Title: Bringing ecosystem services into economic decision making: Land use in the UK

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Abstract: Landscapes generate a wide range of valuable ecosystem services, yet land use decisions often ignore the value of these services. Using the example of the UK, we show the significance of land use change not only for agricultural production but also for emissions and sequestration of greenhouse gases, open-access recreational visits, urban green space and wild species diversity. We use spatially explicit models in conjunction with valuation methods to estimate comparable economic values for these services, taking account of climate change impacts. We show that, while decisions which focus solely upon agriculture reduce overall ecosystem service values, highly significant value increases can be obtained from targeted planning incorporating all potential services and their values, and that this approach also conserves wild species diversity.

One Sentence Summary: Valuation of ecosystem services within land-use planning creates significant gains relative to current, market-dominated, decision making.

Main Text:

The Millennium Ecosystem Assessment (*1*) provided important evidence of the ongoing global degradation of ecosystem services and highlighted the need to incorporate their value into the economic analyses which underpin real-world decision-making. Previous studies have shown that the overall values of unconverted natural habitats can exceed the private benefits following conversion (*2, 3*), that knowledge of landscape heterogeneity and ecological processes can support cost effective land planning (*4-7*), that trade-offs in land-use decisions affect values from ecosystem services and biodiversity at local level (*8, 9*), and that current land use is vulnerable to the impacts of global change (*10, 11*). In the UK National Ecosystem Assessment (NEA) (*12*), a comprehensive assessment of the UK's ecosystems was linked to a systematic, environmental and economic analysis of the benefits they generate. Here we show how taking account of multiple objectives in a changing environment (including, but not restricted to, climate change) fundamentally alters decisions regarding optimal land use. The NEA analyses are based upon highly detailed, spatially-referenced environmental data covering all of Great Britain. These data supported the design and parameterization of models of both the drivers and consequences of land use decisions, incorporating the complexity of the natural environment and its variation across space and time (*13*). Model outputs provide inputs to economic analyses which assess the value of both marketed and non-marketed goods (Table 1).

<Table 1 here>

The NEA specifically addressed the consequences of land use change driven by either just agricultural or a wider set of values, all within the context of ongoing climate change. To assess this, raw data on land use and its determinants were drawn from multiple sources to compile a 40 year dataset, spatially disaggregated at a resolution of 2km grid squares (400ha) or finer across all of Great Britain, forming more than ½ million sets of spatially referenced, time specific, land use records. Data on the determinants of that land use were assembled from multiple sources and included the physical environment (both spatially variable factors such as soil characteristics, slope, etc. and spatio-temporal climate variables such as growing season temperature, precipitation, etc.); policy (both agricultural and relevant environmental measures including subsidies, taxes, activity constraints, etc.); market forces (prices, costs, etc.); and technology (reflected as changes in costs).

Land use change

Land use in the UK is dominated by agriculture which accounts for some 18.3 million hectares or 74.8% of the total surface area (*14*), including not only cropland but also the majority of grassland, mountain, moor and heathland habitats. Agricultural land use was analyzed using integrated environmental-economic models developed to capture spatial and temporal variation in determinants (*15*). These models start from the premise that farmers seek to arrange land use so as to maximize long run profit, subject to the physical-environmental, policy and price conditions they face in a given location and time (*13*). Even within the relatively small area of Great Britain, variation in environmental conditions is sufficient to yield very substantial differences in agricultural productivity and hence land use. These differences are captured by the model along with the variation due to other drivers; the models being verified using rigorous out-of-sample, actual versus predicted, testing (*13*).

The focus of the analysis concerned the consequences of alternative land use futures up till 2060. To assess this, information was needed regarding how drivers of land use change might alter over that period. Some physical environment factors can be treated as fixed (e.g. soil type) but others, most notably climate change, vary temporally and spatially. For these, modeled outputs of variables such as growing season temperatures and precipitation (*16*) were included in our land use models. Certain market drivers were kept constant due to extreme uncertainties (e.g. food prices may well rise due to increased demand from higher population and other pressures; but this may be mitigated by technological advance and behavioral change). Policy-induced changes, such as the consequences of stronger or weaker environmental regulation on both agricultural and other land, were addressed through an expert-based, deliberative process consistent with (*1*). This process generated six plausible future scenarios, each described in terms of changes in regulations, these being either generally applied or spatially focused (Table 2). A rule-based approach was used to generate probabilities for each land cover transition in each cell under each scenario (e.g. transfers of land out of intensive agriculture to support the enhancement of areas of conservation importance, as per (*17, 18*)). Resultant scenarios are summarized in Table 2 and discussed in detail in (*13*).

<Table 2 here>

Response of market-priced goods to land use change

An initial analysis demonstrates the outcome of conventional land-use decision making, which emphasizes market values (e.g. agricultural produce) and ignores non-market ecosystem services. Figure 1 provides maps of the change in the market value of agricultural output from the present day (2010 baseline) to 2060 under alternative climate change and policy scenarios (ignoring any effects from inflation). In Fig. 1A, climate change follows a low greenhouse gas emission path (taken from (*16*)), therefore having relatively little impact on farming during this period, but relatively stronger environmental regulations are imposed (the NW scenario from Table 2), restricting high intensity farming in many areas which results in declines in market agricultural values across much of the country. These relatively strong environmental regulations are maintained in Fig. 1B but climate change now follows a high emissions scenario. While climate change is expected to have mixed consequences for agriculture at a global scale (*18*, *19*), comparison of Figs. 1A and 1B shows that farming in the UK will largely benefit from warmer temperatures. Figure 1C maintains that high emission assumption but now weakens environmental regulations (the WM scenario). This allows land use changes such as the conversion of some currently protected, conservation areas into higher intensity farming, resulting in substantial further increases in agricultural production and corresponding market values.

<Fig. 1 here>

Figure 1 shows that, irrespective of climate change projections, if land use decisions are based on market priced goods alone then a reduction in environmental regulations must always appear justified. Land use change, however, alters not only market-priced agricultural outputs but many other important (but typically non-market) ecosystem services as well.

Response of non-market ecosystem services to land use change

The analysis was extended to include the consequences of land use change for greenhouse gas balance (GHG), open-access recreation, urban greenspace, and wild species diversity (each modeled as per Table 1 and (*13*)). Economic values were estimated for each of these additional impacts (*ibid.*) with the exception of wild species diversity which is difficult to measure accurately using standard economic tools (*15*) and was accordingly assessed using a diversity index (*13*, *20*).

Land use change was then modeled for all scenarios, embracing a variety of combinations of environmental regulation and climate change, with the consequences for all market and non-market ecosystem services (including agricultural outputs) and their value or indices being assessed. Figure 2 presents changes in value from the 2010 baseline under either the weaker environmental regulations of the WM scenario (upper row of Fig. 2) or the stronger regulations of the NW scenario (lower row); with high emission climate change projections being assumed in both cases. Considering agricultural values alone, results are (as per Fig. 1B and 1C) that the weaker environmental regulations of the WM scenario yield higher market values. However, the non-market impacts of land use change illustrated in the rest of Fig. 2 show that, across much of the country, strong environmental policies yield gains in the value of ecosystem services resulting from reduced GHG emissions and enhanced recreation and urban greenspace as well as improvements in species diversity. Temporarily setting aside the non-monetary wild bird diversity index and summing across all other values shows that weaker (stronger) environmental regulations lead to net losses (gains) nationally; a result which reverses the restricted, market value assessment of Fig. 1. It is clear that considering market prices alone can drive decisions for land use that would deprive society of many other benefits from the environment and risk leaving the UK worse rather than better off.

<Fig. 2 here>

Benefits of spatially targeted land use planning

While the two alternative futures shown in Fig. 2 illustrate the importance of bringing ecosystem services into decision making rather than simply relying upon market values, these extremes ignore the potential gains from working with the spatial and temporal heterogeneity of the natural environment and underpinning biophysical processes. This variation makes it unlikely that any single policy will be optimal everywhere (e.g. in Fig. 2 the generally superior NW policy still yields higher GHG emissions in north western Britain than the generally inferior WM scenario), suggesting instead that a move towards a spatially differentiated, targeted approach to decision making will almost inevitably be better.

In order to examine the benefits of spatially explicit decision-making, the outcomes of each scenario were evaluated in each 2km grid square across Great Britain and the scenario which maximized a given objective in that cell was identified (Fig. 3). Results showed that, while a conventional, market dominated, approach to decision making chooses options to maximize agricultural values (Fig. 3A), these policies will reduce overall values (including those from other ecosystem services) from the landscape in many parts of the country (Fig. 3B); notably in upland areas (where agricultural intensification results in substantial net emissions of GHG) and around major cities (where losses of greenbelt land lower recreation values). In comparison, an approach which considers all of those ecosystem services for which robust economic values can be estimated (Fig. 3C) yields net benefits in almost all areas, with the largest gains being in areas of high population (Fig. 3D).

<Fig. 3 here>

To provide an idea of the scale of potential gains, consider the fact that our measure of agricultural profitability (technically, farm gross margins (21)) suggests returns to farming (including subsidies) ranging from £400/ha to in excess of £1000/ha depending on location (see (22)). Our analyses suggest that a targeted approach to land use planning which recognizes both market goods and non-market ecosystem services would increase the net value of land to society by 20% on average, with considerably higher increases arising in certain locations.

Decisions based on all ecosystem services for which robust economic values can be derived (Fig. 3C and 3D) are clearly better than those based only on a conventional pursuit of market priced goods (Fig. 3A and 3B). However, this analysis omits impacts that cannot be reliably monetized; here the effects on wild bird species diversity. We now incorporate our measures of change in wild bird species diversity through the application of a simple constraint requiring that, in each area, any policy which resulted in a reduction in the species diversity index was ruled out for that area (Fig. 3E and 3F). The similarity to Figs. 3C and 3D shows that, when applied in a targeted manner, this constraint has relatively little impact upon which scenario is best; i.e. the ‘opportunity cost' (*17*) of imposing a species conservation constraint is relatively minor. Nevertheless, comparison of Figs. 3C and 3E shows that, in certain areas, the sustainability constraint causes a shift from Scenario NW, which focuses on the enhancement of greenbelt areas for recreation, to Scenario GPL, which focuses on extension of existing areas of conservation value.

National-scale implications

Table 3 presents monetary sums from the analyses of Fig. 3. Even if we only consider agricultural market values, then a targeted approach to maximizing these values (first column of results) can yield a small gain in total values relative to the present situation (a result which is not feasible using single policies applied over all of the country, highlighting the inefficiency of current one-size-fits-all policies, even when they are only assessed in market value terms). However, a targeted approach to optimizing both market and non-market values yields a very major increase in gains (second column of results). Furthermore, placing a targeted biodiversity constraint on the latter approach only marginally reduces these gains (final column), suggesting both that such constraints are a highly effective and efficient solution to conserving wild species diversity, and that land use policies which increase GHG storage and recreation values typically correlate with improvements in such diversity.

<Table 3 here>

Table 3 shows recreation values arising from these changes exceeding those from agriculture. This striking difference does not imply that the total value of recreation is greater than that of food. It comes about because economic analyses such as this evaluate alternatives by focusing not on total values but on the changes in value that these alternatives generate. In a highly developed country such as the UK where food is plentiful and cheap but opportunities for recreational use of the natural environment are somewhat limited it is unsurprising that converting some comparatively small amount of land out of agriculture and into open access recreation yields a relatively modest loss in farm produce value while at the same time generating a much bigger value from increased recreation. This positive disparity will be greater if (as in this analysis) such conversions are spatially targeted so as to maximize net benefits (here by ensuring such land use conversions occur near to urban centers where resulting recreational gains can be huge). However, as progressively more land is converted to recreation so the number of additional visits generated will fall whereas the agricultural loss of each conversion steadily mounts (explaining why only a limited area, typically near to cities, is converted to recreation). Obviously such results would vary substantially if analyses were conducted in very different contexts such as less developed countries where the value of changes in food may be much higher relative to those for recreation.

From potential to practice

Our analysis shows that land use decisions based on market prices alone can reduce the overall value of the sum of agricultural and monetizable ecosystem services at the national scale. Although the economic values provided in Table 3 are subject to certain assumptions (*13*), further work to elaborate significant underpinning processes such as the effects of ecological, biodiversity and other global change factors (*23*-*25*), and to better reflect links between economic valuation of ecosystem services and decisions, seem unlikely to alter this general conclusion. Indeed, if other services such as water resources were added to the analysis, current national estimates of pollution costs (*26*) imply that the differences would be accentuated.

While potential improvements in land use planning would generate social gains sufficient to more than compensate for any associated losses, a new direction for land use decision making does not come without implementation challenges.

A first challenge concerns the mechanics of securing the participation of land managers in delivering land use changes that are unlikely to be privately beneficial. In the UK context, the obvious mechanism through which that goal could be achieved is reform of the EU's Common Agricultural Policy (CAP). Currently, CAP payments to UK farmers are in excess of £3billion per annum (*27*) compared to a total value of UK agriculture of only £5billion per annum (*28*) with the vast majority of those payments (70%) made without consideration of environmental performance. Recasting the CAP as a Payment for Ecosystem Services (PES) mechanism such that farmers are rewarded for the delivery of a broad spectrum of ecosystem services would provide policy makers with a very powerful tool through which to secure beneficial land use change.

A second challenge arises from the need, clearly demonstrated in this research, for that mechanism to allow for spatial targeting, a prescription that stands in sharp contrast to the spatial insensitivity of current CAP payment allocation. Spatial targeting, however, necessarily increases pressures upon decision making and administrative institutions. The key challenge, therefore, is to realize the gains from spatial targeting without overly inflating the costs of policy implementation.

A final challenge concerns how to efficiently target payments when the costs of delivering ecosystem services differ across land managers but are unknown to the funding authority. To that end, recent developments in the design of PES mechanisms suggest that competitive contracting may deliver considerable efficiency gains (*29*).

Principles for future land use analysis and planning

Our results allow us to refine the following principles for future analyses and decision making: (i) The conventional focus upon market priced goods alone can result in decisions which lower overall values; (ii) All the major ecosystem services generated by a change in resource use need to be assessed; (iii) That assessment must recognize spatial and temporal variation in ecosystem services as well as synergistic impacts such as those arising between climate and land use change; (iv) Changes in ecosystem service flows should be valued wherever robust economic values are available; (v) Difficult-to-monetize impacts, such as those upon wild species, should be incorporated through the imposition of sustainability constraints which can then be satisfied in cost-effective ways; (vi) Spatial targeting of policies can generate major gains and, perhaps most importantly; (vii) A range of substantial benefits to society can be realized by bringing natural science and economic information together to inform environmental decision making. Taken together we hope that these principles and their demonstration through the case study illustrate the practical potential for national level, yet spatially sensitive, application of an approach to decision making which places ecosystem services on a level playing field with market priced goods and thereby contributes to the sustainable use of Earth's limited resources.

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Supplementary Materials:

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Fig. 1. Change from 2010 to 2060 in the market value of GB agricultural production under various climate and policy scenarios. (**A**) Under low emissions climate projections (from (*16*)) and strong environmental regulations (NW scenario) conserving environmentally important habitats and restricting farm intensification. (**B**) Under high emissions climate projections (*ibid.*) with the policy scenario as per (A). (**C**) Emissions as per (B) but with weak environmental regulations (WM scenario). All values are adjusted for inflation.

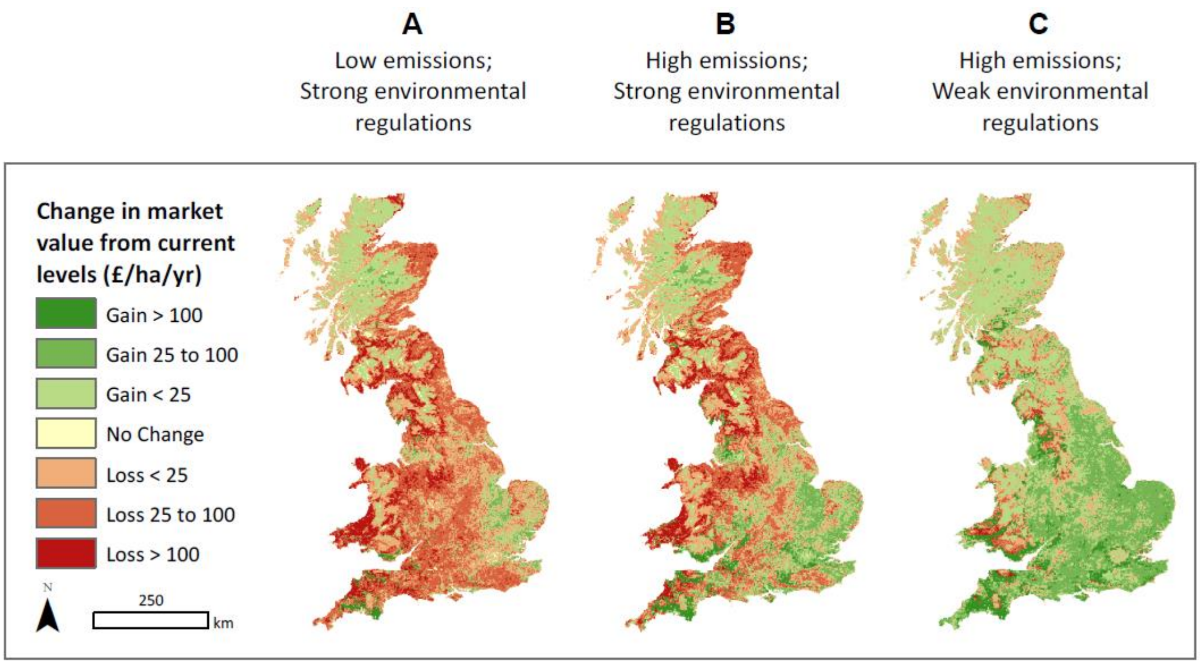

Fig. 2. Spatial distribution of the changes in market and non-market ecosystem service economic values and non-monetary wild species diversity assessments (measured as changes in Simpson's Diversity Index (*13*, *20*) induced by moving from the year 2010 baseline to the WM and NW scenarios for 2060 (all analyses assume high emission climate change projections from (*16*)). (*30*).

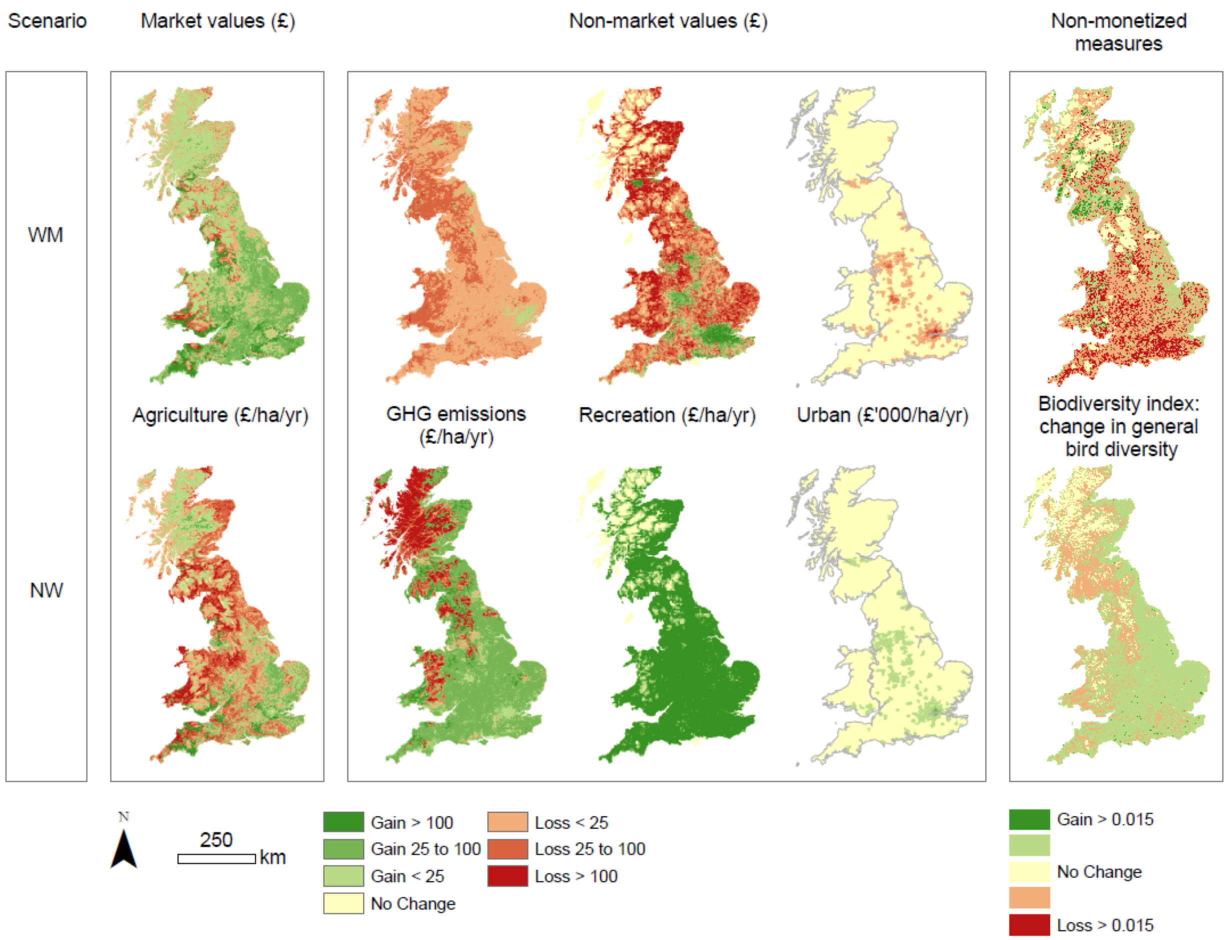

Fig. 3. Optimal scenarios (**A**, **C**, **E**) for each 2km grid square and corresponding changes in value from 2010 to 2060 (**B**, **D**, **F**) in Great Britain under three alternative targeted objectives: (i) Conventional approach maximizing market values only (A and B); (ii) Maximizing the value of all those ecosystem services that can be robustly monetized (C and D); (iii) Maximizing all ecosystem service values but with a constraint so that no scenario which gives a net loss of wild bird diversity is permitted in the area affected (E and F) (all analyses assume low emission, climate change from (*16*)). (*30*).

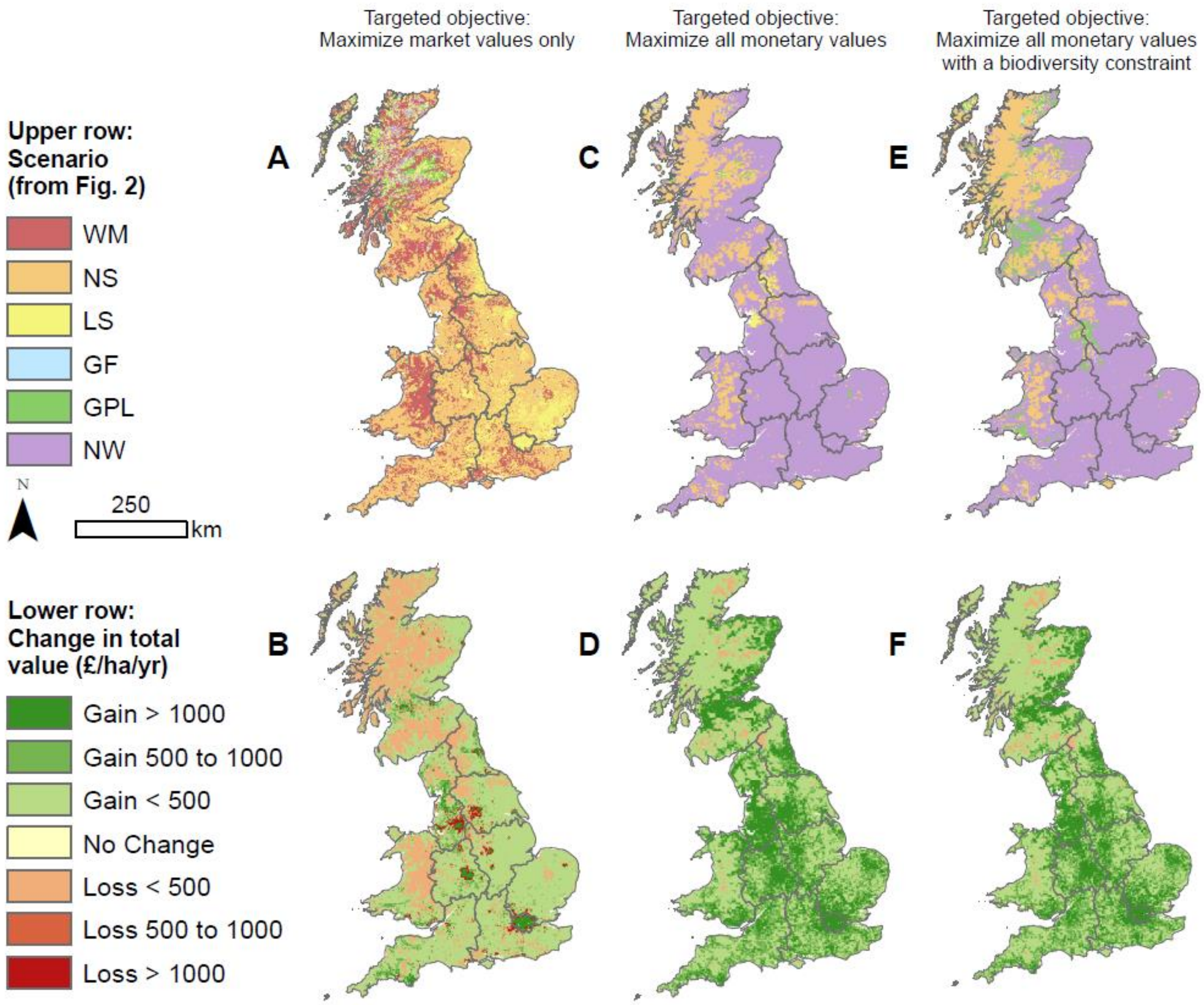

Table 1. Summary of the ecosystem service related goods considered in the analysis (metrics, data, modeling and valuation are fully documented in (*13*))

Ecosystem service related good	Metrics (in year specified)	Main data & sources	Model	Valuation
Agricultural production	Proportion and output of each land use in each 2km grid square.	Land use, soils & physical environment, climate, digital mapping, etc. (*31*-*33*)	Environmental-econometric regression analysis of land use decisions as a function of the local physical environment, prices, costs and policies, based on (*34*)	Market values (*35*)
Greenhouse gases	Net tonnes of CO_2, CH_4 and N_2O per 2km grid square.	Land use predictions, GHG responses (*36*-*38*)	Process models for CO_2, CH_4 and N_2O; conversion to tonnes of CO_2 equivalent based on insulation factors.	Official UK values per tonne CO_2e (*39*)
Recreation	Visitors per 2km grid square.	National survey of over 40,000 households, Census (*40*, *41*)	Regression model of visit count from outset to destination as a function of characteristics of both locations, population socio-economics, etc.	Meta analysis of 300 ecosystem specific valuation estimates
Urban greenspace amenity	Distance to greenspace from each 2km grid square.	Digital mapping Census (*32*, *41*)	Regression model linking distance from households to greenspace sites, their size and quality.	Meta analysis of prior literature examining changes in value with respect to distance

Wild bird species diversity	Wild bird diversity (*20*) per 2km grid square.	Breeding Bird Survey (*42*)	Regression model linking wild bird diversity to land use and location.	Not valued; analysis uses the opportunity cost of avoiding declines

Table 2. Summary of land use change scenarios (details in (*13*)).

Scenario	Environmental regulation and planning policy relative to current	Spatial focusing of changes
Go with the flow (GF)	**Similar:** Policy and regulatory regime as today. Existing patterns of countryside protection relaxed only where economic priorities dominate.	**Unfocussed:** Similar spatial constraints on land use change as today. No expansion of the protected area network.
Nature at work (NW)	**Stronger:** Policy and planning emphasize multi-functional landscapes and the need to maintain ecosystem function.	**Focused:** Greening of urban and peri-urban areas to enhance recreation values.
Green & pleasant land (GPL)	**Stronger:** Agri-environmental schemes strengthened with expansion of stewardship and conservation areas.	**Focused:** Increased extent of existing conservation areas. Creation of functional ecological networks where possible.
Local stewardship (LS)	**Stronger:** Agri-environmental schemes strengthened with expansion of stewardship and conservation areas.	**Unfocussed:** No strong spatial component to changes but protection of areas of national significance continues.
National security (NS)	**Weaker:** Emphasis on increasing UK agricultural production. Environmental regulation and policy is weakened.	**Unfocussed:** Some land use conversion into woodland occurs in areas of lower agricultural values
World markets (WM)	**Weaker:** Environmental regulation and policy is weakened unless they coincide with improved agricultural production.	**Focused:** Losses of greenbelt to urban development, resulting in loss of recreational values. Weaker protection of designated sites and habitats.

Table 3. Change in values across Great Britain from the present day (2010) to 2060 achieved by the targeting of policy options under three decision rules (£millions p.a.; real values in £2010; UKCIP low emission scenario throughout).

Decision component	Maximize market (agricultural) values only (Figs. 3A&B)	Maximize all monetary values (Figs. 3C&D)	Maximize all monetary values with biodiversity constraint (Figs. 3E&F)
Market agricultural value	971	-448	-455
Non-market GHG emissions	-109	1,517	1,510
Non-market recreation	2,550	13,854	12,685
Non-market urban greenspace	-2,520	4,683	4,352
All monetary values	892	19,606	18,092

Supplementary Materials for

Bringing ecosystem services into economic decision making: Land use in the UK

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Bringing ecosystem services into economic decision making: Land use in the UK

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S1: Agricultural land use and the value of farm outputs

S1.1: Overview

The analysis implements a spatially explicit, econometric model of agricultural land use which estimates the statistical relations between land use, livestock numbers and the main determinants of farmers' production decisions, including output and input prices, environmental characteristics and the policy environment. This estimation is underpinned by a theoretically consistent economic model (described in further detail in the next section). Land uses analyzed within the model include cereals, oilseed rape, root crops, temporary grassland, permanent grassland and rough grazing which, together, constitute more than 90% of farmland in Great Britain. The value of this production is assessed using activity specific gross margins, which are given by the difference between revenues and variable costs (i.e. costs which vary proportionally with the quantity produced, e.g. fertilizer costs, labor costs, etc.).

S1.2: Methods

To model agricultural land use decisions we use the structural econometric approach introduced by (*27*). This notes that farms use land in ways which maximize profits by taking into account: the amount of land available to them and its physical environmental characteristics; the costs of inputs needed to produce outputs; the price that those outputs will attain; and a variety of policy and other constraints. More formally, each farmer maximizes profits per unit of land by solving the following constrained optimization problem:

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

where $\pi^L(.)$ are profits per unit of land, **p** is the vector of the prices of the *m* outputs, **w** is the vector of the costs of the *n* inputs, **s** the vector of *h* land share allocations, *L* the total land available and **z** the vector of *k* fixed factors (which includes those physical and environmental characteristics of the land along with policy incentives and constraints, etc.). Adopting commonsense assumptions regarding the attitudes of farmers to prices and costs[*] we allow for the likely non-linearities of applying this approach to a complex real-world environment by specifying the profit function as a Normalized Quadratic (NQ). We indicate with w_n the numeraire good[†]; with $\mathbf{x} = (\mathbf{p}/w_n, \mathbf{w}/w_n)$ the vector of normalized input and output (netput) prices; with $\bar{\pi}^L = \pi^L/w_n$ the normalized profit per unit of land; with $\mathbf{z}^* = (\mathbf{z}, L)$ the vector of fixed factors including policy and environmental drivers; and the total land available is denoted *L*. The NQ profit function is defined as:

[*] Formally the profit function is required to be positively linearly homogenous and strictly convex in input and output prices, i.e. higher prices for an output/input always increase/decrease profits.

[†] The numeraire is the output or input good whose price is used to divide all the prices in the profit function; the resultant "normalized" profit function accounts for inflation and ensures the mathematical properties necessary for a well specified profit function (homogeneity and convexity).

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

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

This representation is very flexible and allows farm profits to be a fully quadratic function (including linear effects, quadratic effects and interactions) of prices, policy and the environmental characteristics of the land. Note that we include only *h*-1 land use shares, since one of these is simply given by one minus the sum of the other shares, and it is therefore redundant. Input and output intensities can be derived via Hotelling's Lemma (a result in microeconomics which relates the supply of a good to the profit made by the producer of that good; see, for example, (*36*). For instance, if x_i indicates the normalized price of cereals, the equation corresponding to cereal yield (y_i^L) can be derived as:

$$\frac{\partial \bar{\pi}^L}{\partial x_i} = y_i^L = \alpha_i + \sum_{j=1}^{m+n-1} \alpha_{ij} x_j + \sum_{j=1}^{h-1} \delta_{ij} s_j + \sum_{j=1}^{k+1} \phi_{ij} z_j^* \ , \qquad (3)$$

Therefore, the output of say, cereals, is a function of input and output prices, policy (e.g. set aside rate, land designation, etc.) and the environmental characteristics of the farm, including for example, slope, soil type and climate variables (more details are given in the next section). There is clearly some pattern of land use which maximizes profits on the farm. The profit maximizing set of land use shares are defined by fixed order conditions of (Eq. 1), as follows:

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

The Lagrange multiplier, λ, corresponds to the land shadow price, or marginal rent (defined as follows). It is assumed that land users will assess the physical characteristics of their land, consider potential land uses and choose those which maximize profits by applying costly resources (e.g. fertilizer) to their land up to the point where the added value of the next unit of resource is equal across all available land (in economic terms this is where marginal rents become equal across all land uses). As shown by (*27*), the system of equations (Eq. 4) can be solved by including the constraint that the sum of the shares is equal to one and deriving a linear system of *h* equations in *h* unknowns which can be solved to obtain the optimal land allocation as a function of **p**, **w**, **z** and *L* as:

$$s_i = \theta_i + \sum_{j=1}^{m+n-1} \theta_{ji} x_j + \sum_{j=1}^{k+1} \eta_{ji} z_j^* \text{ , for } i = 1,...,h\text{-}1, \qquad (5)$$

with θ and η being the vectors of the parameters to be estimated, which are non-linear combinations of the parameters in the NQ profit function (Eq. 2).

As micro-data on land use are often characterized by corner solutions (not all farms cultivate all possible crops) imposing normal disturbances and implementing Maximum Likelihood (ML) or Ordinary Least Squares (OLS) estimation techniques will yield inconsistent estimates of the land use share and input and output intensity equations. We address this issue by specifying a Tobit system of equations, in which the latent shares s_i* (i.e. shares not constrained to be higher than zero) are defined as in (Eq. 5) plus additive Gaussian residual terms. Observed shares are specified as: $s_i = 0$ if $s_i* \leq 0$, $s_i = 1$ if $s_i* \geq 1$ and $s_i = s_i*$ otherwise. This transformation can be interpreted by recalling that the fixed order conditions of the profit maximization problem are equal to the land shadow prices. For this reason, censoring from below (above) implies that the corresponding land use shadow price is lower (higher) than those of alternative uses. 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. To address this issue, we implement the Quasi Maximum Likelihood (QML) algorithm proposed by (*27*). This QML estimator is consistent, allows the estimation of cross-equation correlations and the imposition of cross-equation restrictions.

As mentioned in the introduction, the value of agricultural production is estimated using Farm Gross Margin (FGM). FGM is the measure commonly applied in agricultural economic studies (e.g. (*37*)), this being simply the difference between farm revenues from production and their associated variable costs (i.e. costs which vary proportionally to the quantity produced, e.g. fertilizer costs, labor cost, etc.).

FGM are taken from (*28*) as follows: “cereals” = £290/ha, “root crops” = £2425/ha, “oilseed rape” = £310/ha, “dairy” = £576/head, “beef” = £69/head, “sheep” = £9.3/head, “other land” assumed to have the same FGM/ha of cereals.

S1.3: Data and Materials

The data used to estimate the model are collected on a 2km grid square (400ha) basis and cover the entirety of England, Scotland and Wales (Great Britain; GB) for seventeen, unevenly spaced, years between 1969 and 2006 yielding a dataset of over half a million records. Agricultural land use data include: 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. Livestock count data is also given for dairy cattle, beef cattle and sheep. These data are extracted from the June Agricultural Census (JAC), obtained from the on-line EDINA data source (*24*). As described on the EDINA website, grid-square land use estimates can sometimes overestimate or underestimate the amount of agricultural land within an area, since their collection is based on the location of the main farm house. We correct this feature by rescaling the sum of the different agricultural land use areas assigned to each grid square to match with the total agricultural land derived from the Agricultural Land Classification (ALC) system published by Defra and the Welsh Assembly (data available at: http://www.naturalengland.org.uk/). More details are given in (*27*).

An important attribute of the modeling approach is that it considers all of the drivers determining change in agricultural land use: physical-environmental characteristics; policy determinants and constraints; and economic drivers such as market forces in the shape of product prices and input costs. These were collected in a manner which captured both spatial and temporal variation in all determinants. For example, potential *environmental drivers* include seasonal climate related variables such as temperature and accumulated rainfall. These were initially obtained as 5km grid square values from the UK Met Office web site (www.metoffice.gov.uk) which we then interpolated to our 2km square grid. Other environmental and topographic variables which may influence farmers' decisions include soil depth to rock, volume of stones, and 5 dummy variables representing soil texture (fine, medium fine, medium, coarse, peaty), derived from the 1km raster library of the European Soil Database (*26*), which we aggregate to a 2km square level. Other environmental variables such as altitude and land slope were derived via GIS analysis of the Ordnance Survey Digital Terrain Model (*25*). A similar GIS based approach was adopted for several of the policy drivers with the digital boundaries of various designated areas (such as National Parks, Nitrate Sensitive Areas, Environmentally Sensitive Areas, etc.) being derived from the Defra Magic website (http://magic.defra.gov.uk/). Although all of the above data were spatially explicit this was not the case for the price and cost data. No reliable information on spatial variation in these variables was available although of course the long time series encompassed by our dataset allowed us to control for these factors as annual fixed effects.

S1.4: Predicting performance

After estimation we evaluate the fit and predictive ability of our structural model by implementing out-of-sample forecasting tests. Table S1 reports the Mean Absolute Error (MAE) statistics for the land use share and livestock number equations calculated as the mean absolute value of the difference between the predictions and the actual JAC data. Our approach captures a significant proportion of the variability of the land use variables. For example, the MAE for cereals is approximately one-third of the standard deviation.

Due to the availability of data for the most recent period, this comparison is conducted for the whole of England and Wales in 2004 rather than for all of Great Britain. Since only 5% of the 2004 data are used to estimate the model, this consists of mainly out-of-sample forecasting. Therefore it is an appropriate yardstick to compare models performances avoiding the risk of preferring an over-fitting specification.

An additional test of the predictive ability of our model was undertaken to demonstrate its ability to forecast the spatial pattern of land use. This is provided by first omitting data for 2004 from our analysis after which the model is re-estimated and used to predict values for the omitted period. These predictions are then compared to the actual values observed in the omitted period. Figure S1 maps actual and predicted shares of both cereals and rough grazing for 2004. While there are some minor differences (e.g. for the cereals maps there is evidence of minor over-prediction in the Midlands and under-prediction in Eastern Scotland), the overall pattern is one of very strong spatial predictive performance.

S2: Greenhouse gas (GHG) impacts of land use change: modeling and valuation

S2.1: Overview: Framework of analysis and system boundaries

This analysis estimates the annual greenhouse gas (GHG) fluxes arising from changes in rural land across Great Britain (fluxes arising from urban land use change were excluded from analysis on the assumption that they would be relatively minor). These land use changes were driven by the various scenarios considered under the wider UK-NEA analysis. Each scenario entailed changes within each individual 2km grid square in terms of its area of enclosed farmland, woodland, semi-natural grassland and mountain moorland and heath land (assumed to be analogous to a rough grazing land use) for the baseline year (2010) and 2020, 2040 and 2060. Furthermore, within the agricultural area of each grid square, scenario specific shares of each major arable crop and livestock intensities were estimated (see S1) by applying the NEA structural econometric model of agricultural land use (described in the main paper, in S1 and in (*27*)). Specifically, for the agricultural area of each grid square, the econometric model estimated percentage shares of: cereals, oilseed rape, roots crops, temporary grassland, permanent grassland, rough grazing, coniferous and broadleaf woodland. Livestock numbers (sheep, beef and dairy cattle) were estimated for the grasslands and rough grazing land uses. Further sensitivity was induced by defining two variants of each scenario corresponding to the UKCIP low and high GHG emission projections (*14*).

The resulting scenario/climate induced land use for the baseline and future years were then used to calculate (for each grid square):

(i) the annual changes in potential equilibrium carbon stocks in above- and below-ground biomass across Great Britain between 2010 and 2060 due to changes in land use and;

(ii) the changes in annual emissions of GHGs associated with farm management for each agricultural land use.

This information was then coupled with per tonne carbon equivalent values to yield a spatial assessment of the future costs of emissions from rural land use change across Great Britain. Further detail on all of the above analyses is provided through the following sub-sections.

S2.2: Changes in carbon stocks

The carbon stocks included in this analysis refer to that stored as soil organic carbon (SOC; these being the largest terrestrial carbon stocks in the Great Britain) and in the above- and below-ground biomass (BIOC; the vegetative stock). While various studies have estimated these stocks across the UK under different land uses (*38, 39*), none have done so at the level of spatial disaggregation used in this analysis or considered the impacts of climate change induced land use change.

S2.2.1: Soil organic carbon stocks

Although the focus of our analysis is upon scenario induced changes rather than absolute totals, nevertheless attempts were made to refine a representative baseline level of soil carbon stocks. Soil types were defined as either organic (peat) or non-organic (non-peat) based on the European Soil Database (*26*), as peat soils have the potential to store considerably greater amounts of carbon than non-organic soils and can release large quantities of carbon if change in land use occurs.

National level estimates of average SOC for non-organic soils were used to allow for variation in climatic, hydrological and other characteristics. Specifically these were: 132.6 tC/ha for England, 212.2 tC/ha for Northern Ireland, 187.4 tC/ha for Scotland and 142.3 tC/ha for Wales (*40, 41*). It was assumed that UK organic soils under rough grazing had an average SOC density of 1200 tC/ha (*41, 42*). For each soil type, SOC levels are influenced by land use through its impact on processes such as soil disturbance and nutrient cycling. This was accounted for by applying unique adjustment factors for each land use/soil type combination. Non-organic soils under arable land uses (oilseed rape, cereals, roots crops and other agriculture land uses) were assumed to have 84% of the SOC they would attain under improved grassland (temporary and permanent grassland) while soils under rough grazing (semi natural grassland) were defined as having 33% more SOC than improved grasslands (*40*). In comparison, organic (peat) soils under temporary grass, permanent grass and woodland were assumed to have an average SOC of 580tC, while organic soils under arable land uses were assumed to have long term equilibrium SOC equal to the average non-organic soil SOC of the region within which the soils are located (*40*). All SOC estimates were based on soil depth of 1m.

To check the validity of these model assumptions, our estimate of SOC for the scenario baseline year (2010) was compared to the most comprehensive estimate of UK SOC (*38*). While Bradley et al (*38*) estimated the SOC stock in Great Britain as 4,267 million tC our estimate resulted in 4,357 million tC. The largest discrepancy (6.4%) occurred in Scotland, and is likely to be due to the extensive organic soils found in the region, the difficulty of accurately estimating SOC in organic soils due to issues surrounding soil depths, and technical factors associated with the measurement of SOC in organic soils (*43*).

S2.2.2: Biomass carbon stocks

Average per hectare biomass carbon (BIOC) stocks for baseline woodland extents were taken as 36.84 tC/ha (*39*). Estimates of the BIOC for each agricultural land use were taken from UK based assessments (*39, 40*). These estimates are based on an assumption that annual BIOC on farmland represents a permanent stock while a particular agricultural land use persists. That is, the biomass lost through harvest in one year is assumed to be replaced by new growth in the subsequent year, implying that net accumulation or loss of BIOC only occurs when land use changes. For the baseline year it was estimated that the total agricultural BIOC for Great Britain was 21.46 million tC, this being in broad agreement with other estimates of agricultural biomass carbon stocks (e.g. 22.8 ±5.1 million tC for Great Britain (*41*)). Details of the per hectare SOC and BIOC stock estimates for different land uses and soil types used in the analysis are given in Table S2.

S2.3: Converting from carbon stocks to the annual flow of GHG emissions

The annual net flow of emissions of GHG from land use change is defined as comprising two components:

(i) Annual SOC fluxes due to land use change; for example, the conversion of arable land to permanent pasture will result in the accumulation of SOC, while a switch from rough grazing to permanent grassland is likely to reduce SOC;

(ii) Annual GHG fluxes from the changes in vegetative biomass associated with land use changes.

A lack of data regarding land use prior to the baseline year of 2010 led to the adoption of a conservative, lower bound assumption that non-organic soils were assumed to have a zero annual SOC flow value during the baseline year. For subsequent years mean equilibrium SOC for non-organic soils was assumed to change from the level associated with the previous land use to that associated with the new land use (Table S2). SOC accumulations in such soils were assumed to occur linearly (*43, 44*) over a 100 year period, while SOC emissions were again assumed to be linear although occurring over a 50 year period (*44*). For example, a hectare of non-organic soil in England converted from cereals to permanent grassland was assumed to accumulate 22 tonnes of SOC before it reached a new equilibrium after 100 years, i.e., 0.22tC/ha/yr over the period.

As an aside, in other work we have tested non-linear forms such as logarithmic growth paths (*42*) the linearity assumption is both commonplace (*45*) and, relative to the former, would prove conservative should carbon values subsequently be used within a discounting analysis which places lower weight on delayed carbon storage.

For organic soils, annual flows of SOC were estimated for all years including the baseline as average SOC flow estimates in organic soils are primarily driven by the present agricultural land use rather than changes in land use. For example, annual SOC sequestration rates in organic soils under rough grazing vary from 0.18 tC/ha/yr (*46*) to 0.36 - 0.73 tC/ha/yr (*47*). The average of six estimates found in the literature (0.3 tC/ha/yr) was used and it was further assumed that SOC in organic soils under rough grazing would accumulate this quantity of carbon each year. It was assumed that 1.22 tC/ha/yr and 0.61 tC/ha/yr of SOC would be released from organic soils under arable/horticultural land use and improved grassland, respectively (*48*). The potential for total exhaustion of the organic matter in organic soils is not considered, i.e. such soils will not reach average SOC equilibrium within the time frame (50 years) considered here. We assumed a constant annual release of carbon from organic soils under arable, woodland, horticultural and improved grassland land uses.

Emissions and accumulations of BIOC were based on the change in vegetative biomass arising from a switch in agricultural land use. The change in equilibrium BIOC estimated for each 2km grid was divided by the time period over which the change occurred to provide an estimate of annual vegetative GHG flux due to land use change. Where the modeled annual BIOC was lower than in the preceding year (within a given 2km grid) then it was considered a net emission of GHGs.

Estimates of the annual accumulations of BIOC in the baseline woodland extents were taken from (*44*), based on the assumption that Great Britain woodland planted before 1921 is in carbon balance. In the absence of spatially explicit data regarding the planting date of Great Britain woodland, it was assumed that the post-1921 (carbon accumulating) woodland is distributed evenly across the total Great Britain woodland extent. Consequently, a single (per hectare) estimate of carbon accumulation in woodland was applied to the baseline data. Estimated accumulation of BIOC in woodland planted after then baseline year were taken as 4.61 tC/ha/yr for broadleaf woodland (based on the average of four UK estimates (*39, 49-51*)) and 5.32 tC/ha/yr for coniferous woodland (based on three UK based estimates (*49-51*)). All flows of SOC and BIOC were converted from metric tonnes of carbon to metric tonnes of CO_{2e}.

S2.4: GHG emissions from agricultural activities

Three major agricultural sources of annual, per hectare GHG emissions were considered: (i) energy use for typical farming practices such as tillage, sowing, spraying, harvesting as well as the production, storage and transportation of fertilizers and pesticides (estimates taken from (*52*)), (ii) emissions of N_2O and methane from livestock, i.e., beef cattle, dairy cows and sheep, through the production of manure and enteric fermentation, and (iii) direct emissions of N_2O emissions from the application of artificial fertilizers.

It was assumed that all arable and horticultural crops require annual conventional tillage, sowing and harvesting. Cereals were assumed to receive two fertilizer and two pesticide applications annually, while three fertilizer and five pesticide applications were assumed for oilseed rape and one fertilizer and four pesticide applications for root crops. Permanent and temporary grasslands were assumed to receive a single fertilizer application and a single harvest (including bailing). Temporary grassland was assumed to be conventionally tilled and sown once every four years. Emissions from farming activities associated with the "other agriculture" land use were taken as the average of the other six land uses.

Per head estimates of livestock GHG emissions from enteric fermentation were based on UK species specific emission factors given by (*53*). Estimates for GHG emissions from livestock manure were derived from Beaton (*52*) and Freibauer (*54*). Adjusted emissions estimates were applied to direct deposition of manure on grasslands during grazing periods, while emissions from manure spreading (as fertilizer) from housed livestock were estimated from the average grazing days for different livestock types (*55*). Per head estimates of manure production were converted to area (grid cell) estimates based on the modeled livestock density across Great Britain (as detailed in the main paper and S1). It was further assumed that manure used as fertilizer was utilized within the grid in which it was produced, reducing the requirements for inorganic fertilizers within that grid. Data on per hectare nitrogen requirements for each land use. Specifically, information from (*52*) was used to calculate the inorganic fertilizer input requirement for each 2km grid and this was in turn converted to direct emissions of GHG based on estimated N_2O emissions from the application of inorganic fertilizers (*56*).

Aggregate per hectare GHG emission intensity parameters from agricultural activities and inorganic fertilizer are given in Table S3 while emissions per livestock head appear in Table S4. Total annual GHG emissions within each 2km grid are the sum of the annual SOC and biomass carbon fluxes and the estimated emissions from agricultural activities associated with each land use found within the grid.

This analysis does not represent a complete inventory of greenhouse gas emissions related to land use in Great Britain. The analysis was limited by the information provided by the scenarios and therefore does not account for emissions from land use practices (for example peat extraction) that cannot be inferred from scenario land use data. Estimates of greenhouse gas emissions for forestry practices (such as application of pesticides and energy use in harvesting) were not included as such data were not available. However, informal discussions with the UK Forestry Commission suggested that any errors arising from such omissions are likely to be very minor.

S2.5: Valuing GHG emissions

We use the official UK non-market Marginal Abatement Carbon Cost (MACC) approach providing annual non-traded carbon prices out to 2100 (*32*). This is based on a target constant approach where carbon emissions are assumed to be abated in line with the UK Government's domestic carbon emissions target of at least an 80% cut in GHG emissions by 2050 (*57*). The UK Department of Energy and Climate Change (DECC) non-traded carbon price of £53.70 /tCO2e was used in all analyses (prices were converted to 2010 values using the UK Treasury's GDP deflator (*58*)). The DECC carbon values explicitly differentiate between "traded" carbon prices (i.e. from sectors currently engaged in the European emissions trading schemes) and non-traded carbon prices (for emissions from other sectors including agriculture). DECC therefore assume that traded and non-traded carbon emissions are essentially different commodities. Indeed the official DECC traded 2012 carbon price was £22, implying that the marginal abatement costs are allowed to be higher for the reduction of emissions in the non-traded sectors of the economy than in the traded sectors. For comparison, a recent meta-analysis reports a mean MACC carbon price for 2020 (based on an assumed 20% reduction in carbon emissions by 2020) for non-traded carbon of €126/tCO2 for the UK and €77/tCO2 for the EU (*59*) or, in Sterling values, approximately £109 and £67 respectively. This therefore suggests that the DECC value used in the present study is somewhat conservative. Discounting these values back from 2020 to 2012, DECC's estimate is close to the EU average reported by (*59*) and below that reported for the UK.

The value of £53.70/tCO2e used by DECC is roughly equal to the mean estimate of the shadow price of carbon given by a recent a meta-analysis of 47 studies (yielding 232 estimates) based on a social cost of carbon approach (*59*). The mean cost reported is €49/tCO2 (approximately £42/tCO2 at the current exchange rate) with a 95th percentile cost of €185/tCO2 reflecting a considerable right skew in the distribution of estimates (*59*). Moreover, use of the DECC non-traded carbon price is further justified in the context of this research, because this price provides the monetary basis on which UK policy decisions (including land use decisions) should be made and is therefore directly relevant to UK policy formation on land use and climate change in a way that other carbon price estimates are not.

S3: Modeling and valuing outdoor recreation

S3.1: Overview

The analysis entailed the development of a methodology for spatially sensitive and ecosystem specific prediction of outdoor recreation visits and their value. Data on both outset and destination characteristics were combined with observed outdoor recreation information collected via surveys of over 40 thousand households to yield a trip generation function (TGF) predicting visit numbers. A meta-analysis (MA) of relevant literature was undertaken to predict ecosystem specific per-visit values. Combining the TGF and MA models permits estimation of spatially explicit visit numbers and values under present and potential future land use (including those envisioned under the various land use scenarios of the UK National Ecosystem Assessment; UK-NEA).

S3.2: Data and analysis of the Trip Generation Function (TGF)

Outdoor recreation is one of the major leisure activities of the UK population. Over 2.8 billion outdoor recreational visits were made in England during 2010, entailing direct expenditure of over £20 billion (*33*). The spatial distribution of these visits is highly non-random, a reflection of: the distribution of population and its socio-economic and demographic characteristics; the physical and ecological nature of recreational sites and their location; the availability of substitutes (and complements); and the travel time and other costs involved in visiting sites and substitutes. Thus, every destination will attract different numbers of visitors. Moreover, the ecosystem and other attributes of a site not only affects the number of visits but also alters the on-site experience and hence the value of a visit.

In order to reflect this complexity of spatially- and ecosystem-sensitive recreational behavior, we develop and implement a two-step model of open-access recreational visits and their associated values in Great Britain. In the first step we develop a TGF which explains the count of visits from a given outset area to a given site (the dependent variable) as a function of several independent variables including the characteristics of the outset location (including socioeconomic and demographic characteristics of the population and the availability of potential substitute sites), the characteristics of the destination site (such as the habitat type as defined by (*60*)) and the travel time (and hence cost) of the journey. Data on visit counts and outset and destination locations are obtained from the nationally representative, Monitor of Engagement with the Natural Environment (MENE) survey (*33*). These data are drawn from interviews with 48,514 households across all areas of the country and all seasons from March 2009 to February 2010. The interviews consisted of week-long diary records. These revealed that 20,374 households undertook a recreational trip during the diary week, encompassing visits to more than 15,000 unique locations across England. Zero visit records are also incorporated into the analysis to allow findings to be representative of all households. The same source provides information for calibration from the survey to the total number of visits undertaken nationally per annum.

To permit linkage to nationally representative data on population socio-economic and demographic characteristics (which have been shown to influence trip behavior; see, for example, (*61*) and to facilitate subsequent transferral of our analysis to all areas of the country, the postcode of each interviewed household was first converted to Ordnance Survey (OS) grid reference locations (*62*). These were then linked to their corresponding UK Census Lower Super Output Area (LSOA) (*63*) enabling linkage to corresponding socio-economic and demographic variables (2001 UK Census) (*34*). LSOA-level measures of median gross annual household income are taken from the 2008 Experian Mosaic Public Sector dataset (*64*). LSOA boundaries are drawn in part to ensure homogeneity of area socio-economic and demographic characteristics. Therefore the approximation of household characteristics by LSOA-level data is both reasonable and greatly enhances the transferability of functions across all areas of Great Britain.

Transferability is also enhanced through our treatment of the destination site locations. These were assigned to the standard OS 1km square grid cell in which they are located, yielding a more manageable dataset in which data are grouped from over 15,000 destinations to 7,575 unique grid cells or sites. Travel times from population weighted centroids of all LSOAs to destination sites were calculated using the OS Meridian road network (*65*). This is a GIS dataset consisting of motorways, A-roads, B-roads and minor roads. Data from (*61*) were employed to make allowance for varying average road speeds and urban versus rural congestion. Calculation of travel time is via cost weighted distance functions and details are available from the authors.

The environmental characteristics of destination sites are defined by linking their grid cell locations to habitat proportions derived from the 25 metre resolution Land Cover Map 2000 data (*66*). The Fuller et al. (*66*), habitat types were reclassified to conform to the UK-NEA (*60*) ecosystem categories of: (i) broadleaved woodland; (ii) coniferous woodland; (iii) coast (littoral and supra littoral); (iv) enclosed farmland; (v) freshwater body; (vi) mountain, moorland and heathlands; (vii) estuary (sub littoral); (viii) semi-natural grassland; and (ix) urban and suburban. Each 1km square (and recreational sites therein) was therefore described in terms of proportions of each of the UK-NEA ecosystem categories.

Holding all other factors constant, the number of visits to a specific site from any given outset location will be lower when that outset area is well served by other local substitute sites (*61, 67*). To incorporate this within our TGF analysis a series of GIS derived variables were generated. These assessed the availability of substitute resources by defining circular zones around each LSOA and calculating the percentage of each ecosystem type in that area. Zonal Statistics ++, a module of the 'Hawths Tools' plug-in for ArcGIS was used to query the habitat types in the cells entirely within the search radius. These are converted into percentages of the total search area (1km cells entirely within the search radius which is varied as described subsequently). The radii of the circles defined around each LSOA were varied to allow empirical investigation of the optimal size of the surrounding area which captures the substitution effect. Using an AIC selection criterion to compare models employing differently generated substitution variables indicated that a measure constructed using a 10 km radius around each LSOA population weighted centroid provided the best fit to MENE visitation data. Therefore, this measure of substitute availability is included as an explanatory variable in the TGF.

One issue we highlight here is adjustments to avoid double counting required when adding open-access recreation values and urban greenspace values together (SM4). Both consider the recreational value of urban greenspace areas. To avoid overlap the open-access recreation analysis omitted visits to urban parks where the travel distance from the outset location was less than 3 km (the area generating the large majority of recreation and amenity values in the SM4 analysis). To avoid these observations influencing the prediction of visits the TGF was estimated omitting these trips.

Table S5 reports the best-fitting TGF. As expected, a negative relation with travel time is a highly significant predictor of visits. However, numerous other relationships are observed. The impact of land use (percentage of a given ecosystem habitat within a 1km^2 grid square) is clearly important in determining visits to a potential destination. Marine, coastal and freshwater sites exert the greatest attraction, followed by mountain and woodland sites and with urban locations having a negative impact upon visits (although of course the latter will often have the lowest travel times which boosts the number of visits they generate). Substitute availability around a potential outset location also influences visit numbers. As expected these are negative relationships such that, for example, outset areas with high availability of attractive freshwater resources in that locality tend to yield relatively lower counts of visits to other potential destinations than would otherwise be the case. Interestingly urban locations also yield relatively lower counts suggesting that the residents of such locations might be less willing to travel to other destinations than those in rural locations; a finding which might reflect differing preferences between urban and rural populations (i.e. a self-sorting occurs whereby those that have strong preferences for rural recreation tend to live in rural locations). Relationships with socio-economic and demographic factors are as expected with wealthier and retired groups taking more visits and, as noted elsewhere (*68*), non-white groups revealing lower engagement in outdoor recreation pursuits. Finally the positive relation with outset area population, while unsurprising (larger groups of people yield higher visit counts), should not be over-interpreted. LSOAs are defined such that population does not vary greatly between these areas.

Using the calibration information given in the same data source (*33*), the estimated TGF is used to predict the number of visits to each 1km grid square across Great Britain under both current land use and under the alternative land uses envisioned in each of the UK-NEA scenarios. In each case the resulting estimates of visit numbers are then applied to estimates of the value of each visit (adjusted for the ecosystem profile of each grid square in each scenario) as estimated by our MA model, to which we now turn.

S3.3: Data and analysis of the valuation Meta-Analysis (MA)

In the second step of our analysis, we develop a trip valuation meta-analysis (MA) model to determine the value of a recreational visit adjusted for the ecosystem at the destination sites. The MA model is estimated from a re-analysis of nearly 300 previous recreational valuation estimates obtained from the literature, details of which are available from the authors. In line with previous meta-analyses (e.g. (*67, 69, 70*)), the comparability of valuation estimates is enhanced by controlling for: study year (capturing the impacts of inflation) (deflators were obtained from (*58*)); variations in purchasing power across samples (data on purchasing power parity indices were obtained from (*71*)) ; and methodological differences across studies (the grouping of some estimates within the same study is addressed by estimating the model using cluster-robust standard errors, clustered at the study level). Analysis then identified the variation in recreational values according to the different ecosystems at destination sites. Following analyses of alternative functional forms (available from the authors), Table S6 presents the best fitting MA model.

The MA model provides us with estimates of the per visit recreational value of visiting different habitats. In order to obtain habitat-specific recreational values for use in our analysis we need to choose values for the non-focal variables in our MA model. The variables 'Use value only' and 'RPM & mixed' were set for values derived from stated preference studies of recreational use value as these provide conservative, lower-bound estimates. The sample size variable was set equal to its mean while the survey year variable was set to the most recent year in our dataset to represent state-of-the-art methodological developments in study design.

Results show that mountains, moors, heathlands yield the highest values; grasslands the lowest; while marine, coastal and woodland areas provide intermediate values. Note of course that these reflect only the visitors' perceptions of recreational value rather than say the biodiversity, greenhouse gas storage, or other ecosystem service values of an area which we consider elsewhere.

S3.4: Predicting area-specific recreational values under current and future land use

Recall that, for any given land use (say the current situation) the TGF provides estimates of the number of visits to each potential destination (defined as each 1km grid square across the whole of Great Britain) by taking into account: the habitat characteristics of that destination (defined as proportions of each of the habitat types within that 1km grid square); of surrounding areas; of the spatial distribution of the population and their socioeconomic and demographic characteristics; and of the characteristics of the areas surrounding their home locations. In a similar manner, the MA model yields estimates of the recreational value of a visit to a given location based upon the characteristics of each grid square (defined as above). The product of these two models therefore gives us the expected recreational value of each area under a given land use and population scenario. Summing across any area gives the recreational value of that area (note that one of the desirable features of this methodology is that it can be tailored for decision making at any resolution).

Given this model structure, if we change the land use (as envisioned in the UK-NEA scenarios) or the size or characteristics of the population (again a scenario variable), this will alter both the expected number of visits to each area and the value of those visits. The flexibility of this approach readily allows the consideration of any scenario. The MA model provides us with estimates of the per visit recreational value of visiting different habitats. Results show that mountains, moors, heathlands and greenbelt areas yield the highest values; grasslands the lowest; while marine, coastal and woodland areas provide intermediate values. Note of course that these reflect only the visitors' perceptions of recreational value rather than say the biodiversity, greenhouse gas storage, or other ecosystem service values of an area which we consider elsewhere.

The UK-NEA analysis developed six future scenarios each with a high and low greenhouse gas (GHG) emission variant (for purposes of brevity, in the present paper we focus mainly upon the low emissions variant of each scenario, with high emission variants available from the authors). These scenarios envision different alternative futures for the UK by 2060 arising from changes in land use (partly driven by GHG linked climate change) and in the socio-economic and demographic characteristics of the UK population. The subsequent Supplementary Materials section on scenarios gives details of land use changes in each of these variants. In summary, while some scenarios envisage increases in high quality environmental areas at the expense of agricultural land, others reverse that flow. These land use changes have a direct impact on the availability of land for high quality recreational purposes, with those envisioning relatively larger increases in agriculture likely to impinge upon open-access recreation. However, as indicated in Table S5, changes in income and population are also expected to affect the number of open-access recreational visits (with positive associations in both cases). To further enhance the realism of scenarios upon recreation demand and values, the scenario development process also considered likely changes in both incomes and population, concluding that both were likely to increase over the forecast period to 2060. Accordingly both variables were increased by a relatively modest amount as part of the forecast analysis.

The various changes in land use (and population/income) envisioned under the UK-NEA scenarios were applied to the TGF-MA analyses to yield corresponding estimates of response in recreational values. The spatially explicit basis of the methodology provides outputs for each 1km grid square across Britain. Figure 2 in the main text maps the resulting spatial distribution of changes in recreational values between the baseline year 2010 and scenarios WM and NW while Table S7 sums estimates at national level and calculates per capita equivalents.

The spatial trends illustrated in Fig. 2 immediately illustrate one simple yet vital finding; land use changes yield larger recreational value impacts when they occur near to populations as opposed to in more remote locations. This is hardly surprising given the importance of travel costs in determining visitation behavior, yet it shows that, in determining spending on recreation, location is a vital criterion. The numeric results of Table S7 suggest that the potential for increases in recreational value is substantial (although note that some of the scenarios deliberately consider somewhat extreme land use changes relative to the current situation). However, both this table and Fig. 2 show that policies which entail substantial losses of greenbelt land near to population centers (as per the WM scenario) lead to substantial losses in recreational value. This contrasts with the GF, GPL or (more extreme) NW scenarios where the natural environment is conserved and enhanced. However, one of the central messages of our paper is that land use change generates multiple impacts and decision making should take all of these effects into account if resources are to be used more effectively.

S4: Urban Greenspace

S4.1: Overview

The analysis sought to provide estimates of the value of changes in urban greenspace and link these to changes in the environment and the proximity to households to permit the transferral of analyses and valuations across the UK.

S4.2: Data and Materials

An analysis of the extant literature identified three distinct forms of urban greenspace: Formal Recreation Sites (FRS), Informal Greenspace (IG) and City-Edge Greenspace (CEG). Each of the six UK-NEA scenarios for 2010 to 2060 envisions changes in each of these greenspace types. FRS and IG changes are directly specified. In contrast CEG changes are implied by changes in urban area and population (which together determine the numbers of people located at differing distances to the city edge). Details of these characteristics are presented in Table S8 for each UK-NEA scenario.

In order to assess the implications of these scenario induced changes upon urban populations (and subsequently assess the consequent impact upon the value of greenspace) detailed data on greenspace locations (by type), geographical population distributions and socioeconomic characteristics (both determinants of the value of that greenspace) were assembled for five case study cities: Aberdeen, Bristol, Glasgow, Norwich and Sheffield. These were chosen to embrace the diversity of UK cities in terms of both location and size. Data sources are detailed in Table S9.

The value of Informal Greenspace in the UK has only been the subject of one prior analysis (*72*) and therefore could not be included within a meta-analysis of the wider literature without inducing an identification problem between study and greenspace type. Therefore, the meta-analysis was restricted to studies of the value of Formal Recreation Sites and City-Edge Greenspace alone (with Informal Greenspace considered subsequently). Further exclusions omitted studies which failed to control for the very strong relationship observed between the value of a greenspace and its distance to the valuing household (see, for example, (*73*)). Some 61 valuations were obtained from five studies which conformed to these requirements. No significant difference was found in the value of these two forms of greenspace and so these were jointly modeled using a two-stage double log Heckman selection model where the dependent variable is the natural log of the reported ‘marginal value’ (MV; measured in £) of living one metre closer to the centre of an accessible greenspace. The Heckman selection model was used to allow inclusion of observations with a reported zero marginal value of greenspace (values treated as missing in the log-log model). There was no evidence of a selection bias and the incidence of a zero marginal value is driven by study design rather than fundamentals. Results from this analysis are reported in Table S10.

The model reported in Table 3 confirms the strong negative effect of distance upon values that has been observed in the prior literature. The 'People' variable is measured at the Census Lower Super Output Area (LSOA) level and aggregated to the study level to provide a measure of the pressure upon resources. It is expected that at high levels of such pressure congestion effects will lower the value of those resources.

Based on the estimated coefficients the function to compute the MV of Formal Recreation Sites and City-Edge Greenspace is as per (Eq. 6):

$$MV(Distance, Size, Income, People) = e^{44.53}\frac{Size^{0.5}}{Distance^{0.941}\cdot Income^{2.945}\cdot People^{0.554}} \tag{6}$$

where *Distance* refers to the Euclidean distance in metres between the household and the greenspace in question (either Formal Recreation Sites or City-Edge Greenspace)

; *Income* represents the median gross annual household income at the LSOA level (from (*64*)); and *People* refers to the pressure on greenspace resources exerted by the population of the relevant city.

The value function for Informal Greenspace is based on estimates presented in (*72*) and takes the form given in (Eq. 7):

$$MV = 0.02268\, p^2 - 4.53686\, p + 226.843 \tag{7}$$

where p measures the percentage of Informal Greenspace cover in a $1km^2$ square.

For each of the study cities and each of the six NEA scenarios the change in the value of ecosystem services provided by all three urban greenspace types was first computed at the level of full postcodes and then aggregated to LSOA level. The potential exists for overstating the value of changes in Formal Recreation Sites if we ignore the substitution effect that occurs when more than one site in a household's neighborhood is improved. The presence of substitutes means that improvements in one site will result in lower benefits if other sites are also improved. This 'substitution effect' was allowed for by only counting values induced by changes in the most highly valued park to a given household. Note that this contrasts with the approach given in (*12*) where the sum of value change generated by all sites is reported. However, the latter most likely results in an overestimation of benefit changes due to the omission of any substitution effect between sites. That said, we recognize that the present approach provides a lower bound, conservative estimate of the value induced by changes in formal urban parks.

To extrapolate from our study cities to other urban areas in Britain, for each scenario the natural logarithm of the value change at the LSOA level was regressed on various explanatory variables at both the LSOA level (e.g. median household income) and the city level (e.g. the logarithm of city population). The resulting scenario specific models were then applied to the same variables derived for all British cities with a population of 50,000 and above (based on (*74*)). This extrapolation covered a total of more than 25,000 LSOAs containing just over 15 million urban households and almost 40 million people, representing about two thirds of the population of Great Britain. The results of this analysis are presented in Table S11.

S5: Modeling the diversity of wild bird species

S5.1: Overview

We describe an analysis modeling the linkage between land use and measures of wild species diversity (via a bird-based measure as a proxy) through the modeling of a large and spatially disaggregated dataset. The derived model is subsequently used to identify whether, in each area of Great Britain, our land use policy scenarios result in an increase or decrease in this diversity measure (with scenarios leading to reductions in a given area being ruled out of consideration for land use change in that area, although note that desirable conservation outcomes will depend on the species compositional change of any increase, so other biodiversity metrics might be considered).

S5.2: Materials and methods

While the other impacts of land use change considered in our analysis are measured in terms of their economic value, such measures were not applied to our indicator of biodiversity effects. This is not an indicator of low value; in fact, the reverse is likely to be true as biodiversity plays a number of crucial roles across the ecosystem service hierarchy (*75, 76*). However, one of the major elements of the multiple benefits generated by biodiversity is what economists term the 'non-use' value of continued existence. This is the value generated entirely separately from any use, either direct or indirect, of the driver of that value. This is categorically different from the use-value benefits that humans derive from, say, the pollination services provided by multiple species, or the (again use-value) recreational enjoyment associated with watching wildlife. This categorical difference also causes a major methodological problem for the assessment of non-use existence values in that, unlike use-values, there is generally little or no observable behavior from which underlying values can be measured. Some economists have responded to this challenge through the application of stated preference (SP) techniques that use public surveys to estimate individual willingness-to-pay for the continued existence of species (e.g. (*77*)). However, such applications typically violate two of the major principles for valid SP design (*78*). First, the difficulty of ensuring that survey respondents actually have to pay the amounts they say they will means that most SP studies of existence value rely upon hypothetical scenarios in which respondents have no incentive to tell the truth and indeed may have good reason for misrepresenting their values (either by overstatement if they suspect that no payment will actually be enforced, or conversely understatement if they feel that others will pay and allow them to 'free-ride'). For analyses of the significant differences between stated and actual payments in the context of conservation goods see (*79*) and (*80*). Second, even if respondents wish to report their true values for conservation, the complexity of biodiversity issues means that they may be very uncertain about what their own willingness to pay really is. This can result in framing effects where the design of SP questions can influence valuation responses (see, for example, (*81, 82*)).

While SP studies may not provide robust estimates of the non-use existence value of biodiversity, they confirm that people do not like to see declines in species diversity (*83*). Indeed, this principle has long been reflected in national and international policy (*84-86*). How then might this clear preference be incorporated within economic analyses in the absence of robust values for conserving wild species? One approach is to impose a constraint upon that analysis requiring that, in any area, any option that results in undesirable changes in measures of biodiversity should be ruled out of consideration for adoption in that area. The value of the lost economic activity from imposing that constraint provides us with an estimate of the 'opportunity costs' of maintaining biodiversity. Typically, there will be a number of potential options for land use which comply with that constraint and the one which minimizes those opportunity costs is referred to as the cost-effective solution. While this cost cannot be taken as inferring the benefit value of the biodiversity conserved, it ensures that those values (whatever they are) are preserved and provides an acceptable and risk-averse approach to conservation.

Note that several variants of the aforementioned constraint might be envisaged. The strictest interpretation would be to adopt a constraint against any policy causing losses of biodiversity anywhere in the Great Britain, irrespective of where they occur in the country. This would impose much higher opportunity costs than the variant tested in this paper where we only constrain against the use of a policy within the area in which it causes biodiversity losses, but allow that policy to operate elsewhere. The least restrictive variant would be to ensure that a policy was allowed to operate provided that biodiversity decline did not occur in at least one area, with the size of that area being commensurate with the sustainability of the species concerned. While this variant generates the lowest opportunity cost (because it is the least intrusive upon economic activity) it ignores the (unmeasured) loss of value in areas where declines do occur and crucially relies upon the adequacy of scientific knowledge in assessing the minimum area for long term sustainability of the species concerned. It is therefore a potentially higher-risk strategy than that adopted in this paper.

Operationalizing this approach requires that we have some acceptable measure of biodiversity. The literature on biodiversity indicators is substantial and diverse (see, for example, (*87-90*)). Clearly, no single animal or plant group can ever provide a comprehensive summary of all aspects of biodiversity. However, within the British context there are good reasons for using bird-related measures. Birds very clearly excite the passions of large sections of the British community, as evidence by the Royal Society for the Protection of Birds having a higher UK membership than any other wildlife or environmental organization at over one million members. Birds are one of the most widely observed aspects of British biodiversity, being high in the food chain and considered good indicators of ecosystem health at the landscape scale (e.g. (*91*)). Pragmatically, the availability of monitoring data is better for birds than for any other aspect of UK biodiversity. The BTO/JNCC/RSPB Breeding Bird Survey (BBS, (*35*)) provides monitoring information and habitat data at a 1km square resolution, annually across Britain. The BBS is a line-transect survey of a random sample of 1km squares. Squares are chosen through stratified random sampling, with more squares surveyed in areas with more potential volunteer surveyors. Observers make two early-morning visits to their square between April and June, recording all birds encountered while walking two 1km transects across their square, with the maximum total count for each species per visit used as the annual count for that year. The aim is for each volunteer to survey the same square (or squares) every year. By modeling the well-acknowledged link between bird assemblages and land-use (*92*) we can subsequently examine the consequence for our biodiversity indicator of the various changes in land use envisaged under each of our scenarios.

We adopt Simpson's Diversity Index (*D*; (*18*)) as a well-established general measure of biodiversity assessed across multiple species and calculated in each year as per Eq. 8.

$$D = \frac{1}{\sum_{i=1}^{S} p_i^2} \tag{8}$$

where S = number of bird species recorded at a focal site in that year, Pi = proportion of birds of species i relative to the total number of birds of all species. Our diversity measure was calculated for 96 bird species at 3,468 individual 1km survey squares distributed across Great Britain, using BBS data from 1995-2006. We acknowledge that this index represents only part of the variation in the composition of bird assemblages, but a different summary index or measure of the health of populations of keystone species could equally be used in the constraint if required.

As an aside, even given the obvious interest in birds amongst the British public, the establishment of an ideal measure relevant to the utility generated by wild species remains an open empirical question. Given this, our use of a single index relating to diversity rather than say multiple scores relating to high profile charismatic species (which might plausibly provide a superior link to underlying utility) should be treated as a vehicle for methodological development rather than a definitive measure.

General Linear Models were run using the GENMOD procedure in SAS 9.2 (SAS Institute 2000), with 100km square identity included in every model as a control for gross spatial distribution. Each of the 511 possible combinations of the nine land cover variables (Table S12) was fitted as a separate model. Squared terms were always fitted with the corresponding linear term to allow for curvilinear responses. In order to account for variable survey effort across the UK and to ensure that the model results were equally applicable to all parts of the UK, a weighting variable was included in every model. The country was divided into the standard regions used in the organization of the BBS (N=80), the total number of BBS squares surveyed during each year being divided by the number of squares surveyed in that region during the same year. The weight value for each square used in the models was the mean weight value across the years in which that square was surveyed. The dependent variable was the mean D value (Eq. 8) across all years from 1995-2006, so incorporating data from five years either side of the 2000 Land Cover Map survey. The Akaike Information Criterion (AIC, (*93*)) value was calculated for each model, with the lowest value across models showing the best fit to the data with a parsimonious combination of variables. AIC weights and model-averaged parameter estimates were calculated for each variable, squared term, level of the 100km factor and intercept along with model averaged standard errors, as per Burnham and Anderson (*94*, *95*). See Tables S12 and S13 for details.

The model-averaged parameter estimates were used to calculate predicted diversity values for each 1km grid square across Great Britain, based on CEH Land Cover Map 2000 values and for each of the twelve NEA land-use change scenarios. The difference between these values was calculated to determine whether avian diversity was predicted to rise, fall or remain constant in each square. In accordance with prior expectations, diversity was highest in lowland areas (especially those characterized by high proportions of inland water), which suit generalist species, and lower in coastal and upland habitats where specialist species dominate.

The estimated model was applied to the various land use maps derived under each of the UK-NEA scenarios. Corresponding measures of biodiversity for each scenario (under both high and low emissions variants) were then generated for each 1 km grid square. Figure S2 illustrates the spatial distribution of changes in diversity from the present day under each scenario (assuming high emissions).

The results mapped in Fig. S2 highlight the importance of incorporating spatial variation within any analysis of the biodiversity impacts of land use change. Whilst the model-averaged parameter estimates in Tables S12 and S13 indicate that gross spatial distribution is the strongest driver of bird diversity in these models, every scenario still generated a mix of positive and negative impacts in different areas with land use change, but often with variation across small spatial scales. This highlights the relevance of our constraints approach to bringing biodiversity into economic analyses. However, the intensity and location of impacts varies significantly between scenarios. The extensive, pro-environment NW, GPL and GF scenarios generate increases in our biodiversity measure in most places (shown in green in Figure S2), although there are important exceptions (for example the decline in biodiversity in south-west Scotland under the NW scenario proves to be a notable driver of policy switches when we apply our biodiversity constraint in the analysis reported in the main paper). The NS scenario is interesting in that its focus upon increased agricultural intensification in lowland areas generates corresponding declines in bird diversity, while the uplands generally experience increases in diversity. By far the most serious losses are generated by the intensification of agriculture envisioned under the LS and, to an even greater degree, the WM scenarios, with the latter in particular leading to relatively major losses of biodiversity across most areas of the country. It must be noted, however, that changes in species composition regarded as positive for ecosystem health or from a conservation perspective could be reflected in either positive or negative changes in the diversity index. We would therefore recommend that practical applications of a conservation constraint based on bird assemblages consider a more nuanced index tailored to reflect conservation priorities more directly. This could be achieved by, for example, restricting the species list considered or weighting species' contributions to the index.

Our spatially explicit, modeled linkage between land use and a biodiversity index allows us to operationalize the conservation constraint set out earlier in this section. Specifically, in any area, any policy scenario that results in a reduction in our biodiversity measure is excluded from application within that area. We have only considered six scenarios (arguably twelve given the two emission variants to each) and one biodiversity measure in our analysis, whereas in reality an almost infinite variety of policy permutations are feasible. Nevertheless, inspection of Fig. S2 shows that, even with this restricted analysis, the diversity of policy impacts generated by the natural variation of the environment means that, in any area, our constraint still allows us to pick from a number of options. In short, the cost-effectiveness of our biodiversity conservation constraint can be demonstrated even within this restricted consideration as, in each area, we can choose from a number of options to minimize the opportunity costs of conservation.

S6: Scenarios

S6.1: Overview

The main paper focuses upon the consequences of land use change and its impacts within the UK. While the analysis considers changes in all land uses, the major land use within the UK is, by far, farming.

The agricultural land use model, as summarized in the main paper and described in detail in S1, takes account of a variety of drivers in producing estimates of changes in farm land use. These drivers can broadly be divided into physical-environmental factors and those concerning socio-economics and policy.

Physical-environmental drivers are incorporated into our land use analysis by the inclusion of both those determinants which vary by location (such as soil type, depth to rock, etc.) and those which vary both cross-sectionally and temporally (including climate related variables such as temperature and precipitation). Relationships between these determinants and land use are revealed through the analysis of large, highly detailed databases of measures covering all of the UK at resolutions down to 1km and finer and covering an extended time series (of more than 40 years). Combining these with similar cross-sectional and temporal data on socio-economic and policy determinants (discussed below) and relating these to similar spatially explicit time series data on land use allows us to examine how changes in these drivers are linked to responses in the use of land. We move from simple statistical relationships derived from data to a behavioral econometric analysis in which the objective of change (profit seeking) is specified (see S1). Our analysis is based on consequences of each change in land use upon all analyzed ecosystem services.

Our major empirical focus, however, is not the modeling of historic land use, or even the consequences of change on the other ecosystem services considered. Rather, our objective is to assess the impacts upon both land use and ecosystem services of plausible patterns of change in land use drivers into the future. Specifically we consider predictions over the extended time horizon to 2060. To investigate this we first need some understanding of how underlying drivers will change over this period. In the case of physical-environmental factors, determinants such as soil type and geology, while extremely important drivers of land use change between locations, will, for any given location, not vary over our prediction period. Rather, the major physical-environmental driver of changes in land use over this period will be climate change. To allow for this, temporal and spatial variation in climate is incorporated through direct inclusion of related variables such as growing season temperatures and precipitation; changes in which are taken from the modeled outputs of (*14*). As these outputs are well documented in that source, and their impact upon land use is considered directly in the main paper, we do not discuss them further in this section, the remainder of which is devoted to quantification of the socio-economic and policy drivers of land use change.

Problematic through the prediction of say climate change clearly is, at least it is the product of verifiable relationships the directionality of which is known, albeit with uncertainty. Arguably the task of predicting future agricultural and environmental policy involves even greater complexity. For example, schemes funded under the EU Common Agricultural Policy have at one time sought to increase agricultural areas by paying farmers to rip out hedgerows, only to be reversed by subsequent environmental policies designed to encourage the planting of new hedges (*96-98*). Given the complexities of formally modeling such systems it is simpler to consider a matrix of possibilities where in some cases agricultural polices increase farm area (at the expense of ecosystem service rich environmental areas) while in other cases agricultural areas are reduced (and pro-environmental regulations tightened).

S6.2: Materials and methods

As noted, the scenarios were developed in accord with the methodology set out by (*1*) with full details of the procedures adopted by the UK-NEA provided in (*99*). Scenario development started as a broad based process taking in a wide range of issues, many of which are not pertinent to the land use change focus of the main paper.

Six scenarios were developed. The *Go with the Flow (GF)* scenario provided a comparator case where land use continues to be determined in a manner similar to current policies (i.e. compromise between development/growth and a desire to protect the environment). Compared to this the *Nature at Work (NW)*, *Green and Pleasant Land (GPL)* and *Local Stewardship (LS)* scenarios assume stronger levels environmental regulation. These are further differentiated by: (a) the *NW* scenario having its environmental improvements focused upon the peri-urban greenbelt area where ecosystem service related recreation values are expected to be highest (prompted by government agency and policy initiatives set out in (*100-102*)); (b) the *GPL* scenario having its environmental enhancements focused upon existing areas of nature conservation and biodiversity importance such as designated areas, locations with high levels of broadleaf woodlands, etc., (i.e. a recognition of the importance of enhancing existing conservation lands and moving towards a system of integrated networks of such areas as highlighted by the recent Lawton Review (*15*)) and; (c) the *LS* scenario not explicitly focusing on either of these areas. In contrast to these other cases, both the National Security (NS) and World Markets (WM) scenarios envisage reduced levels of environmental regulation. These are further differentiated in that: (d) the NS scenario maintains current prohibitions on development of peri-urban greenbelt, a restriction which is common to all other scenarios except for; (e) the WM case which allows the development of greenbelt as per recent assessments of the potential for using such land for house building (*103*) and Government policy, as set out in (*104*). Table S14 details the various qualitative attributes used to define each of these scenarios.

In order to translate the general attributes of each scenario into changes in land use, an initial task was to define baseline land cover. Data were taken from the Centre for Ecology and Hydrology UK Land Cover Map 2000 (LCM2000) (*66*). This allowed the proportions of the land cover classes in each 1km × 1km cell of the Ordnance Survey National Grid to be estimated.

A rule-based approach was developed in order to alter baseline land use in accordance with the attributes of each scenario. Each scenario specific rule expressed a probability of change from one land cover type to another taking into account the characteristics of each cell (detailed in (*99, 105*)). So, for example, under the strong environmental regulation scenarios there was a relatively higher probability that the area of a cell under agricultural production would decrease with that changed area of land being replaced by, say, broadleaved woodland (and vice versa for the weak environmental regulation scenarios). These rules also incorporated the spatially targeted aspects of each scenario. For example, under the NW scenario the relatively higher probability of transfers out of agriculture was further enhanced for cells near to population centers (i.e. within peri-urban greenbelt areas).

The rules determining probabilities of change were implemented using a Bayesian Belief Network (BBN) tool (*105, 106*). The characteristics of each 1km × 1km cell were read into the BBN and the rules appropriate to each scenario were applied to generate changes in land use within that cell. Repeating this process across all cells generated a scenario specific change in land use across the country. These outputs were subsequently translated into digital maps using a geographical information system (*107*) which permitted ready integration with the overall analysis described in the main paper. Table S15 provides a national level summary of land uses under each of the scenarios using the UK-NEA categorization under the low climate change variant (figures for high climate change are similar and available from the authors).

Reviewing Table S15 shows that all scenarios still imply that the large majority of UK land stays under agricultural land use. Within these areas the specifics of that land use (e.g. which crop is sown; the intensity of livestock kept) are dictated by applying the characteristics of each cell, under each scenario (together dictating physical environmental characteristics, climate, etc.) to the agricultural land use model detailed in S1 and summarized in the main paper.

While the overall area of agricultural land is fairly stable across most scenarios, it increases under the LS scenario and falls significantly under the NW and NS futures, mainly due to increases in woodland, and under the WM scenario, mainly due to an increase in urban extent. Taken together these scenarios provide an interesting matrix of possibilities for examining the consequence of these alternative futures in terms of their impact on ecosystem services and their corresponding values.

S7: Optimization across scenarios

S7.1: Overview

The analysis reported in the main paper yields values for a series of interlinked ecosystem services for each 2km grid square across Great Britain (although note that with the exception of the Agricultural Census, all data were held at a resolution of 1km or finer). These values relate to agricultural output, net greenhouse gas emissions, recreation and urban greenspace. Non-monetary measures of wild species diversity (proxied by measures relating to bird populations) are also generated at the same resolution. These various values and non-monetary measures vary according to land use which in turn depends upon changes in policy scenario and whether climate change is assumed to follow a low or high emission path. Together these generate twelve permutations although the main paper focuses in the main upon the low emission variants (full results for all permutations are available from the authors) so as to highlight the importance of policy selection for the provision of ecosystem services.

The spatially explicit nature of the methodology developed in the paper allows the analyst to consider each area in turn, focusing down to 2km grid squares if desired (although of course any larger area can readily be accommodated). The methodology also allows the decision maker to compare trade-offs across the multiple dimensions of ecosystem services using the common unit of money values (with the exception of wild-species existence values which, for reasons explained in the main paper, we do not monetize and to which we return below). One of the key results of the paper is that no single policy dominates all others in all locations. Therefore, there are gains to be made by targeting policies to their most appropriate locations. Note that this ignores the increase in institutional and administrative costs associated with targeting. However, discussions with relevant Government officials in Defra and Natural England suggest that the additional costs associated with policy targeting are relatively minor. For example the shift from the untargeted Entry Level Stewardship to the targeted Higher Level

Stewardship scheme involves an increase in administrative costs from around 5% of overall scheme costs to about 10%. It seems very unlikely that such additional costs could overturn the very major gains from policy targeting reported in the main paper.

Within each area the optimal policy is initially taken as being that which delivers the maximum total value of all monetized ecosystem service values. This is subsequently refined to exclude all those policies which result in a decline in our non-monetized measure of wild species diversity. The following section provides details of this optimization process. We describe an analysis modeling the linkage between land use and measures of wild species diversity (proxied by a bird-related measure) through the modeling of a large and spatially disaggregated dataset. The derived model is subsequently used to identify whether, in each area of Great Britain our land use policy scenarios result in an increase or decrease in this diversity measure (with scenarios leading to declines in a given area being ruled out of consideration for land use change in that area).

S7.2: Data and materials

The analysis considered changes from the baseline situation (year 2010) to the situation in 2060 under six land use policy scenarios, each having a low and high greenhouse gas emission variant as described by (*14*). Measures of ecosystem response were: market (agriculture) and non-market (GHG emissions, recreation, urban) monetary values and non-monetary wild species diversity assessments. For simplicity of exposition the main paper changes are presented as differences between the 2060 and baseline situation rather than cumulative totals of interim changes. Similarly, to avoid problems of inflation, all changes are valued in 2010 GB pounds. This section documents the extra processing required for integrating values by means of an optimization procedure.

Spatial data integration requires allocation of data to a common spatial unit for analysis. With a raster grid this is achieved by simple overlay techniques in a Geographical Information System (GIS). To avoid unnecessary smoothing of spatial variation, vector data sets were first disaggregated into smaller component units (a fine resolution raster) and then aggregated to fit the common unit for analysis. A 2km resolution grid structure was common to agriculture and GHG emission value measures. Recreation values and non-monetary biodiversity assessments were generated at 1km resolution and were therefore easily converted to 2km resolution measures.

For the urban greenspace value measure, adjustments were made to convert values estimated at a per-household level to per hectare values. First, values were multiplied by the number of households in each Census LSOA. This was achieved via linkage to demographic data on the number of households in each Census LSOA (provided by (*34*)) and information on the boundaries of those areas (obtained from (*63*)). Next, the area contained by each irregular LSOA boundary was calculated using a GIS and converted to a 20 metre square regular grid to minimize error. These cells were then aggregated to the desired 2km squares.

For each land use change scenario in turn, the GIS data layers (digital maps) for each of the monetary values were overlayed and those values were summed for each 2km grid square. Each grid square was then linked to the corresponding measure of change in wild-species diversity for that scenario. Given our rule that for each grid square, any scenarios which generated a reduction in that diversity measure was to be discarded from further consideration a simple GIS ‘mask’ file was also created identifying each grid square for which a given scenario was to be ruled inadmissible.

S7.3: Methods

The optimum policy option for a 2km grid square was defined as the scenario that produced the maximum positive change in value under a given objective. Three objectives were considered for policy targeting:

1. Optimize market (i.e. agricultural) values alone;

2. Optimize the sum of all monetary values (agricultural, greenhouse gases, recreation and urban greenspace);

3. Prohibit policy options that lead to a decline in bird diversity and then optimize the sum of all monetary values (as per 2).

For each of the above rules the optimal policy was identified for each grid square. Processing was implemented in ESRI’s ArcGIS v9.3 (*107*) using the ModelBuilder feature with custom written rules to permit the above selection across scenarios (available from the authors). A further rule was added to discriminate within the very small minority of cases where two or more scenarios gave the same value by examining and following dominance amongst neighboring cells.

For each objective in turn, values were extracted for the optimal mix of policies across all grid cells. Resultant values can be summarized, tabulated or (as per the main paper) mapped to display the change in values from the baseline to the targeted mix of scenarios selected by the optimization process.

S7.4: From land use change potential to practice: Future challenges.

As discussed in detail in S1, we assess the value of farm production using the standard agricultural economics measure of farm gross margin (FGM), which is the difference between revenues and variable costs (i.e. it excludes long term fixed costs, such as farm buildings, which are less relevant in determining annual activity decisions such as which crop to sow). While FGM per hectare obviously varies from year to year as a result of changes in prices of input and outputs, nevertheless typical FGM values in the UK vary from just over £400/ha to in excess of £1000/ha (see (*108*)). Note that this range includes subsidy values; exclusion of which would reduce the lower end of this range considerably (see (*109*)). Therefore the changes in non-agricultural ecosystem service values generated by the different scenarios are significant with values equivalent to around 20% of FGM occurring frequently and much higher values arising in certain locations.

From an economic perspective, our analysis has shown a potential improvement in welfare arising from changes in land use. Put at its simplest, converting land use in line with certain of the scenarios shown here would benefit society. However, this approach to land use planning generates certain challenges for future decision making and implementation, two of which we highlight here.

A first challenge is that there is great spatial heterogeneity in the various ecosystem service values generated by land use change (monetized and non-monetized). Potential gains from land use change thereby vary not only according to which policy is adopted, but also by location. Figure 3 indicates which scenario is best in each area. However, this spatial variation in policy is at odds with the rather simplistic, one-size-fits-all policies which characterize large areas of the EU, let alone the UK. While our analysis shows that gains from moving to spatially targeted policy making are very substantial, such a shift in approach will increase pressures upon decision making and administrative institutions and their associated costs.

The second challenge identified in this respect concerns the mechanics of securing participation by land managers in land use change schemes. Within a liberal democracy such as the UK, demonstration of clear societal gains arising from a change in the use of a privately owned resource, such as land, is insufficient to ensure that such change is implemented. Either new regulation needs to be enacted or compensation is required. While regulation is a possibility, to date the EU Common Agricultural Policy (CAP) has strongly favored compensation approaches and has sufficient funds to massively transform the shape of rural land use within the UK. In 2012 the CAP had an annual budget in excess of Euro 55 billion (£48 billion), representing well over one third of the total EU budget (*110*). This results in CAP payments to the UK in excess of £3billion per annum (*111*). Comparing this to the total value of UK agriculture of about £5billion per annum (which at about 0.5% of GDP is one of the lowest proportions of any country worldwide (*112*)) shows the very high rate of state support for farming and reveals the massive potential which decision makers have to influence agricultural land use. Currently 70% of support is through direct payments with only the remaining minority linked to environmental and other performance.

Furthermore, recent research within the field of economic ‘game theory’ has highlighted the potential for using self–enforcing competitive contracts as a method to ensure that any spending on PES is highly efficient in terms of the improvements in ecosystem services it yields and the costs to the funding authority (*113-118*). Clearly inviting farmers to name their own price for supplying ecosystem service benefits is unlikely to provide good value for money. However, by offering PES support via competitive contracting, funders can ensure that farmers have a clear incentive not to overstate their requirements for compensation. Furthermore, in cases where environmental improvements generate not only public benefits but also private gains (e.g. where less intensive agricultural methods lead to reductions in the treatment costs faced by water companies), these changes in land use can be induced at no direct cost to the public exchequer.

Innovative decision making which embraces challenges such as targeted policy implementation and competitive contracting offers the potential to implement the socially beneficial changes. These innovations require a change in decision systems, however they do not require increases in public funding of environmental policy or any accompanying increase in taxation. As such they should be welcomed by governments as indicating a new direction for policy implementation.

S8: Land Use Statistics

With respect to the analysis provided in the main paper, perhaps the most important statistic to stress is that agriculture is by far and away the major form of land use in the UK (*119*) extending out to include all enclosed farmland and the majority of other grassland, mountain, moor and heathland habitats. The total agricultural area of the UK is 18.3 million hectares or 74.8% of the total UK land area. A small fraction of this is common grazing land. Arable and horticultural crops cover 4.6 million hectares, making up over a quarter of total farmed land. The remaining croppable area is occupied by 1.2 million hectares of temporary grassland and a small amount of uncropped land. Permanent grassland and sole-right rough grazing account for about 10 million hectares of farmed land. The remainder of land on agricultural holdings has non-agricultural uses such as farm woodland. Table S17 provides national statistics while Table S18 gives a country-level breakdown. After accounting for agriculture, other major land uses include woodland (summarized in Table S19) and urban land (Table S20).

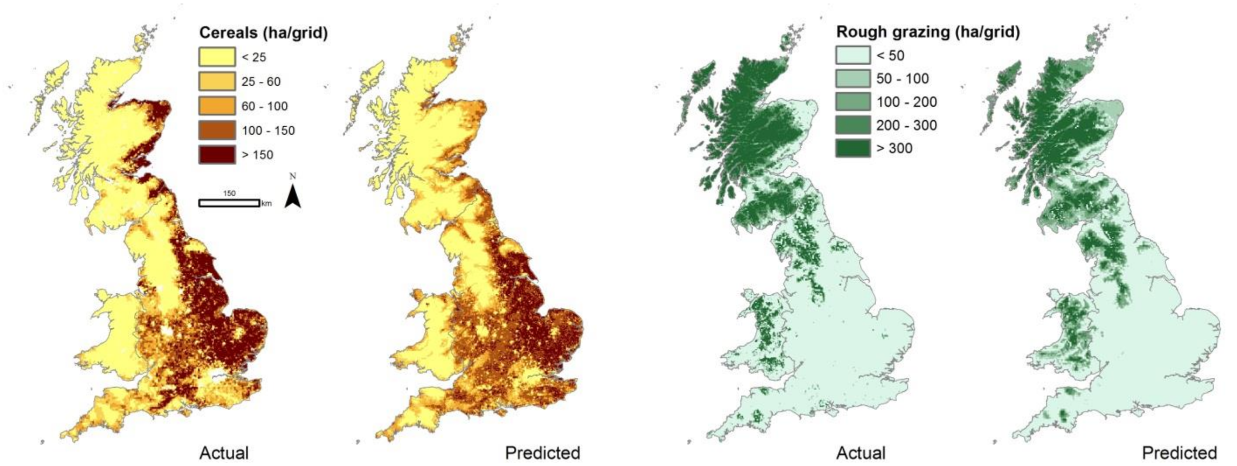

Fig. S1.
Cereals and rough grazing in 2004: model predictions (LHS) and JAC data (RHS) (*23*).

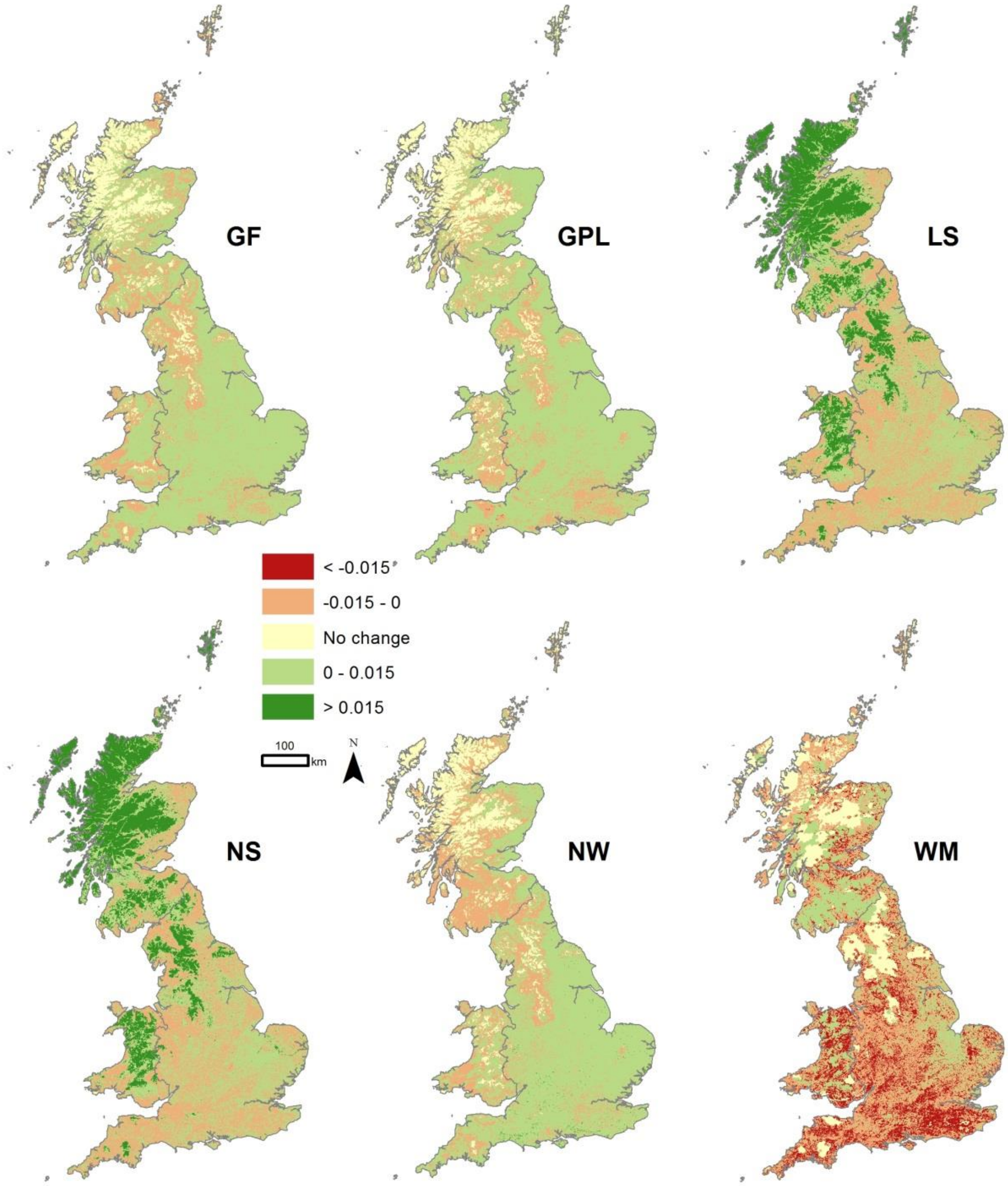

Fig. S2.

Predicted change in Simpson's Diversity Index for British birds arising from a change in land use from the baseline (year 2000) to the land use predicted under each of the UK-NEA Scenarios (assuming a high emissions variant) (*23*).

Table S1.
Forecasting performance.

Variable	MAE Structural	$\hat{s}(x)$
Cereals (ha)	28.77	70.96
Oilseed rape (ha)	8.61	17.54
Root crops (ha)	7.21	17.14
Temporary grassland (ha)	13.07	22.33
Permanent grassland (ha)	50.44	99.03
Rough grazing (ha)	28.53	97.92
Dairy (head)	48.29	92.69
Beef (head)	67.51	120.11
Sheep (head)	442.59	880.55

Notes: Forecasting performance tested on England and Wales data in year 2004 (37980 observations). Only 5% of these observations are used in estimation so this is mainly an out-of-sample forecasting test. MAE = mean absolute error, $\hat{s}(x)$= standard error of the variable.

Table S2.

Estimates of SOC and BIOC for different land uses and non-organic [organic] soils in Great Britain.

Agricultural land use	Carbon stored in above and below ground biomass (tC/ha)	SOC England (tC/ha)	SOC Scotland (tC/ha)	SOC Wales (tC/ha)
Oilseed rape	1.8 [1.8]	111 [133]	157 [187]	120 [142]
Cereals	2.4 [2.4]	111 [133]	157 [187]	120 [142]
Root crops	2.5 [2.5]	111 [133]	157 [187]	120 [142]
Other agriculture	1.4 [1.4]	111 [133]	157 [187]	120 [142]
Temporary grass	0.9 [0.9]	133 [580]	187 [580]	142 [580]
Permanent grass	0.9 [0.9]	133 [580]	187 [580]	142 [580]
Rough grazing	1.7 [2.0]	176 [1200]	249 [1200]	189 [1200]
Woodland	36.8 [36.8]	176 [580]	249 [580]	189 [580]

Sources (*40-42*)

Table S3.

GHG emissions from farm activities related to different agricultural land uses.

Agricultural land use	Emissions from agricultural activities (tCO_2e/ha/yr)	N_2O emissions from inorganic fertilizer applications (tCO_2e/ha/yr)
Cereals	0.55	0.95
Oilseed rape	0.48	1.06
Root crops	0.46	1.01
Temporary grass	0.48	1.27
Permanent grass	0.35	0.89
Rough grazing	0.00	0.00
Other agriculture	0.40	1.03

Sources (*29*, *56*, *121*)

Table S4.
GHG emissions per head from livestock.

Livestock	Enteric fermentation (tCO_2e/head/yr)	Emissions from manure deposited directly onto grasslands (tCO_2e/head/yr)	Emissions from manure used as fertilizer (tCO_2e/head/yr)
Dairy	2.381	0.145	0.016
Beef	1.104	0.086	0.006
Sheep	0.184	0.054	0.001

Sources (*52*, *54*, *121*, *122*)

Table S5.

Trip Generation Function.

	Coefficients	t-stat
One-way trip travel time from outset to site		
Travel time (in minutes)	-0.152***	(-139.7)
Land use variables measured at potential destination		
Log (%Coast at site)	0.165***	(6.301)
Log (%Other marine at site)	0.0905**	(3.216)
Log (%Freshwater at site)	0.0621**	(3.272)
Log (%Mountains at site)	0.0510**	(2.777)
Log (%Woodland at site)	0.0388***	(3.508)
Log (%Grasslands at site)	0.00973	(0.823)
Log (%Urban at site)	-0.176***	(-16.47)
Substitute availability variables measured at potential outset location		
Log (%Coast substitute availability)	-0.0305***	(-3.464)
Log (%Other marine substitute availability)	-0.0348***	(-5.117)
Log (%Freshwater substitute availability)	-0.0689***	(-6.423)
Log (%Mountain substitute availability)	0.00917	(0.860)
Log (%Woodland substitute availability)	-0.0804***	(-3.613)
Log (%Grasslands substitute availability)	0.00922	(0.307)
Log (%Urban substitute availability)	-0.494***	(-28.72)
Demographic variables measured at outset		
Log (Median Household Income) (in pounds)	0.606***	(16.47)
% Retired	0.00842***	(2.735)
% Non-white ethnicity	-0.0170***	(-11.96)
Total population of outset area (no. of people)	0.000259***	(5.592)
Constant	-6.562***	(-16.79)
$\ln(\sigma^2_u)$	-1.065***	(-22.37)
σ_u	0.587***	(42.021)
Observations	3,198,492	

Likelihood-ratio test of $\sigma_u = 0$: chibar2 (01) = 1098.10 Pr>=chibar2 = 0.000

Dependent variable is logarithm of the expected count of visits from an LSOA to a site. Enclosed farmland is set as the base case for both the ‘substitute availability’ and the ‘site characteristic’ variables. The model is estimated using Stata: Version 10. * $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$

Table S6.

Meta-analysis (MA) model of estimates of per person per visit recreational value.

Variable	Variable definition	Coefficient	t-stat
Site characteristics			
Coastal and marine	1 = recreational site valued is coastal or marine; 0 = Grasslands	0.944**	(1.67)
Freshwater & floodplains	1 = recreational site valued is freshwater and floodplain; 0 = Grasslands	0.170	(0.32)
Wetlands	1 = recreational site is wetlands; 0= Grasslands	0.895**	(1.64)
Mountains & heathlands	1 = recreational site valued is mountain or heath; 0 = Grasslands	1.184	(0.90)
Woodlands & forests	1 = recreational site is woodlands & urban forests; 0= Grasslands	0.775*	(1.42)
Urban fringe	1 = recreational site is greenbelt & urban fringe farmlands; 0 = Grasslands	1.248***	(2.29)
Controls for study characteristics			
Survey year	Discrete variable: 1 = survey year is 1975 ... ...29 = survey year is 2008	0.0437***	(2.21)
Sample size	Sample size of study	-0.00547****	(-3.13)
Valuation per household	1 = unit is per household per year; 0= per person per trip	3.043*****	(9.21)
Valuation per year	1 = unit is per person per year; 0 = per person per trip	2.164*****	(6.22)
Other valuation unit	1 = unit is per household/ per person per day/ per month; 0 = per person per trip	2.434*****	(6.85)
Use value only	1 = use value study; 0 = study of combined use and non-use	-0.0373	(-0.14)
Valuation method	1 = revealed preference or mixed methods used; 0 stated preference methods	0.685****	(2.89)
Study of non-UK country	1 = study conducted overseas; 0 otherwise (UK)	0.703****	(2.80)
Constant		-0.420	(-0.71)

Sample size = 297 observations obtained from 98 studies.
R^2 (adj.) value is 0.72
The dependent variable is the logarithm of recreational value/person/trip (£; 2010 prices)
* $p < 0.20$, ** $p < 0.10$, *** $p < 0.05$, **** $p < 0.01$, ***** $p < 0.001$
Grasslands include urban recreation parks, urban greenways, semi-natural grasslands and farmland away from the urban fringe

Table S7.

Change in total recreational value per annum for each country of Great Britain under each of the NEA scenarios (with per capita equivalent).

Country	NEA Scenario					
	GF	GPL	LS	NS	NW	WM
England (£ million)	2,142	3,917	3,347	2,466	9,056	-1,024
Scotland (£ million)	632	1,453	583	688	3,280	-856
Wales (£ million)	179	340	289	291	1,014	-285
GB (£ million)	2,953	5,710	4,219	3,445	13,350	-2,165
GB per capita (£)	13	77	55	30	185	-60

Note: Assumes high GHG emissions scenario (low climate variant values available from authors).

Table S8.

Changes in urban parameters from UK-NEA scenarios for 2010-2060.

Scenario	Change in Formal Recreation Site (FRS) area (%)	Change in Informal Greenspace (IG) area (%)	Change in City-Edge Greenspace (CEG) implied by:	
			Change in Urban Area (%)	Change in Urban Population (%)
NW	39.0	-4.9	-3.0	13.8
GPL	38.9	5.4	0.0	21.7
LS	4.5	2.8	-3.0	0.0
GF	36.2	0.0	3.0	32.2
NS	-34.3	4.8	-3.0	17.2
WM	73.0	20.7	79.0	52.6

Table S9.
Data sources by greenspace type.

Type	Definition	Data Sources	Variables Computed
Formal Recreation Sites (FRS)	Accessible greenspace of at least 1 ha in size including parks, accessible woodlands and recreation grounds.	OS Mastermap Topographic (1:1250) area layer (for Council greenspace) (*64, 123, 124*); UK Forestry Commission Woods For People dataset (*125*); Natural England CROW (1:1250) datasets (*126-128*)	Distance from full postcode centroid (from (*62*)) to FRS centroid in metres. Area measures in hectares.
City-Edge Greenspace (CEG)	Areas of non-urban land directly adjacent to the city-edge.	OS Meridian (1:250,000) DLUA data (*129*); ONS and General Register of Scotland 2001 Census District Area boundaries (*130-132*)	Distance from full postcode centroid (from (*62*)) to city boundary (from (*74*)) in metres. Greenspace defined as 10ha area.
Informal Greenspace (IG)	Features described as being ‘natural’ rather than ‘man made’.	OS Mastermap Topographic (scale 1:1250) area layer (*124*)	Percentage cover per 1km square.

Table S10.

Meta-analysis regression result for Formal Recreation Sites and City-Edge Greenspace using a two stage Heckman procedure.

Valuation equation			*Variable definition*
lnDistance	-0.941***	(0.008)	Log of distance between greenspace and elicitation point (in metres)
lnSize	0.500**	(0.032)	Log of size of the greenspace (in ha)
lnIncome	-2.945**	(0.011)	Log of average annual household income in the study area (in GBP, from *1*).
lnPeople	-0.554**	(0.021)	Natural logarithm of the number of people in the study area
Constant	44.53***	(0.001)	
Selection equation			
lnIncome	-1.196*	(0.068)	See above
Expert	2.685*	(0.051)	Values in original study based on expert judgement (1=yes)
No.Obs	0.000132**	(0.016)	Number of observations in original study
PeerReviewed	1.916	(0.144)	Published in peer-reviewed journal (1=yes)
Constant	10.27	(0.131)	
Mills lambda	1.258	(0.137)	
Observations	61		

Notes: Dependent variable: Natural logarithm of marginal value (lnMValue)

p-values in parentheses; * $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$

Table S11.

The value of changes in urban greenspace under each UK-NEA scenario aggregated for all British cities.

Value	UK-NEA scenario					
	GPL	NW	WM	NS	LS	GF
Total (£Billion p.a.)	2.12	4.76	-18.4	-6.94	1.75	-1.12
Per household (in £)	140	313	-1,210	-457	115	-73.8

Table S12.

Model-averaged parameter estimates, their standard errors and AIC weights after GLM runs of all possible variables for land cover variables.

Variable	Model averaged parameter estimate ± SE	Variable AIC weight
Intercept	2.29306 ± 3.62556	NA
Arable	0.02844 ± 0.01567	0.80
$Arable^2$	-0.00016 ± 0.00011	
Deciduous woodland	0.15126 ± 0.01755	1.00
Deciduous $woodland^2$	-0.00188 ± 0.00026	
Coniferous woodland	0.06773 ± 0.01639	1.00
Coniferous $woodland^2$	-0.00094 ± 0.00013	
Improved grassland	0.05034 ± 0.01437	1.00
Improved $grassland^2$	-0.00038 ± 0.00012	
Unimproved grassland	0.0167 ± 0.01521	1.00
Unimproved $grassland^2$	-0.00063 ± 0.0001	
Urban	0.06978 ± 0.01706	1.00
$Urban^2$	-0.00126 ± 0.00014	
Mountains, heaths and bogs	-0.08134 ± 0.0118	1.00
Mountains, heaths and $bogs^2$	0.00024 ± 0.00011	
Coastal habitat	-0.01183 ± 0.01505	0.58
Inland water	0.01265 ± 0.02121	0.53

Table S13.

Model averaged parameter estimates after GLM runs of all possible land cover variables for levels of 100km Ordnance Survey square class.

Ordnance Survey 100km Square	Model averaged parameter estimate ± SE
HU	6.22 ± 3.46
HY	1.74 ± 3.52
NB	5.49 ± 3.42
NC	6.22 ± 3.40
ND	5.93 ± 3.45
NF	5.49 ± 3.44
NG	5.78 ± 3.41
NH	6.43 ± 3.40
NJ	4.83 ± 3.40
NK	6.28 ± 3.75
NL	3.31 ± 3.69
NM	6.01 ± 3.41
NN	5.60 ± 3.40
NO	5.52 ± 3.40
NR	6.52 ± 3.42
NS	6.05 ± 3.40
NT	5.37 ± 3.40
NU	6.64 ± 3.48
NX	7.00 ± 3.40
NY	6.15 ± 3.40
NZ	7.60 ± 3.41
SD	7.46 ± 3.40
SE	7.04 ± 3.40
SH	7.72 ± 3.40
SJ	8.03 ± 3.40
SK	7.82 ± 3.40
SM	7.43 ± 3.47
SN	8.25 ± 3.40
SO	7.95 ± 3.40
SP	6.99 ± 3.40
SR	9.18 ± 4.55
SS	7.16 ± 3.42
ST	7.40 ± 3.40
SU	6.63 ± 3.40
SW	7.69 ± 3.44
SX	6.83 ± 3.41
SY	6.96 ± 3.45
SZ	6.35 ± 3.50
TA	7.51 ± 3.42
TF	6.69 ± 3.41
TG	7.00 ± 3.45
TL	7.76 ± 3.40
TM	6.75 ± 3.42
TQ	7.08 ± 3.40
TR	7.07 ± 3.47
TV	0.00 ± 0.00

Table S14.

Qualitative-rule base used to define land cover transition matrices for mapping scenario outcomes at the 1km × 1km grid square level. The attributes of each cell shown in the first column define the rule to be applied under a given scenario. These rules were converted into a set of probabilities for a given land cover transition that were operationalized using a Bayesian Belief Network (*105*)*.

			Rule-base for each Scenario					
Criteria	Attribute	Definition	Green and Pleasant Land (GPL)	Nature at Work (NW)	World Markets (WM)	National Security (NS)	Local Stewardship (LS)	Go with the Flow (GF)
Altitude	Upland	*Land > 250m asl; in northern Scotland upland can be almost down to down to sea level though*	Decline in arable, IG, conifer and urban to enhance the landscape biodiversity and aesthetics. BL, SNG and upland habitats all increase as a result.	Similar patterns to GPL although as well as improving biodiversity many of the land cover changes are designed to alleviate flood (> BL, SNG) or improve Regulating ES.	Arable increases slightly although IG declines as animal production becomes more crop-based. Slight decline in BL & SNG to make way for urban growth. Upland habitats decrease slightly due to some conversion to UR. Hi CC increases freshwater as winter flooding becomes difficult and too expensive to manage (the rest to SNG).	Food and timber production very important and CON increases considerably as does arable. Slight decline in IG due to a move towards more efficient food prod (i.e., crop-based protein). BL also slightly increases at the expense of SNG and Upland habitats. Hi CC reduces AR area in UP and more is switched to IG.	SNG and BL two main winners here. Food prod is very important but is managed sustainably and extensively hence the transition to more semi-natural habitats. Upland stays constant but is managed more sustainably.	Slight increases in BL, SNG and Upland reflecting the continuing pattern of ‘softening’ landscapes through agri-env and other conservation grant-aided schemes.
	Lowland	*<250m*	Almost identical patterns to upland (and for the same reasons). Agriculture declines in the UK but is compensated for by much larger imports.	Similar to above although IG declines even more due to it being an inefficient use of land and less meat consumption in the UK. BL also increases more. AR declines slightly to meet Prod. ES demands.	AR increases as a result of a decline in IG (livestock indoors) and a greater need for crop-based animal feed. SNG declines also, some is lost to AR, some to UR. Overall, UR growth is the major lowland winner in the south east and most other land use lose some to it.	AR and conifers increase considerably as does arable. Decline in IG due to a move towards more efficient food prod (i.e., crop-based protein). BL also slightly increases at the expense of SNG. Hi CC reduces AR area in south and more is switched to drought-tolerant conifer.	Similar to above although IG declines slightly (and more under HI CC). Main underlying factor behind land cover changes is a lower demand for food (low pop, less waste) - as a result, SNG increases (but is used for livestock prod too). Loss of AR due to less demand for food.	Continuation of current agri-env policy - slight loss of AR to SNG and BL. Continued conversion of PAWS conifer to BL. Loss of IG as more livestock reared indoors and requires AR crop land. Slight increase in UR as pop continues to rise.

* Abbreviations used: ALC = Agricultural land classification; AR = Arable; ASNW = Ancient semi-natural woodland; BL = Broadleaf woodland; CC = Climate change; CON = Conifer woodland; IG = Improved grassland; PAWS = Planted ancient woodland sites; SNG = Semi-natural grassland; UR = Urban; UP = Upland (aggregation of all the upland cover types given in (*104*));

			Rule-base for each Scenario					
Criteria	Attribute	Definition	Green and Pleasant Land (GPL)	Nature at Work (NW)	World Markets (WM)	National Security (NS)	Local Stewardship (LS)	Go with the Flow (GF)
Woodland potential	ASNWHigh	*Area of land with a density of ASNW or PAWS > 5% of cover in a 10km grid square*	A slightly higher expansion of new woodland near areas of high ASNW but overall new woodland planting is important in both low and high density areas for landscape as well as biodiversity reasons.	ASNWHigh significantly increases BL for conservation/ ecological reasons (and results in lower conifer)	BL woodland stays constant or declines slightly with no ASNW effect on changes. Woodland is abandoned and unmanaged. Some loss to UR growth. Hi CC kills back some vulnerable woods like beech in south.	Increase in BL and huge increase in conifer with little regard to presence or absence of ANSW. ALC more important factor here.	Biodiversity very important in this storyline, as is timber and NTFPs hence increase in traditional native woodland types near existing ANSW woods. Increases in Hi CC to replace AR which struggles with heat and drought.	Presence of ASNW increases likelihood of new BL to improve biodiversity value.
	ASNWLow	*< 5% per 10km grid square*						
Urban influence	Near	*Land ≤5 km or urban boundary*	Distance to urban areas doesn't have a huge influence on land cover transitions (no Urban growth so not an issue)	Distance to urban areas doesn't have a huge influence on land cover transitions except for small urban growth near existing urban.	Near urban is generally converted to urban regardless of land cover type. General spread of urban sprawl.	Generally, proximity to Urban has little effect on other land cover changes.	No influence on land use transitions except for increase in AR (for local peri-urban food prod).	Near urban is more likely to become UR, generally rural areas protected from housing development.
	Far	*Land >5 km from urban boundary*						
Landscape designation	Park	*Land within National Park or AONB*	An important factor which affects changes to semi-natural habitats (increases more in Parks) and productive cover types (decreases less outside parks).	Park designation significantly increases BL for conservation reasons (and results in lower conifer)	Park designation has very little consequence for land cover change. In some areas, UR may increase in Parks as the rich want to live in beautiful areas.	Park designation has very little consequence for land cover change. Recreation and conservation not important in this storyline.	Has major influence - Park areas protect SNG and BL and both increase at expense of AR and IG.	National Parks etc continue to maintain strict planning laws. Conversions of AR and IG to SNG and BL occurs, as does some to Water.
	NotPark	*Land outside National Park or AONB*						
Agricultural Land Class	High	*Grades 1 & 2*	ALC 1 & 2 loses less productive land from arable and IG to other land uses than does ALC 3, 4 & 5. The lowest grade soils gain more in conifers.	High ALC soil that is arable and does not transfer to other land uses as it is important to maintain the most productive land for food. Med and Low ALC significantly increase BL and SNG.	The best soil is protected for AR (ALC 1, 2 and high 3); other soils are more likely to be converted to UR if close to urban areas. Some poor soils will be converted to conifer in from AR or IG or UP	Major determinant factor on AR - the best land is kept or converted to AR; even ALC 3 is protected. Maximizing yield is paramount	High ALC soils kept AR, lower more likely to become BL and SNG through UK. Some will become IG to increase farmland heterogeneity.	High ALC soils kept AR, lower more likely to become BL and SNG through UK.
	Med	*Grades 3a & 3b*						
	Low	*Grades 4 & 5*						

<table>
<tr><th colspan="3"></th><th colspan="6">Rule-base for each Scenario</th></tr>
<tr><th>Criteria</th><th>Attribute</th><th>Definition</th><th>Green and Pleasant Land (GPL)</th><th>Nature at Work (NW)</th><th>World Markets (WM)</th><th>National Security (NS)</th><th>Local Stewardship (LS)</th><th>Go with the Flow (GF)</th></tr>
<tr><td rowspan="2">Change in temperature (UKCIP09)</td><td>Hi – N</td><td>Areas likely to experience a mean change in summer temp of +3 ℃</td><td rowspan="2">Higher temps will affect some land covers types - AR suffers a slight loss with little adaptation capacity (SNG gains here). BL wood also suffers slightly as beech and some oak woods can’t cope with Hi CC in south UK.</td><td rowspan="2">Warmer areas in south of UK sill reduce agric. production slightly although NW loses less AR than others because it is better adapted to CC. Generally speaking, in NW, the diff between Low and HI CC is very small</td><td rowspan="2">Very little adaptation capacity in WM, Hi CC reduces AR area in south (abandoned to SNG or southern hemisphere conifers). Some BL woods suffers and is converted to Conifer.</td><td rowspan="2">Hi CC temps reduce AR production in south east; adaptation capacity (e.g., drought resistant crops) not as prevalent as in NW); switch to conifer or IG in these circumstances.</td><td rowspan="2">Reduces AR but increases native wood planting (not beech or other CC intolerant species). Some IG is converted to SNG because it is more CC tolerant.</td><td rowspan="2">Loss of AR and IG as Hi CC impacts make growing crops more difficult. Some degree of adaptation but not enough to see small transition to either water, BL or SNG</td></tr>
<tr><td>Hi – S</td><td>Areas likely to experience a mean change in summer temp of +4 ℃</td></tr>
<tr><td rowspan="3">Change in precipitation (UKCIP09)</td><td>-40%</td><td>Areas likely to experience a mean change in summer prec. of -40%</td><td rowspan="3">Similar effects as temp on AR and BL (drought compounds heat affect).</td><td rowspan="3">Drier areas in south of UK sill reduce agric. production slightly although NW loses less AR than others because it is better adapted to CC. Generally speaking, in NW, the diff between Low and HI CC is very small</td><td rowspan="3">As for temp.</td><td rowspan="3">As for temp. Drier conditions more likely to result in AR converting to Conifer.</td><td rowspan="3">As for temp.</td><td rowspan="3">Loss of AR and IG as Hi CC impacts make growing crops more difficult.</td></tr>
<tr><td>-30%</td><td>Areas likely to experience a mean change in summer prec. of -30%</td></tr>
<tr><td>-20%</td><td>Areas likely to experience a mean change in summer prec. of -20%</td></tr>
</table>

Table S15.

Mean land use coverage as percentage of the total area of Great Britain under the UK-NEA 2060 scenarios.

Land cover (UK-NEA categories)	Base	GF	NW	GPL	LS	NS	WM
Semi-Natural Grass	17	19	21	23	23	9	14
Mountain, Moor and Heath	16	17	18	17	16	9	13
Enclosed Fields	45	38	29	32	39	44	42
Woodland	12	14	20	16	12	27	11
Freshwater	1	1	2	2	1	1	1
Coastal and sea margins	3	3	3	3	3	3	4
Urban	7	8	7	7	7	7	15
Of which agricultural land (including non-utilized land)	77.8	74.4	68.9	73.0	78.3	62.7	68.8