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Economic Analysis for the UK National Ecosystem Assessment: Synthesis and Scenario Valuation of Changes in Ecosystem Services

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Abstract We combine natural science modelling and valuation techniques to present economic analyses of a variety of land use change scenarios generated for the UK National Ecosystem Assessment. Specifically, the agricultural, greenhouse gas, recreational and urban greenspace impacts of the envisioned land use changes are valued. Particular attention is given to the incorporation of spatial variation in the natural environment and to addressing issues such as biodiversity impacts where reliable values are not available. Results show that the incorporation of ecosystem services and their values within analyses can substantially change decisions.

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1 Introduction

'Ecosystem services' are the contributions which the natural environment makes to human wellbeing. These contributions are both direct, in terms of the sole provision of welfare bearing goods (e.g. animals for those who enjoy watching wildlife), and more often indirect, where ecosystem services combine with human and manufactured capital in the production of goods (e.g. in the case of farming and food production). The so-called 'ecosystem services approach' (Salzman et al. 2001)[1] to decision making seeks to consider all contributions to welfare creation, extending from those derived from conventional human and manufactured capital to include natural capital, and through this determine optimal use of those necessarily limited resources.

The ecosystem services concept and its allied approach to decision making are entirely consistent with economic analysis. The concept is anthropocentric, seeing humans as the arbiters of value; while the decision making approach is a restatement of the often claimed *raison d'être* of environmental economics. However, the emphasis upon integrated assessments inherent in the ecosystem service approach is timely. As we discuss below, practical decision making is often informed by inadequate knowledge regarding the natural environment processes which underpin vast areas of production and welfare creation. This is a serious concern as all decisions concerning natural resources need to be grounded upon a solid natural science foundation, in order to ensure the validity of their conclusions. Given this, the ecosystem services concept can be seen as a welcome reminder that all environmental economic analyses have to be integrated with the natural sciences (and indeed other social sciences). As a logical extension, the "ecosystem services approach" appears nothing short of a requirement for economic decision making to be "done properly"![2]

[1] To our knowledge this is the earliest source of this phrase.
[2] Accepting that, given the limits of economic, social and natural scientific knowledge, no analysis using any methodology can ever be perfect.

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The principles of economic decision making, embodied in practice within cost-benefit analysis (CBA), represents an important and widely used tool for the evaluation of alternatives and the assessment of investments for both environment related and wider projects (see, for example, Boardman et al. 2010). The methodological clarity and common unit comparison attributes of CBA have placed the approach at the centre of many national and international decision making systems (e.g. Hanley 2001; H.M. Treasury 2003; Pearce 1998; Pearce et al. 2006; Hanley and Barbier 2009). However, while the CBA approach is highly attractive in theory, its practice within policy making is the focus of considerable criticism. Some of this criticism surrounds the wider context of decision making (emphasising the need for linkage to other social sciences; see, for example, Vatn and Bromley 1994). But even within the self-imposed confines of resource allocation efficiency, many applications fail to live up to the principles to which CBA aspires (Kopp et al. 1997).

A common deficiency of economic assessments supporting policy is their frequent failure to consider the wider effects of any given investment option. A classic and longstanding example here is the tortuous progress and impacts of the EU Common Agricultural Policy (CAP) which, for decades, has manipulated European farming with the stated intention of raising agricultural production and incomes. Unfortunately, as a result of giving insufficient attention to externalities, the CAP has for much of its time overseen and indeed stimulated increases in diffuse pollution from farms (Howden et al. 2011), accelerated the loss of important habitats for biodiversity (Defra 2010), distorted the price of land and hence reduced the availability of land for other, higher social value, uses (Bateman et al. 2003), etc. Recent attempts to revise the CAP are in substantial part a tacit acceptance of earlier failures to address these issues.

A second common deficiency in practical application of many CBAs is the failure to consider alternative policy options. It is a basic tenet of economic assessment that the opportunity cost of an option, including the net benefits of alternative investments, should be considered within an appraisal. However, policy practice again provides numerous instances of failures to consider such alternatives. For example, transport planning often considers single rather than multiple modes of transport, e.g. by only comparing between different road routes rather than between those and various rail, air and other options (this is a longstanding yet ongoing issue; see DfT 2003; GLA 2006; HS2 Ltd 2010; Castles and Parish 2011).

In analyses of the ecosystem services provided by natural environment resources, these generic problems of not accounting for the wider effects of a specific option and the failure to consider alternative options are supplemented by more specific concerns arising from the application of cost-benefit techniques (MA 2005; Heal et al. 2005; Fisher et al. 2008; TEEB 2009; Pascual et al. 2010; Turner et al. 2010; Bateman et al. 2011). Here, a common practical problem is the failure to incorporate the spatial heterogeneity of such resources within the valuations that form a central element of any CBA. This generates problems for both the validity of those valuations, which fail to capture and reflect variation in the natural environment, and for subsequent policy, which is unable to target scarce resources to their most efficient ends. The common consequence is that assessments, which are allegedly based upon cost-benefit principles, too often yield values which are constant across patently variable areas. A recent example is the Entry Level Stewardship scheme (England Natural 2010a) which offers flat rate payments to farmers across most of England for pro-environmental activities, even though it is clear that the values generated will vary significantly (and in often predictable ways) across the country.

A further, arguably less tractable, problem facing CBAs of environmental resource-based projects arises where the nature of those resources means that certain values are difficult to robustly assess. Perhaps the most high profile example of such difficulties is in respect of biodiversity. Here, use value estimations (e.g. of the pollination services provided by biodiversity[3]) face substantial problems due to gaps in natural science knowledge (Bradbear 2009). Similarly, non-use valuations, which are typically reliant upon stated preference (SP) survey techniques, are beset by challenges associated with respondents having little understanding of, and hence poorly formed preferences for, biodiversity conservation (Morse-Jones et al. 2012). The consequence of using invalid shadow prices for such resources can be severe. Biodiversity provides an archetypal example of stock sustainability concerns, being an asset which, when depleted, can exhibit non-linear threshold effects beyond which stocks can rapidly collapse and exhibit hysteresis,[4], imperfect-reversibility or non-reversibility[5] (Dasgupta and Mäler 2003; Barbier 2011). While a theoretic framework exists for the pricing of such resilience values (Mäler 2008; Mäler et al. 2009), applications remain in their infancy (for a useful exception and illustration of the inherent problems see Walker et al. 2010).

Extending our results presented in Bateman et al. (2013), the present paper seeks to contribute to both the ecosystem service and applied cost benefit literature by addressing all four of the above issues. This objective is tackled through a consistent empirical application; examining the consequences of land use change. In so doing we report the central economic analysis of potential scenarios (defined and discussed subsequently) undertaken for the UK National Ecosystem Assessment (UK-NEA), which in turn provides the major research basis of the UK Natural Environment White Paper (NEWP 2011). The principle contribution of the paper is therefore methodological, seeking to bring together natural science and economic perspectives in a revision of approaches to decision making. However, an important caveat to this work is that it focuses solely upon the flows of services obtained from the natural environment and does not consider changes to the underlying stocks of natural capital. While the incorporation of flow values is clearly a necessary element of a move towards sustainable decision making, it is not of itself sufficient. Incorporation of natural capital stocks is clearly essential. That said, the methodological difficulties of valuing stock resilience, particularly in the face of gaps in our understanding of natural systems, means that accurate flow valuation represents a potential important improvement in decision making.

The first issue to be tackled in such an undertaking is the requirement to consider impacts arising from a given land use scenario, both the market priced effects and various wider externalities (be they either market or non-market). Our analysis of land use change provides a number of important impacts,[6] which together present a range of valuation challenges. As we strive towards a comprehensive approach, the impacts of land use change considered here are as follows[7]:

[3] Note that, although the pollination example is often cited, as here, as a use value, strictly speaking it occupies a lower, more supportive position in the ecosystem service hierarchy. As highlighted by Mace et al. (2012), pollination services are inputs to the production of goods, rather than goods in themselves. Therefore our example is in fact somewhat erroneous although production function methods could be applied to identify the input value of pollination services.

[4] Where levels of depleting pressures (e.g. pollution inputs) need to be reduced well below those which cause the threshold effect before reversibility begins to operate (see Tett et al. 2007).

[5] Typically reversibility refers to natural processes of restoration. However, as a subset of this we can also identify economic non-reversibility where the costs associated with moving to a situation where such restoration can occur and assessed (either correctly or not) as prohibitive (Bateman et al. 2011).

[6] Although, as discussed in our concluding section, we recognise that this assessment is incomplete. As such it is the methodology developed in this paper which, we feel, constitutes its contribution.

[7] While this list is more comprehensive than that considered in many assessments of projects related to land use change (e.g. the CAP example discussed previously), we acknowledge that it is not comprehensive. In particular one substantial omission concerns the impacts of land use change upon the water environment. This particular issue is a focus of attention for the ongoing second phase of the UK-NEA.

- Agricultural food production: Illustrating a market priced good whose output value varies significantly across locations (due to variation in the natural environment) and across time (due to change in policy, prices, climate, technology, etc.);
- Carbon storage and greenhouse gas (GHG) balance: Illustrating a non-market good for which the marginal value is likely to be unaffected by land use changes (because the relatively small area of Great Britain[8] means that even if the entire country was converted into high carbon storage land uses, such as woodland, this would not alter the course of climate change sufficiently to significantly change unit values for further carbon sequestration) and which varies spatially due to natural characteristics (e.g. soil type) as well as land use;
- Open-access recreation: A further non-market good and the value of which varies strongly across locations (being greater near to high populations) and also exhibits diminishing marginal values (while an initial area of recreational land may generate substantial per hectare values these will decline substantially for a second adjacent area) and strong substitution effects (in effect a reflection of the former phenomena);
- Urban greenspace amenity: A non-market good with similar characteristics to open-access recreation;
- Biodiversity: A further, spatially variable, non-market good which, for the reasons discussed previously, poses significant challenges for the measurement of economic values.

While this broader consideration of impacts goes some considerable way towards addressing our first problem, we also need to address the challenge of considering multiple alternative land use scenarios. In order to satisfy this requirement the economic analysis conducted for the UK-NEA was linked with the scenario investigations conducted by social scientists as part of the Assessment (as detailed by Haines-Young 2011 and summarised subsequently).

The third problem of incorporating environmental complexity within our economic analyses is tackled through interdisciplinary research with natural scientists and spatial analysts. This ensures that all of the analyses presented subsequently are based upon spatially explicit natural science models. This in turn reflects the spatial dependence in values which different policy driven land use scenarios would generate, allowing the policy maker to compare different options.

Finally, our empirical example allows us to consider the thorny problem of impacts for which reliable values are not available. Assuming that we accept our prior argument regarding the low validity of SP estimates of willingness to pay for biodiversity, we then investigate the alternative strategy of imposing various sustainability constraints upon our analyses [implicitly designating biodiversity as ‘critical natural capital’ (Turner 1993)]. Potential constraints include avoiding options which entail a reduction in biodiversity, or ruling out any option which results in the loss of a species. By imposing such constraints and hence removing certain options we obtain estimates for the cost-effectiveness of applying these rules. This provides policy makers with decision relevant information regarding the consequences of different options.

The remainder of this paper is organised as follows. In the next section we overview the scenario generation process after which successive sections consider the valuation of each land use change impact: agricultural values (Sect. 3); carbon storage and GHG balance (Sect. 4); open-access recreation (Sect. 5); urban greenspace amenity (Sect. 6); and biodiversity (Sect. 7). All analyses are designed to address the environmental complexity

[8] Note that, while much of the data used in this analysis is collected for all of the UK, data gaps meant that our analysis had to be restricted to Great Britain (i.e. Northern Ireland is omitted from the analysis reported here).

and spatial heterogeneity issues highlighted previously. In Sect. 8 we bring these various analyses together and consider the most favourable scenario. Section 9 concludes.

2 Scenario Generation

The consideration of alternative options required for CBA assessments was provided by undertaking economic analysis of each of the various scenarios generated for land use futures under the UK-NEA (2011). These scenarios considered the consequences for land use of implementing different policy strategies from the present day forward to 2060, the intention being to provide policy makers with an insight into the impacts of these various options and hence guide decisions and policy generation. Given the extreme uncertainties involved in any modelling exercise over such an extended horizon the exercise was conducted by bringing together experts and research users from the fields of natural science, demographics, policy and economics to assess information on past trends and the present situation (quantified in part through the LCM2000 land cover map; CEH 2000) and predict future forecasts under a variety of policy priorities (see details in Haines-Young 2011; Haines-Young et al. 2011). We do not attempt to validate the approach taken to generate these scenarios, nor results obtained (our preference being for a modelled approach, examining the impact of applying specified changes in defined policy levers over shorter periods within which uncertainties are significantly smaller). However, the UK-NEA scenarios do provide a useful test bed for methodological investigation and we prefer to view subsequent results in that light, emphasising the directional trade-offs and rankings of different options.

The expert appraisal approach of Haines-Young et al. (2011) identified six basic scenarios, each describing the consequences of different policy priorities, which they name and describe as follows:

(i) World Markets (WM), where the goal is economic growth and the elimination of trade barriers;
(ii) Nature at Work (NW), where ecosystem services are promoted through the creation of multifunctional landscapes;
(iii) Go with the Flow (GF), where current trends are assumed to continue, and in which current principles and practices are not radically altered;
(iv) Green and Pleasant Land (GPL), where a preservationist attitude to UK ecosystems was taken;
(v) Local Stewardship (LS), where society strives to be sustainable within its immediate surroundings;
(vi) National Security (NS), where the emphasis is placed upon increasing UK production and hence self-sufficiency.

Each of these scenarios was further modified to allow for the impacts of expected climate change under the low and high emission (respectively the SRES B1 and SRES A1FI) projections in the IPCC Special Report on Emissions Scenarios (Nakicenovic and Swart 2000) and subsequently modified under the spatially disaggregated scenarios provided by the United Kingdom Climate Impacts Programme (UKCIP 2009). This provided a high and low emissions variant of each of the scenarios bringing their total number to twelve and providing substantially greater analysis of alternative states than is evidenced under many CBAs. Accordingly, we denote the high and low emission variants of the GF scenario as GF-H and GF-L; repeating this approach to notation for all other scenarios.

The land use implications of each of the twelve scenarios were obtained by taking maps, derived in a geographic information system (GIS), digital maps of current land use, population and related demographic and socio-economic variables and modifying these using the information on trends, forecasts and expert assessment compiled through the process described previously. Summary statistics regarding these drivers and land use changes, disaggregated across the various major habitat types defined by UK-NEA (2011) are presented in Table 1.

To illustrate the differences between scenarios we compare the values in bold, italics, and bold italics from the WM and NW cases (a comparison we repeat throughout this paper) which detail the major land use changes and impacts between these two scenarios. Here, the values in italics indicate substantial increases (be it in land area or population) over the 2010 baseline; values in bold indicate substantial decreases; and values in bold italics show relatively small or no change from the baseline. The WM world envisions a state where regulation of all forms is rolled back resulting in the most substantial increases in population and urban extent of any scenario as environmental and planning restrictions and greenbelt rules are relaxed (as per proposals in H.M. Treasury 2011) and previously protected grasslands and heathlands are lost. By contrast the NW scenario enhances existing regulations ensuring a static urban extent and major increases in grasslands, heathlands and all types of woodland, especially broadleaved as forested areas expand towards the European norm. However, these increases in environmentally important areas result in a significant contraction in farmland. These changes form major determinants of the value changes reported subsequently in this paper.

Wider inspection of Table 1 shows that the scenarios encompass a broad range of losses and gains in major land use types. These land use changes drive the assessments of direct and indirect impacts described in Sects. 3–7 to which we now turn. We discuss the agricultural analysis (Sect. 3) in some detail to demonstrate the spatially sensitive modelling approach used in all subsequent models. These other analyses are generally presented in relatively brief terms with further discussions presented elsewhere in this issue, the exception being our biodiversity models which are presented in detail within this paper.

3 Modelling Change in Agriculture and Its Value

The implications of each scenario option for British agriculture were estimated using the structural model described by Fezzi and Bateman (2011) and Fezzi et al. (2013). These sources provide full details regarding this model but in essence, for each location, the analysis works from a profit function and uses duality theory to derive optimal shares of land use for each of a complete set of agricultural activities. The model is empirically specified to capture both cross-sectional effects (e.g. the influence of location in terms of variation in the physical environment between each area) and temporal change (e.g. variation in prices and policy).

Data for this analysis are drawn from a variety of sources including a panel covering more than 40 years from the Agricultural Census which collects land use shares, livestock numbers and other farm data at a 2×2 km grid (400 ha) basis for the entirety of Great Britain. However, this dataset does not provide profit data. For this reason the empirical focus is restricted to the estimation of land use shares to which farm gross margin (FGM) estimates (obtained from independent sources; see details in Fezzi et al. 2010) can be applied.[9]

[9] FGM is defined as the value of output minus the cost of variable inputs (Nix 2009), i.e. it ignores fixed costs as these tend not to vary over the relatively short periods for which farms make output decisions. Ideally our CBA would employ profit estimates (i.e. including fixed costs) and this is the focus of ongoing work under the second phase of the UK-NEA using data obtained from the UK Farm Business Survey. FGM estimates were obtained from Nix (2009) and Fezzi et al. (2010).

Table 1 Mean land use coverage and population figures for Great Britain: year 2010 baseline and UK NEA 2060 scenarios

Variable	Base	WM-H	WM-L	NW-H	NW-L	GF-H	GF-L	GPL-H	GPL-L	LS-H	LS-L	NS-H	NS-L
Δ population (%)	0	*21*	*21*	***6***	***6***	17	17	2	2	0	0	10	10
Δ real income (%)	0	***2***	***2***	***3***	***3***	1.5	1.5	2	2	0.5	0.5	1	1
% urban	6.7	*14.3*	*14.6*	***6.6***	***6.7***	7.6	8.1	6.7	6.7	6.4	6.5	7.0	6.8
% heathlands	13.8	**11.7**	**11.5**	*16.6*	*15.6*	15.0	14.8	14.6	14.8	14.2	14.1	8.2	8.0
% grasslands	15.9	**13.7**	**13.3**	*20.2*	*20.0*	18.3	17.6	25.3	22.1	21.9	21.5	8.4	8.2
% conifer	5.3	***6.2***	***5.0***	*8.5*	*8.8*	4.2	4.2	3.8	3.8	4.8	4.8	18.9	18.2
% broadleaf	6.3	***5.3***	***5.8***	*10.6*	*10.6*	9.8	9.4	11.1	11.9	7.7	6.7	6.4	7.2
% farmland	43.5	***39.3***	***41.2***	**27.8**	**28.9**	35.5	37.5	29.3	31.5	36.6	38.1	42.0	43.2
% other	8.3	***9.5***	***8.6***	***9.7***	***9.3***	9.5	8.5	9.1	9.1	9.4	8.3	9.1	8.3

Δ = change; variables with names starting "%" refer to percentages of the total area of Great Britain. Values in italics indicate substantial increases over the 2010 baseline; values in bold indicate substantial decreases; values in bold italics show relatively small or no change from the baseline

Farm activity data obtained from the Agricultural Census are combined with annual information on policy (both agricultural and relevant environmental measures), prices, costs and highly detailed data on the geophysical environment (soil characteristics, slope, etc.) and climate. Together this provided over half a million sets of spatially referenced records for the period between 1969 and 2006. Models for optimal land use shares were estimated using techniques which respect the potential for corner solutions (not all farms cultivate all possible crops) and results were tested using out-of-sample, actual versus predicted comparisons.

Scenario analyses, especially for the long horizon required for the UK-NEA, are prone to error if they require that models are extrapolated beyond the range of the data on which they are built. This problem becomes worse when there is great uncertainty regarding the future values of key determinants. For these reasons, the UK-NEA land use scenarios assumed constant real values for agricultural prices and costs throughout the forecast period. While we recognise that this is a strong assumption, the lack of reliable estimates over a future where increases in population, shifts in income distributions and global climate change seem likely to raise demand, yet the potential for substantial technological advance from precision agriculture, genetic modification and adaptation seem likely to cut costs, means that any assumption regarding change in real market prices is open to challenge. In contrast, an assumption of constant real market prices at least provides a useful baseline of predictions from which at very worst the directional change induced by alternative predictions can be inferred. Nevertheless, these substantial uncertainties mean that the absolute value of estimates should be treated with considerable caution and that the relative differences between scenario outcomes are of greater interest.

Despite being the focus of great popular debate, the future path of climate change seems better understood than the economic, demographic and technological issues raised above. Indeed, changes in climate dependent variables (such as growing season precipitation and temperature), predicted by UKCIP (2009) for the UK-NEA scenario period, overlap considerably with the weather variation observed over the more than 40 year length of our panel dataset. These climate predictions are therefore incorporated within our analysis of agricultural land use change. As Fezzi et al. (2013) show, this provides estimates of the shift in the value of farm output due to climate change over that period.

The spatially and temporally explicit nature of the Fezzi et al. analysis provides estimates of the impact of each scenario on a 2 km resolution regular grid across the whole of Great Britain for each year over the UK-NEA time horizon. Full results for agricultural change values across all scenarios and their climate change variants are presented alongside those for all other ecosystem services values in the penultimate section of this paper. Therefore, for ease of comparison, here we restrict our assessment to just the WM and NW scenarios introduced previously. Taken as national totals, and considering the low emission variants initially, the removal of environmental regulation and opening up of greenbelt land under the WM scenario results in an increase in real annual FGM of £490 million per annum. In contrast the reduction in agricultural area implied by the expansion of woodland and other natural environment areas under the NW scenario results in a fall in real annual FGM of some £600 million per annum on average up to 2060.[10] The directional disparity between these two scenarios is hardly surprising but has an immediately important lesson given that the other ecosystem services under assessment all yield non-market values. Ultimately, a single focus upon market priced goods will inevitably favour development over

[10] Our earlier caution regarding the over-interpretation of the absolute value of estimates applies here. Readers are also reminded of the changes in population between scenarios.

environmental conservation or enhancement; in this case the WM scenario generates higher market values than the NW alternative and would be preferred if non-market values are ignored.

While the impacts of climate change are likely to be detrimental at the global level (Schmidhuber and Tubiello 2007), our analysis reveals that, from the narrow perspective of British farming, forecast increases in growing season temperature and reductions in precipitation will actually enhance agriculture, particularly across areas that are presently coldest and wettest. Therefore, in the absence of increased environmental regulation, the WM scenario shows an increase in values under even the low emission variant of climate change. However, given the Fezzi et al. result, it is unsurprising to find that a switch to the high emission variant improves agricultural output values in both scenarios, increasing gains to £1,030 million for the WM case and cutting losses to £130 million per annum for the NW scenario.

4 Modelling Change in Carbon Storage and GHG Balance and Their Value

Land use change almost invariably has implications for GHG emissions. Abson et al. (2013) calculate the implications of UK-NEA scenario land use changes[11] upon three GHGs: methane (CH_4) from livestock (both through the production of manure and enteric fermentation); nitrous oxide (N_2O) from the application of inorganic fertilizers; and carbon dioxide (CO_2) associated with changes in carbon stocks in above and below ground biomass (making allowance for soil type) and from the burning of fossil fuels to power agricultural machinery and production of fertilizers and pesticides. By linking these analyses to the UK-NEA land use scenarios estimates of associated emissions were obtained and converted into CO_2 equivalents.

While the estimation of GHG emissions arising from land use changes is complex, associated uncertainties are dwarfed by the variation in values induced by adopting certain differing approaches to carbon emission valuation given in the literature (Dasgupta 2007; Stern 2007; Nordhaus 2008; DECC 2009). While these differ according to the emission and climate scenarios upon which they are based, approaches to discounting in particular result in very substantial variation in values. Abson et al., approach this problem through a sensitivity analysis across valuation strategies. For simplicity and in the light of the present analysis being focussed upon policy makers, we adopt the UK official non-traded carbon values (DECC 2009) for the results presented in the penultimate section of this paper. However, irrespective of the chosen carbon value function, results for our WM and NW scenario comparison show a consistent directional trade-off between the market value of agricultural products and associated GHG emissions. The increases in agricultural values obtained under the WM scenario are more than offset by the costs of increased GHG; while lower agricultural values under the NW scenario are more than compensated by reductions in the social costs of GHG emissions. In both cases, the ranking of scenarios indicated by market values alone are reversed when even this first non-market externality is included within our analysis. We now move on to consider a further major externality of land use change, recreation.

5 Modelling Change in Open-Access Recreation and Its Value

Outdoor recreation forms one of the main leisure activities enjoyed by the UK population, with more than 2.89 billion visits being made per annum in England alone

[11] Note that, as predicted changes in agricultural land use themselves incorporate expected change in climate variables there is an important feedback element incorporated within this analysis.

(Natural England 2010b). The spatial distribution of these visits is determined in part by: (a) demand pressures such as the distribution of population and their socio-economic and demographic characteristics; and (b) supply issues such as the location of desirable locations, the availability of substitutes and complements and the quality of the transport infrastructure. This means that a given resource located in alternative locations will generate very different numbers of visits and values. In order to address this issue and generate valuations compatible with other assessments in the UK-NEA, Sen et al. (2013) develop a two-step model of open-access recreation visits and associated values.

In the first step of their analysis, Sen et al. build and test a trip generation function which is then used to predict visits from every outset area (aggregations of Census small area records) across Great Britain to a 1×1 km square grid across the nation. This function draws on data from interviews undertaken around the year involving more than 48,000 individuals who together visited over 15,000 unique locations (Natural England 2010b). The incorporation of further, highly detailed, spatially referenced GIS data allows visits to be modelled as a function of the characteristics of the outset location (including population socioeconomic and demographic characteristics, the availability of potential substitutes, etc.), travel time to the destination (taking into account the road network and variation in average speeds) and characteristics of the destination site (including its ecosystem type, the availability of surrounding potential substitutes and complements, etc.).

In the second step of their analysis, Sen et al. develop a trip valuation meta-analysis model to determine the value of predicted visits. This draws upon nearly 300 previous estimates of the value of a recreational visit, examining the determinants of those values including any influence of the ecosystem type of visited sites. This allows generation of an ecosystem-specific value of each visit.

This two stage methodology was applied to each of the UK-NEA land use scenarios. This analysis first provided estimates of the number of visits to each 1 km resolution cell across Great Britain adjusted for location, ecosystem type, road network, population distribution and characteristics and the availability of substitutes and complements. The value per visit for each cell is then estimated by allowing for the mix of ecosystems specified under each scenario. By bringing these together, the spatially and ecosystem sensitive total value of visits is estimated. Differencing from the current land use provides an estimate of the change in recreational values induced by each scenario.

Results of this analysis are presented subsequently. However, an important finding is that, across all scenarios, recreational gains or losses trends are greatest near to population centres. This is hardly surprising given that travel time is a major determinant of visit location, yet it does indicate to decision makers the massive shifts in investment efficiency afforded through a spatially sensitive approach to recreational planning. The efficiency of spending upon improving site quality and facilities is massively modified by the location of the site in question. The importance of location is further borne out in our analysis of urban greenspace values detailed in the following section.

Comparing the WM and NW scenarios revealed pronounced differences in the recreational values generated. The loss of greenbelt land around cities under the WM scenario results in major losses of recreational value in these locations. In contrast the enhancement of the environment emphasised in the NW scenario ensures that there are no losses, with gains being concentrated around areas of high population, reflecting the influence of travel costs in determining recreational choices.

6 Modelling Change in Urban Greenspace Amenity and Its Value

Although covering a relatively small area of the UK, the proximity of urban greenspaces to large populations make them an important source of multiple values including local recreation, pleasant views, cleaner air, etc. (Davies et al. 2011). This value is reflected both in revealed preference hedonic pricing analyses of the determinants of property prices (e.g. Cheshire and Sheppard 1995) and through SP analyses of willingness to pay (e.g. Hanley and Knight 1992). This previous literature is reassessed through the meta-analysis reported by Perino et al. (2013). This allows for the potential of marginal values to vary according to quality by identifying three types of greenspace: formal urban recreation sites; informal urban greenspace; and urban fringe greenspace. Marginal values are then estimated as a function of a variety of determinants including greenspace area, the size and income distribution of the population and location; this latter relationship follows the expected logarithmic distance decay pattern observed in other spatially sensitive valuation studies (e.g. Bateman et al. 2006).

The ultimate objective of this exercise was to transfer the marginal value function to all cities across Great Britain and adjust for the population and income growth and land use change envisioned under each UK-NEA to derive corresponding values for urban greenspace. However, Perino et al. faced a significant data challenge in that detailed spatially explicit information on the size, location and quality (as defined above) of urban greenspace is not available for all UK cities. To address this, an interim step focused on five UK cities representing different categories (based on size and regional location) and for which complete data were available. This complete data requirement included GIS grid-referenced information on greenspace type, size and location (obtained from relevant City councils and national agencies such as Natural England), population and household income distribution (obtained from the UK Census). These data were then manipulated to replicate the changes envisioned under each of the UK-NEA scenarios. By calculating urban greenspace values under each scenario and comparing these to the 2010 baseline the value changes under each scenario were obtained for each city. Extrapolating these estimates to all other cities with a population of 50,000 or more grossed up scenarios values to the country-level.

Perino et al. report estimates for the change in urban greenspace value induced by each UK-NEA scenario and these are discussed subsequently in this paper. However, comparison of the WM and NW scenarios shows that, at least in terms of their impact on urban greenspace, these are polar opposites. While the WM scenario results in major welfare losses arising from the development of greenbelt,[12] the NW world sees an enhancement of greenspace, especially in the greenbelts around British cities, with accompanying welfare gains.

7 Modelling Change in Biodiversity

As noted, given that details of our work on biodiversity modelling are not presented elsewhere in this issue, we provide greater detail on this analysis than accorded to those discussed above.

Biodiversity plays a diversity of roles across the ecosystem service hierarchy (Baumgärtner 2007; Mace et al. 2012), providing supporting and resilience insurance services, direct inputs to the production of a variety of important goods (e.g. soil biodiversity and pollinators both contribute to food production) and both use and non-use values with the former including

[12] Although in a full analysis these would have to be set against the housing benefits generated by such development.

wildlife viewing and the latter generally associated with existence values. The principal challenges to the assessment of use values are often knowledge gaps. For example, although there has been considerable research into the threats and conservation of pollinators (Kremen et al. 2002; Dicks et al. 2010) research into their role as factors of production and hence generators of value is still relatively undeveloped (Bradbear 2009). However, it has been the assessment of non-use values which has attracted the most attention amongst the valuation community.

The lack of observable behaviour means that non-use values are usually only measurable via stated preference techniques. In the case of biodiversity it has long been recognised that reliance upon such approaches will simply reflect preferences for charismatic species (Loomis and White 1996; White et al. 1997, 2001; Christie et al. 2004; Morse-Jones et al. 2012). While in principle this might not be considered a problem (after all CBA seeks to reflect preference based values) nevertheless White et al. (2001) argue that this implies that "attaching too much emphasis to willingness-to-pay studies in nature conservation policy would therefore be at the expense of the less charismatic species and would probably lead to the inappropriate allocation of resources" (p. 165). In essence, allocating resources via preferences may, in the case of biodiversity, lead to outcomes which run contrary to the requirements for sustainable ecosystems. Moreover, we question the likely validity and accuracy of willingness to pay estimates obtained from SP studies of biodiversity non-use values. Such studies typically violate two of the major principles for valid SP design (Carson and Groves 2007). First, the non-use nature of the good in question makes it inherently difficult to ensure that SP valuation questions are consequential and hence incentive compatible. This results in substantial differences between stated and actual payments (as revealed in comparisons between the two; Foster et al. 1997; Pearce 2007). The second issue arises from the previously discussed complexities of the role of biodiversity in ecosystems leading to the possibility that many survey respondents are likely to have poorly formed preferences for non-use goods prior to a valuation survey. This latter factor comes from inconsistent prior knowledge, lack of reflection and unfamiliarity with the task of expressing those preferences in monetary terms;[13] in short, inexperience. Despite attempts to provide information on the goods in question during the course of a valuation survey, a lack of experience has for many years been recognised as a significant reliability problem in SP studies (e.g. Whitehead et al. 1995) with low experience being associated with framing effects and related preference anomalies (Boyle et al. 1993; Bateman et al. 2008).

While stated preference studies may not yield reliable estimates of the value of biodiversity, they do reveal that people do not like to see species become extinct. Indeed, this has long been reflected in policy decisions and legislation nationally and internationally (USDI 1973; CBD 1992; H.M. Government 2007).[14] While this is clearly inferior to a reliable valuation estimate, in the absence of such values a requirement to avoid species extinctions provides a useful constraint to place upon a CBA assessment.[15] Rather than removing the role of economics from this issue, the objective changes to finding the most cost-effective solution

[13] Furthermore, asking survey respondents to express their preferences for biodiversity conservation in a unit (money) which some individuals may see as incommensurate with species existence clearly raises the potential for protest responses (Jorgensen et al. 1999).

[14] Although it should be noted that the last of these three has recently been repealed.

[15] Note that in this paper we adopt a constraint against extinctions irrespective of where they occur in Great Britain. Arguably, individuals might be prepared to countenance a looser requirement that a policy can be sanctioned provided that species are conserved in at least one area within the country. The spatially explicit nature of the methodology developed here is readily suitable to applying such a constraint.

to satisfying this constraint. However, to implement this we need to understand how land use change impacts on biodiversity.

The literature on indicators of biodiversity is long established, extensive and reflects a multitude of opinions (Noss 1990; Lindenmayer et al. 2000; Mace and Baillie 2007; Butchart et al. 2010). Within the UK, however, there are strong arguments for the use of bird related measures as indicators. Birds are one of the most widely observed aspects of UK biodiversity; they are also high in the food chain and are often considered to be good indicators of wider ecosystem health (e.g. Gregory et al. 2005). Birds are more mobile than most other groups, and so will respond to, and reflect, environmental quality at a rather broader scale than mammals or terrestrial insects, for example. This probably makes them better indicators at the landscape scale and less useful locally. However, no single animal or plant group, and especially no small set of variables describing that group, can ever provide a comprehensive summary of all aspects of biodiversity and we do not suggest that they do so. Rather, we note the value that birds have as indicators and make use of the important pragmatic benefit that they are better monitored than any other aspect of UK biodiversity, for example through the annual Breeding Bird Survey (BBS, see Risely et al. 2011) which provides annual monitoring information and basic habitat data at a 1 km square resolution across Britain.

BBS data are used as the basis of two analyses, the first taking a wide view across almost all British bird species, while the second focuses on farmland birds as the group that has suffered the most dramatic declines over the past half century and earlier. In both cases, measures of bird success are modelled as a function of land use as this has a major impact on biodiversity (Michelsen 2008; Polasky et al. 2011). These models are then used to assess the predicted impact on these bird measures as a result of the differing land uses envisioned under each of the UK-NEA scenarios.

7.1 Modelling General Breeding Bird Diversity

A well established general measure of biodiversity is provided by Simpson's Diversity Index (D; Simpson 1949), calculated in each year following Eq. (1).

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

where S = number of bird species recorded at a focal site in that year, P_i = proportion of birds of species i relative to the total number of birds of all species. Our empirical dataset is derived by calculating this diversity measure for some 3,468 BBS 1 km grid squares and 96 bird species across Great Britain. Species data and corresponding diversity measures were then linked through a GIS to the prevalent land use in the survey year.[16]

Investigation of models linking diversity measures to land use revealed a number of quadratic relationships which were incorporated into the specification of the functional form. Estimation proceeded using standard GLM techniques and the best fitting model was identified by inspection of Akaike Information Criterion values (Akaike 1974). Details of the best fitting model are given in Hulme and Siriwardena (2010), however in essence diversity tends to be lower in upland and coastal habitats which are more suited for specialist species whereas the majority of generalist species thrive in lowlands and particularly those with high proportions of inland water.

[16] Regional variations in bird diversity were controlled for by including the 100 km Ordnance Survey grid square in which each BBS square is located within the analysis. A regional bias in survey effort across the UK towards highly populated areas was accounted for by weighting regions with lower survey effort more highly.

The estimated model was then applied to the various land uses specified in each UK-NEA scenario. Changes in the diversity measure were then calculated for each 1 km grid square across Great Britain. Results revealed that, in absolute terms, the change in diversity across scenarios is modest. However, in relative terms the picture is more varied with some scenarios being dominated by losses and others generating substantial gains. Our spatially sensitive approach reveals that all scenarios exhibit strong regional differences reflecting variation in environmental characteristics across areas. Maps of the spatial distribution in diversity changes for the WM and NW scenarios are presented subsequently when we bring together all results. However, in summary the loss of greenbelt areas results in some declines under the WM scenario while the pro-environmental characteristics of the NW scenario results in increases in our general biodiversity measure.

7.2 Modelling Species of Particular Conservation Interest

While our models of simple species diversity provide useful indicators of the general trend of change across scenarios, they do not necessarily reflect the presence or diversity of species of conservation interest. Of particular interest here are farmland birds, both because of their obvious association with land use and changes therein, and because they have been in long term decline. Indeed, changes in farming practices have contributed to a 52% decrease in the farmland bird index for England between 1970 and 2009 (Defra 2010).

Our second biodiversity assessment measure addressed this issue through consideration of a single 'guild'[17] of 19 primarily farmland bird species. Guild richness was measured as the number of these species present in each 10 km grid square in England and Wales, with data from (Gibbons et al. 1993) . Models were developed linking guild richness to data on land use, woodland and urban extent. Percentages of each 10 km grid square utilised for cereals, temporary grassland, coniferous woodland and urban use, along with the mean altitude, were found to be highly significant predictors of measures of the number of farmland bird species present. The analysis was adjusted for spatial autocorrelation using geographically weighted regression techniques (Dugdale 2010a) and then applied across the land use profiles specified under each UK-NEA scenario.

Scenario analysis results for our consideration of farmland birds of particular conservation interest differ noticeably from those obtained from our previous analysis of general species diversity. So, while the extension of agriculture envisioned under the WM scenario results in a reduction in the general (Simpson's) diversity index, it is neutral with regard to change in the number of farmland bird species (although there is considerable local variation). The contrast is extended when we consider the NW scenario which is associated with increases in the general biodiversity measure but results in a mean reduction of one farmland species (although again there is considerable spatial variation).

8 Synthesis

We now bring together the previous analyses to provide a more complete picture of the changes and values associated with each of the UK-NEA scenarios. In essence this is a relatively straightforward task in that the spatially explicit nature of the models developed for each good allows us to simply add the positive or negative values estimated for each good at each location. This imposes a number of implicit assumptions (most obviously linear addi-

[17] Defined as a group in terms of the common foods they consume; in this case primarily seeds and invertebrates.

tivity) most of which we consider in the conclusions to this paper.[18] One issue we highlight here is adjustments to avoid double counting required when adding open-access recreation values Sen et al. (2013) and urban greenspace values together (Perino et al. 2011). 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 Perino et al. analysis).[19] Table 2 summarises results from the various analyses presented previously in this paper, reporting for each good (rows a–e) the value of changes from the baseline induced under the various UK-NEA scenarios (shown in columns with the high and low emission version of each scenario being presented). Row (a) reports changes in the value of our market priced good (agricultural output). Rows (b–d) presents those non-market externalities for which we feel we can estimate defensible monetary values and row (e) sums all of the monetised values, both market and non-market. Rows (f) and (g) provide rankings of these scenarios based upon these various monetised measures. However, as mentioned early in the paper, these results apply to a long forecast period over which underlying assumptions (such as constant real prices across scenarios, specified changes in population between scenarios,[20] etc.) might not hold. Therefore, predicted absolute values should be treated with caution and accorded lower weight than relative differences between scenarios. The remaining rows of the table consider the impact upon these findings of incorporating our non-monetary measures of the biodiversity impacts of each scenario. Rows (h) and (i) provide our two biodiversity impact measures while rows (j–l) show how the incorporation of this information can alter the ranking of scenarios depending upon the decision rule adopted.

The columns of Table 2 are arranged such that our focal WM and NW scenarios, each considered under high then low climate change emission variants, occupy the first four columns of results. As discussed previously, when we only consider the value of market priced agriculture [row (a)] then the deregulated WM scenarios yield gains over the baseline while the NW world results in losses (as before higher climate change resulting in better agricultural values due primarily to the positive impacts on UK food production arising from warmer temperatures). Row (f) provides the ranking of scenarios if we restrict ourselves solely to market prices. Here all scenarios which yield gains are given in italics while those producing losses are shown in bold. The numbers in these cells refer to the ranking of the full 12 scenarios and shows that WM-H gives one of the highest market price outcomes (second only to NS-H) while both the NW scenarios yield very low rankings (with NW-L being the lowest of all). Clearly if, as an unregulated market would dictate, decisions are dominated by priced outputs then the WM scenarios easily outstrip the NW options. This dominance of market priced values over all others reflects not only real world private sector decisions but also the direction of much historic public sector decision making.

Rows (b–d) presents monetary assessments of the non-market values considered in our analyses, starting with the GHG emissions associated with the land use change envisioned under each scenario. Consideration of the WM and NW scenarios reveals a negative correla-

[18] In comparison to interim results given in reports to the UK-NEA project the present analysis adjusts for double counting (see discussion) and utilises and standardises a larger dataset.

[19] To avoid these observations influencing the prediction of visits the trip generation function (Sen et al. this issue) was re-estimated omitting these trips. Full details of this adjustment are given in Sen et al. (2012).

[20] These will of course mean that per capita values need not perfectly follow the pattern of results shown in Table 2. However, the differences due to these changes are relatively minor (as indicated by the figures presented in Table 1).

tion between GHG and agricultural output values[21] with high intensity farming being associated with increased food output but also high emissions (and vice-versa). Similarly, despite growth in population in both scenarios enhancing recreational demand, the WM scenario is associated with negative recreation value outcomes while the protection and enhancement of the natural environment under the NW scenario leads to substantial gains in this respect. Similarly, while a reduction in protection for greenbelt and city parks results in substantial falls in urban greenspace values under the WM world, maintaining such protection in the NW scenario means that increased city populations result in greater values being obtained from these areas.

Row (e) sums all of the preceding values to obtain the monetised value of the changes from baseline induced under each scenario. For the WM and NW scenarios the contrast with values obtained under a mere consideration of market prices is extreme and revealing; the sign of changes is completely reversed. In market price terms the WM scenario dominates its NW counterpart; but when non-market values are also considered this relationship reverses. Furthermore the magnitude of differences alters very substantially, being relatively small for the market price comparison compared to the major differences when all values are considered. This difference is reflected in the rankings presented in rows (f) and (g). Moving from only considering market priced values to including all monetary values can radically change the ranking of options (indeed here this ordering is almost completely reversed); reliance upon market prices alone can lead to major decision failures and consequent serious resource misallocation.

The remainder of Table 2 concerns the incorporation of our non-market assessments of biodiversity impacts into the analysis. Row (h) details the impact upon the species of particular conservation concern (farmland birds) while row (i) indicates relative change in our general (Simpson's) diversity measure. Rows (j–l) report the ranking of scenarios under three increasingly strict constraints. Of these, row (j) shows rankings if we constrain consideration to only those scenarios which both yield positive total monetised values (excluding the WM and NS scenarios) and do not reduce the number of farmland bird species. This has a significant impact on our ranking as it excludes the NW scenario (which reduces farmland in favour of other natural habitats). Such a rule now accords the GPL scenario the highest rank and also provides an estimate of the cost of such a rule, this being the loss incurred by moving away from the NW scenario (which we discuss in the final section of this paper). A stricter rule is investigated in row (k) where we not only require positive total monetary values and no losses of farmland birds, but also impose a requirement that there should be increases in the general biodiversity measure. While this rules out the LS option, the GPL scenario remains the highest ranked. Finally in the last row (l) of the table we recognise that political pressures may make it difficult for governments to introduce measures which actually reduce market priced outputs. Here we impose a rule which supplements all previous requirements with the need for win-win outcomes within both market and non-market good domains. Such a requirement is clearly attractive to decision makers who are concerned about popular characterisations of environmental policy as being anti-economic growth. This final package of constraints rules out all but the GF scenario although it should be noted that this incurs a substantial opportunity cost in terms of a considerable reduction in total monetised values.

[21] Note that this is not always the case across all scenarios with the NS scenario revealing a win-win outcome although its impact upon urban greenspace makes this unattractive overall.

Table 2 Summary impacts for the change from the 2010 baseline to 2060 under each of the UK-NEA scenarios: Great Britain

	Measure	Scenario											
		WM high	WM low	NW high	NW low	GF high	GF low	GPL high	GPL low	LS high	LS low	NS high	NS low
Monetised impacts (*£ millions p.a.; real values*, *£ 2010*)													
(a)	Market agricultural output values[a]	1,030	490	−130	−600	690	260	−30	−340	500	410	1,400	790
(b)	Non-market GHG emissions[b]	−440	−340	230	190	−630	−630	470	470	−790	−920	830	870
(c)	Non-market recreation[c]	−1180	−750	13060	14140	3300	3320	5950	6270	2940	3550	3070	3900
(d)	Non-market urban greenspace[d]	−18,400	−18,400	4,760	4,760	−1,120	−1,120	2,120	2,120	1,750	1,750	−6,940	−6,940
(e)	Total monetised values	−18,990	−19,000	17,920	18,490	2,240	1,830	8,510	8,520	4,400	4,790	−1,640	−1,380
(f)	Rank: market values only	*2*	*6*	10	12	*4*	*8*	9	11	*5*	*7*	*1*	*3*
(g)	Rank: all monetary values	12	11	*2*	*1*	*7*	*8*	*4*	*3*	*6*	*5*	10	9
Including non-monetised impacts[e]													
(h)	Change in farmland bird species[f]	0	0	−1	−1	0	0	0	0	0	0	−1	−1
(i)	Bird diversity (all species)[g]	− − −	+	++	++	++	++	++	++	−	−	++	+++
(j)	Rank: positive welfare values and no farmland bird losses					5	6	2	1	4	3		
(k)	Rank: positive welfare values, no farmland bird losses and general biodiversity gains					3	4	2	1				

Table 2 continued

	Measure	Scenario											
		WM high	WM low	NW high	NW low	GF high	GF low	GPL high	GPL low	LS high	LS low	NS high	NS low
(l)	Rank: positive welfare and market values, no losses of farmland bird species and positive effects on general biodiversity measures					1	2						

Scenarios are as follows: *WM* World Markets; *NW* Nature at Work; *GF* Go with the Flow; *GPL* Green and Pleasant Land; *LS* Local Stewardship; *NS* National Security

All monetary values are in £millions p.a. (in 2010 values)

[a] Change in total GB farm gross margin

[b] Change from baseline year (2010) in annual costs of greenhouse gas (greenhouse gas) emissions from GB terrestrial ecosystems in 2060 under the UK-NEA scenarios (millions £/year); negative values represent increases in annual costs of greenhouse gas emissions. Relative changes were calculated using the change in physical emissions and the 2010 price for carbon. We acknowledge that adopting a 2060 price for carbon would alter these values

[c] Annual value change for all of GB. Adjustments have been made to address double counting with the valuation of urban greenspace (Sen et al. 2012)

[d] Annuity value; negative values indicate losses of urban greenspace amenity value. Using a constant discount rate that is equivalent to the H.M. Treasury (2003) declining discount rate schedule for an annuity with an infinite lifetime ($r = 0.032$)

[e] Note that some commentators prefer to use monetised values for biodiversity. See discussion in UK-NEA Economics chapter (Bateman et al. 2011)

[f] Expected impact on the mean number of species in the seeds and invertebrates guild (including many farmland bird species) present in each 10 km square in England and Wales from 1988 to 2060 (rounded to the nearest whole number). Note that the 2010 baseline has 19 species in this guild (further detail presented in Dugdale 2010b)

[g] Based on relative diversity scores for all bird species with effects coded from largest gains (+ + +) to largest losses (− − −) and further detail presented in Hulme and Siriwardena (2010)

9 Summary, Caveats and Conclusions

The results presented in this paper summarise the application of economic analysis techniques to assessments of the ecosystem service flows generated by a range of land use change scenarios. An interdisciplinary approach is adopted throughout which attempts to incorporate the natural science relationships underpinning the land use changes envisioned in each scenario. The analysis seeks to fully implement CBA rules. These require that we attempt to include all of the major internal (market) and external (non-market) impacts of any land use change. We address the CBA requirement to consider alternative options by considering twelve alternative land use futures under different GHG emission variants. The analysis incorporates the complexity and variability of the natural environment through development and implementation of a spatially explicit approach to modelling each welfare stream. Further, we attempt to ensure a fair assessment by monetising all value streams. Where this cannot be reliably achieved, here in the case of biodiversity, we implement a straightforward constraints-based approach designed to ensure species sustainability.

Three principal results are observed. First, a restricted analysis focussing solely upon market priced goods yields a very different view of which scenario is superior, in contrast to a broader assessment which also considers non-market values. This is hardly surprising but nonetheless provides a highly policy relevant result; social welfare will not be maximised through a decision system which only considers market priced goods and purely financial measures. However, the analysis also shows that a more balanced assessment based upon economic valuations of a wider range of goods is now eminently feasible and sufficiently robust to inform real world decision making. Second, where there are currently limits to the establishment of robust values (as in the case of key biodiversity benefits) a constraints approach allows decision makers to incorporate such goods within analyses. These constraints can be explicitly designed to address concerns regarding threshold levels and hence ensure the sustainability of important natural assets. Third, the methodology developed for this analysis explicitly reveals the spatial variation in values and the different regional responsiveness to alternative policies. This raises the potential for spatially tailored policies designed to enhance the efficiency of resource through the implementation of alternative policies in different areas.

While we contend that this analysis provides a useful contribution to the literature on applied environmental economic analysis, we highlight the assumptions which we implicitly make through adopting such a procedure. Our modelling approach assumes a linear causality between drivers (policy, market forces, technology, cross sectional and temporal environmental change) and consequent land use change and then on to the various goods associated with that change (agricultural food production; GHG emissions; open-access recreation; urban greenspace amenity; and biodiversity). We fully acknowledge that there are a number of undesirable over-simplifications inherent in this approach. Technological change is one such area which might potentially be a major source of error in any assessment of future trends. Similarly, there is a concern as to whether our assessment is extensive enough to provide a basis for robust decision making. Given the complexities of real world environment-economy interactions, truly comprehensive assessment is probably impossible. The obvious rule therefore is to ensure that assessments appraise all of those impacts which might change a decision. While this is inherently difficult to judge a-priori, nevertheless high quality scoping assessments should provide sufficient guidance regarding the key areas to consider. Our own assessment is that there is at least one important omission from our analysis in terms of impacts upon the water environment (arguably a further issue being employment effects). The second phase of research under the UK-NEA seeks to address these omissions.

A further concern regarding our methodology arises from the simplicity of the synthesis analysis. In the preceding section we mention the potential for double counting, however there are a number of further potential challenges to this approach. In particular the linear pathway of effects defined from the drivers of land use change through to its market and external impacts and the flow of values these derive, ignores possibilities of non-linearities, thresholds and feedback effects. Experience shows that these issues may be difficult to detect a-priori and we feel that this is worthy of further consideration. A further issue concerns the treatment of uncertainty within the synthesis analysis. Uncertainty arises both within and particularly between models and to date we have taken no account of this latter factor; an issue of some concern given the obvious potential for error propagation in any form of chained analysis where the outputs of one model become the inputs of another. Ideally uncertainty should be built into all aspects of the synthesis and its consequence examined (through, for example, the use of Monte Carlo analysis) such that we move away from the simple point estimates of the final results and towards estimating distributions of results. Again these issues are central to the ongoing work being conducted under the second phase of the UK-NEA.

A related concern regarding the synthesis analysis is the omission of the institutional costs associated with moving to a more targeted approach to policy making and its application. Clearly a uniform, 'one-size-fits-all' approach to policy making is likely to involve lower institutional costs than does a differentiated, targeted strategy and so the latter costs need to be incorporated within our assessments. That said, the estimated benefits of targeted decision making substantially exceed the entire annual budget (current and capital expenditure) of £2.6 billion for the UK Department for Environment, Food and Rural Affairs (Defra) (H.M. Treasury 2011), suggesting that such an approach would yield strongly positive net benefits, even when institutional costs are considered. Elsewhere we show that combining spatial targeting with an approach which tailors policies to the characteristics of each area can generate substantial gains over uniform policies (Bateman et al. 2013).

One last concern arises from the imposition of constraints upon any CBA; in this instance because of our inability to accurately determine biodiversity values. Arguably this provides a useful insight into cost-effective conservation of current levels of biodiversity. However, a closer examination of the results given in Table 2 gives some cause for concern. In the absence of the sustainability constraint the highest level of all monetised benefits is provided by the NW scenario. However, while the latter achieves this through enhancing valuable natural environments such as heathlands and natural grasslands, the reduction in agriculture required to achieve these changes also results in the reduction of one species of farmland bird. Imposing a constraint to avoid the latter loss means that the next best alternative is the GPL scenario; an option which reduces social gains by over £9,000 million p.a. The problem with this approach is that it is likely that such losses becomes a matter for decision maker judgement; and the track record in this respect suggests that such judgement might well come down on the side of rejecting such constraint. Within the UK the most high profile precedent concerns the case of Twyford Down, a Site of Special Scientific Interest and designated part of the East Hampshire Area of Outstanding Natural Beauty which, despite protests, was in 1994 severed in two by a major motorway (the largest category of road in the UK). In this case, and again in the absence of reliable monetary estimates of the value of conserving the area, a proposal to tunnel the road under the Down was rejected on the grounds that its cost (roughly £400 million at current prices) was too high given the conservation benefits it would yield (POST 1997). Of course our current example concerns the loss of a species which arguably is greater than a partial reduction in one conservation area. Nevertheless the problem becomes obvious; a constraint is, in reality, only binding if society (or rather its decision makers) is prepared to make it so.

Given the above caveats we do not claim that the analysis presented in this paper is definitive. However, we would contend that it demonstrates the potential for incorporating natural science information within economic analyses in a manner which allows more thorough and spatially explicit CBA assessments of decision possibilities. Certainly this potential appears to have been recognised by decision makers with the UK Environment Secretary, Caroline Spelman, stating that "The UK National Ecosystem Assessment is a vital step forward in our ability to understand the true value of nature and how to sustain the benefits it gives us....The findings of this assessment have played a big part in shaping our forthcoming Natural Environment White Paper (NEWP) that will help us revitalise our towns and countryside" (Defra 2011). Indeed the NEWP (2011) has placed recognition of the value of ecosystem services and hence environmental economic analysis at the centre of UK environment policy.

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