

Chapter 13

Ecosystem Services Assessment and Benefit Transfer

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Abstract Ecosystem service assessments aim to integrate the natural environment into decision-making by developing linked biophysical and economic models that demonstrate how changes in the environment affect human welfare. When these analyses inform national level, strategic choices, large-scale analyses are required. Such assessments, embracing multiple ecosystem services, will often rely on the transfer of either economic or biophysical models, or both. This chapter discusses the main concepts of ecosystem service assessments and illustrates the conceptual framework with examples from the UK National Ecosystem Assessment. An analysis of the recreational and carbon values arising from land use changes shows how differences in ecological, socioeconomic or climatic factors result in high spatial heterogeneity in ecosystem services and how this variation can be incorporated within transfer values.

Keywords Ecosystem services · Benefit transfer · Value mapping

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13.1 Introduction

Despite a nearly universal recognition that human society relies on nature for basic needs such as food and fresh air, numerous assessments have shown that management of the natural environment has not been sufficiently integrated to foster sustainable development (e.g., Millennium Ecosystem Assessment (MEA) 2005). A more integrated approach is necessary to ensure that national and local planning agencies maintain the environment such that it can continue to provide benefits to society. The so called “ecosystem services approach” (MEA 2005) seeks to address this need. This approach requires that agencies consider nature and its services at all stages in the decision-making process. At the core of ecosystem service assessments is the objective of incorporating a holistic consideration of ecosystem services and their value into decision-making (e.g., Department for Environment, Food and Rural Affairs (DEFRA) 2007).

The incorporation of sustainable development goals at national levels has propelled interest in large scale assessments of ecosystem services. The increased demand for quantification and valuation of the benefits that nature provides to society has driven environmental economists and social scientists to seek greater cooperation with natural scientists, and vice versa. Integration can also be sought in valuation studies, including benefit transfer, where biophysical values can be considered explicitly. An inclusive, multidisciplinary approach is imperative when multiple ecosystems and their services are considered. Linking biophysical analyses with socioeconomic valuation is vital for assessing situations where tradeoffs and synergies between ecosystem services may occur in the face of changes in ecosystems and biodiversity (The Economics of Ecosystems and Biodiversity (TEEB) 2010).

Inevitably, large scale assessments cannot rely on primary data collection alone. The transfer of models across space is one of the fundamental but challenging building blocks of the methodology of ecosystem assessment. Benefit transfer methods are hence likely to play an important role in ecosystem assessments, and these transfers may involve both biophysical and economic models.

The UK National Ecosystem Assessment (UK NEA) study of 2011 shows the wide scope that such large scale assessments can cover (and the resources they require; in the UK NEA, over five hundred natural scientists worked together intensively with more than fifty social scientists). In this chapter, we use the UK NEA as a case study to demonstrate how large-scale ecosystem assessments can use benefit transfer to provide policy-relevant information for sustainable development decision making. Scenario development and spatial analysis form the basis of ecosystem services assessment, recognizing that ecosystem services are highly context-specific and change over space and time.

Benefit transfer techniques play a central role in the UK NEA. Rather than a small area or local site, the primary “study site” is here represented by countries of the UK. The method applied takes data from different countries and relates them to local characteristics so as to build models which can then be applied to every

location within these countries, as well as to adjacent countries in the UK for which no primary data are available. So in this case the “policy site” is not an independent site but a wider geographical area for which ecosystem services and local variables are likely to be similar. Moreover, the UK NEA acknowledges that ecosystem models are the drivers of values. Therefore, spatially explicit models are developed for both biophysical ecosystem services and their economic value, and transferred across space and time.

In this chapter, the use of benefit transfer approaches to value two particular ecosystem services, carbon sequestration and open-access recreation, will be discussed. These combine biophysical and economic models and make use of different spatial transfer methods. The carbon example shows how biophysical models can be transferred across space to predict CO₂ emission levels in multiple locations, to which economic values can be assigned. The results show spatial variation in the final benefit maps, even though carbon has a fixed price per quantity unit which is unlikely to exhibit diminishing marginal values across the range of provisional levels considered (Bateman et al. 2011). The recreation example demonstrates that both economic and biophysical outputs can vary across space: both visitation numbers and values per trip vary with different habitats. Furthermore, substitution effects across different recreation sites (of both the same and different ecosystem types) need to be incorporated to allow for the likelihood of diminishing marginal values (Bateman et al. 2011). We start this chapter with an introduction of the concepts of ecosystem service assessment and the role of mapping and scenarios. This summary of ecosystem services assessment sheds light on the role of economic valuation of non-market goods. Subsequently, the section on large scale assessments points at the complexity of using primary economic methods (e.g., contingent valuation, travel cost) and introduces the approach of developing spatially explicit, transferable functions for assessing both ecosystem services and their monetary values. The function transfer approach is used to understand and maintain the biophysical link between spatially explicit characteristics of the natural environment and human systems as these jointly determine ecosystem service values. Two examples from the large-scale ecosystem assessment of the UK NEA describe the transfer of ecosystem service values across space. This source of analyses is retained for a final scenario mapping section which provides a formal illustration of transfers across time under alternative policy scenarios.

13.2 Ecosystem Service Assessment

13.2.1 Framework

Working with the framework of ecosystem service assessment requires a stronger focus on natural sciences than is common among environmental economists. One of the key messages of The Economics of Ecosystems and Biodiversity (TEEB) study

is that “any ecosystem assessment should first aim to determine the service delivery in biophysical terms, to provide solid ecological underpinning to the economic valuation or measurement with alternative metrics” (2010, p. 3). This integration allows better accounting for ecosystem functioning and interrelations between ecosystem services in economic analysis, and provides vital information for evaluating the sustainability of systems (Bateman et al. 2011).

Various frameworks, definitions and terminology have been put forward to describe how ecosystems can produce services and goods that are of human benefit, through ecosystem processes and functions as additional capital inputs (e.g. Millennium Ecosystem Assessment 2005; The Economics of Ecosystems and Biodiversity 2010; Bateman et al. 2011). In the UK National Ecosystem Assessment study (2011, p. 12), an ecosystem is defined as “a complex where interactions among the biotic (living) and abiotic (non-living) components of that unit determine its properties and set limits to the types of processes that take place there.” Thus an ecosystem can be regarded as a “stock” of ecosystem assets, which generates a “flow” of ecosystem services (Mäler et al. 2008; Barbier 2009). Conceptualizing ecosystem services using stock and flow notions highlights the importance of the sustainable use of renewable and non-renewable resources. For the former, an optimal harvesting of their services is the key point, whereas for the latter the attention is on optimal depletion and reinvestment (Barbier 2011; Bateman et al. 2011).

The different biological, physical and chemical components of an ecosystem and their interactions determine the functioning of the ecosystem processes from which ecosystem services result (see Fig. 13.1, which expands upon the ecosystem services framework of UK NEA 2011). Fisher and Turner (2008) define ecosystem services as the aspects of ecosystems utilized (actively or passively) to produce human well-being. These services can be subdivided into final services, which directly contribute to the goods that are valued by people, and intermediate services, which underpin the final services. In many cases, these final ecosystem services have to be combined with other resource inputs, such as manufactured or human capital, to generate valuable goods.

In the framework of Fig. 13.1, “goods” can be tangible or non-tangible, and marketed or non-marketed; their main characteristic is that they are at least partly produced by an ecosystem. Most of these goods can be given a [monetary] value to reflect the well-being they provide, using economic valuation methods.¹ The current status of ecosystems and their associated human well-being effects depends on factors related to the demographic, economic and environmental situation as well as the management regime in place. The future development of ecosystem service delivery depends on changes in these drivers. In the UK NEA, alternative policy

¹Although many ecosystem service assessment frameworks highlight intrinsic and community or shared social values, this chapter will focus on the benefits that can be assessed at the level of the individual and expressed in monetary terms. Bateman et al. (2011) discusses cases where reliable monetary values might not be available.

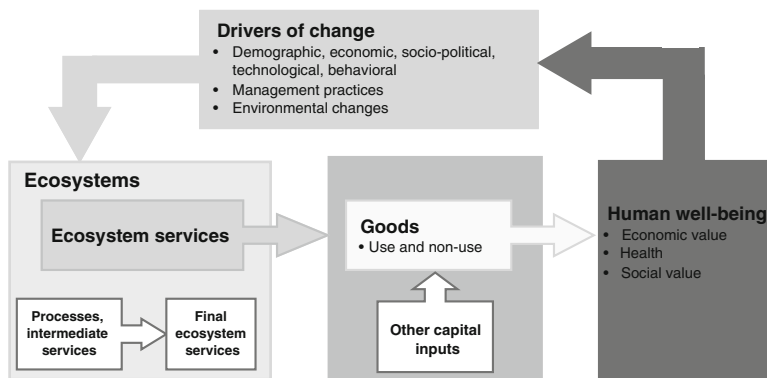


Fig. 13.1 Ecosystem assessment framework

scenarios were developed to demonstrate how human well-being would be affected by different political regimes.

Ecosystem valuation is meaningful only when “marginal changes” are considered. The concept of “marginal changes” is valid if the ecosystem is operating above some safe minimum standard (SMS) which guarantees its functional integrity (Fisher et al. 2008; Turner et al. 2010). Unfortunately, this SMS point is often unknown and the complexity and non-linearity of the interrelated ecosystem processes is often poorly understood. One aspect of ecosystem complexity is that ecosystems may respond to environmental changes in an unpredictable or irreversible way, shifting from one state to another when the SMS threshold is passed. The economics of thresholds are complex and are beyond the purview of this chapter (Johnston and Sutinen 1996; Arrow et al. 2003; Polasky et al. 2011). In addition, the final estimates of the economic benefits of ecosystem services will have confidence intervals that affected not only by uncertainties in economic models, but also variation and model uncertainty in the biophysical assessment of the provision levels of ecosystem services. As a result, because of limited scientific knowledge, wide confidence intervals must be placed on estimates of the change in ecosystem services and physical quantities arising from a change.

The interdependencies between different ecosystem services require careful attention in ecosystem assessments to avoid double counting. Double counting can occur when (a) a service is valued as an intermediate or supporting service as well as a final service, and both values are included in the cost-benefit analysis; or (b) two competing services are valued separately and included in a cost-benefit analysis (Turner et al. 2010). The risk of case (a) can be reduced by using clear definitions of ecosystem services, and focusing on final ecosystem services for valuation (De Groot 2006; Hein et al. 2006). Sufficient understanding of the different processes, functions and services of the ecosystem and their interactions is paramount. The latter case (b) refers to situations where these services cannot be delivered in one “bundle” (TEEB 2010) and have to be traded off. To avoid double counting and

include only final services in the economic valuation, the UK NEA developed matrices of services to qualitatively assess the correlations between services, with +, – or 0 for positive, negative or no correlations, respectively.

13.2.2 Large Scale Assessments

Ecological functioning and economic values are context-, space-, and time-specific, and therefore ecosystem assessments should be spatially and temporally explicit at a scale that is meaningful for decision making (TEEB 2010). Benefits vary across space, along with biophysical characteristics (e.g., the type of land cover, climate, altitude) and socioeconomic factors (e.g., population density and distribution, road network, income, land ownership, land use). The same good can generate very different benefits depending on its context and timing of delivery. Mapping and quantifying the linkages between primary processes, intermediate and final ecosystem services through to beneficial goods is therefore a core component of an ecosystem assessment (Bateman et al. 2011; Fisher et al. 2011).

Fisher et al. (2011) present a spatially explicit ecosystem services approach that is based on creating various model-based maps of stocks, production, flow, beneficiaries, benefits and costs (see Table 13.1).

The first step of the assessment (Fig. 13.2) is an inventory of the ecosystems and their contexts, including those factors that drive environmental change (cf. Fisher et al. 2011). The layer of service production shows what the ecosystem provides and maps the service at the location of production in biophysical units. The related service flow map demonstrates where these services are flowing and can be enjoyed, reflecting that not all services are “consumed” where they are “produced.” For instance, in the case of water quality, one area may collect and purify water while another consumes it. Since services generate value only when there are people to enjoy their benefits, a separate layer highlights the relevant stakeholders and their socio-demographic characteristics from the population. Finally, the biophysical and socioeconomic components are brought together in an economic

Table 13.1 Overview of changes in population, income and land cover under the UK NEA WM and GPL scenarios

	Baseline	GPL	WM
Change in population (%)	0.0	2.0	21.0
Change in real income (%)	0.0	2.0	2.0
Urban (%)	6.7	6.7	14.3
Heathlands (%)	13.8	14.6	11.7
Grasslands (%)	15.9	25.3	13.7
Conifer (%)	5.3	3.8	6.2
Broadleaf (%)	6.3	11.1	5.3
Farmland (%)	43.5	29.3	39.3
Other (%)	8.3	9.1	9.5

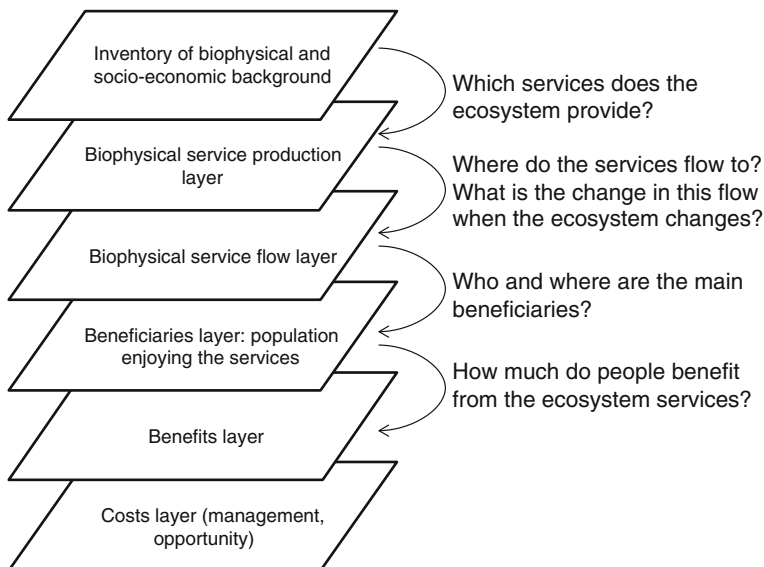


Fig. 13.2 Spatially explicit ecosystem services assessment

valuation exercise to produce maps of benefits as well as costs (e.g. management and opportunity costs).

Mapping the full set of ecosystem services in large-scale, nation-wide assessments requires spatially explicit, transferable models (Bateman et al. 2011). The larger the scale of the assessment, the less likely it is that primary data are available for each area and services of interest. Spatially explicit, transferable models recognize that ecosystem services are context-specific and can be used to transfer analyses to the scale of interest (Brander et al. 2011).

Working from the various information types seen in Fig. 13.2, the modeling framework of an ecosystem assessment is split into:

- (a) Biophysical modeling, in which ecosystem services expressed in biophysical units are linked to explanatory factors reflecting the spatial and temporal context; and
- (b) Economic modeling, in which monetary values per unit of biophysical output are scaled using both spatial and temporal context variables.

These models use input data of both biophysical and socioeconomic processes and characteristics, typically based on geographical information systems (GIS) layers. The two sets of models are combined to assess and map the overall welfare impacts across wider areas and different scales, as required by the level of decision-making. This integration of the economic and ecological models forms an essential part of ecosystem assessment and a departure from environmental economic analyses in which biophysical service provision is taken as a given and ecological heterogeneity largely ignored.

Understanding the spatial distribution of the costs and benefits of changes in land management allows policymakers to spatially target those sites and actions which yield positive net social welfare changes. Overlaying cost and benefit maps also allows for the identification of individual winners and losers, which, combined with socioeconomic information, is important for policymaking as it informs distributional considerations.

One main strength of mapping ecosystem services is its capacity to support scenario analysis. Scenario analysis is a key component of ecosystem assessments. Scenarios reflect hypothetical but internally consistent and (biophysically) plausible story lines with feasible outcomes in terms of land use changes. They are, however, different from forecasts based on time-series analysis. Scenario analysis recognizes that costs and benefits are best measured as a function of changes between counterfactual scenarios (marginal values). Differences between alternative scenarios, resulting from different policy decisions, are often more informative for policymaking than total value estimates (Swetnam et al. 2011). By examining the tradeoffs of alternative future states of the world, the option that offers the highest net benefits to society can be selected and its distributional impacts evaluated. Maps enable the valuation and comparison of benefits and costs related to changes in land use and ecosystem management under alternative options or scenarios in a spatially explicit manner.

The scenario story lines are based on possible changes in the drivers of environmental change, including knowledge and technology, legislation at national and international levels, policies, institutions, governance, societal behavior, markets and incentives, and industrial practice (see Fig. 13.1). Within the spatial analysis framework, these story lines are translated into changes in biophysical outcomes that can be mapped, including future land use maps and population predictions. The models of current behavior are then combined with the future input data layers to predict future ecosystem service benefits, assuming that the functional relationships and parameter estimates remain constant over time.

13.3 Examples from the UK National Ecosystem Assessment

13.3.1 The UK National Ecosystem Assessment

The UK NEA was initiated by the UK government after the publication of the Millennium Ecosystem Assessment study in 2005. It provides a unique synthesis of current knowledge regarding UK ecosystems and explores the inter-linkages among habitats, ecosystem services and biodiversity. This peer-reviewed showcase of the state and value of the UK's natural environmental assets supports decision makers in developing policies that correspond with an ecosystem services approach. The UK NEA makes no claim to be a comprehensive assessment of all services; in part

it highlights knowledge gaps regarding habitats, ecosystems and valuation. The analyses reflect a joint collaboration between scientists from the natural and social sciences, while the wider UK NEA process involved not only academics, but also government, nongovernmental organizations (NGOs) and private sector institutions.

The UK NEA aims to assess major ecosystem services in a spatially explicit manner. The results and outcomes of various scenarios are presented as maps, showing how Scotland, North Ireland, Wales and England might fare in the future under various policy directions and climate-change scenarios. The maps are created by transferring spatially explicit models, estimated by using data from representative areas of the UK over the entire country.

Two examples drawn from the UK NEA are presented. The goal is to illustrate ways in which benefit transfers are combined with large-scale ecosystem assessments to predict future outcomes for human welfare under alternative biophysical and policy scenarios. The first example refers to services provided by agricultural land use to greenhouse gases (GHGs) emissions and the second to open-access recreation sites. These two services were chosen because they represent contrasting examples: whereas GHG values vary only across space because of biophysical differences across land uses, recreational benefits are spatially heterogeneous because both ecological and economic factors affect their economic value.

13.3.2 Greenhouse Gas Emissions

The UK government has set out its GHG emission strategy in the Climate Change Act 2008. The Act aims to reduce carbon emissions by at least 34 % by 2020 and at least 80 % by 2050. The implementation of this Act requires a broad understanding of the terrestrial carbon cycle and its determinants.

The carbon cycle from ecosystems is determined by carbon flow (fluxes) and changes in stocks. Carbon fluxes are determined by carbon emission/sequestration due to changes in carbon stocks by direct emissions from human activities and the natural environment. The carbon stock is the quantity of carbon stored in live biomass, above and below soil, and in the soil as organic carbon, which is primarily composed of various bacteria and fungi. The ability of soils to store carbon depends on many factors such as type of soil, land use, topography, hydrology and climatic factors.

Agricultural management is one of the human activities with a considerable impact on greenhouse gases. Agricultural land uses affect carbon storage, whereas livestock numbers and agricultural activities (e.g., tillage, harvesting) influence carbon fluxes through terrestrial GHG emissions, including methane and nitrous oxide. Agriculture accounts for approximately 77 % of land and roughly 9 % of the UK's net GHG emissions (Thomas et al. 2011). Therefore, sustainable land management and reducing on-farm emissions is part of the implementation plan supporting the UK Climate Change Act.

Forest, woodlands and other (semi-) natural habitats are net sinks for GHG regulation. Over the last 50 years, there has been a slight increase in carbon storage in woodlands in the UK, due to peat land restoration and extensive tree planting projects (Dyson et al. 2009). One of the main objectives of decisions related to future woodland management may be to set land aside for long-term carbon sinks. However, there is no incentive mechanism in place to internalize agricultural GHG emissions into farmers' land use choices. Farmers currently base their land use decisions mainly on agricultural profit maximization objectives. Since farmers influence GHG emissions through agricultural land management and conversion, the inclusion of these land use choices and land-management activities is an integral part of carbon assessment. To capture the spatial variation in the contribution of agricultural activities to climate change through agricultural activities, and therefore GHG changes, a spatially explicit model of farmers' land use decisions is required that reflects the effect of differences in climate and soil conditions across Great Britain (GB). Further, a benefit transfer approach is necessary to assess the impact of farmers' decisions on GHG regulation for the entire UK (including North Ireland) and for valuing future climate and political scenarios. Therefore, the UK NEA GHG case study presents a benefit transfer example of GHG regulation services across space and time. In the UK NEA, the biophysical analysis of GHG emissions consists of a spatially explicit agricultural land use model (Fezzi and Bateman 2011) combined with an assessment of carbon stocks and flows across various habitats and land use types (Abson et al. 2010). Figure 13.3 presents a schematic overview of the model used for transferring the biophysical and economic GHG values across space and time.

The agricultural land use model reflects how climate and land use types influence farmers' profits and therefore the way they use their land. It disaggregates the broad category of "agricultural land" in the GB land cover map into various types of land uses related to different crops and livestock. The land use model considers farmers' outputs produced in GB over the last 40 years, prices of those outputs,

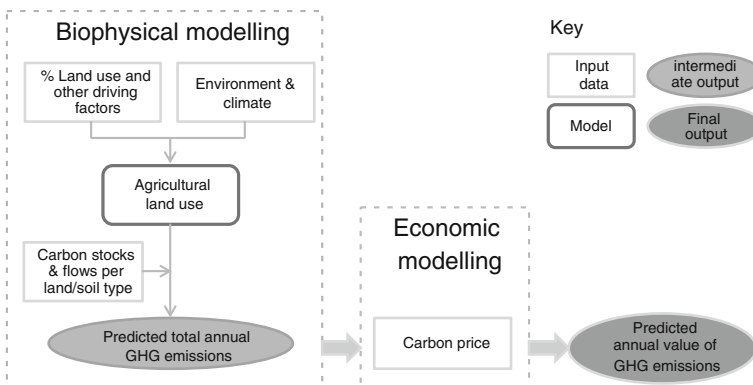


Fig. 13.3 Economic assessment of GHG emissions

costs of inputs, policy and market drivers of farmers' choices and a set of environmental and climate variables. All of these input variables in the land use model are collected at a detailed spatial resolution (2 km grid square) for GB and only partial information are available for North Ireland. The model describes how changes in these factors result in farms allocating different shares of their available land to different activities. The main land categories analyzed are cereals, oilseed rape, root crops, temporary grassland rough grazing and a bundle of other land uses, such as on-farm woodland. The numbers of livestock per grid square are also included in the model to account for direct GHG emissions from agricultural activities. The different land uses as predicted by the land use model are, in turn, associated with different levels of carbon stocks and fluxes. The total area considered in this analysis (farmland forest and woodland) accounts for approximately 88 % of GB terrestrial area, representing the majority of GB land (Abson et al. 2010). Three major categories of GHG emissions were considered and converted to CO₂ equivalents:

1. Direct and indirect emissions from land use and management;
2. Annual flows of carbon from soils due to land use changes;
3. Emissions and accumulations of carbon in terrestrial vegetative biomass.

The carbon fluxes are determined by estimating the emission levels from typical farming practices for different agricultural crops and the manure and enteric fermentation (animal digestive process responsible for methane emissions) due to livestock density. For each crop, a typical farming practice is assumed and the relative CO₂ emissions are calculated for different land shares. Further, changes in land uses are associated with annual GHG fluxes. For example, a conversion of arable land to permanent grass was estimated to produce an average accumulation of soil organic carbon (SOC), whereas a change of rough grazing to permanent grass will imply a loss of SOC. Accumulation of SOC continues until a new equilibrium state is reached; this equilibrium state varies by land use type. The time period over which the change occurs is taken into account in calculating the change in biomass stocks and the relative GHG annual fluxes. Essentially, combining literature findings and case specific assumptions a mean benefit transfer is conducted for determining the GHG quantity in each soil type. More details are available in Abson et al. (2010); the following briefly explains the approach.

The stock of carbon is determined as a function of land uses and woodland density, with estimates of these GHG categories drawn from the literature (e.g., Milne et al. 2001; Bradley et al. 2005; Worrall et al. 2009) and combined with assumptions detailed in Abson et al. (2010). For SOC, a distinction is made between organic and non-organic soils. Abson et al. (2010) assume that peat soils under rough grazing (organic soil) have an average soil carbon density of 1200 tC/ha and that non-peat soils have a density of 224 tC/ha. Furthermore, SOC varies across regions in the UK with, for example, the average SOC (up to 1 m) value in England being 132.6 tC/ha and 212.2 tC/ha in Scotland (Bradley et al. 2005). Other estimates are provided for Wales and North Ireland. Average SOC levels per land

use are adjusted for different regions by assuming that the SOC per land use is proportional to the regional average. For example, crops lands are assumed to have 84 % of the non-peat SOC of the same soils under temporary and grassland (Cruickshank et al. 1998). This implies that given that in England the average SOC estimates for temporary and grassland for non-peat soils is 133 tC/ha, the resulting average SOC for crops is 111 tC/ha. Similar estimates have been produced for the other regions.

The biomass stocks in different land use types are based on estimates from the literature (see Abson et al. 2010 for details), whereas for terrestrial carbon storage in woodlands, estimates given by Thomson et al. (2007) are used. The sum of the SOC and vegetative biomass per each 2 km grid cells represents the UK distribution of carbon stock in terrestrial ecosystems.

The annual GHG emissions from agricultural land in each grid are the sum of the annual soil organic carbon and biomass carbon (crops and woodland) fluxes and the estimated emissions from agricultural activities, where the spatial variation is dictated by the predicted land use shares. To check the validity of the biophysical model results, different out-of-sample tests have been conducted for the land use model (Fezzi and Bateman 2011) and a comprehensive literature review of estimates of carbon stocks and fluxes was carried out. Although satisfactory, the comparison of GHG estimates is less robust than the out-of-sample tests for the land use model, and the mean benefit transfer for GHG quantity could introduce biases into the biophysical model which cannot be tested easily. The results show that the estimated annual GHG emissions from terrestrial ecosystems are roughly 26 million tons of carbon dioxide equivalent in the year 2000. These emission levels are highly heterogeneous across GB, which demonstrates the sensitivity of ecosystem services quantification to spatial and contextual characteristics. Figure 13.4 shows that areas with high impact agricultural practices are mainly in the western coastline of the country.

Based on these underlying biophysical data, a benefit transfer approach is used to predict GHG emissions and resulting economic costs for North Ireland.² As in a standard transfer exercise, England, Wales and Scotland represent the “study sites” and North Ireland the “policy site” for which the annual value of GHG emissions must be predicted.

The biophysical relationship between land uses and carbon stocks and flows is captured in the biophysical model. The annual quantity of GHG emissions in North Ireland is predicted by (a) estimating the land use shares in North Ireland using the land use model combined with secondary data of agricultural drivers and environmental and climatic variables for the policy area, and (b) applying stocks and flows carbon estimates to these values. For step (a), the functional transfer approach is likely to produce low error given that policy and study site present similar farm

²It is worth observing that this exercise aims at transferring biophysical values and not benefits. Therefore under the UK NEA, the correct term for the methodology used would be “value transfer” and not “benefit transfer.”

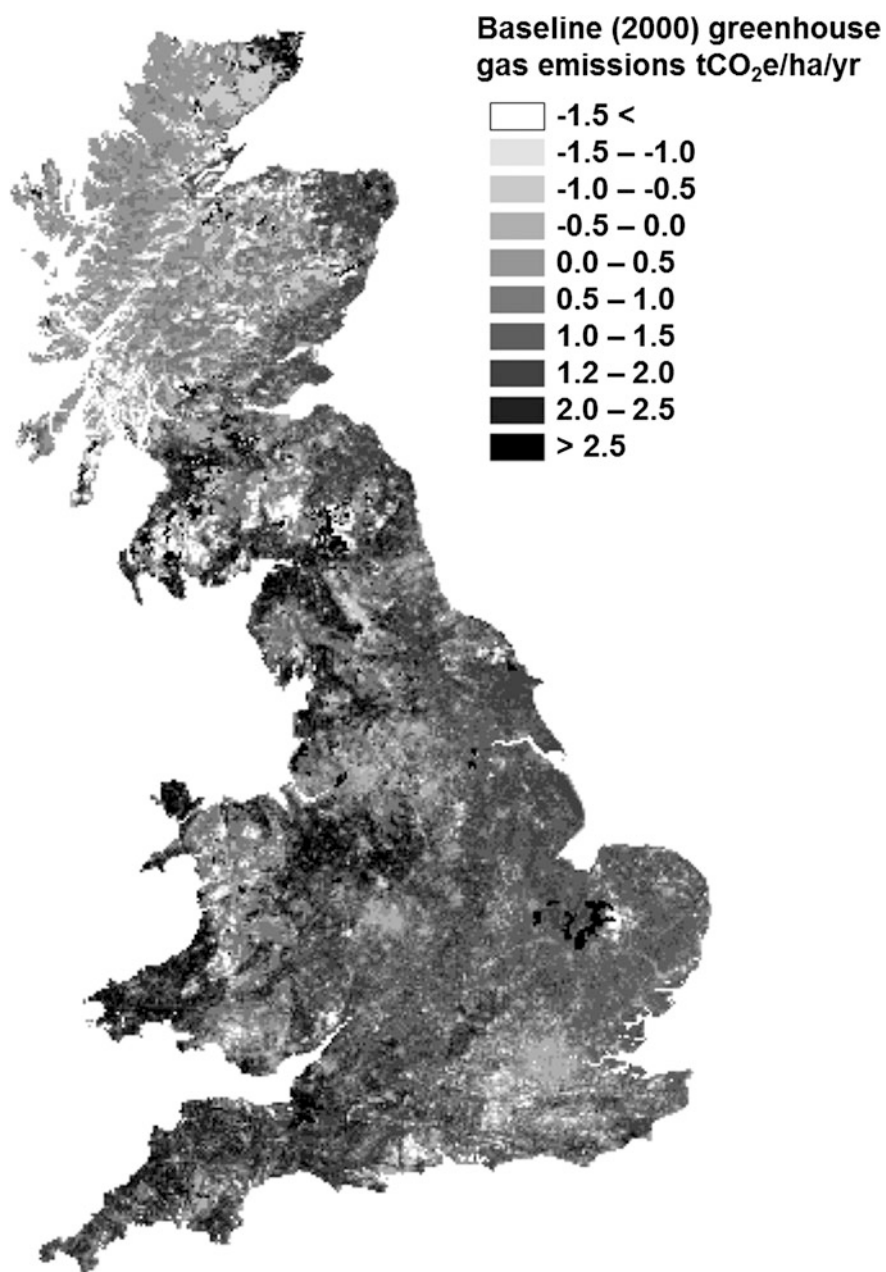


Fig. 13.4 Predicted annual quantity of GHG emissions

management characteristics and technological standards, and the policy site data entering the land use model fall within the range of data of the study site. For the step (b), greater uncertainty exists about the reliability of the mean benefit transfer estimates which do not reflect spatial variability in soil carbon content within the same soil type.

Finally, the economic value of current agricultural GHG emissions is obtained by multiplying the quantity of GHG emissions in each grid cell by the price of CO₂ equivalents. Carbon prices per ton of CO₂ equivalent do not vary across space, because the location at which carbon is sequestered or emitted does not alter the effect on climate change. However, the economic assessment of GHG emissions in terms of the marginal costs of carbon is not a simple task; the welfare impacts of climate changes are influenced by many factors, such as uncertainty in climate change effects, economic consequences of climate change, ecosystems response to climate change, etc. The most commonly applied approaches for estimating carbon prices are the social cost of carbon and marginal abatement cost of carbon (Stern 2007; Department of Energy and Climate Change (DECC) 2009). From the perspective of an economic cost-benefit analysis it is the former value which is greater relevance for welfare evaluations. However, the official non-traded marginal abatement cost of carbon set by the UK DECC (£41.28 per ton of CO₂-equivalent in 2010 prices) falls within the range of published estimates of the social costs of carbon reported by Tol (2010) (whose meta-analysis yields an average value of around £33/tCO₂ with an upper 95 % percentile value of around £123/tCO₂, although the modal value is much lower at just over £9/tCO₂ suggesting that our chosen values, while policy relevant, may be considered to be on the high side from a welfare perspective).

The findings show that average costs from agricultural GHG emissions in GB are £94 per hectare, but regional analysis shows great variability across country, with higher values along the western coastline of England and Wales that are mainly dominated by intensive agricultural practices, principally beef livestock (Fig. 13.5).

The example of GHG emissions shows the importance of linking social sciences and biophysical modeling in ecosystem services assessments and the role that transfer of economic and biophysical models across space plays in such large scale analyses. When the economic value is constant, as it is for the carbon price, the overall carbon values still vary across space following the spatial pattern predicted by the biophysical model.

13.3.3 Recreational Benefits

Recreational opportunities are one of the clearest examples of non-consumptive benefits that the natural environment and ecosystem services provide to human beings. Open-access recreation is valued in excess of £20 billion annually in England alone (Sen et al. 2011). These values are highly variable across space. For

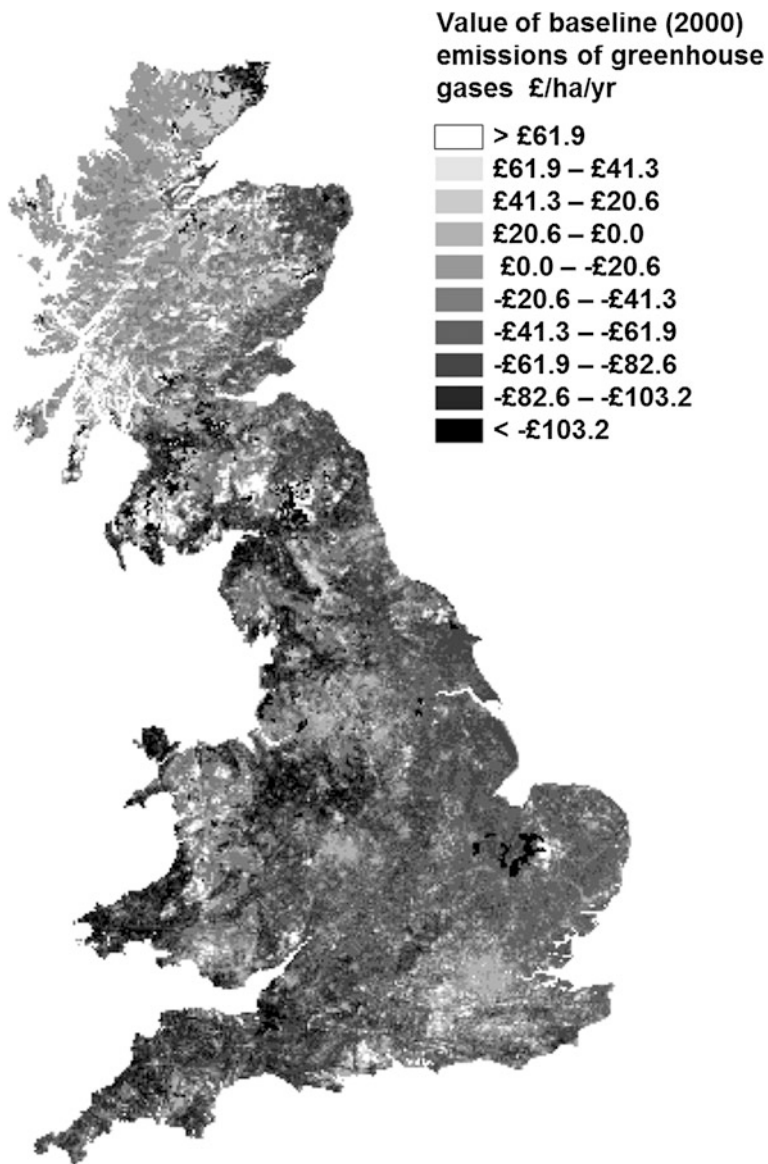


Fig. 13.5 Predicted annual values of GHG emissions

example, the recreation services provided by a river can yield a much higher value when located nearby a highly populated area than for a biophysically similar river located in a remote area. As a consequence, the number of visits to a recreation area is highly non-random and driven by local characteristics. Therefore, different aspects of recreational benefits such as distance to urban area, habitat characteristics

and the availability of substitute recreational sites should be taken into account when valuing open-access resources.

In the UK NEA, the first step of the recreation analysis consisted of an inventory of sites and examination of the factors that determine their existence. Subsequently, the impact of ecosystem services flows to recreational behavior was estimated. The full details of the valuation approach for open-access recreation, schematically depicted in Fig. 13.4, are presented in Sen et al. (2010). The biophysical model is based on a large survey about recreational behavior among more than 45,000 English households. The biophysical output is combined with a meta-analysis on the value per trip to predict the total annual value across different types of habitats. The models are then used in a benefit transfer exercise to predict recreational values in Scotland and Wales.

In this example, the biophysical modelling of ecosystem services consists of two elements: a site prediction function (SPF) and a trip generation function (TGF). The current recreation sites and number of related visits are known and identified by the survey data (Monitor of Engagement with the Natural Environment (MENE) 2010). However, given that sites in non-surveyed areas and under different scenarios are not known, the relationship between site location and habitat type is statistically analyzed. The SPF predicts the number of recreational sites in an area as a function of type of natural resources at the site, the distribution of the population around the site and the travel time from that population to the site. This model is used to predict recreational sites for the policy site areas (Wales and Scotland) or in different states of the world. Next, the TGF models the number of visits from each UK Census Lower Super Output Area to any given recreational site as a function of: population characteristics, availability of potential substitutes, distance and type of habitats. Data for all of these analyses are obtained at different spatial resolutions using GIS (see Sen et al. 2010). The output of the TGF is the predicted number of visits per site which has been multiplied by the number of sites per cell (output of the SPF) to produce the number of visits per week to all 1 km grid square cells across the current estimates of recreational sites in England under the land cover map 2000. These values are the output of the biophysical model and are calibrated with observed visits to sites in England.

The results of the model, reported in Sen et al. (2010), show that the variation in the number of visits is a function of different variables such as location and its main habitat characteristics, road network, population distribution and characteristics, substitutes and complements of different habitats types. Mountains, coasts and freshwater sites and woodlands have a significant positive effect on the number of visits. Retired and richer people have higher levels of participation in recreation activities.

Unlike the fixed prices per unit of carbon, recreational values per trip are likely to be context-specific. Therefore, a meta-analytical regression model was developed based on revealed and stated preference estimates of willingness to pay per person per trip from nearly 250 previous studies on open-access recreational sites

worldwide.³ The trip value is modeled as a function of ecosystem types, controlling for the sample size, valuation method, valuation unit (e.g. household or person) and country. This generates a model of the site-specific willingness to pay value per person per visit, which varies according to the habitat type characteristics of the visited site. Further, details about this meta-analysis study can be found in Sen et al. (2010).

Multiplying the value per trip by the predicted number of visits to a site in that cell produces the annual recreational (or access) value. Although the value of, say, mountain visits is high, the number of visits is low and therefore the annual total value reflects these two components. Further, since cells can contain various types of habitat, the overall habitat value of each cell is obtained by multiplying the coverage of the different habitats by their money measure. For example, given that the value per person per trip to woodland is estimated at £6.10 and that to wetlands £6.88, if in a 1 km cell the coverage is 50 % woodlands and 50 % wetlands, the per trip value of that cell is given by adding £3.44 + £3.05.

The predicted average annual number of visits per each 5 km grid cell is 394,000. This corresponds to over 2.9 billion annual visits, representing more than £8.9 billion in economic benefits for England. These recreational values change according to the natural environment of the area, the availability of substitutes, the infrastructure and the distribution of the population around that area. The models are therefore highly transferable and results can be aggregated across any desired spatial unit (e.g. county, region, and catchment) and scenario. In order to test the robustness of the biophysical results, out-of sample tests have been conducted and an improved version of the biophysical modeling approach is published in Sen et al. (2014).

In the UK NEA, the model has been used for a benefit transfer exercise to predict the annual value of visits to (semi-) natural habitats for the UK, where England is used as the study site and values are predicted for Scotland and Wales. Spatial information on habitat types, travel times and land uses were collected for Scotland and Wales, and coupled with the parameters of the TGF, to predict the annual number of visits to these policy sites.

Figure 13.7 presents the resulting distribution of the TGF model, showing that variation in number of visits is predicted to vary with distance to populated area, habitat and land use types. For example, number of visits to Highland Scotland is relatively low compared to those of England, because the distance to populated areas is high.

In the final step of the analysis, the annual total value of recreational visits is obtained by combining the distribution of recreational visits with the results of the meta-regression model (Fig. 13.6). Since the economic values of outdoor recreation are spatially sensitive, the distribution in Fig. 13.8 differs from that in Fig. 13.7. Figure 13.8 shows that some remote areas, such as the Scottish Highlands, for

³The use of meta-analysis for valuing recreational trips is a well developed area of research and interested readers are referred to Rosenberger and Loomis (2000).

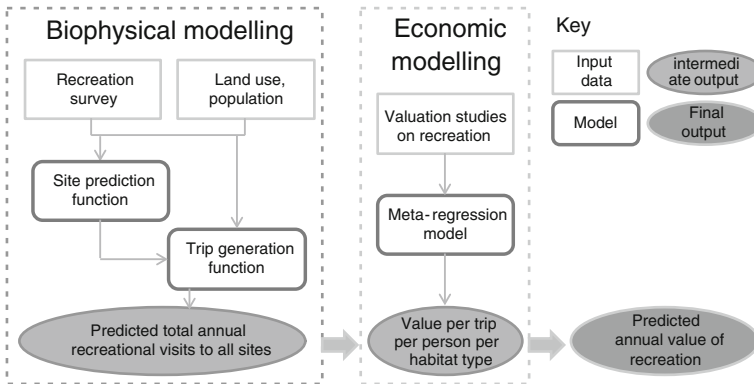


Fig. 13.6 Economic assessment of recreational sites

which the number of visits is relatively low (in the range of 10,000–100,000), are nevertheless associated with relatively higher annual recreational values (greater than £100,000). This is because the habitats in these areas are highly appreciated and therefore the value per visit is higher than for other types of habitat.

13.3.4 Scenario Mapping

13.3.4.1 The Scenarios

The scenario analyses consist of a comparison of ecosystem services in the 2000 baseline (prices in 2010) with various future states in 2060 generated by the UK NEA scenarios team. The baseline is set to a reference year and prices are adjusted to 2010 levels. The scenario analysis uses benefit transfer for valuing ecosystem services under different states of the world by transferring the estimated functions describing both ecosystem services and their values. The scenario analysis proceeds by applying these functions to the same geographical area, but with the physical attributes of that area altered in line with expectations formed through the scenario generation process as described by Haines-Young et al. (2011). The latter study generated a number of scenarios as likely to arise under differing policies formulations. These are further perturbed by climate drivers described by the UK Climate Impacts Programme (UK-CIP) reported in Murphy et al. (2009). For simplicity we focus upon just two scenarios, both of which assume a ‘high emissions’ trajectory.

The first scenario, “Green and Pleasant Land” (GPL), envisions that economic growth is driven mainly by secondary and tertiary sectors. Pressures on rural areas are assumed to be declining as a result of increased concern for the conservation of biodiversity and landscape. Here, as biodiversity preservation is a key objective for policy makers, sometimes habitats will be preserved and conserved primarily to improve the aesthetic appeal of landscape and countryside. Arable lands decline and

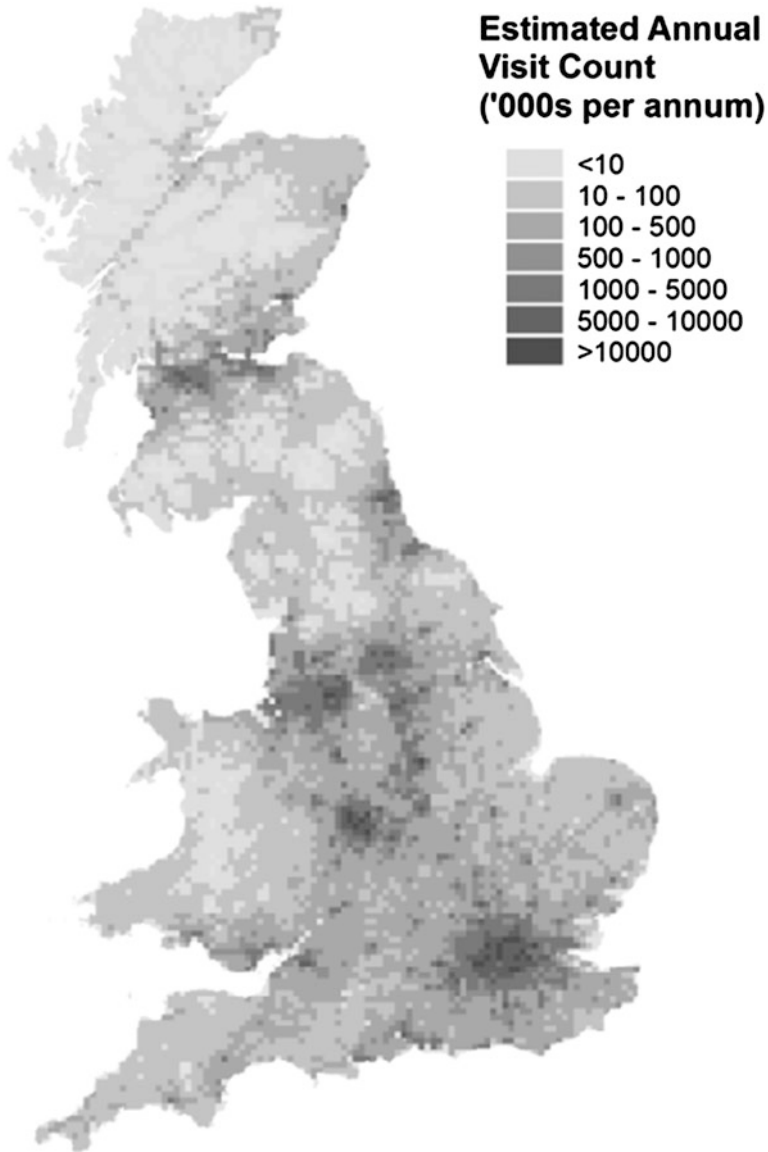


Fig. 13.7 Predicted annual number of visits

the biodiversity and aesthetic values of landscapes are enhanced by increases in improved grassland (temporary or permanent grassland with reduced fertilizer), semi-natural grassland and conifer woodland. This implies a decrease in food production which is compensated for by increased imports to offset the demands of a larger population.

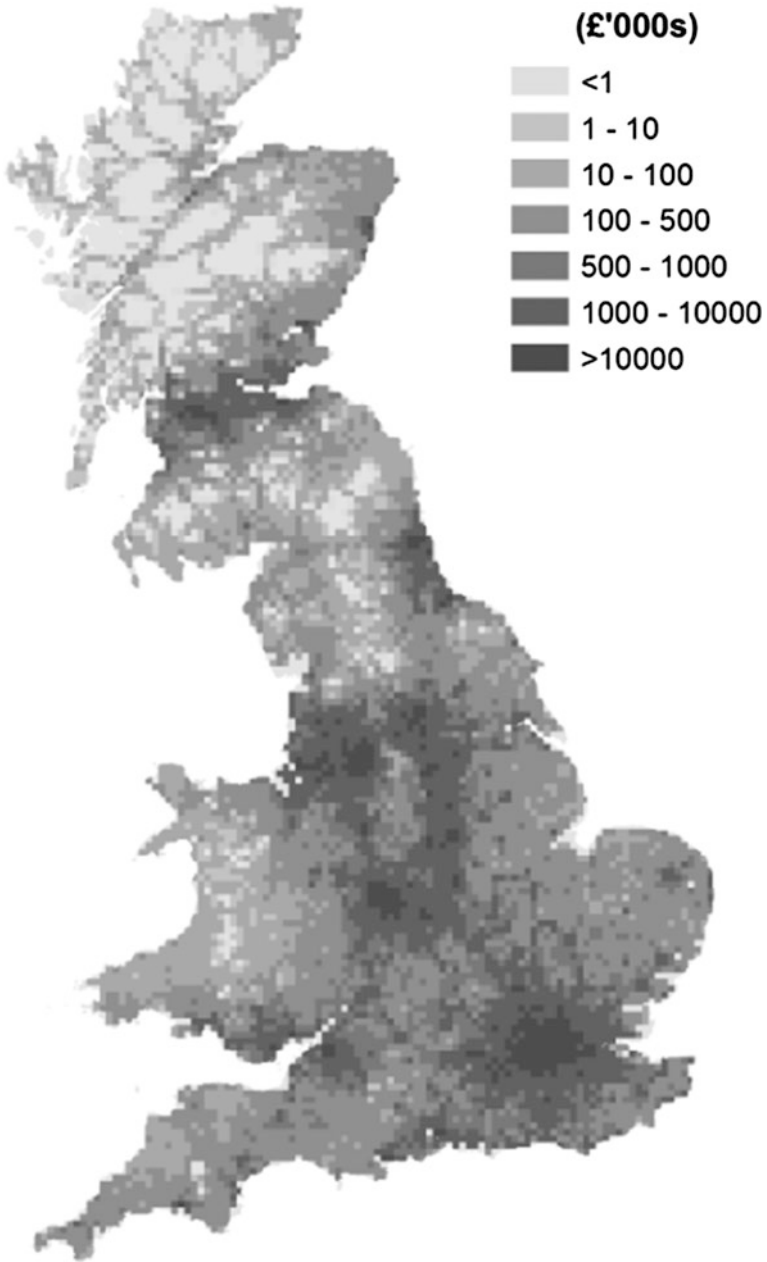


Fig. 13.8 Predicted annual values of recreation

In the second scenario, called “World Market” (WM), a 30 % population increase is envisioned and concomitant changes in land uses are substantial. A complete liberalization of trade is assumed, which implies an end to agricultural subsidies, increases in international trade of resources/goods and reduced rural and urban planning regulation. Consequently, the proportion of arable lands increases and improved grassland and semi-natural areas decrease to accommodate urban growth. Biodiversity declines and technological development is pursued mainly by private companies.

Table 13.1 gives an overview of the land cover and population changes under the WM and GPL scenarios, drawing from results presented in UK NEA (2011). Within the spatial analysis framework, these scenario storylines are translated into future land use maps and population predictions. The models of current behavior are combined with these future input data layers to predict future ecosystem service benefits, assuming that the functional relationships and parameter estimates remain constant over time.

13.3.4.2 GHG Emissions

In this example, the benefit transfer is applied to value changes under the two scenarios reported in Table 13.1. Using the standard nomenclature of benefit transfer, the “study site” is here represented by the current state of the world and the “policy site” is not a different area, but the same area under future foreseen changes. Therefore, changes in Table 13.1 represent the hypothesised values for the scenarios/scenario input valuables available for the “policy site.” A function transfer approach is applied to determine the predicted annual quantity of GHG emissions. The biophysical model predicts changes in agricultural GHG emissions due to land use changes assumed under the GPL and WM scenarios. For woodland planted between 2000 and 2060, average annual flows were assumed following Haines-Young et al. (2011). Carbon stocks and fluxes are calculated using the new land use shares and assumptions presented in Sect. 13.3.2.

Figure 13.9 describes changes in terrestrial ecosystem emissions (tons of CO₂e/ha/year) between the baseline and 2060 under the two scenarios (cf. UK NEA 2011). Darker colors in Fig. 13.7 show where the changes in GHG emissions are going to be most substantial. Scotland and the north of England are predicted to show the highest increase in emissions of agricultural GHG emissions, due to the conversion of rough grazing to more intensive agricultural land uses.

As expected, in the GPL scenario there is a relatively uniform decrease in GHG emission equivalent of roughly 8 million tons of CO₂e/year. This reduction stems mainly from lowland areas, where arable land and improved grasslands are converted to semi-natural and rough grazing. This in turn results in lower density of beef and sheep livestock and therefore lower emissions from fertilizer than in the baseline. In the upland areas, there is a moderate increase in GHG emissions, mainly driven by increased livestock numbers and decreased carbon accumulation

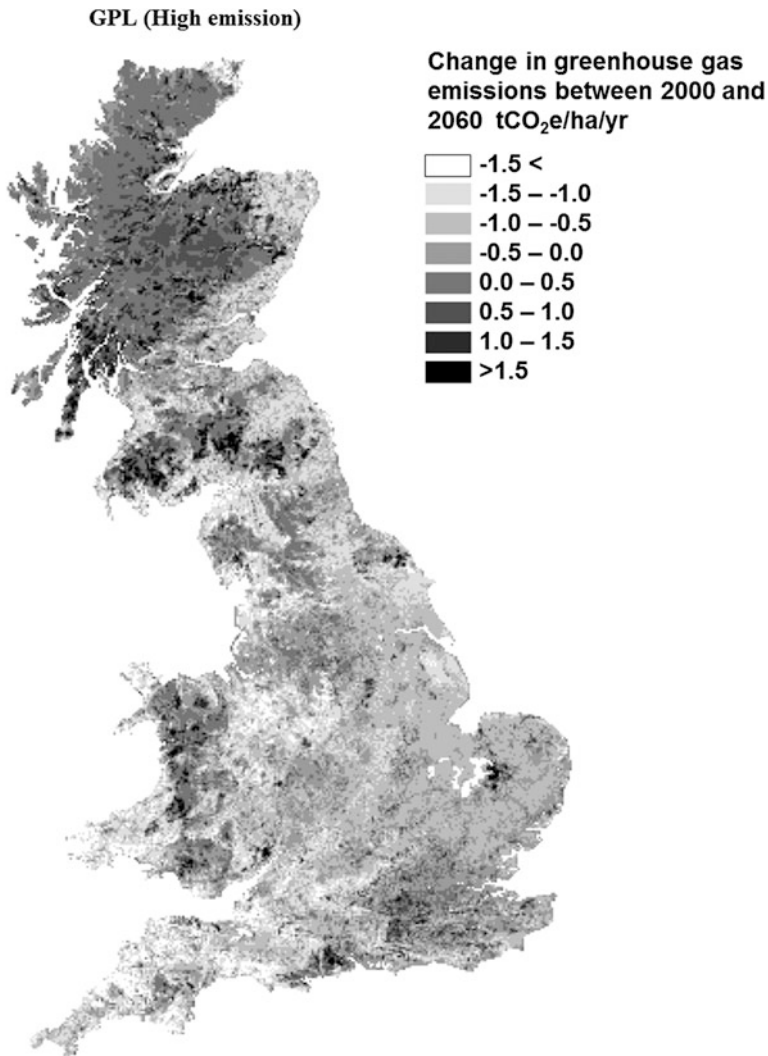


Fig. 13.9 GHG emissions: Scenario analysis and changes between 2060 and baseline. *Left* GPL (high emission). *Right* WM (high emission)

in forests. The latter is assumed to happen because the rate of carbon uptake will decline when numerous conifer plantations reach maturity. Overall, the GPL scenario presents a positive impact in terms of GHG emission reductions.

The WM scenario presents a contrasting result. Here emission levels increase by roughly 6 million tons of CO₂e/year compared to the baseline. The main drivers of this change are reductions in the extent of woodlands, due to the envisioned high pressure of urban expansion, and moderate expansion of arable and dairy production, largely at the expense of semi-natural grasslands.

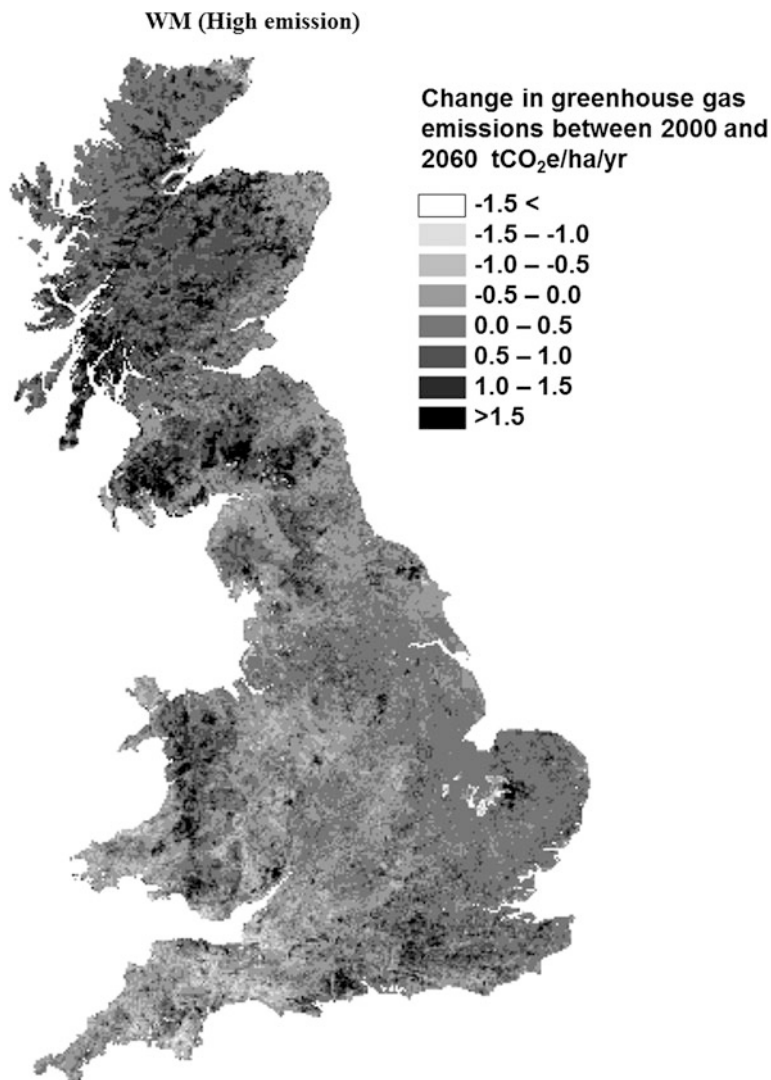


Fig. 13.9 (continued)

The total value of annual GHG emissions is obtained by applying the carbon price to the emission levels under the two scenarios. The results suggest that GB would save more than £2 billion annually in terms of GHG emissions costs under the GPL scenario, while increased emissions under the WM scenario would imply a loss of societal welfare of more than £2 billion annually.

The scenario analysis shows how benefit transfer methods can be used to express significant differences between the GPL and the WM scenario in a spatially explicit manner, and visualize which areas are going to contribute most to the net change in

GHG emissions. Benefit transfer was used to predict values for the future scenarios assuming that the relationships between the explanatory variables and outcome variable as estimated in the biophysical and economic models remain robust over time as does the unexplained variability. Violation of this assumption may invalidate the results of the GHG comparison.

13.3.4.3 Recreation

Again a benefit transfer exercise for the same area across time is performed using the changes described in Table 13.1 and transferring the SPF and TGF function to predict the annual number of visits to recreation sites under the GPL and WM scenarios. The changes in predicted visit numbers for the GPL scenario are visualized in Fig. 13.10, which can be compared with the baseline given in Fig. 13.7.

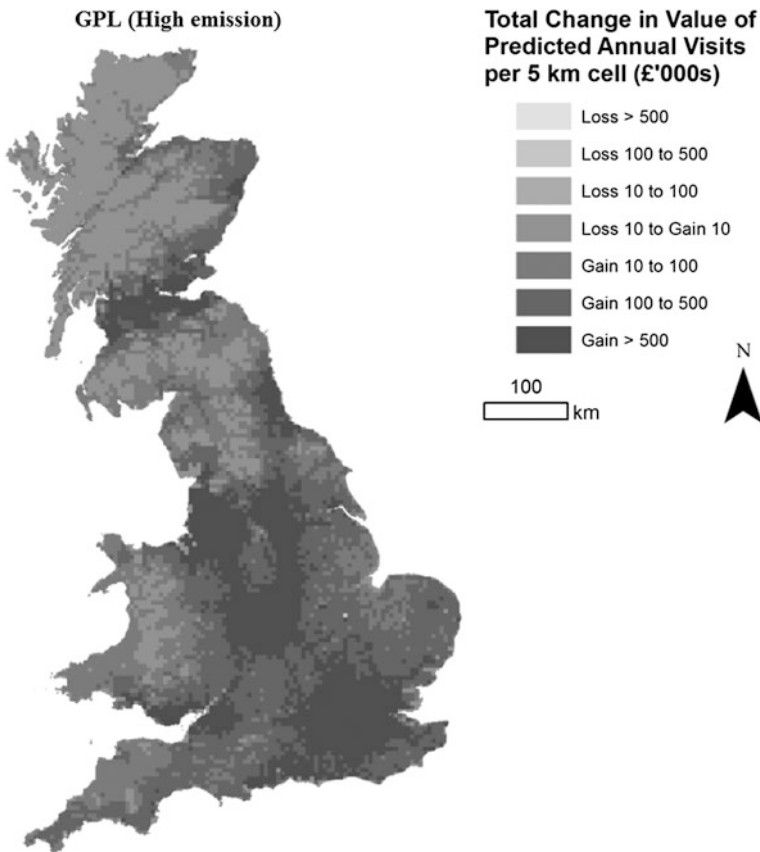


Fig. 13.10 Recreation: Scenario analysis in 2060, changes between 2060 and baseline. *Left* GPL (high emission). *Right* WM (high emission)

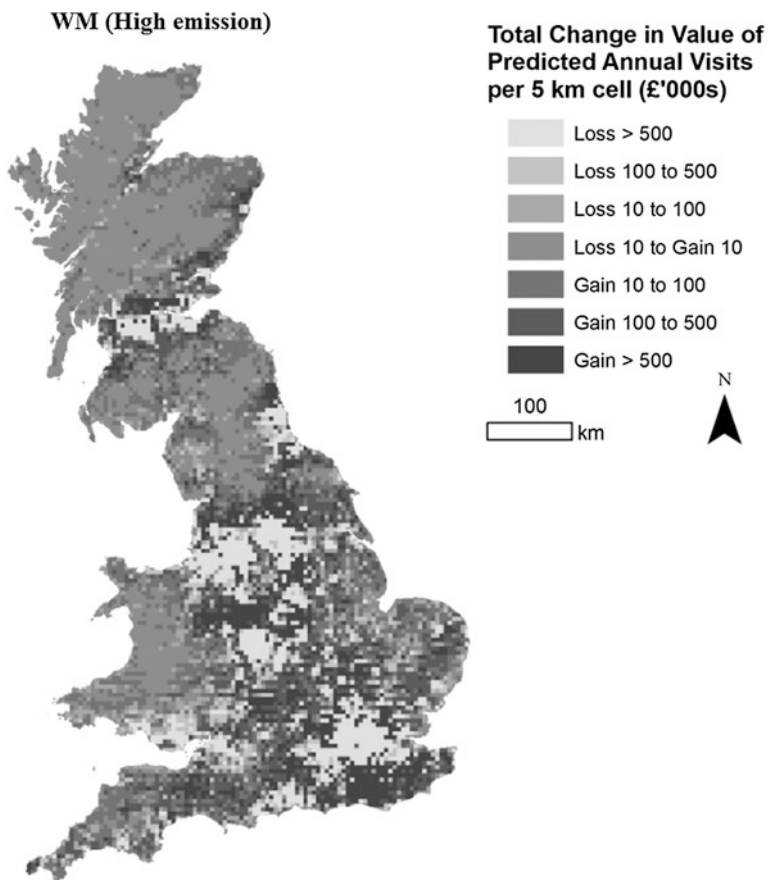


Fig. 13.10 (continued)

The changes in the number of visits occur for a variety of reasons, including changes in the availability of substitutes; variation in habitat characteristics (e.g., change of arable land to improved grassland); and changes in medium household income and population characteristics (e.g., higher proportion of retired people). Under the GPL scenario, the number of visits per year is predicted to increase substantially, mainly around urban areas. Indeed, only remote areas fail to experience increased recreational visit numbers. The values of these changes depicted in Fig. 13.10 are obtained by applying the meta-analytical values per trip to these future visitor numbers, as illustrated (UK NEA 2011). Overall, the GPL scenario delivers a substantial increase in recreational values over the baseline. This effect is driven mainly by a decrease in primary sector production and an increase in aesthetic landscape conservation and protection.

In contrast to the GPL scenario, the distribution of annual recreational values under the WM scenario shows a significant decrease in comparison with the baseline. Major losses are found near to large urban areas and arise due to reductions in urban and peri-urban recreational areas (including green belt) envisioned under the scenario. This loss of resource results in some substitution towards more rural areas, many of which show increased recreational values. However, overall the WM scenario results in major losses of recreational value.

13.3.4.4 Comparison of GHG and Recreation Scenario Analyses

The changes in value due to agricultural GHG emissions and open-access recreation visits under the GPL and WM scenario are summarized in Table 13.2. Values are aggregated for each country in GB and given as total and per capita sums. Results show that the GPL scenario is associated with higher ecosystem benefits per capita (£61 million p.a. for recreation values and £37 million p.a. per GHG emissions), whereas the WM scenario is always associated with annual losses, even though the real income in the two states of the world is assumed to be equal (see Table 13.2).

The numbers in Table 13.2 are the results of the benefit transfer exercises and are based on a range of assumptions, not only those of the scenarios, but also that models can be transferred across time and space (UK NEA 2011). The transfer across time applies the current functional models for GHG and recreation ecosystem services and values, and assumes that the independent variables change over time under two alternative scenarios: GPL and WM. This type of benefit transfer exercise is surrounded with considerable uncertainty and ideally, some information about the confidence intervals around these estimates should be given. However, producing such confidence intervals is difficult given that results are based on a combination of various models, especially when non-linearity in ecosystem

Table 13.2 Total and per capita value of changes in annual ecosystem service values from baseline year for annual recreation visits and GHG emissions under two 2060 UK NEA scenarios

Region	Recreation (million £ p.a.)		GHG (million £ p.a.)	
	GPL	WM	GPL	WM
England	4451	-678	2193	-381
Scotland	556	-61	-52	-1192
Wales	149	-84	268	-101
Great Britain	5156	-823	2410	-1675
GB population (millions)	66	72	66	72
GB per capita values (£ p.a.)	61	-57	37	-23

Note Negative values represent an annual loss. The GHG and recreational value changes are obtained following slightly different approaches: GHG values are based on the difference in GHG quantity and multiplied by 2010 prices, whereas the recreational values reflect the difference in the number of visits for 2000 and 2060 multiplied by the WTP per trip in 2000 prices

functioning may play a role, when there is uncertainty related to the future values of the input variables of the models and uncertainty regarding the stability of preferences over generations of people. The results may be considered as general trends in value change arising from scenarios.

13.4 Conclusions

The two case studies from the UK-NEA presented in this chapter reflect the complexity of large-scale ecosystem services assessment and the central role that benefit transfer methods play in these exercises. Firstly, both ecosystem services and their values are explicitly modeled and the resulting spatially explicit statistical functions are based on highly detailed primary data. Secondly, the transfer takes place at national level using primary models at nation level and transferring that to adjacent areas. Thirdly, the models are used for scenario analysis and transferred across time.

The ecosystem service approach and the UK assessment demonstrate a means to unite natural sciences with economic assessments to estimate the value of changes under different states or development pathways, thereby informing decision-making regarding strategies for improving societal welfare. Ecosystem assessment requires the combination of sound biophysical modeling of ecosystems, services and their processes and interactions, along with detailed socioeconomic analyses of the final ecosystem services and the value that they provide to humans. It is this combination which forms a necessary evolution for environmental valuation from earlier valuation work, where ecosystem functioning was often simplified to very basic levels and interrelations with other ecosystems were often ignored.

The key to ecosystem assessments is that services are not considered in isolation, but in combination, showing where tradeoffs have to be made or synergies can be achieved in ecosystem management. The integration of disciplines in scenario analysis allows for the evaluation of current levels of ecosystem use, and can help elucidate tradeoffs among alternative policy options which can ultimately lead to more sustainable futures with higher ecosystem service benefits. For instance, the analysis of GHG emissions from agricultural land and the quantification of associated costs may be a first step in understanding where emissions reductions can be achieved most efficiently and developing a mechanism to internalize these costs within land-management decisions.

The benefit transfer exercise presented in the chapter relies on spatially sensitive transferable functions for biophysical and economic aspects of ecosystem valuation and ensures that analyses account for the locational context of ecosystem values. Furthermore, in order to minimize errors in modelling and subsequently in transferring ecosystem service values, data from across a large area, in this case GB, at a very fine level of resolution are analysed for different ecosystem types. This suggests that with greater data availability, benefits transfer exercises may be based on spatially explicit models which can better capture variability in ecosystem services.

In order to recognize the importance of spatial context, the UK NEA ecosystem assessments rely heavily on GIS-based maps, visualizing the results of spatially explicit biophysical and economic models. Thus, benefit transfer methods support the incorporation of ecosystem values in policy making, and can provide information about costs and benefits of ecosystem services at a high spatial resolution, even at the national scale of countries the size of the GB.

It should be noted that spatially explicit large scale assessments are complicated by conceptual and practical issues. First, the spatial boundaries of ecosystems and their services are not clear-cut; ecosystems vary widely in spatial scale and their key processes operate across a range of rates that are overlapping in time and space (TEEB 2010). In addition, ecological, economic, social and political boundaries may not match. Second, data availability at the relevant scale or precision may be limited and data collection can be resource-intensive, thereby limiting the accuracy of the analysis or the variables that can be included in spatially explicit models. A high level of GIS information as well as modeling capacity is required. The associated investments in the start-up phase may be considerable, but the results can be used by various stakeholders at any given scale of assessment. At the same time, large scale analyses based on benefit transfers might raise non-trivial questions about the reliability of the predicted values and the related errors. Particularly, where local scale models are applied to larger scales, e.g., to national levels, without (the possibility to do) reliability checks, or vice versa, the assumptions of stability of preferences across space may be challenged. Further, the combination of biophysical and economic models requires that are both well specified and spatially explicit, because where both estimates are associated with large errors, the multiplication of the estimates may introduce considerable transfer bias in the ultimate welfare estimates. Therefore, the results of large-scale ecosystem services assessment based on transferred values may be suitable for initial stages of decision-making, whereas later stages nearing implementation of projects or policies, where higher reliability of value estimates is required, may require more reliability checks or primary valuation studies.

One of the remaining issues in the UK NEA is the assessment of sustainability. Sustainability assessments require the comparison of actual use to regeneration levels, i.e., the impact of service flow changes on the levels of stocks of relevant ecosystem services. In the case of timber use, projections of carbon emissions over time also have to take into account the lifecycle of products made of timber. Both carbon storage and sequestration in woodlands and carbon storage in timber products are excluded in this chapter. Current scientific knowledge is not sufficient for a robust assessment of the sustainability of the current resource use, but this issue is on the list of future research themes.

Of academic and political importance is the need to develop more rigorous testing of the reliability of these large-scale transfers, which may require new primary data collection or temporal stability tests of transferred data. Furthermore, the results presented in this chapter do not consider the confidence intervals of the biophysical and economic models and their effect on transferred values. The use of the benefit transfer for ecosystem services valuation involves a trade-off between

scale of analysis and accuracy. The reliability of benefit transfer across time and space builds upon a range of assumptions. Most notably are the assumed similarity of sites, whereas sites across a nation are likely to vary considerably; the stability of preferences across space, whereas there are likely to be economic and socio-cultural differences between people within a country; and stability of preferences over time, whereas there are likely to be changes in preferences and economic demand over longer timeframes. More localised, short-term decision-making may require more accurate results for which the costs of additional primary data may be justified. The type of large scale assessments, as presented in this chapter, is mainly suitable to inform long-term, strategic policy making at higher levels, using contrasting scenarios to show the direction in which various policies may result in different policy outcomes and associated economic welfare estimates.

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