24, Number 9: 1145-1147, DOI: 10.1007/s10980-009-9392-2

OECD & FAO (Organisation for Economic Cooperation and Development and The Food and Agriculture Organization of the United Nations) (2007) *Agricultural Outlook 2007-2016*, OECD-FAO, Paris, France.

Parsons G., Jakus P., Tomasi T., 1999. A comparison of welfare estimates from four models for linking seasonal recreational trips to multinomial logit models of site choice. *Journal of Environmental Economics and Management*, 38: 143-157.

Phaneuf D.J, Herriges J.A., and Kling C.L., 2000. Estimation and welfare calculations in a generalized corner solution model with an application to recreational demand. *Review of Economic and Statistics*, 82: 83-92.

Pudney S. (1989) *Modelling individual choice: the econometrics of corners, kinks and holes*, Blackwell Publishers, Cambridge.

Randall, A. (1994) A Difficulty with the Travel Cost Method, *Land Economics*, Vol. 70, No. 1: 88-96,

Sanchirico, J.N., Wilen, J.E., 2001. A bioeconomic model of marine reserve creation. *Journal of Environmental Economics and Management* 42: 257–276.

Small K.A., Rosen H.S., (1981) Applied Welfare Economics with Discrete Choice Models, *Econometrica*, 49, 105-130.

Smith M.D., Wilen J.E., 2003. Economics impacts of marine reserves: the importance of spatial behaviour. *Journal of Environmental Economics and Management* 46: 183:206.

TEEB (The Economics of Ecosystems and Biodiversity) (2009) *The Economics of Ecosystems and Biodiversity for National and International Policy Makers*, availa ble at www.teebweb.org

Tobin J. (1958) Estimation for relationships with limited dependent variables, *Econometrica,* vol. 26, pp. 24–36.

Turner, R. K., Morse-Jones, S. and Fisher, B. (2010) Ecosystem valuation, *Annals of the New York Academy of Sciences*, 1185: 79–101. doi: 10.1111/j.1749-6632.2009.05280.x

UKCP (UK Climate Projections) (2009) *UKCP09*, available at http://ukclimateprojections.defra.gov.uk/

UK Technical Advisory Group (UKTAG) – Water Framework Directive (2008) Phase 1 Report - Environmental standards and conditions for surface waters. Available from http://www.wfduk.org/UK_Environmental_Standards/LibraryPublicDocs/UKTAG%20ReportAug%202006UKEnvironmentalStandardsandConditionsFinalReport

van Ittersum, Martin K., Frank Ewert, Thomas Heckelei, Jacques Wery, Johanna Alkan Olsson, Erling Andersen, Irina Bezlepkina, Floor Brouwer, Marcello Donatelli, Guillermo Flichman, Lennart Olsson, Andrea E. Rizzoli, Tamme van der Wal, Jan Erik Wien and Joost Wolf (2008) Integrated assessment of agricultural systems - A component-based framework for the European Union (SEAMLESS), *Agricultural Systems*, Volume 96, Issues 1-3, Pages 150-165, DOI: 10.1016/j.agsy.2007.07.009.

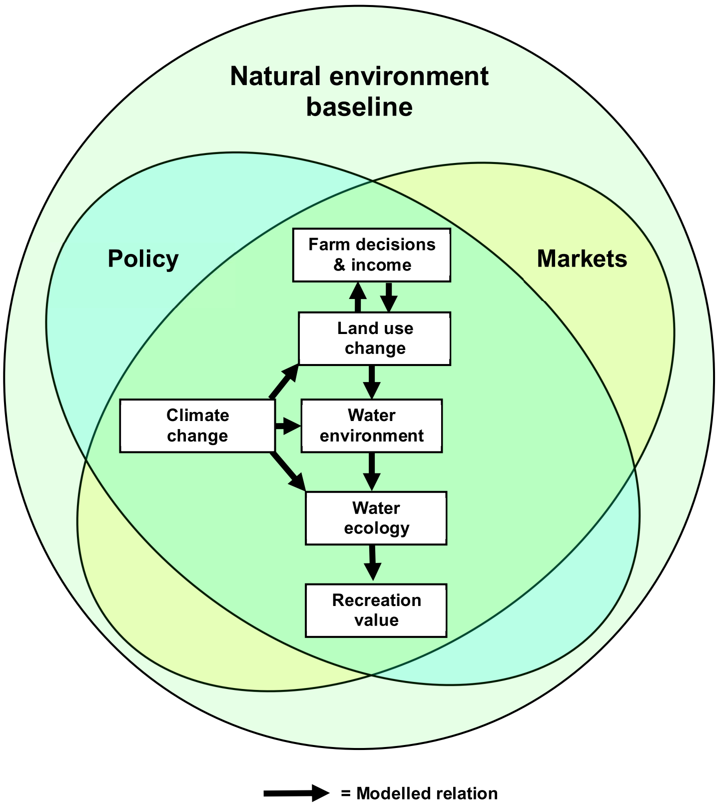
Wallace, K.J. (2007). Classification of ecosystem services: problems and solutions. *Biol. Cons*. 139: 235–246.

Wei, Y., Davidson, B., Chen, D. and White, R., (2009) Balancing the economic, social and environmental dimensions of agro-ecosystems: An integrated modelling approach, *Agriculture, Ecosystems & Environment*, Volume 131, Issues 3-4, June 2009, Pages 263-273, DOI: 10.1016/j.agee.2009.01.021.

Yen S.T., Huang C.L. (2002) Cross-sectional estimation of U.S. demand for beef products: a censored system approach, *Journal of Agricultural and Resource Economics*, vol. 27, pp. 320-334.

Figure

Figure 1: Foci of research; Integrated modeling of the impacts of climate change upon land use and consequent secondary and subsequent effects



Figure

Figure 2: Change in cereals and in FGM under “naive” climate change

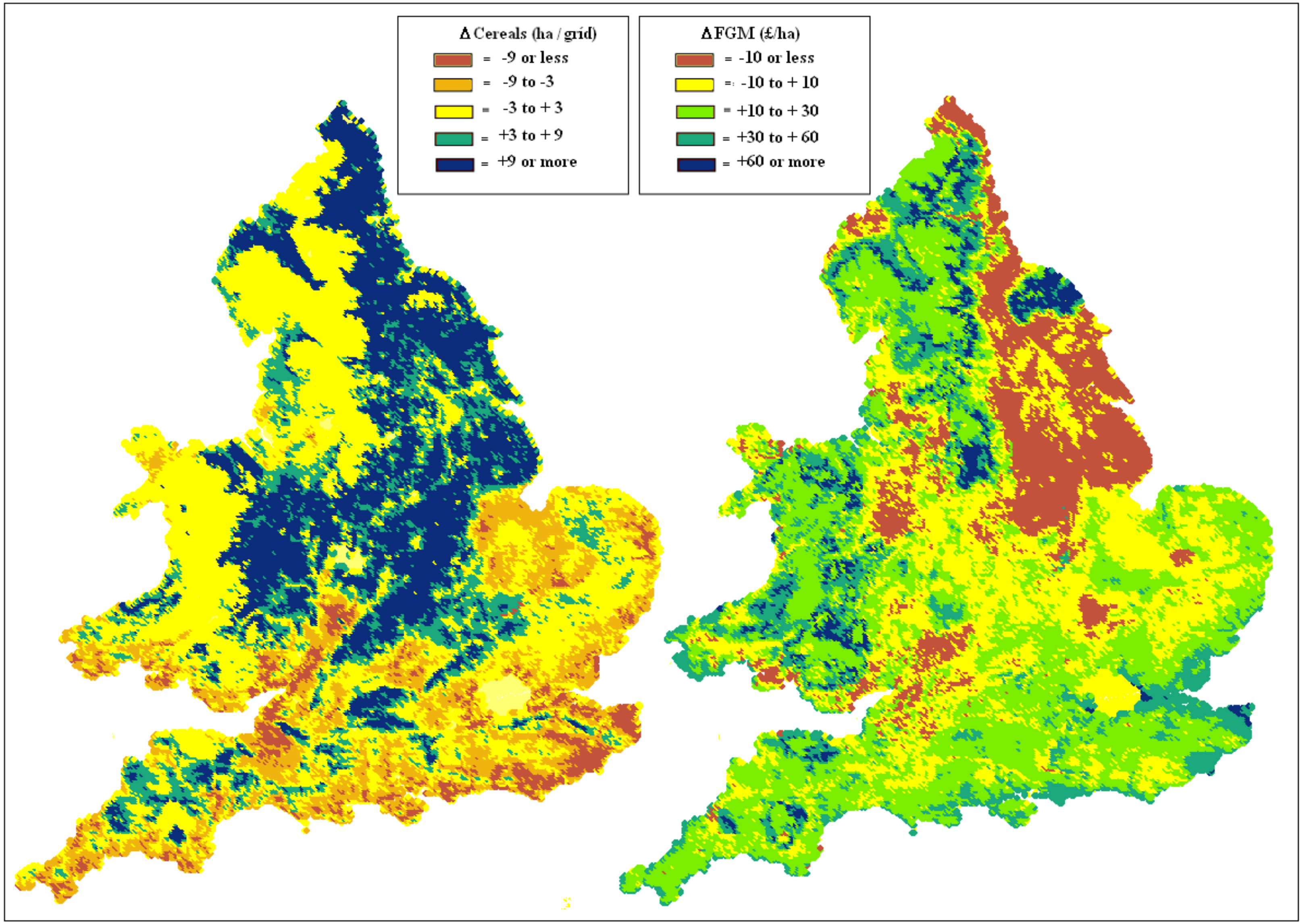


Figure 3: Example of the spatial relationships between two adjacent river basins and buffer zones surrounding water quality monitoring points at their respective outlets.

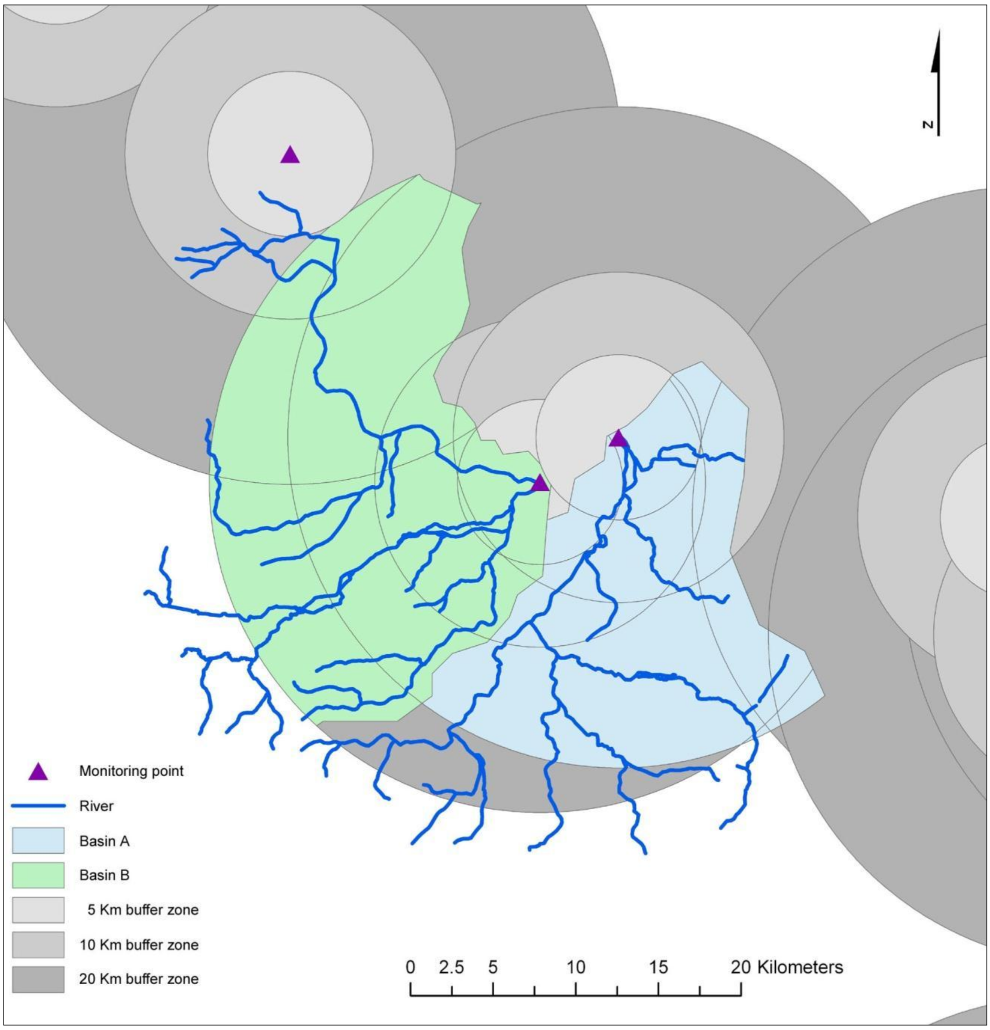


Figure 4: Land use in the River Aire catchment.

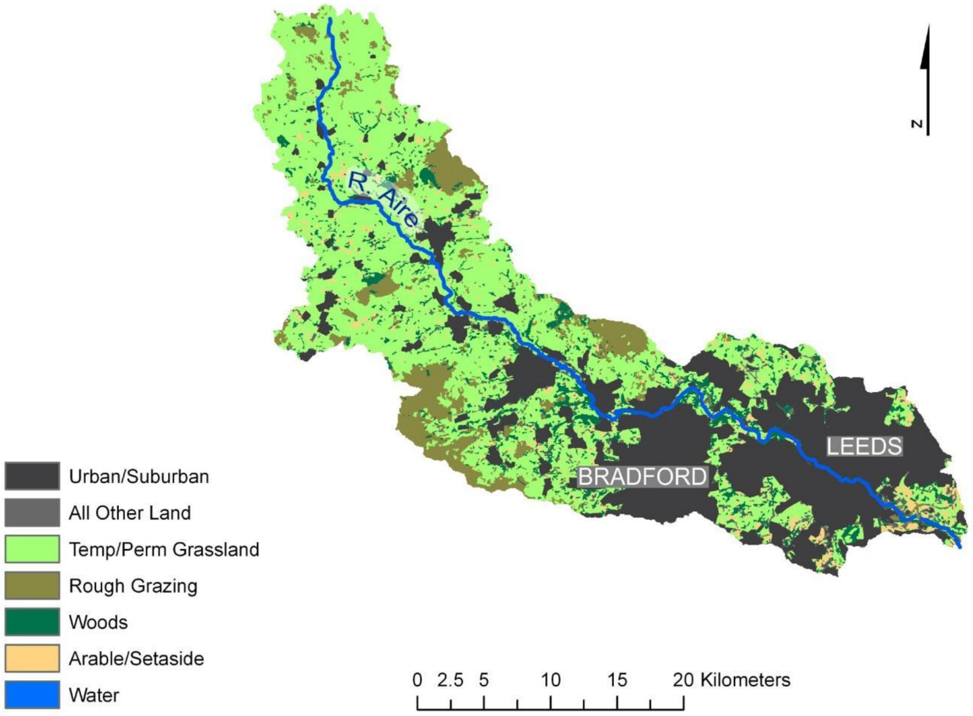
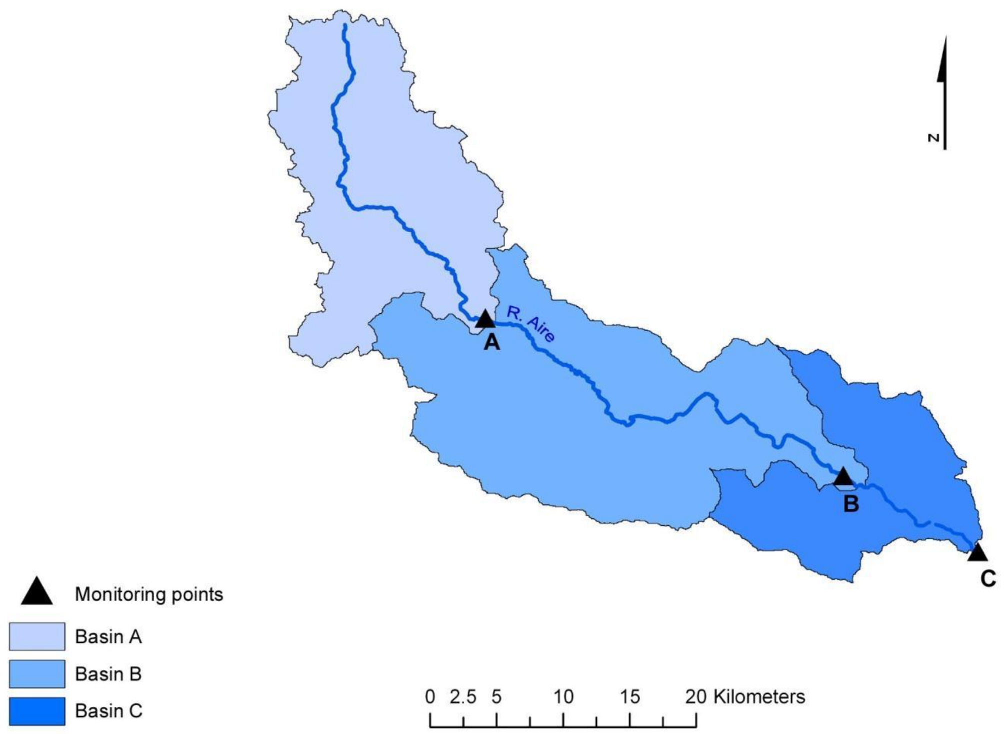


Figure 5: Three sub-basins of the River Aire corresponding to monitoring points A, B and C.



Figure

Figure 6: Sampling area and the quality of recreational access sites

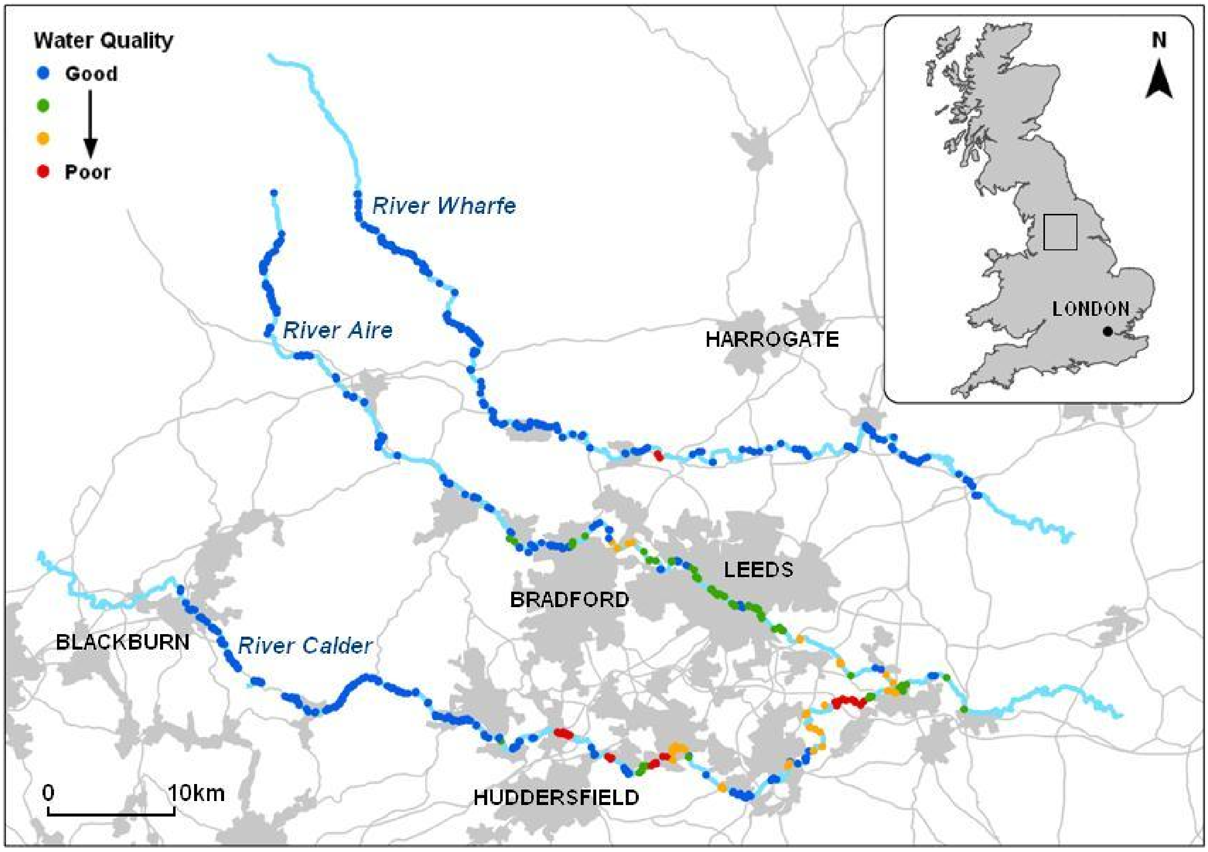


Figure 7: The distribution of per person (left hand panel) and SOA aggregate (right hand panel) value changes for the climate change scenario.

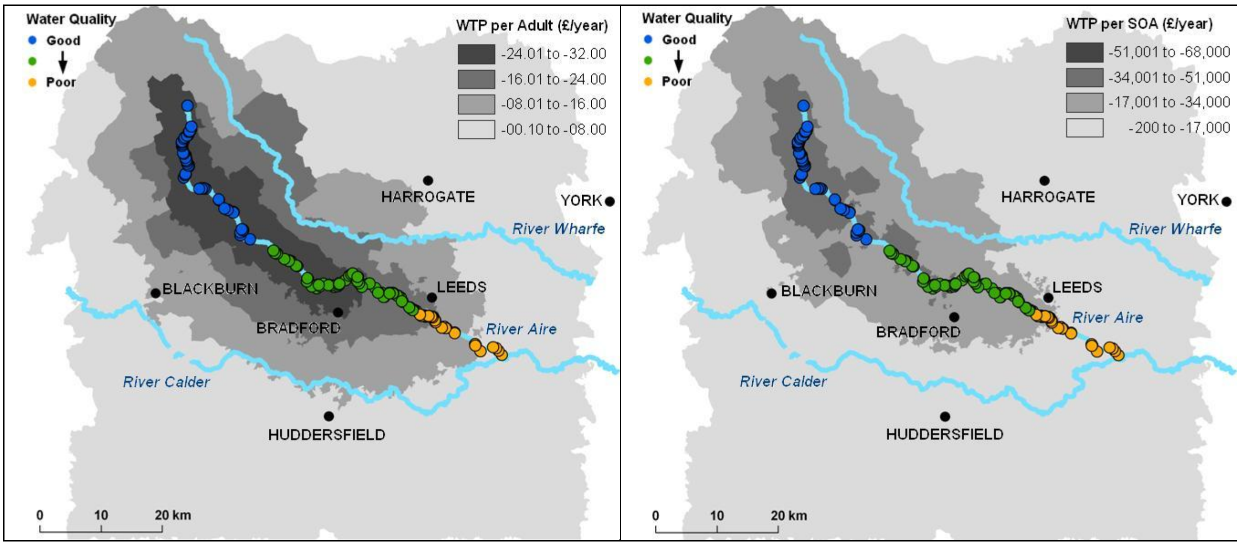
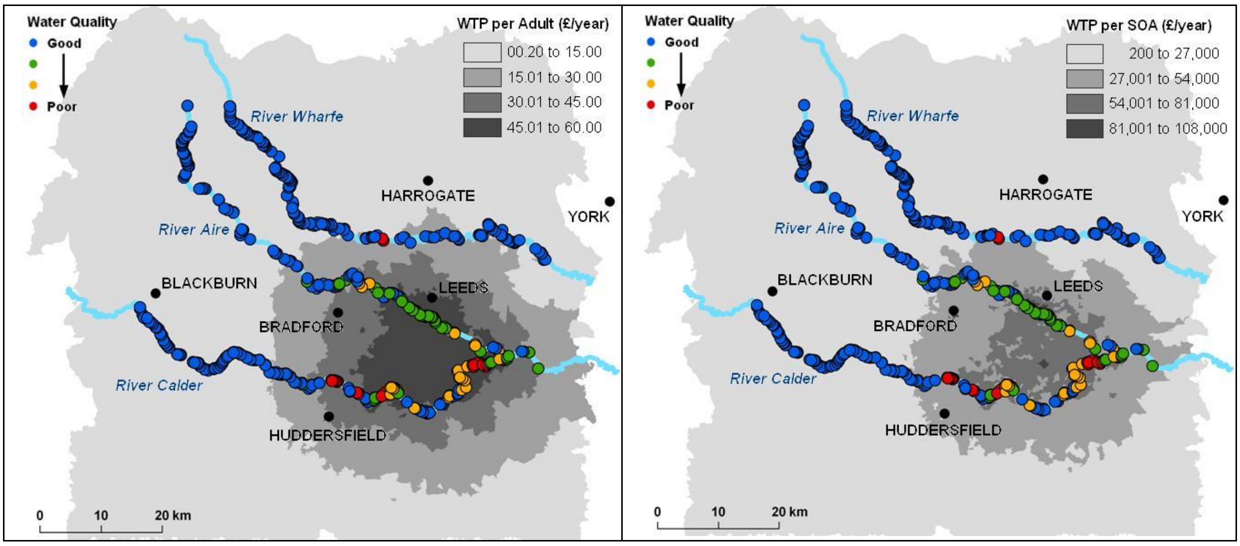


Figure 8: The distribution of per person (left hand panel) and SOA aggregate (right hand panel) value changes for the WFD scenario.



TABLES

For:

Spatially explicit integrated modeling and economic valuation of climate change induced land use change and its indirect effects

Table 1: Descriptive statistics, land uses (ha) and livestock numbers (head) per 2km^2 grid square

	1969	1988	2004		Total			
	$\bar{x}$	$\bar{x}$	$\bar{x}$		$\bar{x}$	$\hat{s}(x)$	Min	Max
Cereals	87.8	94.6	76.4		83.0	77.4	0	347.2
Oilseed Rape	0.1	8.5	13.3		6.9	12.3	0	124.7
Root crops	10.1	9.5	7.5		9.1	18.7	0	186.8
Temp. grassland	41.1	28.8	22.6		29.3	28.7	0	349.5
Perm. grassland	116.7	115.6	112.7		113.0	97.0	0	400
Rough grazing	47.1	39.6	40.5		44.0	100	0	400
Other	22.8	26.6	45.7		37.8	45.6	0	400
Total land	325.6	323.2	318.7		323.1	96.9	1.25	400
Dairy	87.1	71.5	62.0		74.1	99.1	0	1128
Beef	151.4	149.8	89.9		144.9	123.8	0	1221
Sheep	472.2	784.1	323.8		693.6	899.0	0	11289

Notes: only grid squares containing some agricultural land are considered, $\bar{x}$ indicates the sample mean, $\hat{s}(x)$ the sample standard deviation.

Table 2: Land use share equations parameter estimates

	Cereals	**Oilseed rape**	**Root crops**	**Temp. Grassland**	**Perm. Grassland**	**Rough grazing**
$P_{cereals}$	0.134 ***	--	--	-0.044 **	--	--
P_{rape}	--	0.148 ****	--	--	--	--
$P_{rootcrops}$	--	--	0.027 *	--	--	--
$P_{fertilizer}$	-0.111 ***	-0.283 ****	-0.017 *	0.067 ***	-0.018	0.036 *
Set aside rate	-0.425 ****	-0.114 ***	0.003	-0.009	-0.030	-0.025 *
ESA share	-0.033 ****	-0.008 ***	0.000	0.000	0.031 ***	0.032 ***
Park share	-0.019 ***	-0.006	-0.003 ***	-0.018 ***	-0.067 ***	0.041 ***
Urban share	-0.028 **	-0.003	-0.002	0.000	0.061 ***	0.010 *
smore6	-0.087 ***	-0.018 ***	0.000	-0.005	0.131 ***	0.052 ****
Coast	-0.357	-0.505 *	-0.156	1.316 ***	-0.536	1.473 ***
Alt	14.170 ****	3.048 ***	-2.693 ****	-0.787	#	#
atl^2	6.333 ***	1.337 **	-0.494 **	-0.834 *	#	#
alt < 200m	#	#	#	#	-0.057 ****	0.004
alt > 200m	#	#	#	#	0.085 **	-0.156 ***
I(alt > 200m)	#	#	#	#	-25.55 ***	21.96 **
Mwd	4.174 ****	0.079	1.619 ****	0.956 ***	-8.455 ****	-0.582
mwd^2	-1.283 ***	-0.416 ***	0.681 ****	0.147	-1.346 ***	0.271 **
Pt	6.727 ***	1.594 *	0.331 *	-3.419 ***	-23.95 ***	12.46 ***
pt^2	-2.773 **	-1.919 **	0.720 **	3.401 ***	3.969 *	-7.191 ***
Fc	-4.794 *	-7.374 ***	-1.856 ***	0.482	7.165 *	4.394 *
fc^2	16.670 ***	-6.521 ***	2.896 ***	-7.498 ***	-22.22 ***	5.000 ***
Dd	-4.228 ***	1.653 ***	-4.801 ****	4.271 ***	35.45 ****	-6.285 ***
dd^2	2.571 **	-0.233	1.592 ****	-1.506 **	-3.071 *	-1.179 *
Aar	-3.726	-11.57 ****	6.056 ****	3.950 ***	-5.000	9.738 ***
aar^2	-1.269	-7.177 ***	1.701 ****	3.935 ***	-4.537 *	7.246 ***
Trend	0.015	0.282 ****	-0.015 ***	-0.155 ****	-0.101 ***	0.045 ***
Const	38.04 ****	-17.61 ****	6.677 ****	13.34 ****	36.18 ****	-0.884

Notes: to preserve space the residual correlations, the parameters corresponding to the variance equations, to the interactions of the environmental factors are not reported in the Table, but are available under request from the Authors. “--“ = parameters non-significant and therefore removed, “#” = parameter not included in the equation, “*” = t-stat > 2, “**” = t-stat > 3, “***” = t-stat > 4, “****” = t-stat > 10. All variables defined as in Table 1.

Table 3: Land uses and livestock numbers changes and FGM/ha as predicted by our land use model

	FGM/ha	**No climate change**	**Climate change**	**Activity change**
	£/ha	(‘0000 ha)	(‘0000 ha)	%
Cereals	290	298.8	285.4	-4.5
Oilseed Rape	310	41.1	46.6	13.3
Root crops	2400	22.4	16.8	-25.0
T. grassland	0	78.5	83.9	6.9
P. grassland	0	415.4	697.3	67.8
Rough grazing	0	131.4	82.1	-37.5
Other	0	226.7	2.2	-99.0
	£/head	(‘0000 heads)	(‘0000 heads)	(‘0000 heads)
Dairy	570	194.5	219.6	12.9
Beef	70	462.5	506.5	9.5
Sheep	9	2194.2	2632.3	20.0

Table 4: Random effects (GLS) estimates of chlorophyll-*a* concentration (µg/l)

	Coefficient	Standard Error	t-stat
Constant	-3.438401	0.7795196	-4.41
Share of root crops	5.995946	2.821392	2.13
Share of non-agricultural land	0.1239566	0.5439918	0.23
Share of other arable land	0.487991	0.5553501	0.88
Share of temporary grassland	-3.96746	1.707723	-2.32
Number of dairy cows	0.00000873	0.00000234	3.73
Log(Temperature)	1.970617	0.126023	15.64
Log(BFI)	-0.460904	0.2760085	-1.67
Annual average rainfall	-0.000766	0.0002861	-2.68
Log(Suspended sediment)	0.4379061	0.0959914	4.56
R-squared:			
within	0.7924		
between	0.6976		
overall	0.7317		
Number of observations	156		
Number of groups	78		

Table 5: Water quality classifications

Description	Chlorophyll-*a* Threshold	Hime et al., WQL Water Quality color
Hyper-eutrophic	>25 µg/l	Red
Eutrophic	10-25 µg/l	Yellow
Mesotrophic	4-10 µg/l	Green
Oligotrophic	<4 µg/l	Blue

Table 6: Predicted reductions in water quality as a consequence of climate change.

Basin	Present climate		Climate change scenario (+1°C)		Percentage increase in predicted Chlorophyll-*a*	Predicted reduction in WQL classification
	Predicted Chlorophyll-*a* (µg/l)	Corresponding WQL classification	Predicted Chlorophyll-*a* (µg/l)	Corresponding WQL classification		
A	3.39	Blue	5.10	Green	50%	1 class
B	5.63	Green	7.81	Green	39%	No change
C	11.95	Yellow	15.13	Yellow	27%	No change

Table 7: Percentage of all recreational access sites within the study area classified by ecological quality under current situation and climate change

Quality	Present climate	Climate change scenario (+1^{o}C)
Good	76 %	56 %
Medium	11 %	27 %
Poor	13 %	17 %

Table 8: Summary of variables

Variable	Description
Travel cost	Two ways Travel cost defined as: Out-of pocket cost (0.25£ * km) +Adult net income/2000*1/3
Medium water quality	1=if site is green quality; 0 otherwise
Poor water quality	1=if site is below green quality; 0 otherwise
Urban	1=if the predominate land type around the site is urban; 0 otherwise

Table 9: Estimated coefficients from travel cost model

Variable	Coeff (Robust SE)	p-value
Travel cost	-0.16(0.018)	0.0000
Medium water quality	-0.92(0.234)	0.0001
Poor water quality	-1.067(0.221)	0.0000
Urban	0.604(0.14)	0.0000
CSite	-7.43(0.226)	0.0000
COthRiv	-4.41(0.129)	0.0000
CCanal	-3.81(0.079)	0.0000
Clake	-4.13(0.093)	0.0000
COthRe	-2.80(0.071)	0.0000
LL	-488258	