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Research article

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



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ABSTRACT

We present an integrated model of the direct consequences of climate change on land use, and the indirect effects of induced land use change upon the natural environment. The model predicts climate-driven shifts in the profitability of alternative uses of agricultural land. Both the direct impact of climate change and the induced shift in land use patterns will cause secondary effects on the water environment, for which agriculture is the major source of diffuse pollution. We model the impact of changes in such pollution on riverine ecosystems showing that these will be spatially heterogeneous. Moreover, we consider further knock-on effects upon the recreational benefits derived from water environments, which we assess using revealed preference methods. This analysis permits a multi-layered examination of the economic consequences of climate change, assessing the sequence of impacts from climate change through farm gross margins, land use, water quality and recreation, both at the individual and catchment scale.

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1. Introduction

The ecosystem services paradigm (de Groot et al., 2002; Boyd and Banzhaf, 2007; Wallace, 2007; Fisher et al., 2008; Fisher and Turner, 2008) underpins a rapidly improving and expanding body of research linking environmental change to human welfare (e.g. Goldstein et al., 2012; Jackson et al., 2013; Lawler et al., 2014; Guerry et al., 2015). An important and promising avenue of work examines the direct impacts of changes, considering multiple ecosystem services (e.g. Nelson et al., 2009; Rockström et al., 2009)

and at varying spatial scales (van Delden et al., 2011). However, even when multiple ecosystem services are considered, relatively few studies manage to capture the knock-on effects arising along a chain of interconnected ecosystem processes (Milne et al., 2009; Bennett et al., 2009). Inherent complexity and myriad interdependencies within the natural world mean that any change, whether driven by environmental processes or human behavior, is likely to have a 'domino' effect, acting upon a long chain of ecosystem services. For instance, considering livestock production in the Central French Alps, Lamarque et al. (2014) show that the direct impact of climate change on plant functional traits (see de Bello et al., 2010) drives a secondary, indirect impact upon agricultural land management, with implications for ecosystem service provision. Focusing on the United Kingdom (UK), Fezzi et al. (2015) go further by examining the direct consequences of climate change on land use and farm incomes, and the subsequent, indirect effects

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on fresh water quality providing a cost-effectiveness analysis of climate change adaptation strategies. Other studies link predictions of future environmental change with cost-effectiveness analyses of alternative mitigation measures (e.g. Whitehead et al., 2013), but studies which complete the chain of effects in a cost-benefit framework are rare. This poses challenges for investment and policy decisions, since options that maximize net benefits cannot be robustly identified when the full range of impacts is unknown.

This paper contributes towards filling this research gap by using a spatially explicit, integrated modeling approach to examine a chain of related impacts arising from a simple climate change scenario applied to a selected river catchment and, crucially, valuing these impacts in economic terms. Using the UK's River Aire as a case study, we first examine the direct effect of climate change on agricultural land use, as mediated through changes in farming behavior. Resulting land use change (LUC) has repercussions on freshwater quality, as crop selection and livestock intensities impact nutrient run-off into rivers. Finally, we examine how changes in water quality will impact upon outdoor recreation. The integrated analysis elicits economic values throughout the chain of impacts, reporting recreation values at both the individual and catchment level (where our spatially explicit approach delivers a further contribution by revealing the key role which location plays when aggregating from individual to catchment level benefits). Including these secondary (often non-market) impacts is especially important in environmental management as their value can outweigh that of direct impacts, possibly by a large margin (Bateman et al., 2014). Furthermore, the spatially explicit nature of our analysis provides decision makers with the information necessary to permit locational targeting of interventions, enhancing the efficiency of policies to mitigate climate change effects or improve distributional equity.

This paper is organized as follows. The next section introduces the spatially sensitive agricultural land use model that underpins our initial analysis. Section 3 uses this model to examine the likely impact of a stylized climate change scenario (a uniform temperature increase by 1 °C throughout the country) on land use and farm incomes. Section 4 develops a spatially explicit, transferable model for predicting changes in river water quality arising from agricultural LUC within a world of altering climates. These predictions are then refined for use within the UK's River Aire catchment, for which we hold data on recreational demand and associated value. Section 5 develops a revealed preference model, encompassing the changes in the ecological quality of rivers derived previously, to predict the impact of our climate change scenario on recreational values. Section 6 presents the results of our integrated analysis, reporting recreational values at the individual and catchment scales. Section 7 concludes.

2. Land use modeling

This section briefly introduces the agricultural land use model, the data used for its estimation and provides a summary of the main results. For further details, see the supplementary materials and Fezzi and Bateman (2011).

2.1. Specification

Adapting the agricultural land use model developed by Fezzi and Bateman (2011) and expressing the farmer's objective in terms of profit maximization per unit of land, the optimal land use allocation problem can be written as:

$$\pi^L(\mathbf{p}, \mathbf{w}, \mathbf{z}, L) = \max_{s_1, \dots, s_k} \left\{ \pi^L(\mathbf{p}, \mathbf{w}, \mathbf{z}, L, s_1, \dots, s_h) : \sum_{i=1}^h s_i = 1 \right\}, \tag{1}$$

where $\pi^L(\cdot)$ indicates profits per unit (ha) of land, \mathbf{p} the vector of strictly positive output prices, \mathbf{w} the vector of strictly positive input prices, L the total land available (which is kept constant for all analyses), \mathbf{s} the vector of land use shares (with land allocated across h potential land uses), and \mathbf{z} the vector of k other fixed factors (e.g. physical and environmental characteristics, policy incentives and constraints, etc.).

Since the profit per unit of land function is positively linearly homogenous and strictly convex in input and output prices, applying Hotelling's lemma yields output supply (y^L) and input demand (r^L) per area (hereafter referred to as input and output intensities) as:

$$y_i^L(\mathbf{p}, \mathbf{w}, \mathbf{z}, L) = \frac{\partial \pi^L(\mathbf{p}, \mathbf{w}, \mathbf{z}, L)}{\partial p_i} = \frac{\pi^L(\mathbf{p}, \mathbf{w}, \mathbf{z}, L, \bar{s}_1, \dots, \bar{s}_h)}{\partial p_i}, \text{ and} \tag{2.a}$$

$$r_j^L(\mathbf{p}, \mathbf{w}, \mathbf{z}, L) = \frac{\partial \pi^L(\mathbf{p}, \mathbf{w}, \mathbf{z}, L)}{\partial w_j} = \frac{\pi^L(\mathbf{p}, \mathbf{w}, \mathbf{z}, L, \bar{s}_1, \dots, \bar{s}_h)}{\partial w_j} \tag{2.b}$$

where $\bar{s}_1, \dots, \bar{s}_h$ indicates optimal shares.

Since land is allocated optimally when shadow prices (i.e. marginal rents) are equalized across all land uses, optimal land use shares are described by:

$$\frac{\partial \pi^L(\mathbf{p}, \mathbf{w}, \mathbf{z}, L, \bar{s}_1, \dots, \bar{s}_h)}{\partial s_i} = 0, \text{ for } i = 1, \dots, h \tag{3}$$

When these equations are linear in the optimal land allocations (as per Fezzi and Bateman, 2011), including the constraint that the sum of shares must equal unity 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 (see Fezzi and Bateman, 2011).

We specify the empirical profit function per hectare as a Normalized Quadratic (NQ) function. Defining w_n as 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:

$$\begin{aligned} \bar{\pi}^L = & \alpha_0 + \sum_{i=1}^{m+n-1} \alpha_i x_i + \frac{1}{2} \sum_{i=1}^{m+n-1} \sum_{j=1}^{m+n-1} \alpha_{ij} x_i x_j + \sum_{i=1}^{h-1} \beta_i s_i + \frac{1}{2} \sum_{i=1}^{h-1} \\ & \times \sum_{j=1}^{h-1} \beta_{ij} s_i s_j + \sum_{i=1}^{k+1} \gamma_i z_i^* + \frac{1}{2} \sum_{i=1}^{k+1} \sum_{j=1}^{k+1} \gamma_{ij} z_i^* z_j^* + \sum_{i=1}^{m+n-1} \\ & \times \sum_{j=1}^{h-1} \delta_{ij} x_i s_j + \sum_{i=1}^{m+n-1} \sum_{j=1}^{k+1} \varphi_{ij} x_i z_j^* + \sum_{i=1}^{h-1} \sum_{j=1}^{k+1} \phi_{ij} s_i z_j^*, \end{aligned} \tag{4}$$

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 as the residual can be computed by difference and is therefore redundant. Input and output intensities can be derived as in (2), whereas

optimal land use shares can be derived by solving the system (3) of $h-1$ equations with the land additivity constraint $\sum_{j=1}^h s_j = 1$. The resulting equations are linear functions of output prices, input prices, and fixed factors.

2.2. Data and descriptive statistics

In order to model the financial, policy and environmental drivers of LUC, we develop a unique database of land use drivers. Integrating multiple sources of information dating back to the late 1960s we identify six land use types (cereals, oilseed rape, root crops, temporary grassland, permanent grassland and rough grazing) which together cover 88% of UK agricultural land. The remaining 12% is included as a combined 'other' category. Summary information regarding the land use data is provided in the [supplementary materials](#). Further data were extracted for each 2 km Ordnance Survey National Grid square. Data on average annual rainfall, autumn machinery working days, mean potential evapotranspiration, median duration of field capacity, total number of degree days in the growing season and mean elevation were taken from the National Soil Resources Institute LandIS database. We also include the share of agricultural land with slope greater than 6° derived via Geographic Information System (GIS) analysis of the Ordnance Survey Digital Terrain Model (DTM). The model also includes 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, regional output prices are included, using the agricultural output regional price statistics extracted from the UK Farm Business Survey for years 1982–2000, whereas input prices are available at the national level.

2.3. Results

Equation (4) can be estimated via maximum likelihood. Since not all farms cultivate all crops the model is specified as a multivariate Tobit (Tobin, 1958). Fezzi and Bateman (2011) illustrate a computationally-efficient Quasi Maximum Likelihood (QLM) approach which can be used when the number of Tobit equations is high, as in our case (details provided in [supplementary materials](#)). We estimate two censored Tobit systems: the 3 livestock intensity (dairy cattle, beef cattle, sheep) equation system; and the 6 land use shares (cereal, oilseed rape, root crops, temporary grassland, permanent grassland, rough grazing) system. These are the main determinants of the impact of agriculture on the environment. In fact, in a relatively small country such England, there is little variation in fertilizer inputs for a given arable crop, and therefore a single national set of surplus values for such crops is appropriate (Lord et al., 2002). However, nutrient inputs to grassland can vary substantially depending upon the livestock intensity. Therefore, a more accurate approach is to assume that predicted changes in stocking rates indicate changes in fertilizer applications.

Table 1 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, as expected 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 Table 1, 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. Model results are contrasted with the established land use share model (e.g. Wu and Segerson, 1995) in Fezzi et al. (2014a,b) and found to provide superior in and out of sample performance.

3. From climate change to agricultural land use change

The model estimated in the previous section can be used to predict agricultural land use in England and Wales under a variety of conditions. Clearly numerous scenarios could be considered using such a model but in the present analysis we restrict ourselves to consideration of the effects of a climate change induced increase in temperature and hold other determinants constant (an assumption that could be relaxed to examine related effects such as shifts in precipitation; see Fezzi and Bateman, 2015). We first construct a baseline scenario keeping all land use determinants fixed at the level of the last year for which we have observations for the entire case study area (2004). Then we simulate the LUC arising from a simple climate change scenario obtained by holding all land use determinants (prices, policy, urbanization, etc.) constant and increasing average daily temperatures by 1 °C (an increase expected to occur in the UK by about 2030; UKCP, 2009). We can translate the predicted changes in land use into variations in farm income. Profit data are not available at the highly disaggregated spatial resolution used throughout our analysis. Instead we use the related and commonly adopted measure of Farm Gross Margin (FGM), defined as the difference between revenues from agricultural activities and associated variable costs. For illustrative purposes predicted changes can be reported in terms of income by simply applying the average FGM for each activity calculated from the 2004 Farm Business Survey (source: Fezzi et al., 2010) to the land use and livestock data in each 2 km² grid cell. Results are reported in Table 2.

Considering arable production, the climate scenario induces a shift away from cereals and root crops towards more heat 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 (for a detailed discussion, see Fezzi and Bateman, 2011).

A key advantage of our modeling framework is its high spatial sensitivity. The left hand panel of Fig. 1 illustrates the highly spatially heterogeneous nature of changes to the area of cereals. This area increases substantially in northern parts of the country, where warmer temperatures raise cereal yields. Conversely in the south of the country, cereal area decreases as farms substitute into other activities. As illustrated in the right hand panel of Fig. 1, and in line with previous research (e.g. Fezzi et al., 2014a; Fezzi and Bateman, 2015), our results confirm that climate change will generally increase farm incomes. This is in line with expectations given that the increase in temperature is likely to boost yields in the relatively cold and wet UK climate. However, some localized negative effects may be expected, mainly in the East, where higher temperatures limit high-revenue root crops.

4. From land use change to water quality impact

Agricultural LUC alters the flow and quantity of diffuse pollution into rivers. Evaluating the impact of these changes in terms of water quality (the biological status of rivers) requires an understanding of the ecological response induced by LUC. Prior work relates land use change to 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

Table 1
Land use share equations parameter estimates.

Description		Cereals	Oilseed rape	Root crops	Temp. Grassland	Perm. Grassland	Rough grazing
P_{cereals}		0.134 ***	a	a	-0.044 **	a	a
P_{rape}		a	0.148 ****	a	a	a	a
$P_{\text{rootcrops}}$		a	a	0.027 *	a	a	a
$P_{\text{fertilizer}}$		-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 *
<i>Environmental determinants</i>							
smore6	Slope greater than 6°	-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	Altitude	14.170 ****	3.048 ***	-2.693 ****	-0.787	b	b
atl ²	–	6.333 ***	1.337 **	-0.494 **	-0.834 *	b	b
alt <200 m	–	b	b	b	b	-0.057 ****	0.004
alt >200 m	–	b	b	b	b	0.085 **	-0.156 ***
l(alt > 200 m)	–	b	b	b	b	-25.55 ***	21.96 **
Mwd	Machinery working days	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	Potential evapotranspiration	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	Field capacity	-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	Degree days in growing season	-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	Average annual rainfall	-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	Constant	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. Footnotes: a = parameters non-significant and therefore removed; b = parameter not included in the equation, * = t-stat > 2, ** = t-stat > 3, *** = t-stat > 4, **** = t-stat > 10.

et al., 2008, 2010; Hutchins et al., 2009, 2010b). Here we directly model the relationship between land use and its ecological impact on the water environment through the commonly adopted measure of chlorophyll-*a* concentrations. This allows us to assess the overall impact of particular changes in land use rather than relying purely on nutrient models.

We use panel data observations, provided by the Centre for Ecology and Hydrology (CEH) (Davies and Neal, 2007), on the concentration of chlorophyll-*a* at individual monitoring points in rivers across England and Wales. The incidence of chlorophyll-*a*, as represented by summer mean concentration, is a measure of the rate of algal production in a water body, and is a commonly used indicator of water quality in both natural and social science research because it can both identify the risk of eutrophication of aquatic ecosystems and is directly perceived by the general public as a measure of water quality (e.g. Cullen, 1982; Boyer et al., 2009).

As potential predictors of chlorophyll-*a* concentration we consider variables detailing catchment 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 phosphate. Therefore, we expect land use variables to be very important in determining chlorophyll-*a* concentration and overall river ecological quality. Climatic variables considered here include temperature, solar radiation and average annual rainfall levels. Lower temperatures are expected to be associated with lower concentrations of chlorophyll-*a* as lower thermal energy inhibits algal production. Solar 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, which indicates the speed and the 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-*a*, as faster flow rates inhibit algal production and dilute nutrients.

Land use and chlorophyll modeling was undertaken for river basins contributing runoff and leached nutrients at 83 Environment Agency monitoring points along river networks throughout England and Wales during 2006, thus enabling modeled data to be evaluated against observed chlorophyll concentrations. This data also includes river characteristics such as water temperature, base flow index etc. The spatial extent of each contributing river basin was derived using a digital elevation map from the Ordnance Survey Land-Form PANORAMA DTM (www.edina.ac.uk). Using data from the June Agricultural Census (www.edina.ac.uk) and MAGIC Agricultural Land Classification (www.magic.gov.uk) we derived a set of land use variables representing livestock head counts and total areas of land under various agricultural and non-agricultural uses at the basin level.

4.1. The river water quality model

The chlorophyll-*a* concentration ($\mu\text{g/l}$) model is specified as a function of land use, climatic and hydrological variables. Several functional forms were tested, beginning with a simple linear model and progressing through a variety of functional forms (including log-log, log-square and log-square root forms) allowing for non-linear and interaction effects. The best fitting model which also conforms to hydrological theory was

Table 2
Changes in land use, livestock numbers and FGM/ha as predicted by the land use model.

	FGM/ha (1)	Under present climate (2)	Under climate change (3)	Climate induced change in area or livestock (4)
	£/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 head)	('0000 head)	%
Dairy	570	194.5	219.6	12.9
Beef	70	462.5	506.5	9.5
Sheep	9	2194.2	2632.3	20.0

Notes: The table shows, for each land use and livestock type, the predicted impact of a 1 °C increase in mean daily temperature. Column (1) reports FGM values in £/ha and £/head, with predicted total agricultural land use areas and livestock heads for England and Wales (under present climate) reported in Column (2). Column (3) reports total areas and livestock numbers after the 1 °C temperature change, as predicted by our land use model. Column (4) reports percentage changes.

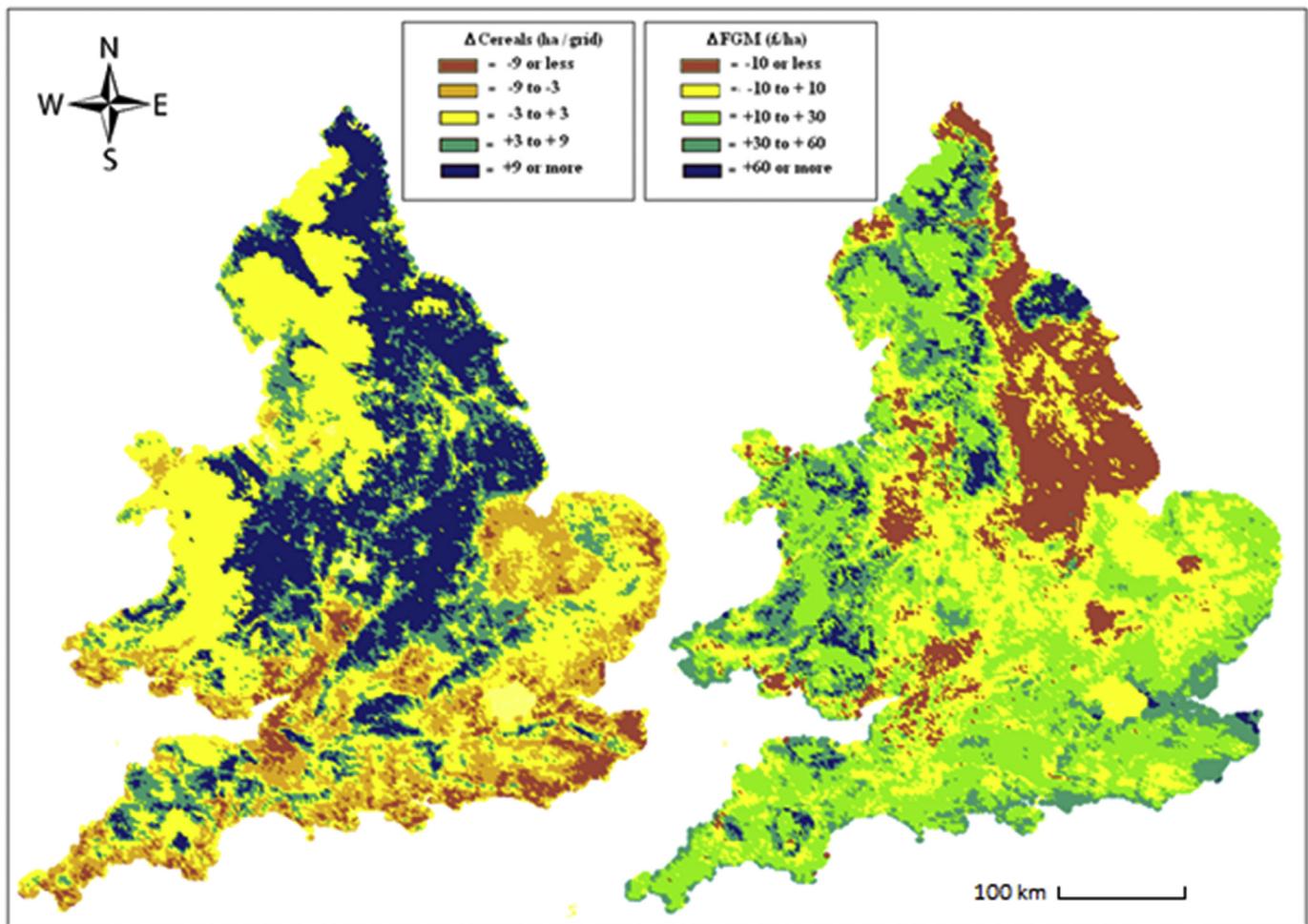


Fig. 1. Change in the spatial distribution of cereals and FGM as a result of climate change (a 1 °C increase in mean daily temperature).

$$\begin{aligned} \log(\text{Chlorophyll})_{i,t} = & \alpha + \beta'_1 \mathbf{s}_{i,t} + \beta'_2 d_{i,t} + \beta_3 \text{Log}(temp_{i,t}) \\ & + \beta_4 \text{Log}(BFI_{i,t}) + \beta_5 \text{AARainfall}_{i,t} \\ & + \beta_6 \text{Log}(SSediment_{i,t}) + \delta_i + e_{i,t} \end{aligned} \quad (5)$$

where i denotes the monitoring point and t indicates whether the observation relates to summer or winter; $\mathbf{s}_{i,t}$ is a vector of shares of

different land uses, $d_{i,t}$ is the number of dairy and beef cattle per square mile in the basin, $temp_{i,t}$ is the average atmospheric temperature, $BFI_{i,t}$ is the base flow index, $AARainfall_{i,t}$ is the annual average rainfall, $SSediment_{i,t}$ indicates the quantity of suspended sediments, δ_i is a residual error term specific to the monitoring point i (random effect) and $e_{i,t}$ is a residual term. Both δ_i and $e_{i,t}$ are assumed to be normally distributed.

The parameters α , β_1 , β_2 , β_3 , β_4 , β_5 and β_6 are estimated via

Generalized Least Squares (GLS) random effects model (see Greene, 2003) with heteroscedasticity consistent standard errors obtained using the White (1980) correction. Results are reported in Table 3. After testing for parameter equivalence, urban and non-agricultural land shares were combined into a single category, while root crops were separated from 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 consistently found to be an insignificant explanatory variable and was dropped from the model. The intensity of dairy cattle was included separately in the model as dairy farms make more intensive use of land and have higher nutrient inputs than other livestock farms whose effect on water quality is primarily captured by the grassland variables. The effect of stocking densities for other livestock, such as sheep, were examined and found to be insignificant; a finding which is echoed in studies examining other water quality measures (Crowther et al., 2011).

The land use share results in Table 3 are as expected. Relative to the rough grazing baseline, the share of root crops has the largest positive association with chlorophyll-*a* concentration while the share of temporary grassland has the largest negative association reflecting the fact that it is the land use which is most typically used for dairy cattle. Once we account for the intensity of dairy cattle the overall impact on chlorophyll-*a* concentration becomes even higher than that of rough grazing. Effects of the shares of non-agricultural and 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 climate variables temperature and precipitation 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, its coefficient represents an elasticity. As its value is greater than one this indicates that chlorophyll concentration is elastic with respect to temperature. The negative coefficients on rainfall and the hydrological variable base flow index (the latter being just insignificant) are consistent with the expectation that a faster flowing river, in which nutrients are flushed through more quickly, leaves less time for algal growth. Conversely a positive, significant but inelastic, relationship is observed with respect to the level of suspended sediment in the river as this assists the transport of nutrients from field to waterway (Ebbert et al., 2005).

4.2. Predicting the ecological impact of climate change: a case study

As noted, the land use and ecological quality models draw upon datasets covering large areas and are both characterized by a high level of spatial accuracy. Derived models therefore encompass a wide degree of data variability and should be generally transferable following standard out-of-sample validation tests (the methodology for which is described, for instance, in Bateman et al., 2002, 2003). 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. For illustration, we consider an area for which we also hold revealed preference data for the recreational value of the freshwater environment; namely the catchment of the River Aire in Yorkshire, shown in Fig. 2. This river basin covers 86,000 ha and encompasses highly heterogeneous land uses, water qualities and socioeconomic characteristics. The western half of the catchment is sparsely populated, with the upland areas being dominated by rough grazing and pastoral agriculture. The remainder of the catchment

Table 3
Random effects (GLS) estimates of chlorophyll-*a* concentration ($\mu\text{g/l}$).

	Coefficient	Standard error	t-stat
Constant	-3.44***	0.78	-4.41
Share of root crops	6.00*	2.82	2.13
Share of non-agricultural land	0.12	0.54	0.23
Share of other arable land	0.49	0.56	0.88
Share of temporary grassland	-3.97*	1.71	-2.32
Number of dairy cattle per square mile	0.009**	0.02	3.73
Log(Temperature)	1.97****	0.13	15.64
Annual average rainfall/1000	-0.77*	0.29	-2.68
Log(BFI)	-0.46	0.28	-1.67
Log(Suspended sediment)	0.44****	0.096	4.56
R-squared:			
Within	0.79		
Between	0.70		
Overall	0.73		
Number of observations	156		
Number of groups	78		

*** = t-stat > 2, **** = t-stat > 3, ***** = t-stat > 4, ***** = t-stat > 10.

includes mixed and arable farming, and some high density urban areas, including the large conurbations of Bradford and Leeds. While these urban areas are obviously unavailable for agriculture, their location is both a major determinant of river ecology and of the recreational values generated by any change in water quality.

Data for the land use model are available at 2 km grid resolution covering the entirety of England and Wales. In contrast data for 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. 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 °C rise in temperature. These monitoring points allow for analysis at the catchment and sub-catchment scale, and their locations (one upstream of the urban areas, one in central Leeds and one at the catchment outlet; indicated as points A, B and C, illustrated in Fig. 3) permit differentiation between the agricultural and urban impacts. The monitoring points are sited according to the physical characteristics of the catchment, taking account of hydrological response units (HRUs) corresponding to areas of land that drain into discrete river stretches (Posen et al., 2011). Aggregations of these HRUs can be thought as sub-catchments, with monitoring stations located at their outlets.

While climate change is expected to increase the market value of agricultural production it also generates non-market costs. The induced LUC is associated with increased nutrient application, higher river pollution and lower ecological quality. This in turn generates a potentially major non-market externality in terms of impacts upon the recreational value of rivers (Freeman et al., 2014). To estimate this value within our revealed preference analysis we need to understand the link between the ecological quality of rivers and the recreational behavior and associated values of visitors. Here we have to allow that there may well not be a simple linear relation between chlorophyll-*a* concentrations and the water quality perceived by recreational visitors (Hunter et al., 2012). To bridge this objective-subjective gap we use a tried and tested 'water quality ladder' (WQL), as developed by Hime et al. (2009). This links $\mu\text{g/l}$ measures of chlorophyll-*a* concentrations and corresponding flora and fauna to a simple four level description of water quality (similar to the scale proposed by UKTAG (2008) for which Hime et al. (2009) provide a conversion table). This scale has been shown to significantly determine individuals' preferences and

willingness to pay for river quality in separate stated preference studies of the Aire and various European rivers (Bateman et al., 2011). Table 4 provides ecological quality descriptions for the WQL and a summary of related recreational activities.

We applied our ecological quality model (Table 3) to predict water quality under the present and future climate scenario. Information on likely changes to predictor variables under our stylized climate scenario was gathered through personal communications with staff at CEH, Wallingford. This suggested that a 1 °C rise in air temperature may cause a greater than proportional increase in water temperature. Note that we assume rainfall to remain fixed at the annual average level and, therefore, the base flow index remains unaltered. Finally, the levels of suspended sediment are assumed to increase by 10%.

Table 5 details predicted water quality under the present and future climate scenario. Comparison of chlorophyll-*a* measures shows that at all three monitoring points we project a decline in future ecological quality arising from both the direct effect of water temperature increases and higher levels of nutrient loading arising from the climate-induced shift in land use. In relative terms this decline is greatest at the upper levels of water quality, where we see a downward shift from pristine to good quality on the WQL scale. In absolute terms, the increase in chlorophyll-*a* is greatest at lower levels of water quality. However, these are less marked in relative

terms and do not breach the boundaries of respective WQL classes.

5. Recreation value impacts

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 water quality changes predicted in Table 5 will affect visitation at sites which are available for recreational access.

5.1. Sample survey and GIS data generation

A large sample survey of households was undertaken to estimate the recreational impact of changes in ecological quality. To capture the distance decay of values away from an improvement site (Bateman et al., 2006), a survey area was defined spanning a 70 km diameter centered on the River Aire, thereby embracing its catchment and surrounding areas. This helped to capture likely substitution effects generated by competing resources such as alternative rivers sites and other outdoor recreational opportunities.

The survey 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.

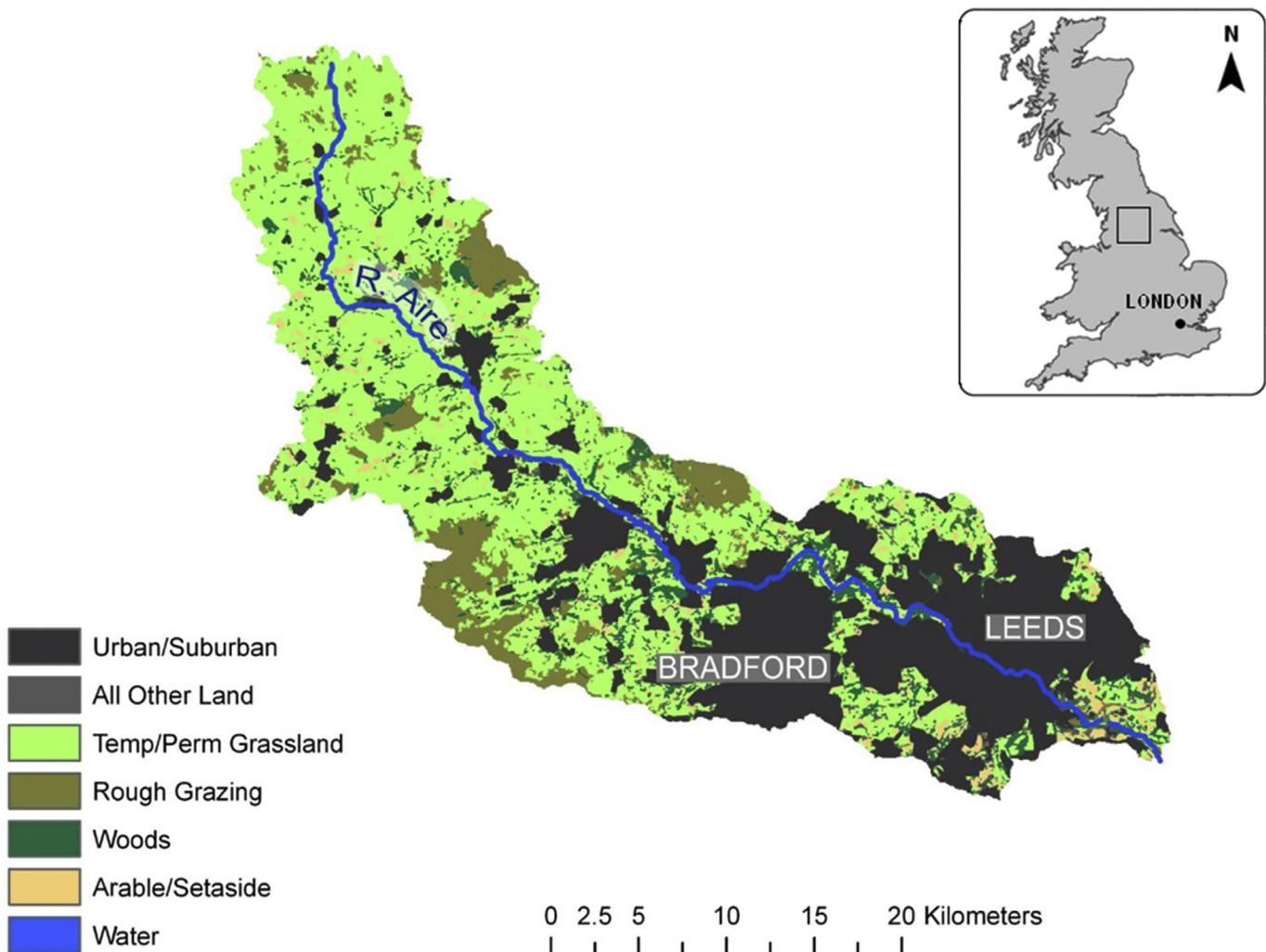


Fig. 2. Land use in the River Aire catchment.

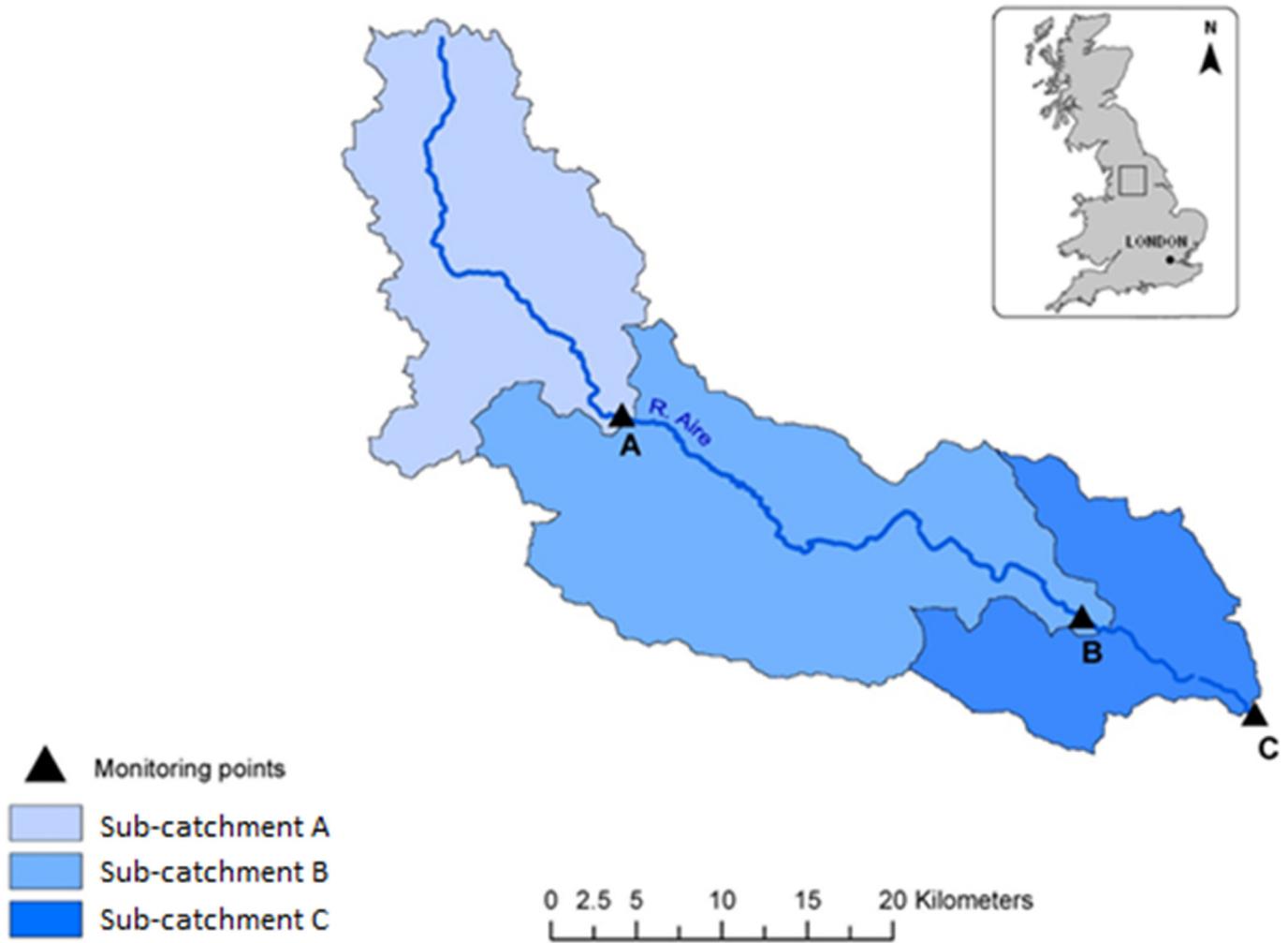


Fig. 3. Three sub-catchments of the River Aire corresponding to monitoring points A, B and C.

During the interview, respondents were shown an interactive map on a computer screen indicating the respondent's home location and all of the surrounding rivers within an area the same size as the full survey area. Respondents indicated on the map the river locations they visit for recreation and the frequency of their visits to each site. We also collected information regarding the total number of all outdoor trips taken in the last 12 months, within this the total number of visits to water bodies and detailed information about the river sites.

Once the interviews were completed, a GIS was used to match site visit locations to real world recreational sites using the four-step process described in the [supplementary materials](#). In total,

531 recreational sites were identified along the studied rivers, which span approximately 230 km in length. The home location of each respondent was identified by linking Ordnance Survey Address Point data to the detailed geographical location information given by the full postcode provided by the respondent. Distance by road (as opposed to straight line) and travel time by car (adjusting for road quality, urban congestion, etc. using the methodology detailed by [Bateman et al., 1996](#)), from each respondent's home to each recreational site, including those not visited by the respondent, were calculated. This allowed us to examine the influence of the availability of substitute sites on the location and number of visits. Information on the environmental characteristics

Table 4
Water quality classifications.

WQL quality	Ecological description	Chlorophyll- <i>a</i> threshold ($\mu\text{g/l}$)	Recreational activities	Flora and fauna
Pristine	Oligotrophic	<4	Fishing, boating & swimming	No algae, all species of fish (game and coarse), water plants and birds, no water turbidity.
Good	Mesotrophic	4–10	Fishing, boating & swimming	No algae, no game fish, many species of coarse fish, water plants and birds, some water turbidity.
Mixed	Eutrophic	10–25	Fishing and boating	Some algae, no game fish, fewer species of coarse fish, water plants and birds. High water turbidity.
Poor	Hyper-eutrophic	>25	None	High levels of algae, no fish, no or few water plants or birds. Highest levels of water turbidity.

Source: [Hime et al. \(2009\)](#).

Table 5
Predicted reductions in water quality as a consequence of climate change.

Sub-catchment	Present climate		Climate change scenario (+1 °C)		% increase in predicted Chlorophyll- <i>a</i>
	Predicted Chlorophyll- <i>a</i> (µg/l)	Corresponding WQL classification	Predicted Chlorophyll- <i>a</i> (µg/l)	Corresponding WQL classification	
A	3.39	Pristine	5.10	Good	50%
B	5.63	Good	7.81	Good	39%
C	11.95	Mixed	15.13	Mixed	27%

of the recreational sites was identified in the GIS using Ordnance Survey MasterMap and CEH Land Cover Map datasets. These provided details of the predominant land use around each of the recreational sites (e.g. urban). The current water quality at each of the river recreational sites was calculated from Environment Agency long-term water quality monitoring data and categorized into the four-point scale given in Hime et al. (2009), as illustrated in Fig. 4.

After removing 2% of respondents due to missing address information or other item non-response from the survey sample, 1782 face-to-face at-home household interviews remained for analysis. Sample characteristics closely matched census data for the study catchment, with 44% of respondents being male, an average household size of 2.6 and average net income of £21,317 per annum (s.d. £11,700). In terms of occupation, 26% of respondents were in full time employment, 13% part-time employed, 33% retired and 7% self-employed with the remaining 21% not in employment.

The purpose of economic analyses of recreation choices is to reveal trade-offs between money and the availability and quality of natural resources. The random utility model (RUM) provides a standard approach for analyzing recreational behavior (Freeman et al., 2014). In the simplest multi-site model the only relevant information required is the site choice made by respondents. However, a change in natural resource quality will affect not only the choice of sites but also the frequency of visits. To address this issue we implement a simplified version of the Morey et al. (1993) approach, as described below. Although there are numerous discrete formulations for modeling site choice, we use a specification which is widely adopted in the literature: the Conditional Logit Model (CLM) with alternative specific constant, based on McFadden (1974).

We specify the utility associated with visiting a recreation site as a function of access costs, water quality levels and other site characteristics. Following Hynes et al. (2009) 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 the respondent's wage (Fezzi et al., 2014b suggests using 3/4 of the average wage rate).

Given the relatively short distance of the recreation sites from the respondents' houses, we assume that every day in a year might, in theory, provide a visit occasion ($T = 365$). Within our sample, we observe the frequencies of visits to the 531 river access points, to other rivers in the sampling area, canals, lakes and other outdoor activities. Finally, we observe the number of times each individual decided not to take outdoor trips, giving that option the index $j = 0$ (opt-out option). In this framework the individual i makes daily choices across the J options available (where $j = 0, 1, 2, \dots, 535$ within which $j = 532, \dots, 535$ represent visits to other rivers, canals, lakes and other outdoor trips). The individual chooses the option with the highest utility on each occasion. We define this utility U_{ijt} that respondent i in period t receives from a visit at site j as the random function:

$$U_{ijt} = v_{ijt} + \varepsilon_{ijt} = f(\mathbf{F}_j, \mathbf{G}_{ij}, \theta) + \varepsilon_{ijt} \quad (6)$$

where \mathbf{F}_j includes site characteristics that are constant across choice

occasions and respondents, \mathbf{G}_{ij} includes respondents' characteristics that change across sites, θ is the parameter vector to be estimated, and ε_{ijt} is a random component that is unobservable to the analyst. This model posits that, given J recreation site options and the possibility of an opt-out, each respondent will choose either to recreate at the site that provides the highest random utility U_{ijt} (or not to recreate if this yields higher utility). Specifying a linear in parameters utility function, Equation (6) can be rewritten for each choice occasion as:

$$U_{ij=1, \dots, 531} = \alpha_j + \eta' \mathbf{F}_j + \beta' \mathbf{G}_{ij} + \varepsilon_{ij}, \text{ for each site within the sampling area} \quad (6.1)$$

$$U_{ij>531 \text{ or } j=0} = \alpha_j + \varepsilon_{ij}, \text{ for the other recreation options and the opt-out} \quad (6.2)$$

where $\theta = (\alpha, \beta, \eta)$. In this structure the utility of visiting sites other than the river stretches 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, this latter choice also captures the utility of leisure and recreation opportunities other than those provided by outdoor trips.

Morey et al. (1993) formulate their repeated recreational choice model assuming that the error term is distributed as a Generalized Extreme Value random term. However, as demonstrated in Scarpa et al. (2005) a similar approach is obtained using an Extreme Value type I error term with alternative specific constants:

$$P[U_{ik} > U_{ij}] = \frac{e^{\alpha_k + \eta' \mathbf{F}_k + \beta' \mathbf{G}_{ik}}}{\sum_j e^{\alpha_j + \eta' \mathbf{F}_j + \beta' \mathbf{G}_{ij}}} \quad (7)$$

The model presents a globally concave likelihood function. Table 6 reports the description of variables and the parameter estimates, obtained via Maximum Likelihood. Note that the two lowest water quality levels (from Table 4) are merged into the category 'poor' while the 'pristine' level is set as the baseline. All variables in Table 6 are highly statistically significant and accord with prior expectations. The "travel cost" variable is significant and negatively signed as expected. The water quality variables are also significant and have expected negative signs indicating utility reductions from the baseline 'pristine' water quality. The positive sign on the "urban" parameter indicates that utility increases if the river site is in an urban area, suggesting that increasing the opportunity to utilize natural resources in highly populated areas might have a greater impact on welfare than in rural areas, possibly because of the lack of similar alternatives in everyday opportunities within cities. However, it is also possible that the positive sign on the urban parameter can be explained by correlations with other facilities available at river sites (e.g. car parks) and opportunities for complementing river recreation experiences with other types of outdoor recreation (e.g. taking children to playgrounds). All alternative

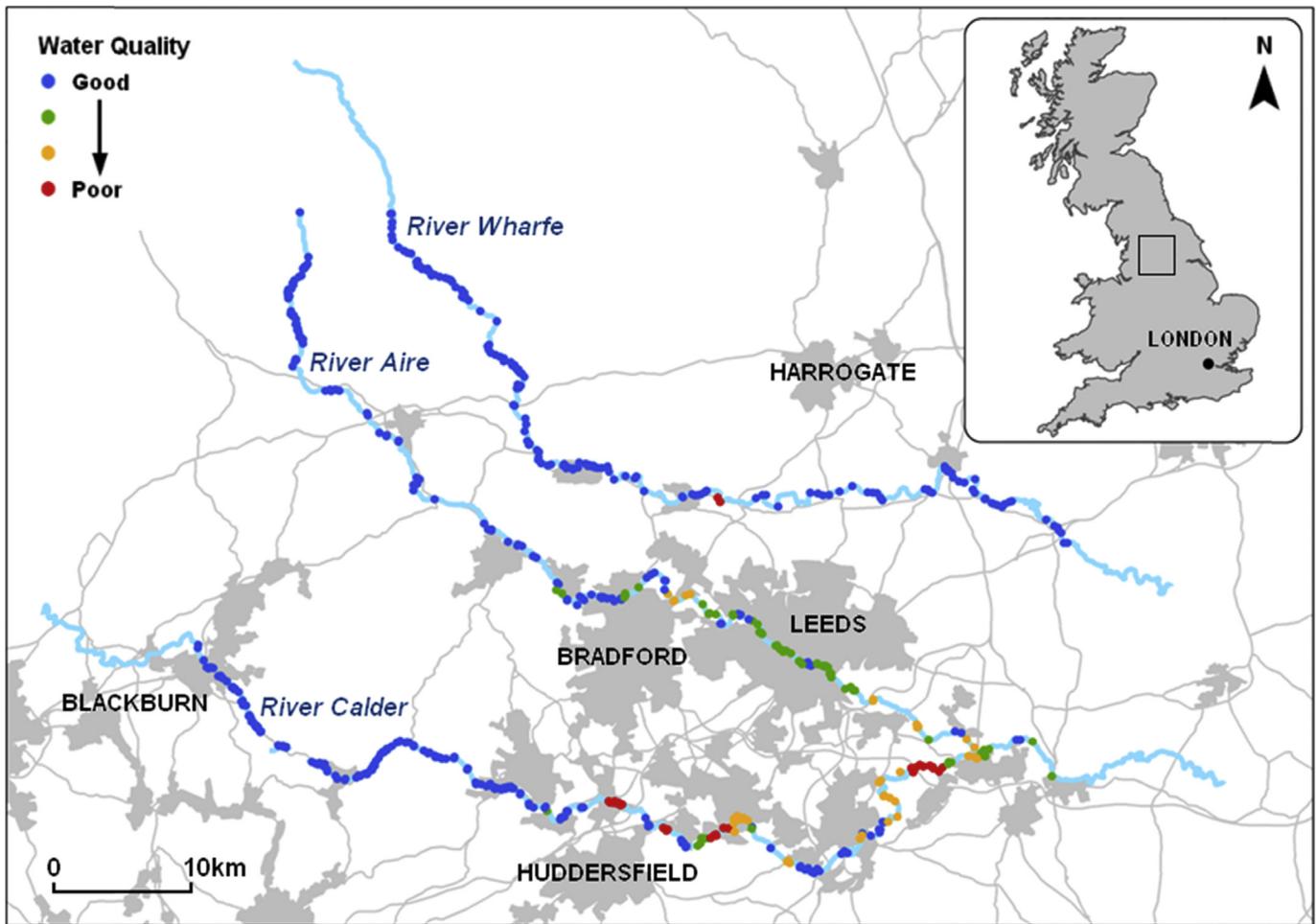


Fig. 4. Sampling area and the quality of river recreational access sites.

specific constants present a negative sign, demonstrating the common sense finding that, over the year, respondents typically choose to spend their time on activities other than outdoor recreation.

Taking the estimated parameters from Table 6 and following, for example, Small and Rosen (1981) or Hanemann (1999), it is possible to derive the welfare impact of changes in water quality across a range of alternative scenarios.

6. Integrated results

Using a highly spatially explicit GIS framework, we have introduced models to track the connections between climate change and induced changes in agricultural land use, river water quality and recreation demand. The current section presents an analysis at both the individual and catchment-wide scales for a simple climate change scenario of 1 °C warming. Here we illustrate how agricultural incomes and water quality are impacted and also how these changes in turn affect the value of river-based recreational activities within the case-study area.

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

The analyses developed above allow us to track how the direct effect of a 1 °C temperature rise, combined with the indirect impact of induced LUC and consequent alterations in diffuse pollution,

would result in a decrease in water quality throughout the case study area. The same analysis suggests that this will in turn reduce the number of good quality water recreation sites.

To estimate the consequences of the climate change scenario we apply the changes in water quality predicted from our ecological model (Table 3) to the parameter estimates given in our travel cost model (Table 6). Results indicate that the average disutility generated by the expected loss in recreational quality, expressed as compensation required per year (i.e. negative WTP; see discussions in Horowitz and McConnell, 2002; Bateman et al., 2009) is equal to £10.44 per person per annum in the case-study area. For decision purposes we then aggregate these individual estimates up to assess losses for the entire study area.

The Supplementary Materials to this paper assess the value of policy action to both mitigate these losses and attain pristine ecological and chemical status through full implementation of the EU Water Framework Directive (WFD) (European Commission, 2000).

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

In order to aggregate our recreation values we require distance calculations from all possible recreation sites to all households (not just those sampled in our survey) within the study area. We also require socioeconomic characteristics for all households to allow for variations in wage rate within the travel cost calculation. Our

Table 6
Estimated coefficients from travel cost model.

Variable	Description	Coeff.	(Robust SE)
Travel cost	Two ways Travel cost defined as: out of pocket cost (0.25£ * km) + value of time	-0.16	(0.018)****
Good water quality	1 = if site is good quality; 0 otherwise	-0.92	(0.234) **
Poor water quality	1 = if site is below good quality (mixed or poor quality); 0 otherwise	-1.07	(0.221) ***
Urban	1 = if the predominant land type around the site is urban; 0 otherwise	0.60	(0.14) ***
CSite	Alternative specific constant (ASC) river sites within the sampled area	-7.43	(0.226) ****
COthRiv	ASC for other river sites	-4.41	(0.129) ****
CCanal	ASC for canals	-3.81	(0.079) ****
Clake	ASC for lakes	-4.13	(0.093) ****
COthRe	ASC for other outdoor recreational sites	-2.80	(0.071) ****
LL	-488,258		

Note: **** = t-stat > 3, ***** = t-stat > 4, ***** = t-stat > 10.

GIS-based methodology allows us to perform these calculations with minimal simplification, working, in this instance, with UK Census Super Output Areas (SOAs).

The spatial distribution of changes in recreation values under the climate change scenario is illustrated in Fig. 5. Here the left hand panel presents the distribution of changes in per person values while the SOA aggregated values are given in the right hand panel. The site colors illustrate the estimated quality in the baseline situation. The climate change scenario induces a reduction in water quality along the length of the River Aire although impacts are most marked in the western area of the catchment, where quality declines from Pristine (sites shown as blue in Fig. 5) to Good quality. Aggregating across the entire case study area yields an estimate of the total loss in recreation value induced by climate change of approximately £26 million p.a. These losses, as one might expect, are concentrated in the western area of the catchment. Although population is relatively low here, this is the principal location where declines in water quality occur. The eastern area of the catchment does not suffer such an appreciable decline and, therefore, its aggregate values are relatively low.

Fig. 5 suggests that climate change effects present spatially

heterogeneous welfare impacts which should not be ignored if we wish to ensure efficient allocation of resources to mitigate impacts. However, policies, such as the WFD, tend to ignore the spatial distribution of benefits such that funds are spent in an untargeted and inefficient manner resulting in resources generating highly variable net benefits.

7. Conclusions

We present a unified series of models examining the direct, secondary and further effects of a given driver acting upon natural capital resources. The specific case study concerns the impact of climate change upon agricultural profitability, inducing shifts in land use and consequent impacts upon diffuse pollution, water and ecological quality and the recreational value of affected waterways. We model each of these relationships and calculate the resultant impact upon non-market recreational values. In so doing we demonstrate the use of spatial analytic techniques for incorporating biophysical data within environmental economic analyses.

In line with previous research (e.g. Fezzi et al., 2015; Fezzi and Bateman, 2015), results show that climate change is likely to

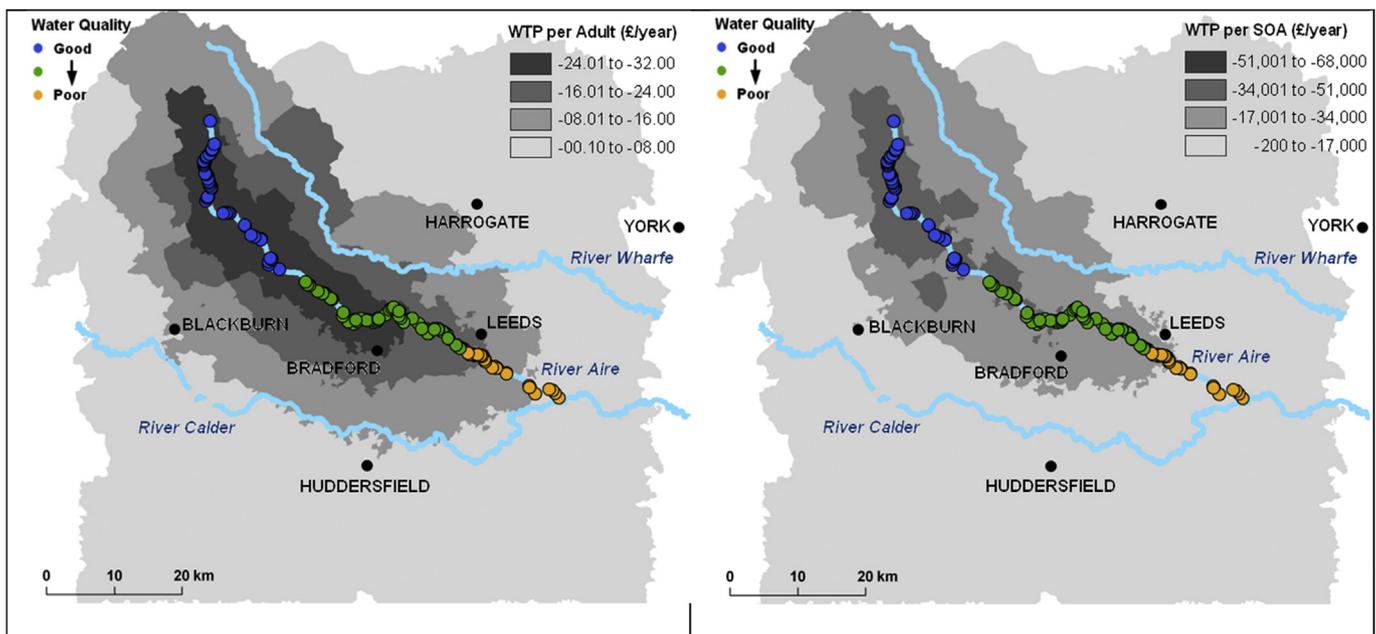


Fig. 5. The spatial distribution of per person (left hand panel) and SOA aggregate (right hand panel) changes in recreation value under the climate change scenario. Notes: Colors represent predicted site qualities in the baseline scenario where: blue = Pristine; green = Good; yellow = Mixed. Under the climate change scenario, water quality at all currently Pristine sites declines to Good quality. Water quality is as follows: blue = Pristine; green = Good; yellow = Mixed; red = Poor. Water quality definitions given in Table 4. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

generate spatially variable impacts upon both land use and resulting farm incomes. In some areas, it will generate income gains while other areas will experience losses. This pattern directly reflects the diverse and heterogeneous nature of UK agriculture. While acknowledging that the direct market impacts of climate change on UK agriculture may generally be positive, a central contribution of this research has been to demonstrate that focusing solely on these direct impacts paints a highly incomplete picture of the net impact of climate change. In our case study of the UK's River Aire, climate change both directly and via induced agricultural LUC yields a general decline in ecological quality which in turn reduces recreational values substantially.

The complexity of human–environment relationships means that examining only direct impacts may well be insufficient in order to guide efficient policy making. Research needs to extend analyses to embrace all major consequences of change. Depending upon the case study in question, this may require substantial integration of data, modeling and analytical techniques as well as a willingness to combine natural science, social science and economic perspective. No single discipline is sufficient to tackle the challenge of integrated environmental–economic decision making.

The modular nature of our methodology means that it can be readily extended. The agricultural, hydrological and recreation models presented here can easily be updated as improved versions become available. Moreover, the approach is highly suitable for extension to incorporate additional elements describing impacts on, say, greenhouse gases and biodiversity, or natural hazard and flood risk while expansion to cover alternative or wider geographic areas is straightforward (Bateman et al., 2013). An interesting and timely extension would be to consider how recent policy changes (e.g. WFD or Common Agricultural Policy) interact with climate change to drive on-farm decision making and the subsequent impacts. While complex, we believe that such methodologies are vital to addressing the complexities of the real world and bring them within the remit of economic analysis.

Finally, some caution is needed in interpreting our results. We used a simplified climate scenario for illustrative purposes. Future research should incorporate more sophisticated climate models and also account for seasonal variation. Moreover, timescale is an important element of the relationship between land use change and water quality. Given the interactions with groundwater reservoirs, there is some lag between land use change and impacts on water quality (Hutchins et al., 2010a; Stalnacke et al., 2004) and, therefore, the direct effect of climate on water environments may arise considerably later or last longer. Another important area for further development is considering measures of uncertainty. Our approach, being a fully statistical and econometric approach, can be used to produce confidence intervals for all the estimates and explore how uncertainty propagates throughout the models via Monte Carlo simulation. While this issue has not been explored in the present analysis, it is certainly a promising avenue for further research.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at <http://dx.doi.org/10.1016/j.jenvman.2016.06.020>.

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