

Chapter 22:

Economic Values from Ecosystems

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Key Findings

The contribution that ecosystem services make to the national economy in terms of a sustained flow of income is very substantial. The continued maintenance of this natural capital stock is critically important for the future prospects of a thriving 'green' economy. The sustainable development goal will not be achievable without a more efficient and effective management of ecosystems encompassing economic appraisal principles and practice.

It is clear **that a body of theoretically sound methodologies now exists for the valuation of most (if not all) ecosystem service flows** (i.e. the flow of values which ecosystems deliver to individuals). This methodology is consistent with the Conceptual Framework of the UK National Ecosystem Assessment (Chapter 2) and has been clarified in supporting papers (see Bateman *et al.* 2011a). This methodology extends, but is consistent with, standard decision analysis principles set down by HM Treasury and is expected to be highly compatible with the aims and objectives of the forthcoming Environment White Paper.

In line with standard economic analysis, **the methodology that has been developed rejects attempts to estimate the total value of ecosystem services.** Many of these services are essential to continued human existence and total values are therefore underestimates of infinity. However, real world decisions typically involve incremental changes and require choices between options. **Our economic analysis therefore examines the value of observed trends and feasible, policy-relevant changes.** It also adopts a precautionary approach, given the uncertainties shrouding the necessary and sufficient conditions for continued 'healthy', functioning ecosystems under the pressures of environmental change.

Our economic analysis provides a bridge from the ecosystem habitat focus of the natural science elements of the UK National Ecosystem Assessment (UK NEA) to consideration of the goods and services those ecosystems provide and the values these yield to individuals. **The analysis has highlighted the considerable value provided by a broad range of ecosystem service flows** (see **Table 22.27** for a summary). These include: the contribution of ecosystem services to the production of both terrestrial and marine foods; the direct and indirect use value of biodiversity in underpinning and delivering ecosystem services; timber production; carbon sequestration, storage and greenhouse gas (GHG) flux; water quality and quantity; inland and coastal flood protection; pollution remediation; energy and raw materials; employment; sporting and game; landscape values and the amenity value of nature; the amenity value of the climate; the amenity value of urban greenspace; environmental education and knowledge; the health effects of the environment; and recreation and tourism. Collectively, this service flow makes a vital contribution to the wealth and well-being of the UK. While information gaps mean that we cannot estimate values for all services, **those values that are reported are substantial and underline the vital role which the natural environment plays in supporting current human wealth creation and well-being and in offering the foundations for a sustainable future economy.**

The **detailed ecosystem service valuations** presented in the main body of this chapter are broadly categorised into those that assess past trends and those that consider likely future scenarios. Considering the first category, there has been relatively little work which has adjusted for the value of manufactured and human capital in ecosystem service-related output values. This means that many of the estimates in this category are liable to overstate the contribution of ecosystem services to resultant values. Nevertheless, ecosystem inputs are often vital to the production of such goods and accepting this caveat, we highlight the following examples for the UK:

- The **value of UK fish landings** is about £600 million per annum (p.a.), while that of aquaculture (fish and shellfish farming) is around £350 million p.a..
- **Biodiversity pollination services** are estimated at £430 million p.a.
- Willingness to pay (WTP) estimates of **the non-use (existence) value of terrestrial biodiversity** range from £540 million to £1,262 million p.a. and for **marine biodiversity**, estimates of around £1,700 million p.a. have been reported. However, as noted below, there is debate regarding such estimates. Legacy values are around £90 million p.a.
- **Timber values** are just under £100 million p.a.
- The **water quality** benefits of inland wetlands may be as high as £1,500 million p.a., while planned river quality improvements may generate values up to £1,100 million p.a. However, climate change-induced losses of water availability are valued at £350 million to £490 million p.a..
- The costs associated with changing agricultural land use to reduce nutrient loadings into rivers are substantially smaller than the benefits which consequent reductions in **diffuse water pollution** would bring (however, the former costs are concentrated within rural communities, while benefits are distributed across a mainly urban society).
- The **amenity value** of all wetland types, including coastal, is around £1.3 billion p.a.
- **Renewable fuels** currently meet 3% of UK energy demand and 7% of electricity generation.
- Marine-based biotic **raw materials** are worth £95 million p.a.
- The UK **aggregates** industry is worth £4,800 million p.a., of which more than £100 million comes from the marine environment.

- The environment generates substantial **educational benefits** each year.
- The total value of **net carbon sequestered** currently by UK woodlands is estimated at £680 million p.a.
- There are also **substantial costs** arising from activities which deplete ecosystem services. For example, considering the previous result regarding carbon sequestration by woodlands, this is completely negated by **GHG emissions** from UK agriculture, which are currently around £4,300 million p.a. Similarly, the average annual **cost of flooding** is about £1,400 million, although this can rise as high as £3,200 million in extreme years. These costs need to be added to WTP to avoid intangible costs of £120 million p.a.

Moving to consider valuations based upon future trends and scenarios, this draws upon new work undertaken for the UK NEA, most of which isolates the role of changes in ecosystem –and wider environmental – services in the estimation of values. Highlights here include the following:

- **Changes in climate services are likely to have marked impacts upon agricultural land use**, although the value implications of these changes will vary across the country. Forecast increases in temperature and shifts in rainfall patterns may well improve the agricultural potential of currently challenging upland areas, resulting in increases in incomes in much of upland England, Northern Ireland, Scotland and Wales. Impacts upon lowland areas, including most of southern England, depend crucially upon changes in technology such that under current forecasts, incomes are liable to decline in these areas. However, it is likely that this will stimulate technological change which would alter predictions for these areas.
- The **increase in agricultural productivity** in upland Britain is **likely to stimulate a corresponding rise in agricultural carbon emissions** in those areas. Full economic costing of these emissions would cancel out a substantial portion of the benefits of higher agricultural outputs.
- Changes in land use will have a **significant impact upon biodiversity**. Indicators such as the number of farmland bird species suggest that at best, agricultural land use changes will have a neutral effect, while at worst, there is the likelihood of local extinctions.
- **Ecosystem services have a major impact upon outdoor recreation values**. There are over 3,000 million recreational visits p.a. generating a social value in excess of £10,000 million p.a. (see details in Chapter 26). The recreational value of ecosystems varies not only with their type but, more significantly, with their location. Economic valuation shows that a modestly sized, physically identical, nature recreation site can generate values of between £1,000 and £65,000 p.a., depending purely upon location.
- **Urban greenspace amenity values** range from losses of £1,900 million p.a. to gains of £2,300 million p.a., depending on the policy context.
- Again, there are also substantial costs arising from activities which deplete ecosystem services. For example, **climate change is likely to increase the frequency and intensity of flooding events**, with annual costs rising to more than £20,000 million (in 2010 prices) by 2060 under extreme scenarios.

We conclude our key messages with two caveats. First, while we report values for a wide array of ecosystem services, there are limits to the ability of economics to capture all values associated with ecosystem services. In particular, this applies to certain shared social values, especially those which are not evident in observable behaviour. An example of this might be **the spiritual value of the environment**, especially where this is linked solely to the knowledge of pristine or intact environments (this issue is addressed more fully in Chapter 16). Related to this, while we have included estimates of the use-related values of biodiversity, **there is debate regarding our ability to derive robust monetary estimates of the non-use (existence) value of biodiversity**. Currently these can only be estimated using stated preference methods. While such methods fit conventional economic principles for non-market environmental goods for which individuals hold well-formed economic preferences, commentators are not in agreement as to whether preferences for the non-use (existence) value of biodiversity conform to these requirements. While **some argue that stated preference valuation methods are applicable, and can include collective value estimations via group-based elicitation methods, others reject this and instead argue for natural science determined strategies for safeguarding biodiversity** (possibly including biodiversity offsets), **with economic assessments being confined to cost-effectiveness analysis of competing strategies**.

Our second caveat recognises that **a vital area for future investigation is the incorporation of stocks of natural resources into economic analyses. This is essential in order to ensure that ongoing and future flows of ecosystem service values are sustainable**. While theoretical approaches to the economic valuation of stocks are established (Bateman *et al.* 2011a), there is a significant dearth of information on the size of stocks and, equally importantly, how they may deplete as economic activity changes. The potential for thresholds beyond which stocks might more rapidly deplete, or even collapse, needs to be recognised along with the potential for imperfect restoration or irreversible loss. **Addressing this problem requires the establishment of an integrated decision analysis and support community, uniting different disciplines of the natural sciences with economists, risk analysts and other social scientists**. Although initial moves to establish such a community are underway (see www.valuing-nature.net/), it remains in its infancy and further development of such intellectual capital is a clear requirement if the UK is to move towards ensuring efficient, sustainable and equitable management of the natural environment.

22.1 Introduction

In keeping with the UK National Ecosystem Assessment (UK NEA) Conceptual Framework set out in Chapter 2, in this chapter we move from consideration of ecosystem types and the services they provide, to focus instead upon the contribution which these services make to human well-being. Specifically, this chapter presents an economic assessment of this contribution following the methodology set out for the UK NEA in Bateman *et al.* (2011a), which in turn rests upon a wealth of prior literature covering the application of economic analysis to ecosystem assessments. Given the diverse audience addressed by the UK NEA, we open this chapter with an overview of that methodology, the key issues which it addresses, and its limitations. The remainder of the chapter presents a summary of the published literature focused on the economic analyses of ecosystem service values, combined with new analyses which have been prepared partly or wholly for the UK NEA initiative. The new material covers the following topics: the value of environmental legacy giving (Section 22.3.3.2); a meta-analysis¹ of wetland ecosystem values (Section 22.3.3.1, 22.3.6 and 22.3.8); the health effects of broadly defined UK habitats (Section 22.3.16); the CSERGE (Centre for Global and Economic Research on the Global Environment) land use change model (Section 22.3.17.2, 22.3.17.3 and 22.3.17.4); carbon storage modelling for the UK (Section 22.3.18.2); the value of agricultural climate regulation (Section 22.3.18.3 and 22.3.18.4); cost-effective biodiversity conservation (Section 22.3.19); education and environmental knowledge (Section 22.3.15); informal recreation (Section 22.3.20.1); urban greenspace amenity (Section 22.3.21); and the amenity value of nature (Section 22.3.14). Space limitations mean that full details of these analyses cannot be presented within this chapter and the reader is directed to the UK NEA website (<http://uknea.unep-wcmc.org/>) for detailed reports compiled by the UK NEA Economics team (Abson *et al.* 2010; Beaumont *et al.* 2010; Dugdale, 2010; Fezzi *et al.* 2011; Hulme & Siriwardena 2010; Maddison, 2010; Morling *et al.* 2010; Morris & Camino, 2010; Mourato *et al.* 2010; Perino *et al.* 2010; Sen *et al.* 2010; Termansen *et al.* 2010; Tinch, 2010; Tinch *et al.* 2010; and Valatin & Starling 2010).

Note that this chapter deliberately adopts a broad remit, considering not only biotic ecosystem services (those involving living organisms), but also encompassing a brief overview of certain abiotic services of the natural environment, such as renewable energy. It also briefly considers wider issues such

as raw material, energy and ecosystem-related employment. This is to illustrate the flexibility of the approach adopted and through this, to argue for a wider application of this approach beyond purely biotic ecosystem services. We recognise that these additional discussions go beyond the remit of other analyses in the UK NEA, but feel that they constitute a useful case for the extension of the principles underpinning the ecosystem services approach, contributing to a possible harmonising of methods across all related fields of decision making. The literature review (Section 22.3) also contains links to financial value data and their interpretation in the natural science chapters of the UK NEA (Chapters 4–16). Appendix 22.1 further broadens its scope to consider the macroeconomic implications of adopting the ecosystem service approach to decision analysis and policy formation.

Overall, the chapter makes the case that ecosystems and their services are economically very significant at the national scale (see **Table 22.27** for a summary). The conservation and efficient management of the natural capital stock and the flows of value that ecosystems represent can provide a solid foundation for a sustainable and thriving 'green' economy. Equally, inefficient management and overexploitation of natural capital may well inhibit future prospects for sustainable growth (by imposing unnecessary costs) over the medium- to long-term future. A full recognition of the wealth of services provided by ecosystems can also underpin efforts to improve well-being (e.g. health, cultural heritage and diversity, social cohesion) in society at large. Long-term economic growth prospects will be substantially conditioned by both natural and social capital stock/flow maintenance.

22.2 Methodological Summary²

The crucial role which managed and unmanaged natural systems play in underpinning economic activity and human well-being is of growing concern as evidence mounts of the increasing pressures being placed upon such systems by human activity (GEF 1998; Chapin *et al.* 2000; Koziell 2001; MA 2005; CBD 2006; Loreau *et al.* 2006). One reflection of that concern is the recent undertaking of major assessments of the status of the services provided by ecosystems (see, for example, MA 2005 or TEEB 2010³). Economic analysis is an

1 A meta-analysis entails the combined re-analysis of previous studies.

2 This section draws heavily upon Bateman *et al.* (2011a).

3 In response to review requests, we can contrast the UK NEA with the studies undertaken under The Economics of Ecosystems and Biodiversity (TEEB) initiative. While TEEB considers the global value of certain ecosystem services, the UK NEA, as its name implies, focuses almost exclusively upon the UK. Each has its own specific advantages. TEEB is intended to support international negotiations within the global-political sphere and has a particular interest in the relationship between ecosystem services and poverty. However, the complexity of global environmental issues and the lack of valuation and other data at a worldwide level mean that the empirical focus of TEEB is necessarily confined to a selection of services, notably: the carbon storage value of forests; fisheries; and coral reefs. In contrast the national level focus of the UK NEA permits a more comprehensive assessment of relevant ecosystem services and focuses upon practical decision making. The restriction of the NEA to the UK also avoids some (if not all) of the more extreme data and knowledge gaps which inevitably arise across the global context. However, in many respects the fundamental principles of both TEEB and the UK NEA are similar. Both recognise that "successful environmental protection needs to be grounded in sound economics" (TEEB 2010, p.3) and attempt to move from previous considerations of total value to more policy-relevant assessments of the marginal value of ecosystem-related goods and the benefits generated from alternative strategies for change.

increasing feature of such undertakings and has prompted a rapidly expanding literature regarding the implementation of such analyses (see, for example, Bockstael *et al.* 2000; Balmford *et al.* 2002; De Groot *et al.* 2002; Howarth & Farber 2002; Heal *et al.* 2005; Barbier 2007; Boyd & Banzhaf 2007; Wallace 2007; Finnoff & Tschirhart 2008; Fisher *et al.* 2008, 2009; Mäler *et al.* 2008; Tschirhart 2009; Liu *et al.* 2010; Turner *et al.* 2010; Bateman *et al.* 2011a). This literature forms the methodological basis of the economic analysis conducted for the UK NEA. Some of the concerns raised by critics of the economic approach to ecosystem services assessment (O'Neil 2001; Sagoff 2011) are also addressed in this chapter.

Ecosystem service assessments and accompanying economic analyses can be roughly divided into two types.⁴ 'Sustainability analyses' typically assess the stocks of natural assets,⁵ while 'programme evaluation' analyses seek to ascertain the value of the flow of ecosystem services provided by those assets. Each type of analysis has its various uses. For example, sustainability analyses may inform macro-level policy formation while programme evaluations might be used to support calculations underpinning payments for ecosystem services (Defra 2010b). However, both require information regarding the value of ecosystem services and it is this task which forms the focus of the economic analysis conducted for the UK NEA, leaving the assessment of natural asset stock levels mainly for future consideration.⁶ This is not an entirely satisfactory situation. Arguably, the focus on flows rather than stocks is perfectly acceptable provided that we are operating safely above any thresholds below which stocks (and hence the sustainability of flows) might collapse. Even when this is not the case, flow analyses can be perfectly acceptable, provided that the values used reflect the long-term stream of benefits to society and incorporate the value of any depletion of stocks (such assessments are properly termed 'shadow values').⁷ However, there is a lack of data on and understanding of threshold levels for different stocks of services. In the absence of that information, analysis of ecosystem flow values is, it is argued, a major improvement

over conventional decision making, but work on thresholds is an important future supplement to that analysis. It is not accepted that the complete absence of economic monetary data in ecosystem management and decision making is an acceptable situation (for contrary perspectives, see O'Neil 2001 and Sagoff 2011). The underpinning of the economic analysis conducted for the UK NEA is provided by the Conceptual Framework set out in Chapter 2. Within it, at any given point in time, an ecosystem is defined by its structure and processes. These processes are inherently complex and any attempt to value both the primary supporting services (say the weathering processes which lie at the heart of soil formation) and higher processes (such as the contribution of soil quality to food production) risks the possibility of generating double counting errors. Therefore Fisher *et al.* (2008, 2009) argue that economic analyses should focus upon the 'final ecosystem services' which are the last link in the chain of natural processes which contribute to human well-being by inputting to the production of goods.⁸ Our use of the term 'goods' goes well beyond the common conception of market-priced items to include non-market contributors to well-being, be they physical or non-physical (pure experiential) objects.⁹ While some of these goods come straight from the natural world without the intervention of humans (e.g. the visual amenity of beautiful natural landscapes), many other items (e.g. intensive food production) require some inputs of manufactured or other human capital. In the latter cases it is vital to isolate the contribution of the natural environment to the production of those goods, as failing to do so ignores human and manufactured capital inputs and so risks overstating the value of ecosystem services and undermining the credibility of such analyses.¹⁰ Once isolated, economic analyses seek to assess this value in monetary terms, applying methods which are summarised in Section 22.2.1. However, as acknowledged in the Conceptual Framework of the UK NEA (Chapter 2), not all of the benefits derived from ecosystem services are necessarily amenable to economic valuation

4 We are grateful to Sir Partha Dasgupta for highlighting this distinction and suggesting these terms.

5 Much of the empirical literature concerning sustainability analyses has focused upon assessing historic development paths through adjustments of national income accounts (Bartelmus 2001, 2008; UN 2003; Hamilton & Ruta 2009). An underpinning theoretical framework for sustainability analyses is provided through the notion of 'Comprehensive Wealth', which considers the ecological stocks from which all ecosystem service flows are generated and corresponding economic values derived (Dasgupta & Mäler 2000; Arrow *et al.* 2007; Mäler *et al.* 2008; Dasgupta 2009). See also Turner (1999) on the notion of the 'primary' or 'glue' values that healthy, functioning ecosystems possess.

6 Both the natural science and economic analysis bases for sustainability analyses are less developed than that for flow valuations. In particular, accurate sustainability analyses require an understanding not only of the scale of stocks and rates of depletion but also of any threshold effects (points beyond which further depletion may result in accelerated reductions in stocks which may be imperfectly reversible, hysteretic (i.e. reversible but only when the rate of depletion is first very substantially lowered; see references listed for further discussion, or completely irreversible; see Brock & Starrett, 2003; Mäler *et al.* 2003; Rockström *et al.* 2009). In the review presented in Bateman *et al.* (2011a) we consider three potential strategies for incorporating sustainability concerns into economic appraisals of projects and programmes: i) assessment of how future depletion of ecosystem stocks might increase the marginal social value of corresponding services (see also: Gerlagh & van der Zwann 2002; Hoel & Sterner 2007; Sterner & Persson 2008; Pascal *et al.* 2009); ii) incorporation of the insurance value of maintaining ecosystem resilience (see Mäler 2008; Mäler *et al.* 2009; Walker *et al.* 2010) and iii) the use of safe minimum standards as a means of preserving stocks of ecosystem assets (see Farmer & Randall 1998; Randall 2007). To date none of these analyses have been conducted within the UK and this is one of the empirical foci of the recently established Valuing Nature Network (www.valuing-nature.net/), which seeks to bring together natural scientists, economists, other social scientists and the policy community to improve the valuation of ecosystem service flows, facilitate sustainability analyses and incorporate these various assessments within decision-making protocols.

7 Note that the use of such shadow values is also fundamental to sustainability analyses such as green accounting exercises (see, for example, Dasgupta 2009; Hamilton & Ruta 2009; and Mäler *et al.* 2009).

8 Of course, there is a potential problem here if the primary value and hence sustainability of supporting systems is ignored and only the value of final ecosystem services is considered; hence our earlier discussions of the need for ancillary sustainability analyses.

9 So a beautiful woodland landscape generates amenity views which are a good to the outdoor walker as much as a piece of timber is a good to the home improver. As this example illustrates, some goods are mutually exclusive of others.

10 This is achieved by examining how production of goods varies as inputs of final ecosystem services and other capital are varied at different rates. Natural variation across different areas and across time will often provide a good source of such data (see discussion in Bateman *et al.* 2011a).

(examples include environmentally related social norms which condition, for example, symbolically important landscapes or the spiritual value of the natural world). The debate over the individual value and collective value distinction and the use of non-monetary assessment methods are described in Chapter 16.

22.2.1 Valuing Ecosystem Services

The value of some change in the provision of a good is, within economic analyses, assessed in terms of the change in well-being that it generates; this value is often referred to as a 'benefit' ('cost') if it raises (lowers) well-being. Note that we draw a sharp distinction between the terms 'good' and 'benefit' to highlight the fact that the same good can generate very different benefit values depending on its context (e.g. location) and timing of delivery. For example, considering the spatial context of a good, a woodland situated on the edge of a major city will generate much greater recreational benefits than a physically identical woodland situated in a remote area.¹¹ Note also that some goods generate instrumental 'use value' (e.g. the value of timber to a carpenter), while others deliver 'non-use value' (e.g. the knowledge that biodiversity is being conserved even if the person expressing that value does not observe the species concerned).

In considering the task of valuing ecosystem services an important distinction needs to be drawn between the terms 'value' and 'price'. That they are not, in fact, equivalent is easy to demonstrate. Consider a walk in a local park. The market price of such recreation is likely to be zero as there are no entrance fees and anyone can simply walk in. However, the very fact that people do indeed spend their valuable time in parks shows that this is not a zero value good. In fact the price of a good is simply that portion of its value which is realised within the marketplace. Now in some cases, price may be a perfectly acceptable approximation to value, particularly where all the inputs to the production of a good are privately owned, that good is produced in a competitive market,¹² and where there is not large-scale intervention by governments or other authorities.¹³ Indeed, even when these latter distortions do arise, economists can often adjust for their influence. However, as the park recreation example shows, market price can, in some cases, be a poor approximation of value. Indeed, this divergence can often be substantial and is a characteristic of many of the goods produced by the natural environment.

Economists have developed a variety of methods for estimating the value of goods whose market prices are either imperfect reflections of that value or non-existent. These methods are designed to span the range of valuation challenges raised by the application of economic analyses to the complexity of the natural environment. Application guidelines are discussed in detail through a variety of reviews¹⁴ and **Table 22.1** provides only a brief summary of the available techniques.

It was noted earlier that market prices can, in some cases, provide an acceptable starting point for valuation (e.g. Cairns 2002). However, adjustment should always be made to correct for market distortions such as taxes and subsidies (which are effectively merely transfers from one part of society to another) as well as for non-competitive practices (Freeman 1991; Dasgupta 2009; Nicholson *et al.* 2009). Related to this approach is the factor input or production function method (see Barbier 2000, 2007; Freeman 2003; and Hanley & Barbier, 2009). As discussed previously, this examines the contribution of all of the inputs used to produce a good in terms of the value they add.¹⁵ This approach can be applied to a range of market (consumption) goods, but has also been used for valuing regulatory and 'protection' goods (examples of the latter including flooding and extreme weather protection).¹⁶ All of these approaches infer values by examining linkages with (adjusted) market-priced goods. This tactic is also used in the examination of potential value losses in terms of avoided damage costs or behaviour and expenditure intended to avert such damages.¹⁷ However, we have excluded the use of restoration or replacement costs as a proxy for the value of ecosystem services. Although there are a few interesting examples of such studies, such as the study of the New York City drinking water source in the Catskills Mountains discussed by Chichilinsky & Heal (1998), many economists consider that such methods should be used with caution (Ellis & Fisher 1987; Barbier 1994, 2007; Heal 2000; Freeman 2003), due to the suspicion that restoration or replacement costs may bear little resemblance to the values they approximate.¹⁸ That said, in cases where cost-benefit assessment is not feasible (say, because of a lack of robust benefit estimates), not required (for example, because of regulations requiring compensatory offsetting shadow projects), or even not permitted (say, because of legislation requiring certain actions), then cost information becomes a vital informational input to cost-effectiveness analyses.¹⁹

11 Of course biodiversity might be inversely related to urban proximity. Analysing such trade-offs is the essence of environmental economics.

12 Typically, the less competitive a market, the more any individual producer can exert pressure upon price.

13 Interventions such as government subsidies or taxation can distort prices from their competitive market levels.

14 See, for example, Champ *et al.* (2003), Bateman *et al.* (2002a), Freeman (2003), Pagiola *et al.* (2004), Heal *et al.* (2005), Kanninen (2006), Barbier (2007), Bateman (2007), and Hanley & Barbier (2009).

15 Examples of production function-based valuations of ecosystem services include: multi-purpose woodlands (Bateman *et al.* 2003; Boscolo & Vincent 2003; Nalle *et al.* 2004); marine nutrient balance (Gren *et al.* 1997; Knowler & Barbier 2005; Smith 2007), pollination (Ricketts *et al.* 2004); power generation (Considine & Larson 2006); fisheries (Rodwell *et al.* 2002; Sumaila 2002; Barbier 2003, 2007); watershed protection (Kaiser & Roumasset 2002; Hansen & Hellerstein 2007).

16 Examples include the storm protection values of mangroves in Thailand (Barbier 2007) and hurricanes along the US Atlantic and Gulf coasts (Costanza *et al.* 2008).

17 Note that the averting behaviour method could also be viewed as a variant of the revealed preference approach discussed subsequently.

18 Note that we are not rejecting the use of costs within the process of determining values. For example, cost-based payment vehicles are a standard element of many stated preference willingness to pay studies. Costs may also be useful indicators of value where variations in the level of costs can be related to the level of purchases of such services (again revealing values). Rather what we are cautioning against is the inference that costs can directly approximate benefits in the absence of these further data and analyses.

19 Cost-effectiveness analyses compare alternative options for delivering a specified outcome with the most efficient option typically being preferred.

Table 22.1 Various valuation methods applied to ecosystem services. Source: Bateman *et al.* (2011a).

Valuation method	Value types	Overview of method	Common types of applications	Examples of ecosystem services valued	Example studies
Adjusted market prices	Use	Market prices adjusted for distortions such as taxes, subsidies and non-competitive practices.	Food; forest products; Research & Development benefits.	Crops; livestock; multi-purpose woodland.	Godoy <i>et al.</i> (1993); Bateman <i>et al.</i> (2003)
Production function methods	Use	Estimation of production functions to isolate the effect of ecosystem services as inputs to the production process.	Environmental impacts on economic activities and livelihoods, including damage costs avoided, due to ecological regulatory and habitat functions.	Maintenance of beneficial species; maintenance of arable land and agricultural productivity; support for aquaculture; prevention of damage from erosion and siltation; groundwater recharge; drainage and natural irrigation; storm protection; flood mitigation.	Ellis & Fisher (1987); Barbier (2007)
Damage cost avoided	Use	Calculates the costs which are avoided by not allowing ecosystem services to degrade.	Storm damage; supplies of clean water; climate change.	Drainage and natural irrigation; storm protection; flood mitigation.	Kim & Dixon (1986); Badola & Hussain (2005)
Averting behaviour	Use	Examination of expenditures to avoid damage.	Environmental impacts on human health.	Pollution control and detoxification.	Rosado <i>et al.</i> (2000).
Revealed preference methods	Use	Examines the expenditure made on ecosystem-related goods, e.g. travel costs for recreation; hedonic (typically property) prices in low noise areas.	Recreation; environmental impacts on residential property and human health.	Maintenance of beneficial species; productive ecosystems and biodiversity; storm protection; flood mitigation; air quality; peace and quiet; workplace risk.	See Bockstael & McConnell (2006) for the travel cost method and Day <i>et al.</i> (2007) for hedonic pricing.
Stated preference methods	Use and non-use	Uses surveys to ask individuals to make choices between different levels of environmental goods at different prices to reveal their willingness to pay for those goods.	Recreation; environmental quality; impacts on human health; conservation benefits.	Water quality; species conservation; flood prevention; air quality; peace and quiet.	See Carson <i>et al.</i> (2003) for contingent valuation and Adamowicz <i>et al.</i> (1994) for discrete choice experiment approach.

The methods described above might appear straightforward. However, this is somewhat deceptive. Recall that the task of the economist is to estimate the value of goods in terms of the welfare they generate, rather than simply their market price. As mentioned, it is only under a set of fairly restrictive assumptions that we can take market price as a direct estimate of value (recall the park recreation example) and the adjustment process from the former to the latter is far from straightforward. However, even this route becomes impassable for goods which are devoid of market prices such as outdoor, open-access recreation, or peace and quiet. Revealed preference methods provide an approach to the valuation of goods such as these where an individual can only enjoy some non-market environmental good through the consumption of some market-priced private good. Here, economists make use of the ‘weak complementarity’ concept introduced by Mäler (1974) to examine how much individuals are prepared to spend on

the private good in order to enjoy the environmental good, thereby revealing the value of the latter. A number of variants of the revealed preference approach exist. For example, the travel cost method examines the expenditure and time that individuals are prepared to give up to visit environmental recreation areas. Similarly, the hedonic property price method typically examines the premium which people are prepared to pay in order to purchase houses in areas of higher environmental quality (e.g. quieter, less polluted neighbourhoods, and locations near parks). By controlling for other determinants (e.g. the number of bedrooms in a property), such purchases reveal the values people hold for these environmental goods.²⁰

While revealed preference techniques tend to be applicable to a relatively narrow range of goods, stated preference approaches such as contingent valuation and discrete choice experiment methods (see **Table 22.1**) should, in theory, be applicable to a wide range of ecosystem service goods,²¹ and

20 Notice that the hedonic property price approach examines the value of a flow of services as capitalised within house prices. A related approach is to model the relationship between the price of land and its attributes. Examples of such ‘Ricardian’ analyses include Mendelsohn *et al.* (1994), Schenkler *et al.* (2005), Seo *et al.* (2009) and Fezzi *et al.* (2010b). While revealed preference methods have been widely applied, they have various drawbacks and limitations. They often require a number of assumptions to hold as well as copious amounts of data and intensive statistical analysis.

21 The stated preference literature is vast but for a few examples focused upon ecosystem services: Naylor and Drew (1998), Rolfe *et al.* (2000), Hearne & Salinas (2002), Carlsson *et al.* (2003), Hanley *et al.* (2003), Huybers & Bennett (2003), Othman *et al.* (2004), Naidoo & Adamowicz (2005), Banzhaf *et al.* (2006), and Luisetti *et al.* (2011a,b).

typically they are the only option available for estimating non-use values.²² Such methods are defensible in cases where respondents have clear prior preferences for the goods in question or can discover economically consistent preferences within the course of the survey exercise. Where this is not the case, elicited values may not provide a sound basis for decision analysis. Such problems are most likely to occur when individuals have little experience, or poor understanding, of the goods in question (Bateman *et al.* 2008 2010a).²³ Therefore, while stated preferences may provide sound valuations for many goods, the further we move to consider indirect use and pure non-use values, the more likely we are to encounter problems.

While a number of solutions have been proposed for the problem of valuing low experience, non-use goods (Christie *et al.* 2006; Bateman *et al.* 2009b), we have to consider those cases where such values cannot be established to any acceptable degree of validity. The question of what should be done in such cases has generated much debate, but one approach is the adoption of 'safe minimum standards' to ensure the sustainability of resources (such as the continued existence of species) which are not amenable to valuation (Farmer & Randall, 1998). This would not negate the need for economic analysis, which would still play an important role in the identification of cost-effective approaches to ensuring the maintenance of sustainable ecosystems.²⁴

While much of the valuation literature consists of original research conducted for a variety of purposes, real world policy decisions often face time and resource constraints which preclude the undertaking of new field studies. To remedy this, a substantial literature has developed examining techniques for transferring values from original source to new policy situations. The value transfer literature embraces a number of approaches.²⁵ The simplest technique is to search for a prior source valuation study which addresses a good and context which approximates that of the policy application

and apply the value from the former to the latter.²⁶ This simple approach, often referred to as mean value transfer (because typically it is the average value which is transferred) is defensible, provided that source and policy good and context are highly similar. However, the limitations of source valuation studies mean that this is often not the case. In such cases, one option is to attempt to adjust the source values by incorporating differences between the source and policy contexts (e.g. differences in good characteristics, changes in valuing populations and their characteristics, different use costs or substitute/complement availability). One approach to such adjustment is to undertake a meta-analysis of results from previous studies, relating values to the characteristics of those studies and the goods and contexts valued. Such an analysis typically yields a regression model linking values to the characteristics captured in the available source data. As shown by Brander *et al.* (2006), the analyst can then apply the characteristics of a particular policy case to this model to estimate the relevant value.²⁷ An alternative approach to adjusting from source to policy values is to undertake a set of prior studies specifically designed to capture the effect of factors known to influence values, such as variation in the level of ecosystem service or changes in the spatial location of those services. Data from these studies are then analysed to yield a transferable, spatially explicit value function. The characteristics of any policy relevant site can then be fed into this model to estimate its corresponding value.

22.2.2 Total and Marginal Values

While the literature on ecosystem service valuations is developing rapidly, it highlights a variety of caveats regarding the application of such methods. Of these, one of the most serious problems facing the effective and robust valuation of ecosystem services is that there are gaps in our understanding of the underpinning science relating those services to the production of goods.²⁸ In addition, there is

22 Notice that we deliberately eschew the term 'intrinsic value'. The word 'intrinsic' is defined by the Merriam-Webster dictionary as 'belonging to the essential nature or constitution of a thing'. Therefore the intrinsic value of, say, an endangered British bird such as the bittern (*Botaurus stellaris*) (Eaton *et al.* 2009) belongs to the bittern and is not reliant in principle on human perception. Of course, humans can and do hold values for bitterns. These can include the use value held by birdwatchers and the non-use values which a wider group hold for the continued existence of the bittern as a species. However, these are anthropocentric rather than intrinsic values. Some would argue for notions of human-assigned intrinsic values (e.g. Hargrove 1992) but from a conventional economic perspective, many so-called 'intrinsic' values would instead be reclassified as non-use existence values. True intrinsic values (e.g. the value of the bittern to the bittern) could be protected by a property rights approach which makes it illegal to harm the species concerned. However, in reality such rules are more likely to be enacted and maintained when they are actually supported by anthropocentric non-use values. The issue of how far society is prepared to go to protect so-called sacrosanct rights is an interesting topic of ongoing heated debate.

23 A related problem is where variants of the stated preference approach provide survey respondents with heuristic cues (simple rules of thumb) regarding response strategies (Bateman *et al.* 2009b).

24 A related strategy, the implementation of offsetting compensatory 'shadow' projects validated for their ecological suitability (Klassen & Botterweg 1976; Pearce *et al.* 1990; FR 1995), would also generally require cost-effectiveness analyses. For an example of a cost-effective approach to species preservation, see Bateman *et al.* (2009c) and contrast this to the highly variable stated preference values for these projects given in Bateman *et al.* (2010a).

25 Examples of value transfers (sometimes called benefit transfers, although this is confusing as these techniques can also be applied to costs) and related meta-analyses for environmental goods include Desvousges *et al.* (1992); Bergland *et al.* (1995); Carson *et al.* (1996); Downing & Ozuna (1996); Brouwer & Spaninks (1999); Brouwer *et al.* (1999); Brouwer (2000); Barton (2002); Bateman & Jones (2003); Muthke & Holm-Mueller (2004); Ready *et al.* (2004); Brouwer & Bateman (2005); Johnston *et al.* (2005, 2006); Moeltner *et al.* (2007); Navrud & Ready (2007); Zandersen *et al.* (2007); Leon-Gonzalez & Scarpa (2008); Lindhjem & Navrud (2008); Johnston & Duke (2009); TEEB (2009, 2010); and Bateman *et al.* (2010c, 2011b).

26 Transfer databases such as The Environmental Valuation Reference Inventory (EVRI) have been developed to assist the search process for such applications.

27 Although it is important that such meta-analyses take into account any effect exerted upon values by the choice of valuation methodology in the source studies (see Bateman & Jones 2003).

28 Two problems are particularly highlighted: i) the availability of quantified data on changes in the provision of services over time and space under different scenarios; ii) quantified understanding of the interactions between ecosystems and their services, particularly under novel general stressors such as global climate change. These issues will require concerted action and high degrees of collaboration between the natural and social sciences.

a paucity of valuation studies and available data regarding the values of these goods. A further complex, yet important, aspect of the ecosystem service valuation problem is that even when overall stocks are at or above sustainable levels, the size of any given stock of natural assets may affect the value of changes in associated service flows. This can be illustrated in part through reference to the highly cited study by Costanza *et al.* (1997), which attempted to provide value estimates for the total stock of all ecosystem services globally. While that paper very substantially raised awareness of the application of economics to ecosystem assessments, particularly within the natural science community, the focus upon valuing total stocks has been criticised on a number of grounds (e.g. Heal *et al.* 2005).²⁹ In particular, very few policy decisions relate to total losses of ecosystem services. Instead, most decisions concern incremental, often relatively modest changes in natural assets and their service flows. Economic valuation of such changes requires an initial understanding of the value of changing a single unit of a stock. Economists refer to this as the 'marginal' value of the ecosystem service in question. Of course, if the value of a marginal unit is constant, then it is straightforward to go from valuing a single unit to valuing whatever number of units a given policy will create or destroy. However, an interesting phenomenon is that for many goods and services, marginal values will change with the total size of the stock, even when the overall stock level is above sustainable levels. **Figure 22.1** illustrates the relevant point here by contrasting the two cases: the first concerning the marginal benefit (i.e. the per unit value) of reducing climate change by increasing carbon storage; the second showing the marginal benefit of increasing the area of recreational greenspace. In both cases, we postulate a situation where there is a policy which changes land use so as to increase the provision of both carbon storage and land for recreation (e.g. through the creation of woodlands, which in turn generate both carbon storage and recreational visits).

Figure 22.1a shows a (virtually) constant level for the marginal value of carbon storage throughout the range of feasible projects within the UK. This reflects the simple fact that, using existing technologies whereby the bulk of terrestrial carbon storage is held in living biomass and soils, the UK is simply not big enough to capture sufficient carbon to significantly reduce the problem of climate change to the level where the marginal benefits of further carbon capture change. Only if carbon sequestration were to be undertaken on a truly global scale would it begin to significantly affect the potentially damaging effects of climate change and hence reduce the marginal value of further carbon capture. Here then, the total benefit value of the envisioned provision change is estimated by multiplying the (constant) marginal benefit of carbon capture by the increase in provision between the baseline and alternative scenario.

A more complex situation is shown **Figure 22.1b**, which concerns increases in the area of recreational land. Within any given area, while an initial provision of recreational land may be highly valued, once that is provided, further (marginal) units of such land in that area generate progressively lower increases in recreational value.³⁰ This pattern of diminishing marginal values is a characteristic of many goods (even carbon capture would exhibit such a pattern once climate change began to be significantly ameliorated).

The two parts of **Figure 22.1** also reflect the role of location in determining values. While the benefits of storing a tonne of carbon are spatially unconstrained (all individuals gain from this good), the benefits of increasing the size of a given recreational area are highly spatially confined, being disproportionately captured by those who live near to the site. This of course means that locating recreational sites near to population centres can substantially increase their value. Bateman *et al.* (2006) discuss the concept of 'distance decay' in such values. Note also that this raises the possibility of localised losses of stocks occurring even when regional, national or global stocks are maintained. This is likely to generate high spatial specificity in marginal values.

Figure 22.1b also illustrates why it may be unwise to attempt to estimate the total value of ecosystem stocks rather than the value of specified changes. A total value would be given by summing all of the values underneath the marginal value curve back to a level of zero provision. However, such a situation (e.g. the disappearance of all recreational land) may be highly unlikely to occur. Equally

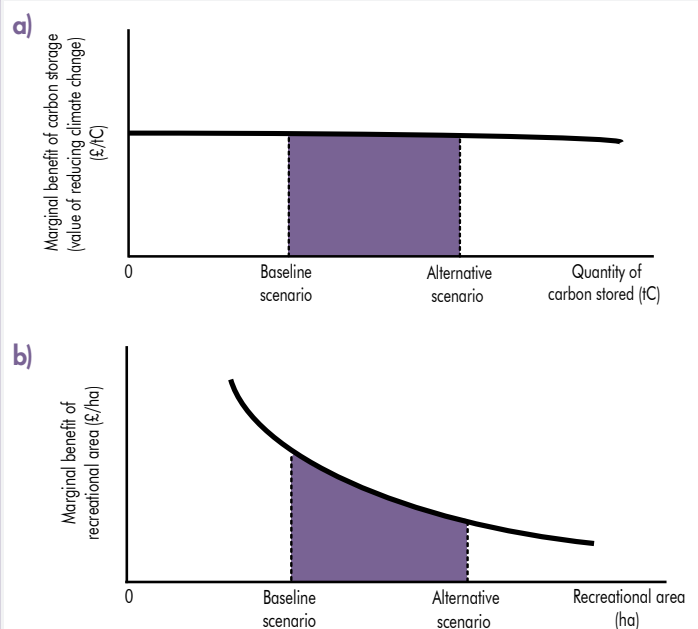


Figure 22.1 Marginal value curves for two goods: a) carbon storage (tonnes of carbon, tC) and b) recreational area (hectares, ha).

29 Note that while they do not provide solutions to these problems, Costanza *et al.* (1997) are aware of these issues and raise these within the discussion of their findings.

30 The Brander *et al.* (2006) meta-analysis of wetland valuation studies provides an example of such a case with per hectare values diminishing as the overall size of a wetland area increases.

importantly, it moves the calculation through areas of the marginal value curve which are entirely unsupported by data. Extrapolation out of the range of existing data is likely to generate unreliably high values.³¹ One common alternative to this approach is to use the current level of marginal benefits and hold this constant for the calculation of total values. However, just as the former approach is likely to generate overestimates of value, this latter method ignores the shape of the marginal value curve and is liable to lead to underestimates of total value. Both options are unattractive and unnecessary. The focus upon changes in value between feasible, policy-relevant scenarios is much more useful for decision purposes. Accordingly, this is the approach adopted for the UK NEA, which argues that for the valuation of any good we require:

- i) understanding of the change in provision of the good under consideration (i.e. the change in the number of units being provided) given changes in the environment, policies and societal trends;
- ii) a robust and reliable estimate of the marginal (i.e. per unit) value; and
- iii) knowledge of how ii) might alter as i) changes.

22.2.3 Discounting

So far in our discussions we have said nothing of the additional complications which arise where benefits and/or costs do not all occur in the present period but instead arise at some future time. This raises the issue of 'discounting': the process by which economic analyses reflect the preferences of individuals by reducing the present-day value of future costs and benefits, with this reduction increasing in intensity the further into the future we go.

The discounting procedure is based upon both theoretical and empirical arguments that individuals have a preference for receiving benefits sooner rather than later. This means that social values encapsulate within them conceptions of the impact of changes in the stock of all

assets (including natural assets) upon intergenerational well-being. However, both the form and rate of the discounting procedure are the subject of intense and very long-standing controversy.³² A critical element of this debate centres on whether, in selecting the social discount rate, a descriptive or prescriptive approach should be used (IPCC 1996; Dietz *et al.* 2007; Stern 2007). Put another way, should investments in natural assets be appraised purely in the light of information about preferences for the future as revealed in actual economic decisions, or is there room for the practitioner to make alternative moral judgments such as support for intergenerational equity?³³

Interestingly, recent discussions surrounding discounting have also broken new ground with the growing recognition that some environmental problems, such as climate change, are truly 'non-marginal' in the sense that this problem could end up shifting the global development path, say with 'business as usual' emissions of greenhouse gases (GHGs)³⁴ possibly leading to considerably lower future consumption levels than now (Hoel & Sterner 2007; Weitzman 2007; Dietz 2010). Indeed, the corresponding notion that the socially appropriate discount rate for short-term effects might differ from that relevant to long-term impacts (such as climate change) has caught hold in official practical guidance (e.g. HM Treasury 2003).³⁵ This results in the concept of time-varying discounting, where discount rates fall for more delayed costs and benefits (i.e. giving them greater emphasis in present values than if the short-term rate were maintained throughout an assessment).

22.2.4 Principles of Economic Analysis for Ecosystem Service Assessments: A Summary and Illustration

The methodology discussed so far in Section 22.2 allows us to define four key principles for the economic analysis of ecosystem services: *integration*, *valuation*, *efficiency*, and *distribution*. In this final discussion before presenting the key economic research undertaken for the UK NEA, we briefly

31 Note that it may indeed be that large reductions in a resource will involve losses of value which are very high. However, such reductions may begin to take analyses beyond the realm of marginal changes within which conventional economic assessments typically reside. A significant complication to this arises where we consider local rather than regional or national assessments. A given reduction in a resource might be nationally marginal but locally non-marginal, especially in areas with low stocks of the resource in question. A further issue is the possible non-marginal cumulative effects of individually marginal changes. This further emphasises the need, stressed at the outset of this chapter, to supplement consideration of the value of flows with stock assessments. This becomes even more important for resources with non-linear depletion paths, i.e. those which exhibit threshold effects whereby further exploitation leads to a rapid acceleration in stock depletion (e.g. when long-term overfishing suddenly breaches the capacity of the stock to replenish itself, leading to population crashes). Further complications include the problem of hysteresis in attempts to replenish depleted stocks. This arises for resources for which rates of exploitation have to be massively reduced before any recovery of stock levels begin. The extreme case here is when there is irreversible depletion of a stock. This irreversibility may be either physical or economic, the latter referring to cases where the costs of restoration become prohibitive. These issues are overviewed by Bateman *et al.* (2011a).

32 This is nowhere more evident than in the debate surrounding the recent Stern Review on the economics of climate change (Stern 2007). Subsequent argument has focused on the evidence that underpinned the central conclusion of the Review that "the benefits of strong, early action far outweigh the economic costs of not acting" (page VX). In particular, the focus of much of this discussion has been on the way in which this conclusion was driven by choices made in setting the social discount rate (that rate which is relevant for decisions made on behalf of, and reflecting the wishes of, society – it differs and is typically markedly lower than the market discount rate which reflects private investment decisions), including all of the fundamental reasons for discounting: pure time preference, the utility value of future increments in consumption and the extent to which it can be assumed that future consumption will be higher than consumption today (see, for example, Dasgupta 2007, Nordhaus 2007, Weitzman 2007).

33 Stern (2007) adopts a strong intergenerational equity position (and also addresses the problem of potentially non-marginal effects) through a very low discount rate giving a relatively high weight to future costs and benefits. However, Nordhaus (2007) and Weitzman (2007) argue that there is little evidence that such an approach is reflected in people's actual behaviour and choices and, thus, the empirical evidence suggests that the pure rate of time preference should take a higher value. Resolving such debates is far from straightforward and entails questions on which, to quote Beckerman & Hepburn (2007) "reasonable minds may differ" (p198).

34 When talking about GHG emissions the term carbon (or tonnes of carbon) is often used as shorthand for carbon dioxide (CO₂) or the equivalent of other GHGs (CO₂e) in the atmosphere. For the sake of expediency we will follow this convention here.

35 For a variety of views on the discounting debate see Groom *et al.* (2005), Dietz & Hepburn (2010) and Dasgupta (2001).

expand upon these principles before illustrating them via a couple of case studies.

Integration. The bedrock of an economic analysis of ecosystem services has to be an architecture of highly integrated natural science and economic modelling. Clearly, one cannot value any ecosystem service if the basic relationships determining the provision of that service are not understood and embedded within the analysis. This analysis needs to embrace the variation in the quantity and quality of ecosystem services across differing locations (spatial heterogeneity). This often arises as a result of underlying variation in the natural environment across different areas.

Valuation. While financial analysts are solely interested in the prices of marketed goods, true economic analyses value the full gamut of goods and services which contribute to human well-being, irrespective of whether or not those drivers of welfare are traded in markets. Appropriate application of the valuation methods summarised above allows the analyst to move from decisions which are dominated by market prices to ones which are supported and informed by social values. Again, marginal values may differ between locations, for example in response to changes in the quality of ecosystem services in different areas. Importantly, spatial variation can substantially affect the level of demand for a given service (e.g. demand for recreation sites will change with proximity to population centres) and this needs to be reflected in the aggregate value of changes in the supply of ecosystem services.

Efficiency. Efficient use of resources is always desirable, but especially so in times of austerity. Economic assessments are crucial when identifying efficient options for resource use as they allow the decision maker to compare across alternative options. Where resources are constrained, efficiency analysis allows the identification of optimal investments in ecosystem service provision in terms of their net benefits (benefits minus costs).

Distribution. Although many economic analyses apply an efficiency-based rule that the option offering the highest net benefit should generally be recommended, decision makers need to know about which groups gain or lose from these alternatives. Concerns regarding the perceived equity of different policy options will often play a major role in determining which alternative is adopted. Economic analyses have the potential to contribute significantly to such decisions if they are extended to assess the incidence of benefits and costs across society, both now and at future points in time.³⁶

A brief illustration of these methodological principles and techniques is provided by considering a case study concerning the issue of land use change (Section 22.3.17). Drawing on Bateman *et al.* (2002b, 2003) and Bateman (2009), we consider an economic analysis of a potential change from

farmland to woodland in Wales. The policy motivation for such an analysis comes from the fact that farming receives a higher rate of public subsidy than woodland, and that while most agricultural outputs have market prices (however imperfect), this is not true of various of the major benefits of woodland (notably open-access recreation and carbon storage). This raises the possibility of a welfare-inefficient situation in which we have a relative excess of farmland as opposed to woodland that justifies policy interest in such an analysis.

Given our first principle of economic analysis for ecosystem service assessment, the underpinning requirement of any such study is to ensure that we have an integrated understanding of the natural environment and the economic forces which dictate the possible agricultural and woodland uses for the full study area. This requires the integration of a long time series of highly detailed, spatially explicit information from across the study area. These data capture variation across time and space, encompassing issues such as local changes in soil characteristics and slope, fertiliser application and labour inputs, as well as more macro-level variables such as temperature, rainfall, the price of outputs and inputs, and subsidy levels. These data are brought together within integrated environmental-economic models which embrace both the physical and economic considerations required for informed decision making.

Figure 22.2 illustrates the outputs of such an environmental-economic analysis through a series of maps, all but the last of which show the annual social value of the various benefit streams which arise from the land use decision under consideration (while a separate analysis allows a contrast with the private values which determine land use in the absence of any policy intervention).³⁷ The first map in **Figure 22.2** shows the social value of agricultural output.³⁸ This is derived from an integrated environmental-economic model which reflects the highly heterogeneous nature of Wales, as shown in the relatively low values in the central upland areas, where poor soils and low temperature limit productivity, and the comparatively higher values in areas such as the lowland south west, where excellent soils and warm, moist conditions produce excellent yields.

Our second principle of economic analysis is now brought into play as we reject simple market prices in favour of estimating social values by adjusting prices to reflect subsidies and other transfers. A similar integrated analysis underpins the woodland timber values illustrated in the second map. Here, integrated models incorporating natural environment factors (such as tree species, soils, slope, topographic shelter, aspect), together with economic determinants (such as planting regime, management, genetic improvement), are combined to determine timber yield and, through further analysis, its social value (again based upon

36 While agricultural values are typically given in annual terms, for ease of comparison the long term discounted net present value of woodland has been annualised. For details of this and the private values of land use, see Bateman *et al.* (2003).

37 Official guidelines given in HM Treasury (2003) discuss both conventional and distributionally adjusted cost-benefit analyses. Although we consider distributional issues within our analysis of urban greenspace values (Section 22.3.21), generally there is a paucity of cost-benefit consideration of such concerns, suggesting that this may be a fruitful area for future research.

38 There are multiple agricultural sectors with the highest value dairy farming sector being illustrated here. For a comparison across sectors and between the social and private (farm gate) value of agriculture see Bateman *et al.* (2003).

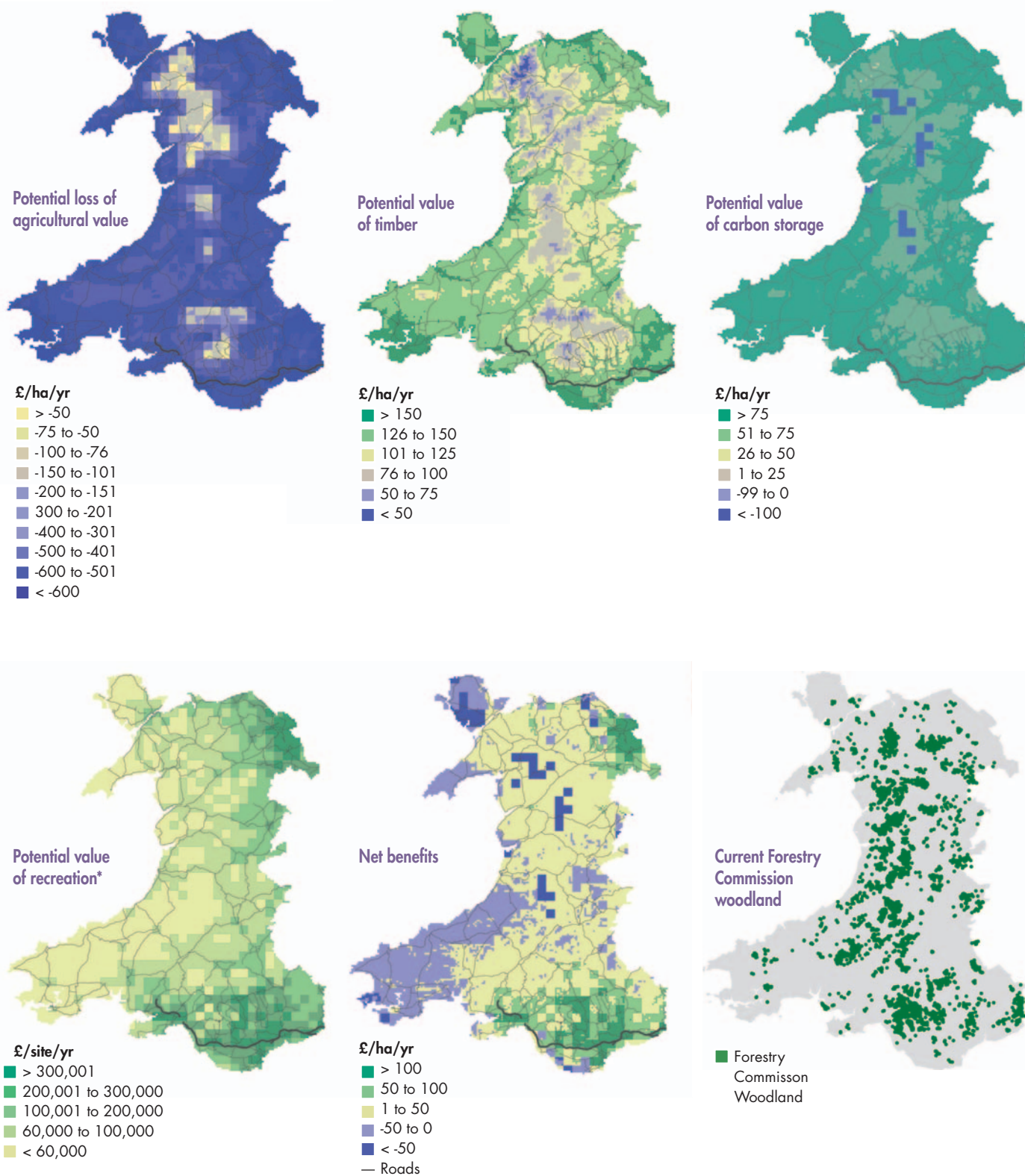


Figure 22.2 Economic values that would arise from a change of land use from farming to multi-purpose woodland in Wales (£ per year). *Unlike other values which are on a per hectare basis, the recreation is valued using one site per 5 km grid; this captures the fact that once a woodland site is established the per hectare recreational value of establishing a second site is not constant but diminishes significantly and to err on the side of caution we take that marginal value as being zero. Source: adapted from Bateman *et al.* (2002, 2003) and Bateman (2009) and reproduced with permission from Elsevier © (2009).

adjusted market prices). These values echo those of the agricultural sector, being higher in more favourable, lowland locations. Notice that the map covers the entire non-urban extent of Wales, indicating the timber values that would be achieved in each location, irrespective of its present use.

The third map of **Figure 22.2** illustrates net carbon storage values, combining the effects of both above- and below-ground biomass, soil carbon gains and losses and the effects of post-felling carbon emissions across different species and end uses. Whereas both of the previous value streams (agricultural produce and timber) involved adjusted market prices, here social values for carbon sequestration are taken from the literature on the value of avoiding damaging climate change (although the official UK policy value could be used as an alternative to this). Note that the values follow a generally similar pattern to those of timber, except for some very significant negative values in peatland areas (highlighted later in **Figure 22.2**) where the planting of forests dries out wetlands and results in net carbon release rather than storage.

The fourth map in **Figure 22.2** illustrates the value of recreation which would be generated through the establishment of woodlands. Here, the initial modelling phase requires information on the travel patterns of recreationists so as to capture the influence of population distribution and road infrastructure upon likely demand for visits to woodlands in differing locations. Values might be obtained through either revealed or stated preference methods or through some meta-analyses or value transfer exercise (as in this case). While the agricultural, timber and carbon storage values described previously all exhibit reasonably constant marginal values (as per **Figure 22.1a**), this is not the case for recreation, which is likely to exhibit diminishing marginal values (as per **Figure 22.1b**). So, in any given area, while an initial woodland area might generate substantial marginal recreation benefits, planting further woodland in the same area will yield lower marginal benefits.³⁹

The fifth map of **Figure 22.2** summarises all previous analyses by detailing the net benefits arising from a move from agriculture into woodland. Here the green areas indicate locations where woodland provides a higher shadow value than agriculture, while yellow and purple areas indicate locations where agriculture provides a higher value. It is interesting to note that the areas which generate the highest shadow values from conversion into woodland are in the north east and south east, a result which reflects the high populations in these areas and consequent elevated recreational values arising from afforestation. In contrast, the most negative shadow values from such conversion are shown by the purple areas corresponding to upland peats where afforestation causes major losses of soil carbon. This then provides the analysis of efficient resource allocation, which is our third principle of economic analysis for

ecosystem service assessments. It shows that there should be a major reshaping of land use in Wales which would introduce woodlands into lowland urban fringe areas.

This also provides the basic information for the consideration of distributional issues, which is our final principle for such economic analyses. One can see that the major beneficiaries of any such change would be urban populations. Whether or not this would be accompanied by losses for the rural farming community depends crucially upon how such change is implemented. Given that this change allows for net social gains, there is clearly scope for implementation via incentives; in effect, compensating farmers for facilitating such change. Given the massive ongoing reorganisation of the European Union Common Agricultural Policy (CAP), which gives great emphasis to the natural environment and the provision of ecosystem services, there is clearly scope here to avoid the inequity of one relatively small group losing out to provide benefits to the majority. However, economic analysis can only provide the raw information for such decisions, which are ultimately political.

The geographic distribution of net benefit shadow values is in sharp contrast with the actual distribution of forests shown as the dark green areas in the final map. The latter is driven primarily by market forces alone and hence ignores the carbon sequestration and recreational values and fails to adjust to the social values of farming and timber shown at the start of this figure. On the basis of market prices only being considered, agriculture outperforms woodland in all lowland areas, pushing forestry up the hill to low productivity areas where land prices are lower. This results in a distribution of woodland which is in marked contrast to its true social value; a finding which underlines the importance of using integrated environmental-economic analyses as the basis for decision making.

22.2.5 Methodological Summary

As Section 2 has shown, there is a growing research and policy interest in the application of economic analysis within ecosystem service assessments as a guide for decision making. Such analyses have to deal with the complexities of both the natural world and individual preferences and values for the goods to which it contributes. They are most applicable when decision contexts are framed in such a way as to highlight the welfare gains and losses stimulated by marginal changes in the provision of ecosystem services. Such changes are typically spatially explicit, providing an argument against straightforward aggregation valuation exercises. They must also be carefully scrutinised from an interdisciplinary perspective for the possible presence of threshold effects. A number of methods have been developed to address these complexities, and these form the tools employed within the various economic analyses presented

39 Similarly, existing forests constitute recreational substitutes for subsequent woodlands, lowering the marginal values of the latter (see, for example, Jones *et al.* 2010). In effect, while the map shown is valid for any initial decision and helps guide the optimal location for land use change, the analysis needs to be repeated after any such change to allow for these substitution effects. However, automation of this analysis makes this a straightforward operation. Note that in reality many ecosystem service goods exhibit non-linear marginal value functions. The marginal recreational values of a tiny woodland may be trivial and can initially increase with size but eventually exhibit declining marginal values. The same is likely to be true of landscape amenity benefits although this may well not coincide with the function for recreation i.e. the optimal size of woodland for recreation will differ from that for landscape amenity and the objective for the decision maker will be to maximise the overall net benefit.

in Section 22.3. Section 22.3 is organised so as to present reviews of previously published literature in Section 22.3.1 to 22.3.14. The remainder Section 22.3 (i.e. Section 22.3.15 to 22.3.21) presents valuation work specifically conducted for the UK NEA.

22.3 Ecosystem Service Valuations

The UK NEA Economics team undertook a wide-ranging review of ecosystems services derived for all UK natural habitats, considering the goods these generate and, where possible, their resultant values. These are, wherever possible, estimates of economic value. But where full economic valuation is unavailable simpler financial costings are included in order to give an indication of market impacts. Full details are given in the UK NEA economic reports referred to in Section 22.1; some financial/economic information is also included in a number of the UK NEA ecosystem science chapters (Chapter 5 to Chapter 16). In addition, work on the CSERGE SEER (Social and Environmental Economic Research) programme was accelerated to provide the analyses of agricultural food production, recreation, bird biodiversity (with the British Trust for Ornithology) and urban greenspace amenity.⁴⁰ This work is outlined in Section 22.3.15 to 22.3.21.

22.3.1 Non-agricultural Food Production

22.3.1.1 Marine food production⁴¹

The Marine environment plays a major role in food production. **Figure 22.3** details the weight and value of

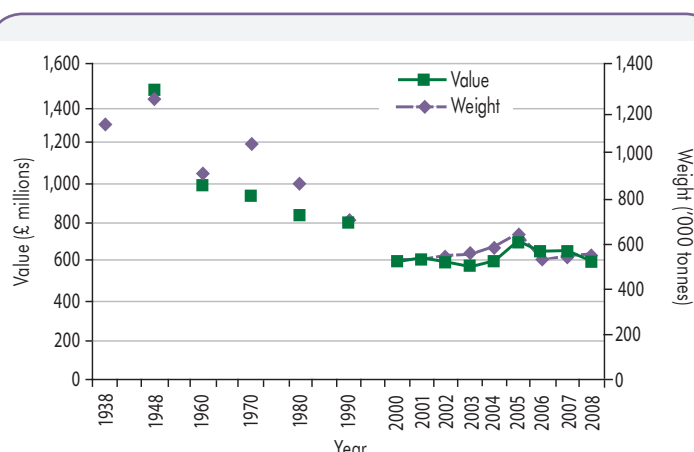


Figure 22.3 Landings into the UK by UK and foreign vessels: 1938 to 2008 adjusted to 2008 prices using the Retail Price Index. Source: data extracted from MMO (2010).

total landings of pelagic and demersal finfish and shellfish into the UK by domestic and foreign vessels from 1938 to the present day.

Noting the uneven time axis of **Figure 22.3**, we can observe a marked decline in landings throughout the second half of the 20th Century to a more stable trend in recent years. Although landings have clearly declined over the period shown, this has been only marginally reflected in prices, which are influenced by readily available imports and the introduction of alternative fish species over time. This has meant that the value of landings has roughly tracked their weight, falling from £1,465 million/yr in 1938 (in 2008 prices) to £596 million in 2008. While much of this is due to the inputs of the natural environment, a lack of data meant that it was not possible to separate out ecosystem services from other inputs to the value of fish.

One area that has seen considerable expansion is the farming or culturing of aquatic organisms (fish, molluscs, crustaceans and plants). Collectively known as aquaculture, this sector has increased dramatically in the UK, with the financial value of fish and shellfish farming rising by 132% over the period 2000–2006 (CEFAS 2008). In 2007, turnover from finfish farming in the UK was £327 million, while shellfish farming generated £23 million (Saunders 2010; CEFAS 2008).

The sustainability of UK fish stocks. The steadily growing influence of EU fisheries policies means that the landings data do not reflect the size and sustainability of UK fish stocks. With regard to stock analysis and sustainable extraction level, 18 species of finfish are routinely monitored and used to create a sustainability index for marine finfish stocks around the UK. This is not representative of the UK fisheries provisioning service, but does provide useful data for discussion, and also highlights the lack of UK-wide species stock data. Armstrong & Holmes (2010) report that for 2008, 50% of assessed UK stocks were at full reproductive capacity and were being harvested sustainably, an increase from 5% to 15% in the 1990s, and from 20% to 40% in 2000. While this is a positive trend, a number of scientifically assessed UK stocks continue to be fished at levels considered to be unsustainable, the majority are fished at rates well above the values expected to provide the highest long-term yield, and a number of other commercially important species remain unassessed due to inadequacies in the available data.

As fish stocks have declined, there has been an increase in the levels of human and technological inputs to substitute for the decreasing natural capital (i.e. fish) to maintain landings. Indeed, Thurstan *et al.* (2010) report that despite changes in the size of the fishing fleet, technological advancements, and improvements in fishing efficiency, UK bottom trawl landings per unit of fishing power (LPUP) have reduced by 94% over the past 118 years. The authors suggest that this decrease in LPUP reflects a decrease in fish stocks and indicates that fish catch globally has only remained stable in recent years because of an increase in fishing effort.

⁴⁰ Social and Environmental Economic Research (SEER) into Multi-Objective Land Use Decision Making. Funded by the Economic and Social Research Council (ESRC); Funder Ref: RES-060-25-0063. The UK NEA and SEER objectives are coincident in several respects and so the latter was rescheduled to help inform the former. The work on urban greenspace amenity was conducted in collaboration with Grischa Perino, Barnaby Andrews and Andreas Kontoleon.

⁴¹ This Section draws on Beaumont *et al.* (2010).

Aquaculture is a financially significant and growing sector (Chapter 12; Chapter 15). In 2007, turnover from finfish farming in the UK was £327 million, while shellfish farming generated £28 million (CEFAS 2008). Marine aquaculture contributes around 21% of the finfish and shellfish supplied to the fish processing sector (CEFAS 2008). The UK fish processing sector in total generated a financial gross value added flow of £490 million in 2007, within which aquaculture contributed around £105 million. A full economic assessment of the marine food production sector is not available, but it would need to account for, among other things, the externalities (e.g. possible impacts of pollution, effects on wild populations) of fish farming and not just its financial, value added contribution.

Given the complexity of the social and natural drivers affecting fisheries, it is very difficult to make any future projections beyond the next few years, and even these are prone to significant error. It is, however, widely agreed that the demand for fish will increase globally, although fish consumption rates within the EU are expected to remain stable. Wild capture fish landings are expected to show limited or no growth (and may even decline as many stocks are overexploited), with the increased demand for fish protein being met through aquaculture. An additional variable is climate change, which has been shown to alter fish community structure through changes in distribution, migration, recruitment and growth (Walther *et al.* 2002).

In order to move from the simple accounting approaches outlined above to a true economic analysis, we need to introduce the concept of a resource 'rent'. For fisheries, this is the difference between the total costs faced by those who fish and the total revenues arising from fish landings.⁴² As exploitation rates are increased, so this resource 'rent' declines. In a recent study, Cunningham *et al.* (2010) estimate the annual rent earned by Britain's fishing fleet at around £50 million per annum (p.a.) (although they acknowledge that this estimate is highly uncertain). However, the same authors claim that a reduction in fishing effort would both reduce total costs and allow stocks and hence total revenue to recover, such that annual rents might increase more than ten-fold.⁴³ Up until the latter part of the last century, UK fisheries were effectively open-access resources and as such, highly susceptible to the 'tragedy of the commons' (Hardin 1968) problem of overexploitation. Unfortunately, the excess fishing capacity built up historically still persists to some degree, resulting in excess fishing effort reducing rents. Cunningham *et al.* (2010) argue for a shift to a 'wealth'-based approach in which rental values are optimised by reducing excess capacity. This would fit well with a move towards sustainable management of natural stocks and the service flows they generate.

42 Note that total costs include what is termed 'normal profits', i.e. those that would be made if the fishery was being overexploited to the point where total revenues declined to equal total cost.

43 Cunningham *et al.* (2010) estimate that British fish stocks have the potential to produce resource rents in the order of £573 million p.a. Using a discount rate of 9% they estimate that the capitalised value of such rents would be £6.4 billion. Such inefficient over-exploitation is a characteristic of global fisheries. The World Bank & FAO (2009) 'Sunken Billions' report estimates that the difference between the potential and actual net economic benefits from marine fisheries is in the order of \$50 billion per year; equivalent to more than half the value of the global seafood trade.

44 This Section draws in part from Valatin & Starling (2010).

45 This Section draws in part from Morling *et al.* (2010).

22.3.1.2 Woodland-related food production⁴⁴

There is a burgeoning national (e.g. RS 2003) and international (e.g. Marshall *et al.* 2006) literature on the issue of recognising (and increasingly valuing) non-timber forest products (NTFP). In essence, NTFP include all the products obtainable from forest other than timber. While internationally this can include a very wide variety of products, within the UK the major value streams focus around wild foods such as mushrooms, berries and certain wild animals, of which one of the major groups is the variety of deer which now use woodland as a major habitat.

Six species of deer are currently found in the wild in the UK. Although data on UK deer populations and their change over time is generally sparse and approximate (see Hunt 2003; Ward 2005; Ward *et al.* 2008; Dolman *et al.* 2010), there is general agreement that wild deer populations have been increasing and now approach around 1.5 million animals (Spence & Wentworth 2009). Deer are associated with a range of ecosystem services, including recreational values associated with wildlife viewing (see subsequent discussions). They are also associated with various costs, including negative impacts on wood production (although estimates range from negligible costs up to £57 per ha; White *et al.* 2004), damage to gardens, and road accidents (Langbein 2006; Langbein & Putnam, 2006). However, increasing deer populations have led to a rise in culling and a consequent increase in UK venison supplies. No firm data are available on the annual value of this service flow, but one estimate from 2004 puts it at over £24 million p.a. (Tinch *et al.* 2010), although this primarily refers to stalking rather than venison values. A further £5 million p.a. in venison revenue is generated through the culling of deer by shooting estates purely for purposes of population control. The future value of this service is more difficult to forecast as, while culling has roughly doubled over the past 25 years to around 60,000 annually in Scotland, so venison prices have declined by almost 75% over the same period (MacMillan & Phillip 2010). Note, however that this is due in part to increasing import penetration (Munro 2003; MacMillan & Phillip 2010).

22.3.2 Biodiversity: Use Values⁴⁵

The Convention on Biological Diversity (CBD 1992) defines what is commonly referred to as biodiversity as "the variability among living organisms from all sources including, *inter alia*, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems" (Article 2, p.5). This definition has subsequently been broadened to embrace the diversity (a measure of variation between genes, species and ecosystems),

composition and relative abundance of living things. This complexity of definition is mirrored by the diverse roles of biodiversity within ecosystem services. Within this section we consider the variety of use-related values generated by biodiversity, while Section 22.3.3 considers non-use values.

Use values can be subdivided into two broad types:

- The role of biodiversity in the direct delivery of ecosystem services.
- The role of biodiversity in underpinning ecosystem service delivery.

We discuss each of these in turn below.

22.3.2.1 The role of biodiversity in the direct delivery of ecosystem services

Pollination, fertilisation and pest reduction effects upon food production. Evidence on the relationship between biodiversity and ecosystem service delivery is mixed. However, while some studies show little association (Anderson *et al.* 2008; Naidoo *et al.* 2008), in the greater number of experiments to date, increased rates of the ecosystem processes underlying ecosystem services are associated with increased numbers of species (Hooper *et al.* 2005; Hector & Bagchi 2007). In a recent meta-analysis of 446 studies of the impact of biodiversity on primary production, 319 of which involved primary producer manipulations or measurements, Balvanera *et al.* (2006) found that there is “clear evidence that biodiversity has positive effects on most ecosystem services” and, specifically, that there is a clear effect of biodiversity on productivity. Most of the evidence for this association is drawn from overseas. For example, Ricketts *et al.* (2004) estimated that pollination services to coffee plantations in Costa Rica can be worth up to nearly USD\$400/ha/yr (approximately £220/ha/yr at 2004 rates), or about 7% of farm income. However, evidence for the UK is scarce. An exception is provided by research for the UK NEA (outlined in Chapter 14) that estimates 20% of the UK cropped area comprises pollinator-dependent crops and note that a high proportion of wild, flowering plants depend on insect pollination for reproduction. This is considered a conservative estimate of the value of pollinators to UK agriculture of £430 million p.a. (see also POST 2011).⁴⁶ Similarly Bianchi *et al.* (2006) review the considerable evidence regarding the pest control services of biodiversity, noting that this appears highest in diverse landscapes. Valuations of this service are not provided, but appear potentially substantial.

As our brief discussion of threshold effects indicates, evidence of a valuable stock of ecosystem services, such as pollination, need not necessitate any policy action unless there is reason to believe that this stock is under threat. Certainly, there is evidence that proximity of semi-natural habitats is correlated with pollinator visits to crops (Tinch 2010). Furthermore, there has been an extremely large contraction of semi-natural and natural habitats (since the 1930s, some

97% of enclosed Neutral and Calcareous Grasslands in the UK have been lost; Fuller 1987). However, the evidence that this contraction has resulted in any fall in agricultural productivity is less clear. That is not to say that we are not close to a tipping point, but further high spatial resolution research is required looking at the mosaic of different land cover types before a definite assessment of any threshold effects becomes clear. Until then we are unable to say how much of the above pollination value might be at threat.

Maintaining genetic diversity. Maintaining crops’ wild relatives, rare breeds and landraces offers potential benefits to domesticated crops as well as insurance-type values. While there is a range of potential benefits to conserving such genetic diversity and international examples suggest that associated values can be substantial (Poysa 1993; Newton *et al.* 2010), the only evidence available from the UK to demonstrate the marginal values associated with their conservation are internal Department for Environment, Food and Rural Affairs (Defra) estimates in respect of the Millennium Seedbank (pers comm., Mallika Ishwaran, Defra 2011; taken from Defra’s Spending Review business case). Here, under various assumptions,⁴⁷ the value of genetic material in species in the seedbank likely to be extinct by 2050 gives a return of 26:1 on investment.

Bioprospecting. If biodiversity harbours potentially valuable species or compounds as yet undiscovered, bioprospecting may be an economically rewarding activity. Consequently, bioprospecting focuses on the world’s biodiversity hotspots. The marginal pharmaceutical value of a species is estimated to be moderate or small in biodiversity hotspots. Some commentators suggests that terrestrial values for the UK are likely to be relatively small (Morling *et al.* 2010), although marine values might be more substantial (Lloyd-Evans 2005). However, recent work from the Joint Nature Conservation Committee (JNCC 2011) provides at least one example of potentially significant terrestrial bioprospecting values in the form of treatments for Alzheimer’s Disease being derived from daffodils (*Narcissus pseudonarcissus*) and snowdrops (*Galanthus nivalis*). Given that treatment of dementia costs the UK economy £23 billion/yr (JNCC 2011), the potential value of such ecosystem services is clearly highly substantial and worthy of further investigation.

Biodiversity-related recreation. The direct appreciation of wildlife can generate substantial benefits, as evidenced by the widespread participation in activities such as birdwatching and the high price paid for certain flower bulbs from wild stock (e.g. snowdrops). These may be valued through observed behaviour (e.g. applying the travel cost method to valuing nature watching trips or estimating values through membership fees). The issue of recreation is addressed in Section 22.3.20. While that analysis examines evidence of habitat-related variation in recreation values, we acknowledge that this can only provide a relatively weak proxy for any biodiversity element in these values.

⁴⁶ See also the Insect Pollinators Initiative: www.bbsrc.ac.uk/web/FILES/PreviousAwards/pollinators-biesmeijer.pdf. The UK agricultural sector as a whole was worth £6.6 billion in 2009 and approximately 20% of the UK’s cropped area comprises pollinator dependent crops (pers comm., Mallika Ishwaran, Defra, 2011).

⁴⁷ These are: that all seeds are equally likely to be stored, and go extinct; that all seeds are equally likely to be those contributing to the economic markets depending on genetic resources; that the seedbank at Millennium Seedbank at Kew Gardens holds the only examples of seeds if they do go extinct; that extinction rates are those given by the Millennium Ecosystem Assessment (MA 2005).

22.3.2.2 The role of biodiversity in underpinning ecosystem service delivery

Morling *et al.* (2010) argue that there is evidence to suggest that increased rates of the ecosystem processes underlying ecosystem services are associated with increased numbers of species or genes. There are also a number of examples where simplification of ecosystems has potentially led to a net loss of services. However, valuation of such services requires an understanding of the following concepts:

- The infrastructure, or primary, value of biodiversity is related to the fact that some combinations of ecosystem structure and composition are necessary to ensure the 'healthy' functioning of the system.
- The insurance hypothesis states that enhanced biodiversity insures ecosystems against declines in their functioning because the more species there are, the greater the guarantee that some will continue to function, even if others fail.
- The resilience hypothesis may be characterised as an ecosystem's flexibility to reconfigure itself in the face of external shocks. It suggests that biodiversity per se may also have economic benefits if species richness enables an ecosystem, currently in a desirable state, to resist or recover from perturbations.

While there is evidence from both terrestrial and marine ecosystems that lends support to the insurance and resilience hypotheses (Morling *et al.* 2010; Beaumont *et al.* 2010), there is little information with which to quantify the magnitude of these values within the UK or the habitats and services for which they are most applicable. Empirical research is limited by gaps in our understanding of the underpinning science and a consequent lack of relevant data alluding to the primary value of ecosystems.

22.3.3 Biodiversity: Non-use Values⁴⁸

While there is substantial anecdotal evidence of non-use (existence and bequest) values associated with maintaining biodiversity, the estimation of associated values is somewhat problematic.⁴⁹ Unlike use values, we cannot observe behaviour regarding non-use values, neither are they reflected in productivity. Some commentators have argued that a lower boundary estimate of values might be provided by the payments provided by policies designed to promote biodiversity. Certainly such amounts are substantial and usually related to opportunity costs (e.g. the profits forgone by farmers when they agree to take on biodiversity schemes). For example, payments of £280/ha are available for additional Semi-natural Grasslands (Morling *et al.* 2010), while the Rural Development Plan for England (which is a development of the CAP agri-environmental schemes) will run from 2007 to 2013 with a budget of £3.9 billion. However, the use of public policy costs as a proxy measure

of biodiversity values has to be handled with caution, with the potential circularity of the valuation process being recognised. Given this, some would argue for the application of estimates of individual preference, with the most common approach to assessing the non-use value of biodiversity being via stated preference studies.

22.3.3.1 The Non-use Value of Biodiversity: Stated Preference Estimates

Stated preference (SP) valuations of what are principally non-use benefits typically fail to provide values at a UK level. However, one exception is the assessment of the benefits associated with the Environmental Stewardship scheme provided by Boatman *et al.* (2010); also see Christie *et al.* (2008). Unfortunately, results are reported for the joint bundle of both wildlife and landscape benefits and seem likely to also include elements of perceived use value. However, accepting that this cannot all be assigned to non-use biodiversity value and that it only applies to agricultural land within the Stewardship scheme (although this is likely to be a large proportion of farmland), nevertheless the UK-level sums estimated are substantial, ranging from £540 million to £1,262 million p.a. with a mid-range estimate of £845 million p.a. (all adjusted to 2010 prices). More recently Christie *et al.* (2010) estimate the value of the UK Biodiversity Action Plan (BAP) at £1,366 million p.a. Mallika Ishwaran, Defra (pers comm., 2011) contrasts this with a BAP cost estimate of £564 million p.a. (GHK Consulting 2010) to yield a benefit:cost ratio for conserving biodiversity of approximately 2.5:1.

Further national level SP estimates for terrestrial biodiversity include a value of £320 million p.a. to prevent the decline of nine bird species in the UK (Foster *et al.* 1998) and an estimated biodiversity value for British forests of £480 million p.a. (Willis *et al.* 2003; all values adjusted to 2010 prices). Leaving the terrestrial environment, McVittie & Moran (2010) use an SP analysis to estimate a UK value for halting the ongoing loss of marine biodiversity (through the introduction of a UK-wide marine conservation zone) of £1,714 million p.a. The same authors note that this benefit value easily outweighs the associated costs of such a scheme. Arguably, one of the areas where biodiversity non-use values have been most closely studied using SP methods is in relation to wetlands, to which we now turn.

Meta-analysis of stated preference estimates of biodiversity non-use values: the case of wetlands.⁵⁰ The perceived high cost of undertaking SP research, while in itself a subject of some controversy, has resulted in a considerable number of meta-analysis and related studies seeking to draw out generic findings and valuations from the literature.⁵¹ One of the sources of ecosystem services most frequently subject to such analyses is wetland habitats (see Brouwer *et al.* 1999; Woodward & Wui 2001; Brander *et al.* 2006, 2008).

⁴⁸ This Section draws in part from Morling *et al.* (2010).

⁴⁹ As their names suggest, existence value is that benefit which individuals gain from the pure knowledge that some entity (e.g. some species) will continue to exist, while bequest value is associated with passing on a stock of benefits to others (typically future generations although one might include present others here). Note that neither value category involves direct use of the resource by the valuing individual, hence they are 'non-use' values.

⁵⁰ This Section draws on Morris & Camino (2010).

⁵¹ The costs of any study, SP or otherwise, should always be assessed in cost-benefit terms taking into account the value of extra information they provide.

Wetlands deliver a number of important ecosystem service-related goods and so a single meta-analysis can provide a range of valuation estimates relevant to the UK NEA. Morris & Camino (2010) conclude that the recent meta-analyses of wetland valuation provided by Brander *et al.* (2008) provide the most appropriate value transfer function for valuation of UK wetland goods. The Brander *et al.* (2008) study draws upon 264 valuations from 78 European sites. Morris & Camino's (2010) reworking of the Brander *et al.* (2008) meta-analysis provides values for five ecosystem service-related goods:

- Biodiversity
- Water quality improvement
- Surface and groundwater supply
- Flood control and storm buffering
- Amenity and aesthetics.

For completeness, we present valuations for all of these goods within **Table 22.2**, although only biodiversity values are discussed here, with other values being discussed subsequently in this chapter.

Table 22.2 is divided into separate assessments for inland and coastal wetlands, reflecting the finding that in all cases, values for the latter exceed those for the former. Considering biodiversity values, these were principally non-use and are expressed as additions over a default value for wetlands which do not provide significant biodiversity habitat. Therefore, considering inland wetlands, the first result reported indicates that on average the meta-analysis of SP valuations estimates that a wetland which affords good quality biodiversity habitat generates a value of £454/ha/

yr more than one which does not offer such habitat. The second column calculates the total annual value of these (mainly) non-use biodiversity values on the assumption that all UK inland wetlands provide good quality biodiversity habitat. While this is clearly an upper bound assumption, it is true that most wetlands are indeed highly biodiverse areas (note that Morris & Camino (2010) considerably extend this analysis by calculating total values for UK inland and coastal wetlands, disaggregating these down to individual country levels and supplementing them with detailed case studies). However, this only tells us about the status quo situation, not the value arising from changes induced by policy or other drivers. To assess this we require a marginal value for a change in the area of such biodiverse wetlands. This is provided in the third column of each block of values. In both cases we see, as expected, that the value of such a marginal hectare of wetland is lower than the average value. This reflects the diminishing marginal values associated with increases in almost any good, including biodiversity. It is these values, of £304/ha of inland wetland and £1,866/ha of coastal wetland, which should be applied to any proposed change in the area of these habitats. As noted, we discuss the other values given in this table subsequently.

Stated preference estimates of the non-use value of biodiversity: caveats. The SP literature therefore suggests that the non-use value of biodiversity is substantial. However, some reservations can be identified regarding the use of SP methods for estimating these non-use values. Arguably, an invalid critique is that such studies can yield values which may be inconsistent with natural science assessments of what is required for sustainability. Stated preference studies reveal

Table 22.2 Estimated average, total and marginal values for specified ecosystem service-related goods provided by inland and coastal wetlands in the UK.* Source: Morris & Camino (2010).

Wetland type	UK Inland Wetlands			UK Coastal Wetlands		
No. of sites [†]	1,519			693		
Total area (hectares; ha)	601,550			274,613		
Ecosystem service-related goods	Total value of service assuming it is present in all UK inland wetlands [‡] (£ million/yr)	Average value of service where present (addition to default value) [§] (£/ha/yr)	Marginal value of service when provided by an additional hectare of new wetland [§] (£/ha/yr)	Total value of service assuming it is present in all UK coastal wetlands [‡] (£ million/yr)	Average value of service where present (addition to default value) [§] (£/ha/yr)	Marginal value of service when provided by an additional hectare of new wetland [§] (£/ha/yr)
Biodiversity	273	454	304	1,275	2,786	1,866
Water quality improvement	263	436	292	1,245	2,676	1,793
Surface and groundwater supply	2	2	1	514	16	12
Flood control and storm buffering	366	608	407	1,534	3,730	2,498
Amenity and aesthetics	204	339	227	1,081	2,080	1,394

* Values are area-weighted estimates for all UK inland wetland sites using the Brander *et al.* (2008) benefit function and CORINE land use data sets. All values are given in (£, 2010) prices.

[†] Data on the number and area of wetlands are drawn from the European CORINE Land Cover Maps (Morris & Camino 2010).

[‡] Default total value of the existing inland wetland stock, assuming that none of the ecosystem services in the table apply, is £182 million/year for UK inland wetlands and £509 million/year for UK coastal wetlands.

[§] Default average values (where all of the ecosystem services specified in this table do not apply) are £303/ha/year for UK inland wetlands and £1,856/ha/year for UK coastal wetlands.

[§] The per hectare value of services associated with additional new wetlands is lower than the average per hectare value of existing wetlands. This reflects the diminishing marginal value of additional wetlands.

the unsurprising result that individuals attach much higher values to charismatic megafauna such as larger mammals or familiar birds rather than small reptiles and amphibians (Morse-Jones *et al.* 2010). Similarly, habitats yielding high amenity values, such as water meadows, are valued more than, say, mudflats (Bateman *et al.* 2009a). Of course, from a natural science perspective, lowly amphibians and mudflats might form a vital element in the food and habitat webs which ultimately support those animals which are considered of greater value. This, however, is not a problem which can be laid at the door of SP techniques; rather, these appear to be reasonable representations of preferences which may have little to do with sustainability requirements.

A more pertinent critique is that SP assessments assume that, at the point of expressing willingness to pay (WTP) amounts, the SP respondent comprehends biodiversity goods in the same absolute sense that they would comprehend everyday goods. While SP studies can certainly enhance comprehension through the provision of appropriate information,⁵² there is evidence that in some biodiversity and animal welfare valuation studies respondents may not have the stable preferences required for economic valuations (see, for example, Bateman *et al.* 2008), resulting in stated values which are malleable (Loomes & Sugden 2002) and may not provide robust evidence regarding true underlying WTP (Cameron 1992; Harrison 1995; Kahn *et al.* 2001; Christie 2007).⁵³ Morris & Camino (2010) discuss at length the caveats that need to be borne in mind when working with meta-analyses of SP valuations. A more fundamental critique of the applicability of all economic approaches within this area is given by Craig *et al.* (1993).⁵⁴

22.3.3.2 The non-use value of biodiversity: legacy values

While there is no ideal measure of the non-use value of biodiversity, an alternative to SP studies is provided by examining actual payments for non-use-related wildlife conservation.⁵⁵

Pearce (2007) notes that private donations to charities are relatively small (in part because of the transaction costs individuals face in banding together), and instead focuses upon UK overseas expenditure on biodiversity of roughly £65 million p.a. (at 2010 prices). However, the policy-led determination of such amounts means that they cannot be taken as a robust estimate of values. A more robust, although very much lower bound source of individualistic valuations, is provided by examining legacies to environmental

charities. Legacies can be argued to represent a pure non-use value: individuals leaving a charitable bequest to an environmental organisation in a will, for the purposes of supporting their conservation activities, will not experience the benefits of this work.

Mourato *et al.* (2010) examine the value of legacies to the largest environmental charities in the UK: The National Trust, the Royal Society for the Protection of Birds (RSPB), and the National Trust for Scotland. Atkinson *et al.* (2009) estimate that in 2007, only 6% of all deaths in Britain resulted in a charitable bequest (with this percentage rising considerably with the size of the estate). But despite the relatively small proportion of estates leaving a charitable bequest, legacies are a major source of income for charities. In 2008/09, charitable giving by individuals was almost £6 billion to the top 500 fundraising charities (Pharoah 2010). Legacies represent almost one-quarter of this total (£1.4 billion), with almost three-quarters of charities reporting income from legacies. Although environmental charities rank seventh in terms of total fundraised income, they rank fourth in terms of legacy income (within the top 500 charities in the UK) after cancer, animals and general social welfare charities. Legacy income is an important source of revenue for environmental charities, comprising almost 30% of all their fundraising income. Overall, the total legacy income earned by environmental charities in 2008/09 was £97 million, which represents 7% of all charitable legacies (Pharoah 2010).

Table 22.3 details the top five environmental charities according to the fundraised and legacy income they earned in 2008/09. Three of these charities (The National Trust,

Table 22.3 Fundraised and legacy income of top five environmental charities (2008/09). Source: data extracted from Pharoah (2010).

Environmental charity	Legacy income (£ million and % of total fundraised income)		Total fundraised income (£ million)	Rank within top 500 charities
The National Trust	42.8	44%	97.8	12
Royal Society for the Protection of Birds	26.6	41%	64.9	16
WWF UK	8.1	22%	37.4	32
The Woodland Trust	8.2	40%	20.6	58
National Trust for Scotland	4.0	21%	18.8	61

52 Note that an association between the information provided and SP values is not an indication of bias in the latter values; indeed, we would expect such a link and observe this in everyday values (Munro & Hanley 1999). Furthermore, different forms of what is objectively the same information can substantially hamper or enhance its comprehension (Bateman *et al.* 2009b). However, what is not consistent with economic theory is where values based upon the same information vary purely because of the way that questions are framed (Loomes & Sugden 2002).

53 Note that much of the existing literature does not conform to best practice guidelines (e.g. Bateman *et al.* 2002a) and therefore cannot be taken as clear evidence of the non-applicability of stated preference methods for valuing non-use values for biodiversity.

54 We are grateful to Nigel Cooper for highlighting this critique.

55 In lieu of biodiversity values, Morling *et al.* (2010) consider the cost of managing biodiversity on the strong assumption that the political biodiversity targets and legal mechanisms that have been brought in to support biodiversity are a reflection of public preferences. Annual costs for the UK at 2010 prices are as follows: Biodiversity Action Plans = £837 million (although this contrasts the previously cited BAP cost estimate of £564 million p.a. given by GHK Consulting (2010); additional costs for protected areas = £217 million; marine biodiversity costs = £63 million. This gives a total UK cost for these biodiversity initiatives of £1,117 million p.a. However, the assertion that policy spending is a good indicator of underlying benefit values is a very strong assumption and may well not hold. Given this, we do not argue that this should be taken as a robust indicator of non-use biodiversity value.

RSPB and WWF UK) rank within the top 50 largest charities in the UK. Environmental legacy income is considerable, with the National Trust attracting the largest number of legacies, constituting some 44% of their total fundraised income at almost £43 million (Pharoah 2010). Had donors intended their legacy income to be spent on National Trust countryside, RSPB reserves or National Trust for Scotland countryside, we would have been able to estimate a legacy-based non-use value of around £219/ha of National Trust countryside, £190/ha of RSPB reserve and £53/ha of National Trust for Scotland's Scottish countryside for 2008/09, respectively. However, donors' preferences about the allocation of their legacies are not known and these figures are therefore liable to overstate the environmental component of these legacies. That said, further analysis suggests that for the two largest environmental charities (National Trust and RSPB) the total value of annual legacies has increased significantly over the last two decades and the proportion of estates leaving a legacy to environmental causes has risen, even in the light of falling death rates.

Legacies are interesting proxies for non-use values in that they are observable in the market and not reliant upon SP data. But clearly, they capture only one element of environmental non-use values, i.e. those that are reflected in the marketplace at the time of death. Further research is needed to ascertain the magnitude of the non-use values that are not reflected in the market. Moreover, there are major knowledge gaps in our analysis. In general, very little is known about charitable bequests in the UK. Data on charitable bequests, estates and demographic characteristics of donors is not easily accessible, particularly for analysis over time. Equally, comprehensive data on charitable giving over time, from the perspective of the recipient organisations, and covering a wide range of organisations, is not freely available.

22.3.4 Timber Production

The total quantity of wood produced in the UK has risen substantially over the past three decades, more than tripling since the mid-1970s to over 8 million green tonnes currently, as Coniferous Woodlands planted in the 20th Century have matured. Forecasts suggest that UK softwood production from existing woodlands will continue to rise over the next decade, and then decline until the mid-2050s (Valatin & Starling 2010). However, during the same period, world softwood timber prices have collapsed from £35/tonne in the early 1970s to about £12/tonne at present (all at 2010 prices). This appears to follow a longer term downward trend. Given that domestically produced wood accounts for under one-fifth of the total used in Britain, there does not seem to be a purely timber-based case for a domestic forest sector on social value grounds (although clearly there is a private financial case for such production and a reduction of imports may reduce transport-based GHG emissions). However, the case is much stronger when we consider the wider values

of UK woodland in relation to ecosystem services, with recreation and carbon storage values being particularly substantial and both exceeding timber values (recreation and carbon storage values are considered subsequently in this chapter). The increasing significance of such ecosystem service values in the case of broadleaved woodland is reflected in a halving of hardwood production since the mid-1970s, reflecting a shift in management objectives by state sector bodies including the Forestry Commission away from timber production and towards the provision of multiple ecosystem services.

22.3.5 Carbon Storage and Greenhouse Gas Flux: Marine and Coastal Margins⁵⁶

22.3.5.1 Coastal Margins

Biomass and sediments in Coastal Margins and the Marine environment raise the potential for sequestration or release of GHGs. In the case of Coastal Margin habitats, carbon sequestration is primarily provided by Sand Dunes, Saltmarsh and uncultivated Machair, although carbon sequestration rates are not available for the latter. The second half of the 20th Century has seen a reduction in the area of both Sand Dunes and Saltmarsh in the UK, with the former falling most rapidly. These trends are expected to continue through the first half of the present century and overall, are expected to result in declines in sequestration within UK Sand Dunes of more than 80,000 tonnes of carbon dioxide per year (tCO₂/yr) and within Saltmarshes of around 35,000 tCO₂/yr.⁵⁷

Applying the Department of Energy and Climate Change (DECC 2009) carbon sequestration values (which are based on avoided damage costs calculations as discussed in Section 22.2.1) to these estimates allows us to derive marginal (per ha) values for changes in storage within these coastal land categories. Wide variations in storage capacity estimates mean that for Sand Dunes, these values range from £32/ha/yr to just over £240/ha/yr, whereas the higher sequestration capacities of Saltmarsh yield values ranging from £60/ha/yr to around £620/ha/yr. Combining these marginal values with data on expected changes in areas for each habitat type yields suggests that in 2010 UK Sand Dunes will sequester carbon at a rate of nearly £8 million p.a. Despite the expectation that the area of sand dune will reduce over the next half century, the roughly six-fold increase in the planned DECC carbon sequestration value between 2004 and 2060 means that by 2060, UK Sand Dunes are expected to sequester nearly £40 million of carbon p.a. (in 2010 prices).⁵⁸ A similar pattern arises with UK Saltmarsh, with a shrinking area being offset by a rising carbon price to yield an increasing annual value. Annual values for carbon sequestration in UK Saltmarsh are expected to rise from just under £11 million in 2010 to over £63 million p.a. in 2060 (again at 2010 prices). The spatial distribution of these values is uneven, with most Sand Dune sequestration occurring in Scotland, and the majority of carbon fixing by

⁵⁶ This Section draws in part from Beaumont *et al.* (2010).

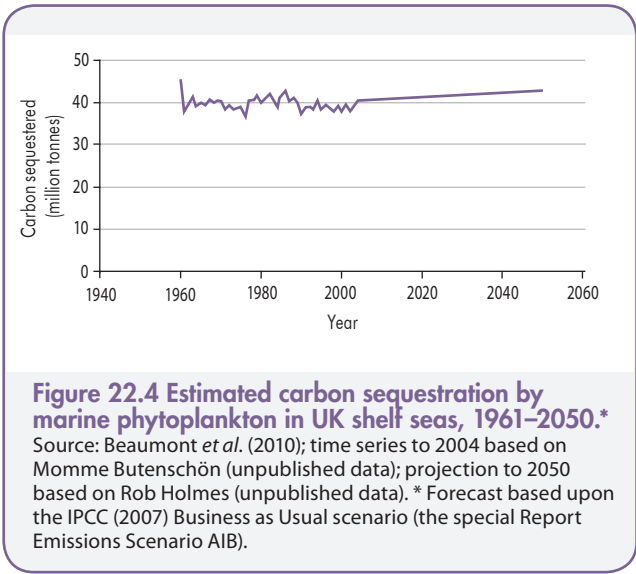
⁵⁷ Sand Dune estimates from data and forecasts for the period 1900–2060 (Jones *et al.* 2010). Saltmarsh estimates from data and forecasts for the period 1945–2060 (Jones *et al.* 2004, 2008, 2010; Beaumont *et al.* 2010).

⁵⁸ The stock of carbon in Coastal Margin vegetation and soils is estimated to be at least 6.8 megatonnes of carbon. However, there are insufficient data to determine how this may change.

Saltmarsh arising in England.⁵⁹ A reorientation of coastal protection and defence policy in recent years has meant that a number of new saltmarshes have been created on the eastern coast of England. Economic assessments of this so-called managed realignment policy are presented later in this chapter. **Table 22.4** summarises the various results concerning Coastal Margin sequestration of carbon.

22.3.5.2 Carbon sequestration in Marine habitats

The Marine habitat plays a significant role in the global carbon cycle although, as detailed in Chapter 12, there are minimal data readily available to quantify the extent of this role, or indeed even the total stock of carbon stored within the Marine habitat. What is clear is that, at any point in time, large amounts of carbon are stored in marine phytoplankton (Davis 2007). **Figure 22.4** details estimates of the historical levels of this storage in UK shelf seas from 1961, together with a forecast out to 2050. Analysis suggests that there may be some growth in forecast levels, but that, at present, there is no clearly significant trend. However, even if this were proven, it would not illuminate whether or not there is any net change in carbon storage over time. For marine carbon to be considered permanently sequestered it must either sink to the deep ocean, via the ‘biological carbon pump’, or be buried in the benthic environment. The UK waters assessed in this analysis are primarily shallow shelf seas and the currents in these waters mean that it is unlikely that the carbon fixed by primary productivity in UK waters will be transported to the deep oceans. It is also unlikely that the carbon will be buried in the benthic environment as the carbon is more likely to be labile (subject to change), and therefore more accessible and likely to be ‘processed’ and kept within the marine ecosystem. That said, the massive



levels of carbon involved in these processes suggests that further research into the processes and any underlying trends may be worthwhile.

22.3.6 Water Quantity and Quality⁶⁰

Freshwater habitats, comprising open waters, wetlands and floodplains, provide a range of ecosystem services associated with the provisioning and regulation of water quantity and quality. In turn, they generate a range of final goods, including for example public water supply, water for habitats, recreation, amenity and heritage.⁶¹ These aspects of freshwater ecosystems are also considered in other sections of this chapter.

22.3.6.1 Water quantity

The freshwater ecosystem regulates the provision of water for human use. Water is vital to life and hence it is not meaningful to try and put finite estimates on its total value. However, at least in the UK, there is no feasible scenario in which a total value for water would be needed for decision making. Instead economic analysis focuses upon feasible marginal changes in supplies.

About 22 billion cubic metres (m³) of water are abstracted in the UK each year, 52% from rivers and lakes, 11% from groundwater and about 37% from tidal waters (mainly used for cooling; EA 2009e; SEPA 2004). Of the 13 billion m³/yr extracted from non-tidal sources in England and Wales, about half is used for public water supply. A further third is used for electricity power generation. Industry takes about 10% and aquaculture and amenity about 9%. Spray irrigation accounts for less than 1% of total abstraction, but this is concentrated in the relatively dry Anglian water region in summer. Total reported abstraction quantities have remained more or less constant over the last 15 years (EA 2010).

Prices charged for abstraction do not reflect the full value of water, either in its natural state or in any particular

Table 22.4 Summary of the quantity and value of Coastal Margin carbon storage (tonnes of carbon dioxide; tCO₂). Values assessed as avoided damage costs in 2010 prices. Source: Beaumont <i>et al.</i> (2010).	
Units	Estimates
Quantities (t CO ₂ /yr)	Sand Dunes: decrease of 80,168 tCO ₂ /yr
	Saltmarsh: decrease of 34,774 tCO ₂ /yr
Marginal values (£/ha/yr)*	Sand Dunes: sequestration value £32.25–241.49/ha/yr
	Saltmarsh: sequestration value £60.63–622.30/ha/yr
National (UK) values (£/yr)	Sand Dunes: 2010: £7.98 million/yr; 2060: £39.13 million/yr (an increase of £31.15 million/yr)
	Saltmarsh: 2010: £11.93 million/yr; 2060: £63.22 million/yr (an increase of £51.29 million/yr)
	2010: Value of carbon dioxide sequestration: £268/ha
	2060: Value of carbon dioxide sequestration: £1,420/ha
* These marginal values imply a total UK stock value from Sand Dunes, Saltmarsh and Machair of £1,282 million in 2010 prices. However, given that changes to this entire area are not credible, this is not a policy relevant value.	

59 Calculated using the mid-range carbon price and mid-range sequestration rate.

60 This Section draws in part from Morris & Camino (2010).

61 See Table 9.1 in Chapter 9.

applications. Rather, they reflect the cost of managing the licensing system and there is concern that this leads to inefficient use. Water prices vary from £0.003 to £0.06/m³ for abstracted raw water, through to £1.50/m³ for metered, treated, potable water piped to households. These cost-based prices grossly underestimate the very considerable consumer surplus that water users enjoy over and above the prices paid for this essential good.

The Scottish Government provides the most comprehensive assessment of water values and these are thought to be broadly indicative for the UK in general (SEPA 2004; Moran & Dann 2008).⁶² As demonstrated in **Table 22.5**, the value of water varies considerably between uses. The marginal value for treated water ranges from £0.50/m³ to £1.20/m³. For raw water, the marginal value for irrigation water ranges between £0.23/m³ and £1.38/m³ for the Scottish case, comparable with values well in excess of £1.5/m³ for irrigated potato and salad crops in eastern England (Knox *et al.* 1999; Morris *et al.* 2004). Marginal values for raw water vary considerably according to industrial processes, highest where high water quality is required for the chemicals industry and whisky manufacturing. The energy sector shows relatively low marginal values for water used for cooling but for large throughputs. The value of water for hydropower is particularly sensitive to assumptions about the economic price of energy and the cost of alternative sources. **Table 22.5** also shows the relative use of abstracted water across the sectors, but it is not clear whether the estimates are entirely comparable between the countries of the UK.

Fresh water has a value *in situ* in the natural environment, supporting the range of services referred to elsewhere in this

chapter, such as biodiversity, recreation and property values. A survey in southern England of household WTP to leave water in the environment in situations where abstraction could lead to environmental damage produced an estimate of £0.30/m³ per day in 2010 prices (Jacobs 2008).

However, while natural habitats are obviously the source of such supplies, it is unclear how these are liable to change and what the implications are for water provisioning. For example, Tinch *et al.* (2010) note that mountainous areas are major providers of water but there is no clear association between changes in the natural environment in these areas and water supply levels. Rather, the major contributors to variation in water quantity supplies in such regions are due to human and manufactured capital inputs such as damming. Such values cannot readily be attributed to ecosystems. They can, however, indicate the value of services provided by Freshwaters where their supply, for a variety of reasons, is limited.

There is concern about how development pressures, exacerbated by climate change, could affect the capacity of Freshwater ecosystems to provide sufficient water for people. Reduction in the amount of water available for abstraction could result in i) the loss of value from some water uses and/or ii) extra costs of providing water from alternative sources or adopting water saving technologies. 'Unsecured' sources such as for irrigation and industrial/mineral washing are likely to be most vulnerable to variations in supply. This may justify additional expense of securing water by, for example, winter storage reservoirs. High value uses of water, such as those associated with public water supply, clearly justify relatively high investment to improve water security.

Table 22.5 Estimates of the value of water use.* Source: water value use data is from SEPA (2004); valuation assumptions and estimated abstraction data for Scotland is from Moran & Dann (2008); estimated abstracted data for England and Wales is from the Environment Agency (2010). Note the abstraction estimates are not comparable. Amounts of water in cubic metres (m³).

Sector	Water value in use for Scotland (2004 prices)	Valuation assumptions: MV (marginal values); AV (average values); TV (total values)	Estimated abstraction in Scotland (million m ³ /year)	Estimated abstraction in England and Wales (million m ³ /year)
Households (treated water)	50–120 pence/m ³	MV for treated water only, based on WTP [†] estimate	876	6,038
Agriculture–irrigation	23–138 pence/m ³	MV based on value added	57	72 (+19 for non-irrigation uses)
Aquaculture	0.126 pence/m ³	AV assumes avoided cost of waste disposal	1,582	1,203
Salmon angling	£175/day	TV benefit transfer estimate	-	-
Industry	4–37.5 pence/m ³ (e.g. 16 pence/m ³ paper and pulp industries; 35 pence/m ³ chemical industries)	MV benefit transfer from Canadian industry study	675 chemicals, food, textiles and paper	1,151
Energy	0.049–0.817 pence/m ³	MV comparative cost of alternative energy sourcing: coal, gas, windpower	23,755 hydro throughput; Non-hydro 3,783 including tidal	4,012 non-tidal 6,672 tidal

* All monetary values derived from Scottish data.

[†] Willingness to pay.

62 One unpublished yet interesting contrast is provided by NERA economic consultants for Thames Water which estimates the value of lost output in London from water-use restrictions during the 2000 droughts at around £174 million a day. This impact would be expected to increase as a result of climate change where, under a medium emissions scenario, summer mean precipitation in the south east is expected to fall by 23%, creating the imperative for more efficient water management.

Measures to secure water for nature conservation may be justified, especially in protected areas. Failure to restrict abstraction in the face of declining Freshwater resources would compromise the non-market ecosystems services referred to elsewhere in this chapter.

In the long term, the economic value of freshwater provisioning will reflect the costs of achieving an appropriate balance of the demand for and supply of water. On the demand side, the Environment Agency reports that measures such as compulsory metering to reduce household water consumption by a target of 15% (from 150 to 130 litres/day) could cost between £1.40 and £1.6/m³ (EA 2009e). By comparison, options to enhance freshwater supply appear more expensive, namely surface and ground water development (£1–£5/m³), reservoirs (£3–£10/m³) and desalinisation (£4–£8/m³). A detailed review of water supply options (Mott MacDonald 1998), however, estimated incremental average costs for reservoir development ranging between £0.21/m³ and £1.36/m³ of water delivered in a given year in 2010 prices, assuming a 50% annual utilisation rate.

Increased investments may be required in future in order to avoid pressures on Freshwater habitats associated with changes in climate and/or demographics.⁶³ A moderate climate change scenario could reduce water available for immediate abstraction by 10% by 2060, equivalent to about 1.4 billion m³/yr for the UK at current levels of abstraction. Assuming water storage and transfer costs of between £1.0 and £1.5/m³ for large-scale provision, securing this amount of water would cost about £1.4 to £2.1 billion/yr for the whole UK population. (This assumes that there are similar abstraction rates across the nation, equivalent to about £23 to £35/yr/capita of population affected). These investment costs could be higher if the climate change impact is greater and the growth in water demand is unconstrained. While these figures do not estimate the value of water services provided by Freshwater ecosystems, they indicate the equivalent cost of securing water supplies for

use while maintaining the non-market ecosystem services of rivers, lakes and aquifers. In some cases, investments in supply enhancement and regulation may also achieve environmental enhancement.

One assessment of the potential marginal value of changes in ecosystems upon water supply is provided by Morris & Camino (2010). **Table 22.6** details estimates of average, total and marginal values for surface and groundwater supply provided by inland and coastal wetlands in the UK. However, while these are significant, amounting to more than £0.5 billion p.a., the marginal values associated with expansions of wetlands appear relatively minor. It is noted that inland wetlands, particularly, help to reduce variations in water flows and levels.

22.3.8.2 Water quality

Water quality is a major determinant of the capacity of the Freshwater ecosystems to provide a range of market and non-market services. It is important here to distinguish between the total value of water quality and the value of a marginal change in quality. As discussed below, the quality of most water bodies in the UK is moderate to good, according to the EU Water Framework Directive (WFD) classification. Much of the discussion below refers to a change in quality around the current position, recognising the significant ongoing measures to protect water quality by the water industry and others. Clearly, a major deterioration in the quality of a freshwater body could result in complete loss of some ecosystem services and final goods, such as drinking water, irrigated crops, bathing and fishing, or require major expenditure to mitigate the consequences of loss of quality. Within the limits of the available information, the assessment here focuses on selected marginal changes from the current situation, mostly associated with the WFD.

Market benefits associated with water quality. The quality of water that is abstracted and used will obviously affect a range of market benefits for particular sectors and groups such as water companies, those involved in

Table 22.6 Estimated average, total and marginal values for surface and groundwater supply provided by inland and coastal wetlands in the UK.* All values are given in £, 2010 prices. Source: Morris & Camino (2010).

Ecosystem service-related goods	No. of sites [†]	Total area (ha)	Average value of service where present (addition to default value) [‡] (£/ha/year)	Total value of service assuming it is present in all UK inland/coastal wetlands [§] (£ million/year)	Marginal value of service when provided by an additional hectare of new wetland [§] (£/ha/year)
UK inland wetlands	1,519	601,550	2	2	1
UK coastal wetlands	693	274,613	16	514	12

* Values are area-weighted estimates for all UK inland wetland sites using the Brander *et al.* (2008) benefit function and CORINE land use data sets.

[†] Data on the number and area of wetlands were drawn from the European CORINE Land Cover Maps (Morris & Camino 2010).

[‡] Default average values (where all of the ecosystem services specified in this table do not apply) are £303/ha/yr for UK inland wetlands and £1,856/ha/yr for UK coastal wetlands.

[§] In contrast, the default total value of the existing inland wetland stock, assuming that none of the ecosystem services in the table apply, is £182 million/yr for UK inland wetlands and £509 million/yr for UK coastal wetlands.

[§] The per hectare value of services associated with additional new wetlands is lower than the average per hectare value of existing wetlands. This reflects the diminishing marginal value of additional wetlands.

63 For example, the Environment Agency forecast change in water demand for England and Wales for the 2050s ranging from -4% through to +35% according to different scenarios: www.environment-agency.gov.uk/research/library/publications/40731.aspx.

commercial fisheries and those providing recreation and tourism services (Entec 2008; University of Brighton 2008). Household drinking water supplies are routinely treated to bring them up to potable standards. Both common sense and empirical studies have confirmed the massive health benefits of such treatment. Ecosystems contribute to these benefits by improving water quality through natural processes such as the filtration services provided by healthy soils. That said, it is argued that the economic benefits of such services should be measured in terms of a reduction in treatment costs rather than attempting any estimation of the benefits of avoided ill health.

Assessment of the avoided remediation costs of water purification which may come about by environmental improvement is complicated, as necessary information is typically considered as confidential by private water utilities (Andrews 2003; Knapp 2005). However, Lovett *et al.* (2006) draw upon work by the Environment Agency (EA 2002) and Pretty *et al.* (2003) to provide a lower bound estimate of the annual cost of treating UK drinking water to meet EU nitrate standards of at least £13 million and note that this is expected to rise further in the future. A more recent report published by UK Water Industry Research (UKWIR 2004) summarises the costs incurred by the UK water supply industry in response to a range of groundwater quality problems (arising from nitrates, pesticides and other chemicals, salinity, metals, bacteria and so on) during the years 1975–2004. Total capital (CAPEX) and operating (OPEX) expenditure associated with these problems is estimated at £754 million (2003 prices). In addition, Lovett *et al.* (2006) estimate capital expenditure by water companies to reduce nitrate levels in ground and surface water of about £300–£400 million during the Fourth Asset Management Plan (AMP4) investment period ending in 2009, although the authors again note the difficulties of obtaining accurate costing data from a privatised water industry. Working from these and other sources, Lovett *et al.* (2006) estimate costs of around £8/person/yr to treat nitrate problems in affected areas.

Further variations in treatment costs can arise at a local level if specific issues arise due to ecosystem influences. Numerous natural habitats such as upland and peatland areas contribute both positively and negatively to water quality, and hence to the costs and benefits accruing to water users. In particular, the management of peatlands can influence water colouration. Colour problems due to run-off of dissolved organic carbon have increased over the last 20–30 years. The practice of moorland ‘gripping’ (digging and enlarging drainage ditches) may have contributed to this problem. Avoided cost calculations can be made of the benefits of reducing colouration problems by blocking drains to reduce peat wastage. These will vary on a catchment-to-catchment basis and are not known at a national level. However, one study showed benefits from avoided costs of treatment were around £5 million over 10 years. As we note subsequently, these are likely to be dwarfed by the non-market benefits of avoiding such problems as discolouration.

While information is incomplete, the evidence which is available suggests that the direct market benefits associated with the incremental changes in water quality to be achieved under the WFD are unlikely to be significant in total. They are also difficult to estimate at a national level using available data (Defra 2010a). It is noted, however, that a major loss of water quality would seriously compromise the market-based services provided by freshwater ecosystems and for some purposes, would be similar to a curtailment in water supply.

Non-market benefits associated with water quality.

Turning to consider the non-market values of water quality in rivers and lakes, these are typically estimated by examining the benefits associated with improving quality back to natural levels (i.e. in effect, these are estimates of the value of losses currently being experienced under present lower quality).⁶⁴

In a major study undertaken for Defra as part of their preparations to implement the WFD, NERA Economic Consulting use a mixture of contingent valuation and choice experiment methods to estimate the value that households in England and Wales ascribe to water quality as it affects biodiversity (in terms of fish and other aquatic life), aesthetic quality (viewing, clarity, smell, insects) and recreation (suitability for providing relaxation, recreational activities in and near streams) (NERA 2007). Estimates of WTP for water quality varied according to the methods of elicitation,⁶⁵ with mean WTP thought by NERA (2007) to lie between £45 and £168 per household p.a. for improving water quality in 95% of rivers and lakes to ‘good quality standards’. Allocation of values across different levels of improvement is given in **Table 22.7**, which also reports aggregate benefits across England and Wales of £1,140 million p.a. The greatest proportion of extra benefits is associated with improvements from moderate to good water quality. This reflects not only the greater share of water bodies in this improvement category but also, as expected, the relatively high values for improvements in more populous areas.

Drawing on the preceding analysis, the Environment Agency has compiled estimates of the benefits of improvements in water quality per kilometre for the main river basins in England and Wales. Average benefits are £15.6/km, £18.6/km and £34.2/km for improvements that lift water quality from low to medium, from medium to high and from low to high respectively. Benefits per kilometre are much greater than these average values in river basins with higher population densities.

Another perspective on freshwater quality is given by the estimated annual equivalent expenditure of £1.1 billion/yr (in 2008 prices) to meet WFD quality targets over the next 43 years through to 2052. Reflecting pressures and vulnerabilities, most of this expense is associated with supporting water abstraction and discharges (£889 million/yr), habitat and fisheries (£160 million/yr), urban drainage and reservoir safety (£91 million/yr) and agricultural pollution (£57 million/yr).

64 There are actually four theoretically acceptable economic measures of welfare change: WTP for a gain; WTP to avoid a loss; willingness to accept compensation to forgo a gain; willingness to accept compensation for a loss. Terminology and theoretical and empirical comparison of measures is explored by Bateman *et al.* (2000).

65 For a discussion of WTP elicitation effects see Bateman *et al.* (1995).

Table 22.7 Non-market benefits associated with improvements in water quality in rivers and lakes in England and Wales in 2009. Source: Morris & Camino (2010).

Initial quality status of water bodies: rivers and lakes	Benefit of planned improvement in water quality to be achieved in the period 2009–2015 (£ million/yr)	Remaining benefits associated with achieving Good quality status post 2015 (£ million/yr)	Total benefits of improvement to Good quality status (£ million/yr)	Distribution of extra benefits of water quality improvement by class (%)
Moderate	46.4	720	766.4	67%
Poor	26.3	273.8	300.1	26%
Bad	9.1	55.7	64.8	6%
Not known	0.7	8.1	8.8	1%
Total			1,140.0	100%

It is recognised that the preceding figures do not indicate the value of the total benefits of non-market goods associated with freshwater quality. Rather, they indicate in broad terms the expected benefits of services associated with achieving given increments in water quality about current quality levels, and a (potential) revealed willingness to incur costs to obtain these incremental benefits. Neither do they tell us about WTP to avoid the loss of non-market benefits if there were considerably lower standards of water quality in UK Freshwaters, other than suggesting that these are likely to be very significant.

One attempt to consider both the benefits and costs of changes in water quality is provided through the work of Fezzi *et al.* (2008, 2010a) and Bateman *et al.* (2010b). Fezzi *et al.* (2008) draw on the prior work of Cuttle *et al.* (2007) to consider the costs of a variety of measures to reduce farm diffuse nutrient pollution of waterways (the agricultural sector being the principle source of such pollution). Fezzi *et al.* (2008) estimate that measures such as lowering livestock dietary nitrogen and phosphorus intakes could increase farm costs by up to £46/cow p.a. (due to the need to find alternative foodstuffs) and reduce revenues by as much as 8% (in the poultry sector where cuts in nutrient intake reduce productivity). Fezzi *et al.* (2010a), extend this work to develop an integrated hydrological-economic analysis combining data from the Farm Business Survey with models of nutrient leaching and in-stream processes. This enables them to estimate the indirect costs to farms of changing activities in order to reduce their diffuse nutrient pollution. The effectiveness of competing strategies was assessed in terms of both nutrient loading and in-stream concentrations, with the latter being more relevant to the ecological impacts central to policies such as the WFD. While Fezzi *et al.* (2008) estimate that mean costs of reducing nutrient pollution via a 20% reduction in fertiliser application exceeded more than £100/ha in the worst affected sector (dairy), Fezzi *et al.* (2010a) show (in a study of a catchment within the Humber Basin) that alternatives such as the targeted conversion of arable areas into grassland could more than halve the impact of pollution reductions upon farm incomes.

Of course, the costs associated with reducing water pollution need to be set against the benefits. Bateman *et al.*

(2010b) build on the prior work of Fezzi *et al.* (2010a), to conduct a benefit valuation study in the Humber Basin. Data were collected from more than 2,000 households detailing their outdoor recreational behaviour across the year. By recording both the trip outset and destination locations, a travel cost analysis was conducted to examine the influences upon trip choice. Focusing upon water-based recreation, Bateman *et al.* (2010b) show that, after controlling for other determinants as diverse as travel time, the presence of local pubs, and recreational facilities, significantly more visits are made to rivers with higher water quality.⁶⁶ Bateman *et al.* (2010b) relate this model to the level of improvement in river water quality that was shown by Fezzi *et al.* (2010a) to be feasible through farm land use change. They estimate that in the study area considered (the Aire catchment which covers much of Leeds, most of Bradford and areas upstream of the confluence with the River Calder), the benefits of improving water quality to pristine levels (as defined under the WFD) were of the order of £12.5 million p.a. This was contrasted with the costs of land use change in the Humber catchment assessed by Fezzi *et al.* (2010a) of just over £5.5 million p.a. Given the considerable excess of benefits over costs in this case, it would seem likely that such a scheme would pass most assessments. However, there is a distributional issue to be addressed here, in that the costs of such a scheme would impact upon a small rural sector of society, whereas the benefits would be dispersed across the mainly urban population of visitors. Clearly, there is the potential for a compensated trade-off leading to social gain here. However, without such compensation the potential for inequality is obvious.

A further cost-benefit result can be approximated by contrasting the costs associated with combating discolouration problems with the benefits derived from such actions. Bateman & Georgiou (2010) report findings from a contingent valuation study of such benefits, showing that average WTP per household, in order to avoid one day of discolouration problems, was £5.40. Comparison with costs presented previously suggests that such schemes are likely to pass cost-benefit tests.

Turning away from rivers, wetlands are also a major provider of water quality improvement benefits through their ability to recycle nutrients. **Table 22.8** uses a value transfer

⁶⁶ Interestingly this is not a simple linear relation; potential visitors are indifferent to variation at the lower end of the quality scale. In other words, there is a lower threshold which water quality must exceed before visitor numbers increase. Thereafter the relationship is approximately linear, with increases in water quality leading to higher visitor numbers.

Table 22.8 Estimated average, total and marginal values for water quality improvements provided by inland and coastal wetlands in the UK.* All values are given in £, 2010 prices. Source: Morris & Camino (2010).

Ecosystem service-related goods	No. of sites [†]	Total area (ha)	Average value of service where present (addition to default value) [‡] (£/ha/year)	Total value of service assuming it is present in all UK inland/coastal wetlands [‡] (£ million/year)	Marginal value of service when provided by an additional hectare of new wetland [§] (£/ha/year)
UK inland wetlands	1,519	601,550	436	263	292
UK coastal wetlands	693	274,613	2,676	1,245	1,793

* Values are area-weighted estimates for all UK inland wetland sites using the Brander *et al.* (2008) benefit function and CORINE land use data sets.

[†] Data on the number and area of wetlands were drawn from the European CORINE Land Cover Maps (Morris & Camino, 2010).

[‡] Default average values (where all of the ecosystem services specified in this table do not apply) are £303/ha/yr for UK inland wetlands and £1,856/ha/yr for UK coastal wetlands.

[§] In contrast, the default total value of the existing inland wetland stock, assuming that none of the ecosystem services in the table apply, is £182 million/yr for UK inland wetlands and £509 million/yr for UK coastal wetlands.

[§] The per hectare value of services associated with additional new wetlands is lower than the average per hectare value of existing wetlands. This reflects the diminishing marginal value of additional wetlands.

function to estimate average, total and marginal values for water quality improvements provided by inland and coastal wetlands in the UK. These can be substantial, amounting to £1,500 million p.a. Notice, however, that the marginal values associated with expansions of wetlands are significantly lower than present average benefits, reflecting the diminishing marginal benefits of increases in such resources.

Clearly, Freshwater ecosystems play a central role in supporting human welfare. They are also a focal point for conflicts that arise when there are competing human demands for water as an essential natural resource. The analysis here (and that covered in other sections of this chapter that deal with water-related benefits such as biodiversity, recreation and amenity) is known to be incomplete in terms of the full identity and valuation of benefits. For such a critical resource, data on the value of water resources and related services appear fragmented and incomplete, in spite of the very considerable advances made recently under the WFD. This is an important area of work for the future.

22.3.7 Flood Protection: Inland⁶⁷

Ecosystems can play a major role in flood control. Approximately £1 billion/yr is spent on flood risk management (EA 2009a,b). However, in recent years, flooding has become more problematic in the UK (Pitt 2008). In the UK as a whole, probably over 5 million properties are exposed to low to moderate probability of river and coastal flooding (between 0.5% and 1.3% chance of flooding each year) and the average annual cost of flooding in the UK is about £1.4 billion (EA 2009a,b). However, extreme flooding events can generate much higher costs, with the 2007 floods in England resulting in estimated costs of £3.2 billion (Chatterton *et al.* 2010) with two-thirds of this being borne directly by households and businesses. This leads to a strong case for investment in flood defences, both natural and man-made, with Defra's Spending Review suggesting an average benefit-cost ratio of 8:1 (pers comm., Mallika Ishwaran, Defra, 2011).

Direct intangible impacts on flood victims include stress and health risks. A survey of households (RPA & FHRC 2004)

showed a weighted average WTP of £200/household/yr to avoid the intangible costs associated with a 1% per year chance of flooding, equivalent to a present value sum of about £5,000 over 50 years. Evidence from the 2007 floods suggests this is probably an underestimate. There are currently about 600,000 households in the UK at serious risk of flooding (FFCD 2004). This equates to a WTP to avoid intangible costs of £120 million/yr.

The link between ecosystems and flooding can be demonstrated via two examples. First, the climate can be seen as an ecosystem service and hence, deterioration in the climate should be seen as a relevant value for the UK NEA. Second, changes to the extent and management of certain terrestrial habitats can lead to flooding-related values, whether benefits or costs.

Climate change could double numbers of households exposed to serious risk for the UK by 2060 (EA 2009d). Looking forward to 2080, the Foresight Future Flooding Project (FFCD 2004) identified a possible increase in the annual river and coastal flood damage costs to property of £14–£19 billion (in 2004 prices) under future consumption-oriented scenarios in the absence of additional measures to control flood risk (**Table 22.9**). This is equivalent to about £17–£23 billion in 2010 prices: or about £11–£17 billion/yr in 2060 (the UK NEA time horizon), assuming a linear increase in damage cost over time. Incremental flood damage costs were estimated at £0.5–£3.8 billion for 2080 and £0.4–£3.4 billion in 2060 (all figures at 2010 prices) under sustainability oriented scenarios, reflecting a combination of reduced flood probability and damage costs. Additional costs were identified for urban flooding unconnected with river and coastal sources.

Climate-induced increases in flood damage will also impact upon agricultural land. The average cost of a flood occurring at any time within a given year on intensively farmed Grade 1 agricultural land (£1,220/ha) is much higher than on extensively grazed grade 4 land (£160/ha), with costs rising for summer flooding (Posthumus *et al.* 2009). Where flooding results in permanent abandonment, land prices of up to £15,000/ha can apply (Defra 2009; RICS 2010).

67 This Section draws in part from Morris & Camino (2010).

There are about 1.34 million hectares of agricultural land at risk of flooding in England and Wales, of which 62% are liable to flooding by rivers only, 23% by sea only and 15% by both. About 421,500 ha currently benefit from flood defences in England and Wales, of which 70,000 ha (17% of total) are grade 1 and 2, and 424,000 benefit from coastal defences, of which 158,000 ha (37%) are grade 1 and 2. About 1.28 million hectares in England and Wales also benefit from pumped drainage to avoid either flooding or waterlogging; over 90% of this land is used for agriculture, and one-third is located in the Anglian region.

An assessment of land use, estimated flood damage costs, and flood return periods in years for defended and undefended areas of England and Wales (Roca *et al.* 2010) shows that flood defence reduces expected annual damage costs from river flooding by £5.2 million, and from coastal flooding by £117.7 million. These estimates, however, undervalue the considerable associated benefits of land drainage and the management of water levels for farming. Estimates are not available for other parts of the UK at the time of writing.

Land use management clearly impacts upon the probability of flooding of adjacent or downstream property, although robust national estimates of associated values are not available. Nevertheless, some wetland values are available. While Tinch *et al.* (2010) argue that the ability of peatlands to act as flood buffers may be overstated, European evidence suggests that wetlands can be a major provider of flood control values, depending on their location.

Table 22.10 employs findings from a value transfer model to provide estimates of average, total and marginal values for these benefits, as provided by inland wetlands in the UK. These are substantial, although the marginal values associated with expansions of wetlands are somewhat lower than present average benefits, reflecting the diminishing marginal benefits of increases in such resources.

22.3.8 Flood Protection: Coastal

The majority of UK coastal defence is provided by the natural environment, with only 18% protected by defence works and artificial beaches. Of course, much of this natural defence can effectively be omitted from decision making where there is no significant danger of flooding (e.g. high, non-eroding cliffs). While this provides a clear flood defence value, effectively we can treat such defences as infinite

Table 22.9 Estimated annual economic flood damage to residential and commercial properties for the UK under current (2000) and future (2080) scenarios according to Foresight Flood Defence (2004 prices). Source: FFCD (2004).

Flood source	Current flooding costs Year 2000 (£ million/yr)	Costs under consumption-oriented scenarios* (£ million/yr)	Costs under sustainability oriented scenarios† (£ million/yr)
River and coastal	1,088	15,175–20,600	1,508–4,820
Intra-urban	270	5,100–7,900	740–1,870
Total	1,358	20,275–28,500	2,248–6,690

* National Enterprise and World Market scenarios.

† Local Stewardship and Global Sustainability scenarios.

and any value calculations as mere mental gymnastics. However, there are many other areas of the country where topography means that there is a real risk of sea flooding. Here the natural environment can provide a very valuable service.

In assessing the net annual value of any flood defence option one needs to consider three factors:

- the frequency of any flooding which will occur under this option (virtually no defence scheme is perfect);
- the damage that would occur in any such flood; and
- the costs of building (where appropriate) and maintaining that flood defence option.

Consideration of items i) and ii) allow estimation of the expected flood damage under a defence option. This can then be added to the defence costs given at iii). One could then repeat the analysis for a situation in which the defence disappears. Obviously, this reduces maintenance and other costs, but is likely to increase the damage costs. If the latter outweighs the former, there is a case for retaining that defence, although one would then wish to consider further defence options, typically opting for the one which yields the largest net benefits relative to other options.

While there are numerous case studies of local defence schemes, to date there is no national level assessment that would allow a comparison of natural versus man-made defence values (Beaumont *et al.* 2010). Indeed, even at a more

Table 22.10 Estimated average, total and marginal values for inland flood control provided by wetlands in the UK.* All values are given in £, 2010 prices. Source: Morris & Camino (2010).

Ecosystem service-related goods	No. of sites†	Total area (ha)	Average value of service where present (addition to default value)‡ (£/ha/yr)	Total value of service, assuming it is present in all UK inland wetlands§ (£ million/yr)	Marginal value of service when provided by an additional hectare of new wetland§ (£/ha/yr)
UK inland wetlands	1,519	601,550	608	366	407

* Values are area-weighted estimates for all UK inland wetland sites using the Brander *et al.* (2008) benefit function and CORINE land use data sets.

† Data on the number and area of wetlands were drawn from the European CORINE Land Cover Maps (Morris & Camino 2010).

‡ Default average values (where all of the ecosystem services specified in this table do not apply) are £303/ha/yr for UK inland wetlands.

§ In contrast, the default total value of the existing inland wetland stock, assuming that none of the ecosystem services in the table apply, is £182 million/yr for UK inland wetlands.

§ The per hectare value of services associated with additional new wetlands is lower than the average per hectare value of existing wetlands. This reflects the diminishing marginal value of additional wetlands.

local level, with the exception of managed realignment scheme assessments (Turner *et al.* 2007; Luisetti *et al.* 2011a), studies tend to focus not on the net benefits of natural versus built defences, but instead simply on the cost of the latter, arguing that these costs are saved when natural defences are used. For example, King & Lester (1995) estimate that an 80 m wide saltmarsh can save from £2,600 to £4,600 per metre of seawall that does not have to be constructed. Obviously, such costs do not reflect the net benefits of different defence options.

Although no national estimates of the value of Coastal Margin ecosystems for flood defence currently exist, there are examples in the literature of methods that could be applied if such a study were to be undertaken. Penning-Roswell *et al.* (2010) and Defra (2009) provide some damage-cost analysis and Eftic (2010) considers the use of value transfers. However, a key requirement for such valuation would be a quantitative assessment of flood risk for the entire UK coastline. This seems a useful direction for future research. Such an approach could draw on the method of Costanza *et al.* (2008), who estimate the spatial value of coastal wetlands for hurricane protection. Through a two-step regression analysis, they explore the relationship between hurricane damage, wind speed and wetland area, and combine this with data on annual hurricane frequency to derive an estimate of the annual value of wetlands to hurricane protection. Unfortunately, however, they do not compare the values calculated to other forms of coastal defence.

Building on the meta-analysis of SP studies undertaken by Brander *et al.* (2006), Morris & Camino (2010) show that wetlands are a major provider of coastal storm surge protection benefits. **Table 22.11** provides estimates of average, total and marginal values for these benefits as provided by coastal wetlands in the UK. These are substantial, at more than £1.5 billion/yr. While the marginal values associated with expansions of wetlands are somewhat lower than present average benefits, reflecting the diminishing marginal benefits of increases in such resources, these are, nevertheless, still highly significant values. This underlines the argument that in many cases, coastal wetlands yield storm protection values which exceed the opportunity cost of not converting such areas to agricultural production.

Coastal saltmarshes can provide a range of services in addition to carbon storage and have more recently been utilised as a component in a new, more flexible approach to

coastal erosion and flood management strategy. So-called 'managed realignment' schemes have been designated to replace/augment hard engineering coastal defences on the east coast of England. Economic cost-benefit appraisal of a selection of managed realignment schemes indicates that such investments may be efficient; however, their spatial location is critically important, both in terms of the ecosystem services generated and the human beneficiaries. While there are 'win-win' policy opportunities, managed realignment is not sustainable as a generic solution to the complex problem of 'defending' Coastal Margins under the threat of climate change.

Managed realignment typically involves the deliberate breaching of existing sea defences, with the land behind them consequentially being flooded. Such projects result in the creation or restoration of saltmarshes, which, it is claimed, may provide a sustainable flood defence approach to dissipating wave energy. Such 'soft' defences allow the intertidal habitat to naturally move inland, thereby creating opportunities for biodiversity enhancement, amenity and recreation (i.e. a diversity of ecosystem services). Note, however, that this will of course be dependent on how successfully saltmarsh communities can re-establish.

A number of appraisals of potential or implemented managed realignment schemes have been reported in the literature. For example, a case study of the Alkborough Flats in the Humber estuary (Everard 2009; also Chapter 11) aimed to both reduce flood risk and provide physical compensation for habitat lost elsewhere in the estuary. The Environment Agency argues that this case study shows that, given the value of the ecosystem services generated following an ecosystem restoration, managed realignment innovations can result in 'win-win' solutions. One of the key results of the report is that the annual loss of food production (opportunity cost of realignment) was compensated for by the higher value of fibre related to the sale of rare breed genetic stock sheep and cattle farmed on the reclaimed marshes. The economic value of commercial fishing was also considered to be a potentially significant research gap. The valuation approach followed in this case study differs from that used by Turner *et al.* (2007) and Luisetti *et al.* (2011a,b) to value similar schemes around the Humber and Blackwater estuaries respectively (see below). For the Alkborough Flats case study, supporting services and regulatory services were assessed as being

Table 22.11 Estimated average, total and marginal values for storm buffering and flood control provided by coastal wetlands in the UK.* All values are given in £, 2010 prices. Source: Morris & Camino (2010).

Ecosystem service-related goods	No. of sites [†]	Total area (ha)	Average value of service where present (addition to default value) [‡] (£/ha/yr)	Total value of service, assuming it is present in all UK inland wetlands [§] (£ million/yr)	Marginal value of service when provided by an additional hectare of new wetland [§] (£/ha/yr)
UK coastal wetlands	693	274,613	3,730	1,534	2,498

* Values are area-weighted estimates for all UK inland wetland sites using the Brander *et al.* (2008) benefit function and CORINE land use data sets.

† Data on the number and area of wetlands were drawn from the European CORINE Land Cover Maps (Morris & Camino 2010).

‡ Default average values (where all of the ecosystem services specified in this table do not apply) are £1,856/ha/yr for UK coastal wetlands.

§ In contrast, the default total value of the existing inland wetland stock, assuming that none of the ecosystem services in the table apply, is £509 million/yr for UK coastal wetlands.

§ The per hectare value of services associated with additional new wetlands is lower than the average per hectare value of existing wetlands. This reflects the diminishing marginal value of additional wetlands.

worth just under £1 million p.a. (excluding possible flood regulation function value), and included in the aggregated gross benefit calculation. While a full investigation of the whole services production and delivery 'system' is to be commended, there is a risk of double counting problems due to the addition of both supporting service values and the value of those services they support.

Published research has highlighted the fact that managed realignment policy needs to be appraised across a more extensive spatial and temporal scale than has been the case in the traditional scheme-by-scheme coastal management system. Whole estuaries or multiple coastal cells should be treated as a single 'project' encompassing a number of realignment sites. Although in some estuaries along the English east coast some experimental managed realignment schemes have already been implemented, the approach continues to be controversial because previously reclaimed coastal land (usually agricultural land) is sacrificed in order to reduce the threats of coastal erosion and flooding (RCEP 2010). The value of agricultural land may increase over time as food security concerns rise up the political agenda.

A best practice appraisal approach first requires the identification of all sites that are likely to generate low opportunity costs and the minimum of social justice or ethical concerns. In this policy context it is feasible to apply an efficiency-based cost-benefit analysis, with the expectation that this may provide decisive information for policy choice (Randall 2002; Turner *et al.* 2007). It is also necessary to demonstrate, as was the case in the Blackwater case study (Luisetti *et al.* 2011a,b) and in analyses completed in the Humber estuary (Turner *et al.* 2007), that there has been no reduction in the level of protection (*vis-à-vis* hard defences) where new saltmarshes were put in place.

In their study of managed realignment on the Blackwater estuary, Luisetti *et al.* (2011a,b) provide economic values for the sites considered and examine issues of location and ecosystem services. They show three important results: i) that the values of users or potential users of the area are higher than those of non-users; ii) that the values held by both groups decay with increasing distance from the managed realignment site; and iii) that values increase with the size of the proposed wetland, but at a declining rate (a result echoing the diminishing marginal values mentioned in our methodological overview—Section 22.2.2). These relationships mean that the value of any managed realignment site will not be constant, but will vary according to location. Factors i) and ii) mean that a site located nearer to population centres is likely to generate higher values than an otherwise comparable site located in some remote place. Factor iii) means that we cannot use simple constant per hectare values to estimate the value of such schemes. However, all of these factors are in line with expectations and can be quantified, providing that a sufficient number of high quality, comparable valuation studies are undertaken. This requires study designs which are specifically orientated towards the production of generalised and transferable value functions.

Although studies such as Luisetti *et al.* (2011a,b) show that some realignment schemes and soft defences can pass economic analyses, for many stretches of coastline, hard

defences will continue to be required for the foreseeable future because of the scale and significance of the economic and social assets that are at risk. This means that we cannot claim that managed realignment will always offer 'win-win' solutions. Although general principles for analysis can be identified, the costs and benefits of differing options will vary by location and will require individual consideration.

22.3.9 Pollution Remediation

Tinch *et al.* (2010) argue that habitats such as Mountains, Moorlands and Heaths may provide a substantial pollution remediation service, noting that they assimilate air pollutants such as sulphur dioxide and nitrogen oxides. Similarly, in Chapter 8 it is noted that woodlands and trees can intercept pollution from point sources, and capture diffuse pollution (including both ground and atmospheric pollution), thereby helping to reduce ambient concentrations and limit the spread of pollutants. One of the few studies to value such pollution remediation services in the UK is Powe & Willis (2004) who state, for example, that trees in Britain absorb 0.4–0.6 million tonnes of particulates (PM₁₀) a year. They include an estimate of the annual value of pollution remediation services by Britain's trees (associated both with absorption of particulates and of sulphur dioxide) of £0.9 million. Based upon associated net health benefit (reduced morbidity and mortality) estimates, the latter is closely related to other types of health benefits considered subsequently.

It seems likely that ecosystem service values for pollution remediation are substantial. Yet there was little evidence available on the value of these services or how they may vary due to habitat change. It seems likely, therefore, that this is an area which requires further research.

22.3.10 Energy and Raw Materials

22.3.10.1 Energy

The focus of the UK NEA has been upon biotic ecosystem services and their value. However, there is no reason why the principles of the ecosystem services approach should not be extended to embrace the wider contribution of the natural environment to human well-being, and indeed, such extension is argued for elsewhere (Bateman *et al.* 2011a). Two areas of extension seem to be of particular importance for consideration within a future expanded assessment: energy and abiotic raw materials.

The energy contribution of the natural environment is likely to expand globally in line with development needs. Fossil fuels currently dominate global energy markets. Market prices represent a good starting point for estimating the underlying economic value of fossil fuel extraction, but adjustments may need to be made for subsidies, taxes and the exercise of market power. The latter is particularly important in global oil markets. The market value of UK consumption of fossil fuels was £112 billion in 2009 (DECC 2010), of which £35 billion comprises tax and duties. Fossil fuels met 90% of UK energy demand in 2009 (DECC 2010). Two concerns are typically highlighted in consideration of fossil fuels: externalities and sustainability. The externality issue is particularly pertinent in respect of the contributions

of fossil fuels to global climate change through atmospheric emissions of carbon dioxide and other GHGs. Clearly, the costs associated with such emissions must be considered within any economic analysis of such services, and these impose a substantial penalty on fossil fuels. Fossil fuel extraction and use also give rise to a range of other environmental externalities associated with air pollution, water use and the disposal of solid wastes. The sustainability issue arises because fossil fuels are physically non-renewable. However, this highlights the fact that we are looking at the maintenance of services rather than the physical constitution of any given asset. So we might run down conventional oil reserves yet maintain the service of energy provision by increasing stocks of alternative energy resources.

This brings us to consider renewable energy sources such as solar, wind and wave power and energy crops. After a slow start, the deployment of renewable energy is starting to expand rapidly. Renewables met 3% of UK energy demand in 2009 and 7% of electricity generation needs. Estimating the value of the renewable contribution is complicated by the level of subsidy associated with the Renewables Obligation and, more recently, Feed-in Tariffs for smaller generation.⁶⁸ The current value of renewable energy supply is dwarfed by that of fossil fuels. However, the supply of renewables will grow considerably if policy ambitions and forecasts are realised. For example, a recent study predicts large rises in demand for wood fibre in the UK over the period 2007–2025, mainly due to government policies and incentives to encourage the use of woodfuel (JCC 2010). Overall renewable fuels are typically associated with very low levels of externality and are inherently sustainable, making them attractive options for long-term development.

Of course, a further alternative energy source is provided by nuclear power, which supplied 17% of the UK's electricity generation needs in 2009 (DECC 2010). While providing a low emission alternative to fossil fuel, the nuclear power sector raises unique issues regarding risk and long-term waste storage and decommissioning costs.⁶⁹

22.3.10.2 Raw materials

The annual value of marine-based biotic raw materials, including fish meal, fish oil and seaweed, is estimated to exceed £95.1 million p.a. (2010 prices). The value of non-biotic services arising from the Marine environment is huge, as summarised in **Table 22.12**. However, these are not investigated in detail in this report as they are not 'true' ecosystem services, and have been well documented elsewhere (Pugh 2008; Saunders 2010).

Terrestrial abiotic resources are also generally excluded from analyses, although they are of substantial value. For example, the UK aggregates industry is worth in the region of £4.8 billion annually and is almost exclusively supplied by natural resources. However, one resource that was considered was the value of peat extraction for supply to gardeners and horticulturalists. UK production fell from about 1.8 million m³ in 2001 to 0.94 million m³ in 2009. However, while this most

Table 22.12 Review of UK per annum values of abiotic commercial activities occurring in the Marine and Coastal Margin environments. GVA = gross value added; n/r = not reported.

Marine and Coastal Margin services	Pugh (2008) (GVA, £ million)*	Saunders <i>et al.</i> (2010) (£ million, 2008)
Oil and gas	19,845	36,814
Aggregates	114	31
Cooling water	n/r	100
Salt	n/r	4
Ship and boat building	1,223	n/r
Marine equipment and materials	3,268	n/r
Marine renewable energy	10	62
Construction	228	n/r
Shipping operations	3,399	7,100
Ports	5,045	n/r
Navigation and safety	150	n/r
Cables	2,705	n/r
Business services	2,086	n/r
Licence and rental	90	n/r
Defence	2,814	300

* Price base varies from 2004 to 2006. See Beaumont *et al.* (2010) for details.

recent output was worth about £9.7 million p.a., it resulted in the release of about 400,000 tonnes of carbon dioxide, which had an external cost of around £20 million using a DECC price in 2009 of £50 per tonne of carbon dioxide. Given this net social cost, there is a policy target for ending the use of peat in gardening products by 2020.

22.3.11 Employment

While it is certainly the case that large numbers of jobs are connected to ecosystem services, the argument that these should be counted as a distinct and robust economic benefit of such services is less clear cut. The economic approach to appraising benefit values rests upon considering trade-offs and in the case of employment benefits, the key issue concerns the opportunity costs of alternative employment. A good example of this thinking is provided by the case of forestry.

It has been argued that creating jobs in forestry is a good way to stem the ongoing trend of rural depopulation and combat the psychological and other economic costs of rural unemployment. However, numerous studies have suggested that forestry is a relatively expensive and inefficient method of providing rural employment, particularly when compared to agriculture (HM Treasury 1972; Laxton & Whitby 1986; NAO 1986; Evans 1987; Johnson & Price 1987). Therefore, while forestry expansion might be justified on a number of grounds, employment does not appear to be one of them. Such conclusions have been disputed by noting that since the 1990s,

68 DECC Feed-In Tariffs support small scale (less than 5 MW), low carbon electricity generation schemes, while the Renewables Obligation mandates the partial use of low carbon energy options such as wind and biomass sources.

69 Construction, containment and disposal emissions mean that this cannot be described as a zero-emission option, although clearly, carbon release is far lower than for fossil fuels.

employment in forestry has been falling and productivity rising (Thompson 1990; FICGB 1992; FC 2001). However, coincident rises in the efficiency of the most likely alternative form of rural employment, agriculture, means that the economic case for arguing that there is a major employment benefit from ecosystem services remains to be proven.

A stronger argument may well be made in terms of the benefits of ecosystem service-related employment in terms of cultural and social cohesion in marginalised and remote rural communities. For example, in 2005 more than 31,500 people were employed in the fish catching, processing and aquaculture sector in the UK, with many of these jobs located in remote coastal regions of Scotland, Wales and south-west England. While some of this employment might be transferred to other sectors if fisheries were to decline further, previous experience of translocations from remote communities dominated by single industries suggests that there are genuine net benefits in this respect. Similar arguments can be made regarding upland farming, remote forestry, employment on grouse moors and the like. An in-depth analysis would be required to estimate such benefits in economic terms and find out whether there is any robust linkage to ecosystem service levels. However, ultimately it may well be that the magnitude of any such values is dependent, in considerable part, not only upon the individuals concerned but also upon wider social preferences regarding the maintenance of such remote rural communities and the landscapes they work. While the case for conventional economic appraisal rests on the criterion of efficiency, employment and related social impacts raise equity and social justice concerns, which will be important components of the policy- and decision-making process.

22.3.12 Game and Associated Landscape Values

A substantial area of UK moorlands, most noticeably in Scotland, is managed for shooting. While ecosystem services are clearly an important input to be considered in the valuation of such activities, data are not available to permit us to isolate the value of such services separately from the human capital and other inputs required to generate sporting activities. However, it is unlikely that net values are substantial. As an example, while gross expenditure on grouse shooting in Scotland is estimated at between £5.8 and £12.6 million (FAI 2010, adjusted to 2010 prices), only 43% of Scottish sporting estates actually make a profit (Tinch *et al.* 2010). Valatin & Starling (2010) estimate mean stalking revenues of up to £3/ha (2010 prices) for English woodlands, based upon data for Forestry Commission land, although they recognise that these may be somewhat higher in Scotland.⁷⁰

An economic assessment of an undertaking should consider all of its externalities, positive and negative. Clearly blood sports excite strongly negative passions amongst some in society. However, proponents point out that much lowland woodland, especially in England, has been maintained as such

precisely because of sporting interests and so provides vital wildlife habitat which would not be economically sustainable without sporting revenues. Indeed, many in the blood sport fraternity argue that positive contributions to biodiversity are provided not only in terms of habitat but also directly through the culling of what are now considered pest species such as deer and therefore they are a necessary substitute for the historic loss of top predators such as wolves that previously kept deer densities in check. Similarly, the management of Mountain, Moorland and Heath habitats for grouse shooting is a direct driver of the open landscapes which are valued by many in society. This example can be extended further through allied management practices such as heather burning and raptor control to highlight the complexity of issues that are raised by grouse moor management practices. It is interesting that many of these habitats, including the agricultural areas which dominate the majority of the UK, yield landscapes which are in fact not natural, but are perceived as such by a population accustomed to such environments. This raises an interesting point that what people value about landscapes is in part dictated by what is familiar, rather than simply some innate preference.

22.3.13 Amenity Value of the Climate⁷¹

As noted previously, there are no constraints against (and good reasons for) extending the principles of the ecosystems service approach to the wider set of benefits and costs which are provided by the environment. Hence, we here consider the extent to which the climate delivers amenity benefits or disbenefits quite separately from the other impacts it is likely to deliver.

Whilst the case for the existence of a relationship between climate and well-being seems clear, in practice the nature of that relationship is liable to be complex. People may feel happier inhabiting warmer climates or indeed find that they need to spend less in order to achieve the same level of well-being. But this change in temperature may influence other determinants of well-being such as prices, incomes and even ecosystem availability, especially if these changes are not locally confined but global, as seems very likely to be the case. Therefore, as in the case of urban greenspace amenity, we are faced with a potentially complex set of highly correlated goods which cannot readily be untangled. Given this, all we can reasonably do is to value a subset of the possible impacts of climate change and furthermore, stop short of attributing the relationship between climate and value to particular causes, for example the reduction of heating expenditure or the existence of particular landscapes.

It may not be immediately apparent how climate fits conceptually within the ecosystem services framework. This, however, is readily understood by noting that households combine marketed and environmental goods in order to produce 'service flows' of direct value to themselves. Climate is an input to the households' production functions in the same way that pollination services, genetic diversity and indeed climate are inputs to agriculturalists' production functions.

70 Inspection of shooting offers on the Shooting4All website (www.shooting4all.com) suggest current rental values are in the region of £20/ha/yr although these can vary substantially according to location and site quality. Comparison with values quoted by Crockford *et al.* (1987) suggests that these have not varied greatly in real terms for some time.

71 This Section draws on Maddison (2010).

Most valuations of climate amenities have been undertaken through revealed preference studies, mainly considering property purchases across very varied climates.⁷² Such hedonic pricing studies typically relate large numbers of house sale records to characteristics of the properties concerned, their access to facilities and workplaces, local neighbourhood and environmental conditions. By including climate variables in the analysis and examining how these are related to variation in house prices, a valuation of climate amenities can be obtained. By using spatial variation in climate as an analogue for future climate, such exercises assume perfect adaptation. The phrase ‘perfect adaptation’ means that households have made all cost-effective adjustments. The question is whether it is reasonable to assume that households are able to adapt perfectly over the period in question. If not, any benefits will be overestimated and any costs underestimated.

While such studies have been conducted for Great Britain (GB), and are discussed below, revealed preference methods do face a practical challenge when applied within the GB context. Although GB is characterised by different climates, these differences are much less pronounced than in many countries. However, for the purposes of revealed preference valuation, this more restricted range of climates is not helpful as, ideally, the analyst wishes to observe behaviour under a wide variety of conditions. Therefore most revealed preference analyses of climate amenity values have been conducted in large, climatically diverse countries such as the USA. Imprecision is likely to be a greater problem in a GB study. Consequently, the literature has recently been extended to consider a first life-satisfaction analysis of global climate amenity values. Here, survey respondents are asked to place their life satisfaction typically on a 1–10 scale. By analysing the impacts which income has upon life satisfaction and contrasting these with the impacts of other factors, including climate, trade-offs between money and climate can be inferred and valuations obtained.

The relevant international literature indicates two important characteristics of the resultant valuations; first, that they possess wide ranges of uncertainty, and second, that the central and upper end of those ranges include some very high values. Both of these characteristics are present within estimates of the value of climate amenity in GB. The finding that such values have the potential to be very high is not surprising, given the ubiquity of the climate. That the range of value estimates is very wide is a less desirable aspect of the literature, but again not surprising given our previous comments on the relatively restricted range of climatic conditions in GB (although the weather changes frequently, in global terms the range of climates experienced nationally is relatively small).

Accepting the above caveats, the literature reports both a revealed (hedonic pricing) and life-satisfaction preference assessment of climate amenities in GB under a common scenario: Intergovernmental Panel on Climate Change (IPCC) A1B under which there is rapid global economic growth, especially in developing nations (IPCC 2007). While this scenario is expected to generate major damages and economic losses at a global scale, and these and more direct effects may very adversely impact upon well-being in the UK, in terms purely of climate amenity alone, both studies suggest that the most probable change in climate associated with the A1B emissions scenario will bring significant benefits to the population of GB.

Results from the revealed preference study (based upon observed behaviour) suggest that climate amenity benefits in GB, averaged over the time period 2030–2059, are just over £21 billion p.a. These gains are estimated using the current climate as a counterfactual, that is, it represents the value of such a change in climate if it occurred today. The life-satisfaction approach, while detecting major welfare losses in many countries of the world, also predicts that global warming will actually generate climate amenity benefits within GB which, calculated in the same manner as previously, are estimated at just over £69 billion p.a. (equivalent to £1,130/person/yr) by 2030–2059. This analysis, however, suggests that richer societies care less about the climate and that as temperatures exceed those expected for 2030–2059, they will eventually result in losses rather than benefits.⁷³

It is important to bear in mind that these estimates only consider climate amenities and their findings and have to be offset against the potentially very significant losses which could impact upon GB due to the international impacts of climate change, and the impact on prices and incomes. Neither do these estimates account for extreme events associated with changes in the distribution of climate variables, or, as noted, the short-run costs of adaptation. It is not possible to argue that climate change is a ‘good thing’ for GB based on analysing only a subset of the impacts and holding everything else constant.

22.3.14 The Amenity Value of Nature⁷⁴

There is a long tradition of using hedonic pricing studies to estimate the value of a wide range of environmental amenities and disamenities as they are reflected in local property prices (Sheppard 1999). Using this approach, a novel study was undertaken for the UK NEA to estimate the amenity value associated with proximity to habitats, designated areas, heritage sites, domestic gardens and other natural amenities. The analysis considered over 1 million housing transactions from across England for the period 1996–2008. Information on sales prices and the internal

72 Hedonic studies simultaneously examine differences in wage rates paid to workers in different areas. An alternative revealed preference approach is applied by Maddison (2003), who examines household expenditures across areas with differing climates, considering how much individuals have to pay to modify their environments where they are adverse (e.g. heating and cooling costs). Arguably, this will only yield a lower bound assessment of climate amenity values, as a number of the benefits of pleasant climates will not be reflected in these expenditures.

73 Studies have shown that survey respondents tend to overestimate the beneficial impacts on their well-being which warmer climates will have (more precisely, they fail to allow for the extent to which they are likely to adapt to new situations; see Schkade & Kahneman 1998). However, this should not be a problem for the life satisfaction approach, which does not directly ask respondents for their perceptions of future or different environments, but rather assessed whether satisfaction scores differ across groups, including those exposed to different climates. However, as with any analysis, the potential for correlation between climate and some related factor cannot be entirely ruled out.

74 This Section draws upon Mourato *et al.* (2010).

characteristics of these houses (e.g. property type, floor area, tenure, age, number of bathrooms, number of bedrooms) was combined with data on their proximity to a variety of built environment facilities (e.g. distance to transport infrastructure, distance to the centre of the local labour market, local school quality, land area of ward, population density) and natural environment characteristics including:

- the proportion of the local area classified as Marine and Coastal Margins; Freshwaters – Openwaters, Wetlands and Floodplains; Mountains, Moorlands and Heaths; Semi-natural Grasslands; Enclosed Farmland; Coniferous Woodland; Broadleaved Mixed and Yew Woodland; Urban areas; and inland bare ground;
- the proportion of the local area which is made up of private gardens, greenspace and water features;
- the proportion of green belt and National Park land in the census ward in which a house is located; and
- the distance to various natural and environmental amenities, such as coastline, rivers, National Parks and National Trust properties.

While internal characteristics such as house size and number of bedrooms or proximity to places of work, have a major influence on the price of a property, the analysis showed that, after allowing for these, the local environment exerted highly significant effects on house prices; in other words, homeowners reflect their values for better environments through the amounts they are prepared to pay for houses which enjoy higher levels of environmental quality. In this manner, the hedonic pricing technique allows us to see the prices that homeowners implicitly pay for those environmental improvements. Because these ‘implicit prices’ are amounts that homeowners pay at the time of purchase, they reflect the stream of benefits purchasers expect to receive into the future rather than just the benefits obtained during the purchase year (i.e. they are capitalised present values). However, these are not perfect indicators of value as they reflect not only individuals’ underlying values but also the conditions of the local housing market. It might be that in some areas there is a good supply of high quality environments, while in others there is not; this may not change people’s value for such environments, but it will alter the implicit price they have to pay to enjoy these benefits in differing areas.⁷⁵ Nevertheless, these implicit prices represent a major advance over making decisions without any such information on the benefits of better environments and the disamenity of degraded areas.

Table 22.13 summarises these ‘implicit prices’ of environmental amenities in England. Results for all of England (column 1) reveal that many of the land use and land cover variables are highly statistically significant in influencing house prices and represent quite large implied economic effects. Domestic gardens, greenspace and areas of water within the census ward all attract a similar positive price premium, with a 1 percentage point increase in one of these land use shares increasing house prices by around 1%. Translating these into monetary implicit prices indicates capitalised values of around £2,000 for these land

use changes at the mean transaction price of £194,000. Regarding land cover shares (within 1 km squares) there is a strong positive effect from i) Freshwaters – Openwaters, Wetlands and Floodplain locations, ii) Broadleaved Mixed and Yew Woodland, iii) Coniferous Woodland and iv) Enclosed Farmland, with a 1 percentage point increase in the share of these types of land cover attracting house price premiums of 0.4% (on average £768), 0.19% (£377), 0.12% (£227) and 0.06% (£113) respectively.

We find that increasing distance from natural amenities such as rivers, National Parks or National Trust sites is associated with a fall in house prices. It is easy to misinterpret these relationships by extrapolating them outside the sample from which they were estimated. However, a simple example indicates the magnitude of some effects. So, while the data is not accurate enough to allow analysis of precisely what can be seen from any given house, moving from a property near to (but without a direct view of) a river to one, say, 1 km away, will lower the price of otherwise identical properties by some 0.9% (or, on average, £1,750). Clearly, homeowners place substantial values upon such environmental amenities.

We can use this analysis to predict the house price differentials that can be attributed to variations in the level of environmental amenities across England. This is achieved by effectively ignoring (holding constant) differences in house types and non-environmental characteristics across areas and only looking at the impact on house prices arising from variations in environmental quality. The resulting predictions therefore show the variation in prices around the mean in England as a result of environmental quality. These are mapped in **Figure 22.5**, with those areas in which environmental quality has the strongest positive impact on house prices being shaded in green, while negative impacts are shown in purple. Given that the mean house price in 2008 was just under £200,000, then this implies that in areas of the highest environmental amenity values, implicit prices were up to £68,000 higher than might be expected on average. Annualised over a long time horizon, this is equivalent to nearly £2,000/yr at the Treasury discount rate. These highest values are seen in areas such as the Lake District, Northumberland, the North York Moors, the Pennines, Dartmoor and Exmoor.

Returning to **Table 22.13**, columns 2–4 show the implicit prices (capitalised) for grouped Government Office Regions in England. These are derived from separate regression models for each regional group sample, with reported implicit prices based on the mean 2008 house price in each sample (reported in the last row of the table). Looking across these columns, although the results are qualitatively similar, it is evident that there are differences in the capitalised values and significance of the various environmental amenities according to region. While the ward land use shares of gardens, greenspace and water have remarkably similar implicit prices across regions, a notable difference is the greater importance of National Park designation in the Midlands regions (the Peak District and Broads National Parks), but lesser importance of National Trust sites. It is also evident that the value of Freshwater,

⁷⁵ A few studies have extended their analyses from implicit prices to underlying values. For example, Day *et al.* (2007) provide estimates of the underlying benefits of reducing road and rail noise in urban locations.

Table 22.13 Implicit prices by region (£, capitalised values).[†] Statistically significant results are indicated by:
***p<0.01, **p<0.05, *p<0.10. Source: Mourato *et al.* (2010).

	All England	London, South East and West	Midlands and East	North, North West and Yorkshire
Ward share of[‡]				
Domestic gardens	1,970***	1,769***	1,955***	2,487***
Greenspace	2,020***	2,068***	1,200***	1,773***
Water	1,886***	1,794***	1,179***	1,911***
Domestic buildings	4,242***	4,796***	610	2,292**
Other buildings	5,244***	5,955***	2,858***	4,593
Green belt	41	19	81	17
National Park	94	-184*	256***	131
Ward area (+10 km square)	0.017***	0.034***	0.013**	0.009***
Distance to[§]				
Coastline	-275	-56	-94	-348
Rivers	-1,751*	-2,446	-2,711***	-884
National Parks	-461***	-348**	-188	-782***
Nature reserves	-143	-1,322	632	-402
National Trust properties	-1,347***	-3,596***	-212	-1,117***
Land cover share in 1 km square[§]				
Marine and Coastal Margins	70	138	53	58
Freshwaters – Openwaters, Wetlands and Floodplains	768***	1,332***	36	233
Mountains, Moorlands and Heaths	166	-155	-258	832***
Semi-natural Grasslands	-27	6	-32	-191**
Enclosed Farmland	113***	123***	32	71**
Coniferous Woodland	227*	305***	307	-131
Broadleaved Woodland	377***	495***	412***	240*
Inland bare ground	-738***	-1,055***	-111	-479**
Sample size	1,013,125	476,846	341,527	194,752
Mean house price 2008	£194,040	£243,850	£181,058	£158,095

[†] The table reports implicit prices evaluated at regional mean prices. The analysis covers a sample of housing transactions in England, 1996–2008. Variables which are not of focal interest are considered in the analysis but omitted from the table (e.g. the impact of extra bedrooms).

[‡] 'Ward share of' shows the implicit prices for a 1 percentage point increase in the share of land in a specified use in the census ward containing the property. For gardens, greenspace, water, domestic and other buildings the omitted category is 'other land uses'.

[§] 'Distance to' variables shows the implicit prices associated with an increase of 1 km to the specified amenity.

[§] 'Land cover share in 1 km square' shows implicit prices for a 1 percentage point increase in the share of the specified land cover in the 1 km square containing the property (≈ 10,000 m² within nearest 1 million m²). Omitted category is 'Urban'.

Wetlands and Floodplain locations is driven predominantly by London and the south of England. Coniferous woodland attracts value in the regions other than the north, but Broadleaved woodland attracts a positive premium everywhere. Although Mountain, Moorland and Heath cover had no significant effect on prices in England as a whole, we see that it attracts a substantial positive premium in those locations where this land cover is predominantly found, i.e. the north, North West and Yorkshire.

Further restricting the sample to major metropolitan regions (not shown in **Table 22.13**) leads to a pattern of results that is broadly similar to those discussed above for England. Some effects become more significant, particularly those related to distance to coastline, rivers and National Parks and, as might be expected, green belt designation becomes more important. The results indicate implicit

prices amounting to around £5,800 for houses in green belt locations (although these are much higher in some areas), which offer access to cities, coupled with tight restrictions on housing supply.

While there are limitations to this analysis (discussed in Mourato *et al.* 2010), overall we conclude that there is substantial value attached to a number of natural habitats, designations, heritage sites, private gardens and local environmental amenities. While there is evidence of some substantial differences across regions, generally the underlying preferences for these amenities seems robust and may well be broadly transferable across the UK.

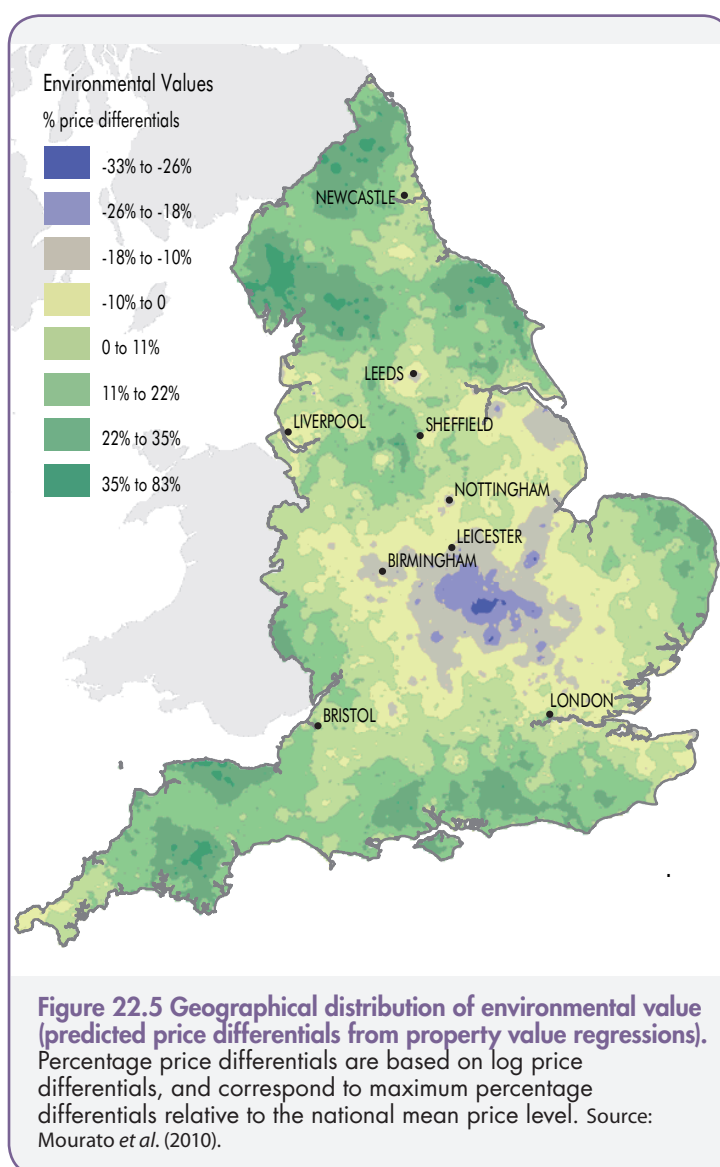
One limitation with the hedonic pricing approach is that it only reflects values which are embodied within property prices. A concern, then, is that this may underestimate the amenity and landscape aesthetics value of more remote

environments. Certainly some of the latter value will be captured within our prior assessments of outdoor recreation values. But still there is the risk that some values, especially residual non-use benefits, may be omitted. There is a clear need for an integrated assessment which addresses such omissions in a coherent framework which also avoids double counting. In the meantime we are forced to rely upon a mixture of assessments which risk both of these problems. Accepting that this may be an issue, there is nevertheless considerable evidence of amenity and aesthetic landscape values associated with various ecosystems. As a purely illustrative example of such benefits, **Table 22.14** provides estimates of average, total and marginal values for these as provided by inland and coastal wetlands in the UK. These are substantial, potentially amounting to roughly £1.3 billion p.a. Marginal values associated with expansions of Wetlands are also significant, although somewhat lower than present average benefits, reflecting the diminishing marginal benefits of increases in such resources.

Clearly there is a concern regarding the potential for overlap and hence double counting between the estimates provided by Mourato *et al.* (2010) and those from Morris & Camino (2010). However, ignoring these different sources would risk significant underestimation of ecosystem service benefits. In short, these values appear very substantial, yet there is a need for an integrated assessment of these benefits.

22.3.15 Education and Environmental Knowledge⁷⁶

Engaging with nature can lead to increased environmental knowledge. A novel accounting study of the investment value of environmental knowledge was undertaken for the UK NEA.⁷⁷ Given the importance of such knowledge within the education process, this study focused on environmental knowledge accumulation within the formalised education system for school-age children. Specifically, we consider two types of ecological knowledge experience related respectively to indoor and outdoor learning: i) the environmental knowledge embodied in successful student outcomes in GCSE and A-level examination in geography and biology,



at the end of the school year 2009/10, in England; and, ii) nature-related school trips, taking place outside the school, as well as 'citizen science' projects taking place within (and

Table 22.14 Estimated average, total and marginal values for amenity and aesthetics provided by inland and coastal wetlands in the UK.* All values are given in £, 2010 prices. Source: Morris & Camino (2010).

Ecosystem service-related goods	No. of sites [†]	Total area (ha)	Average value of service where present (addition to default value) [‡] (£/ha/yr)	Total value of service, assuming it is present in all UK inland/coastal wetlands [§] (£ million/yr)	Marginal value of service when provided by an additional hectare of new wetland [§] (£/ha/yr)
UK inland wetlands	1,519	601,550	339	204	227
UK coastal wetlands	693	274,613	2,080	1,081	1,394

* Values are area-weighted estimates for all UK inland wetland sites using the Brander *et al.* (2008) benefit function and CORINE land use data sets.

† Data on the number and area of wetlands were drawn from the European CORINE Land Cover Maps (Morris & Camino 2010).

‡ Default average values (where all of the ecosystem services specified in this table do not apply) are £303/ha/yr for UK inland wetlands and £1,856/ha/yr for UK coastal wetlands.

§ In contrast, the default total value of the existing inland wetland stock, assuming that none of the ecosystem services in the table apply, is £182 million/yr for UK inland wetlands and £509 million/yr for UK coastal wetlands.

§ The per hectare value of services associated with additional new wetlands is lower than the average per hectare value of existing wetlands. This reflects the diminishing marginal value of additional wetlands.

76 This Section draws in part from Mourato *et al.* (2010).

77 As noted by one reviewer, this method has not to date been subjected to rigorous academic peer review and so is offered with that caveat in mind.

around) school grounds. Obviously, such an assessment can at best provide only a very lower bound investigation of such values and important omissions such as the contribution of ecosystem services to the tertiary education sector require further analysis.

22.3.15.1 Environmental knowledge embodied in successful student outcomes

An economic interpretation of environmental learning experiences is that they are one element of the output of the education sector and hence, per the pioneering work of Jorgenson & Fraumeni (1989, 1992), an investment in human capital. Core to that method is the calculation of the present value of (lifetime) earnings from spending an additional year in formal education.

Mourato *et al.* (2010) follow Jorgenson & Fraumeni (1989, 1992) to define three groups of school pupils for the academic year 2009/10: i) those who do not attain qualifications; ii) those who attain GCSEs in the grade range from A to C and iii) those who pass A-level exams.⁷⁸ Assuming initial earnings for group i) being at the current minimum wage for 16–18-year-olds of £3.64/hour then, following Dearden (1999), Dearden *et al.* (2000) and Blundell *et al.* (1999, 2004), it is assumed that, relative to group i), the wage rate for group ii) is 15% higher and that of group iii) is 22% higher. Numbers in each group are taken from ONS (2009).⁷⁹ Using these data and assuming a retirement age for all groups of 68 (reflecting expected changes in this age), we estimate future earnings from age 16 for group i), from age 17 for group ii) and from age 19 for group iii) assuming incomes growth of 1.5%. Present values of these income streams are calculated using a HM Treasury discount rate of 3.5%.

We then seek to identify the environmental component of this educational attainment and its value. We focus on geography and biology as the fields of study where, at school level, there is formal evidence of significant environmental components to the curriculum, either in guidelines provided by national curricula and/or official examination boards. Determining the precise weight that ecological education

has in these studies is clearly contentious and subject to variation across schools. Nevertheless, on the basis of consulted documentation (AQA 2009, 2010; Edexcel 2008a,b), we assume that the weights reflecting the ecological components to be the following: GCSE geography = 0.15; GCSE biology = 0.25; GCSE (basic) science = 0.08; A-level geography = 0.15; and A-level biology = 0.25.

Results are provided in **Table 22.15**. The left-hand side of the table gives the number of students accomplishing specified examination outcomes. The right-hand side gives corresponding values. These are the product of pupil numbers and the ‘adjusted’ present values for representative individuals achieving the relevant qualifications (as estimated above) in 2010. Our tentative findings indicate that the annual value of environmental knowledge embodied in successful student outcomes in (relevant) GCSE and A-level examinations at the end of the academic year 2009/10 is substantial, at just over £2.1 billion. However, some caution is needed in interpreting these results. The data that we provide cannot be interpreted as the net benefit of the production of environmental knowledge (i.e. relative to other forms of education). Ours is purely an accounting framework that attempts, in a very approximate way, to identify some portion of the environmental component of school education. Nevertheless, we would argue that the findings are instructive, not least in indicating, in explicit terms, that the value of this environmental knowledge is possibly substantial.

22.3.15.2 Environmental knowledge embodied in nature-related school trips

Environmental education also occurs outside the classroom and Mourato *et al.* (2010) consider case studies of both school trips to UK nature reserves and a national ‘citizen-science’ project as follows:

- There is no central record of the number of school trips to nature reserves and related environmental resources annually. However, during the 2009/10 school year just over 50 RSPB reserves played host to nearly 2,000 school trips involving over 57,000 students and staff. Valuations of travel costs and travel time suggest an economic expenditure value ranging from just under £850,000 to just over £1.3 million for these trips alone.
- Taking the RSPB Big School Birdwatch as one example of a citizen science project, in 2010 some 75,500 people participated (69,101 children and 6,275 adults) from 1,986 schools. Utilising a similar methodology to the previous case study gives a value of this time of about £375,000 or £188 per participating school.

Neither of these case studies provide true economic valuations of educational benefits concerned, reporting instead just the ‘cost of investment’ involved in these undertakings. Nevertheless, assuming that these undertakings were deemed to be value for money, such costs should provide a lower

Table 22.15 The annual value of environmental knowledge in GCSE and A-level attainment for school leavers in 2010.*

Source: Mourato *et al.* (2010).

	Candidates ('000)		Value of environmental knowledge (£ million/yr)		
	GCSE	A-level	GCSE	A-level	Total
Geography	118.2	29.2	426.9	134.7	561.6
Biology	110.2	52.7	663.4	405.9	1,069.2
Science	258.4	n/a	497.8	n/a	756.2
Total	486.8	81.9	1,588.1	540.6	2,128.7

* The values refer to successful candidates who would have received their results in these GCSEs and A-levels in the summer of 2010.

78 Those attaining higher educational qualifications (containing environmental knowledge in our accounting year) are not considered in this analysis as it stands. However, inclusion of this further increment in ecological knowledge, in principle, could be incorporated as a further (net) investment.

79 Note that these need not equate to labour market participation figures.

bound minimum of the values concerned. This suggests that there may be a substantial underlying value within the much larger number of total school trips and citizen science projects undertaken each year.

22.3.16 Health⁸⁰

Environmental quality and proximity to natural amenities is increasingly being recognised as having substantial effects on physical and mental health, both directly and indirectly (e.g. Bird 2004). Broadly this can happen in three ways. First, the absence of environmental quality can directly impact upon human health. Second, natural settings can act as a catalyst for healthy behaviour, leading, for example, to increases in physical exercise, which affect both physical and mental health (Pretty *et al.* 2007; Barton & Pretty 2010). Third, simple exposure to the natural environment, such as having a view of a tree or grass from a window, can be beneficial, improving mental health status (Pretty *et al.* 2005) and physical health (Ulrich 1984). Health outcomes in this respect can be disaggregated into two categories: reductions in mortality and reductions in morbidity (including physical and mental health).

The focus upon ecosystem services underpinning the UK NEA means that the major emphasis of our analysis is upon the second and third pathways mentioned above. However, some consideration of the direct impact of poor environmental quality upon health is worthwhile, if only for completeness. A key example of such a pathway is the issue of air quality. The chronic health effects of particulate matter alone is estimated to cost around £15 billion per year (IGCB 2010).⁸¹ In comparison, action to address this cost can be highly cost effective. For example the latest vehicle emission standards are estimated to reduce the value of this health impact by around £1 billion annually at a cost of around £350 million p.a. (IGCB 2010).⁸² A second example is the issue of noise which is estimated to incur health costs of around £2 billion p.a. (and wider costs of a further £5–£7 billion⁸³; IGCB(N) 2010). Here the costs of noise mitigation measures vary substantially according to local circumstances but in many areas the benefit:cost ratio is strongly positive (pers comm., Mallika Ishwaran, Defra 2011).⁸⁴

Returning to our ecosystem focus, Mourato *et al.* (2010) conducted a preliminary investigation of the valuation of the impacts of marginal changes in the provision of natural habitats and greenspaces on physical and mental health. They address both of the pathways identified above: i) health improvements arising from additional exercise created by the provision of natural habitats and green settings; and ii) health benefits arising from more passive forms of contact with nature such as viewing nature or being within natural spaces.

22.3.16.1 Value of the health benefits of green exercise

Willis (2005) identifies three key steps in the valuation of the health benefits of 'created exercise' due to additional provision of greenspace: i) measuring the physical and mental health impact of exercise; ii) valuing the health benefits of exercise; and iii) estimating the probability of additional exercise with changes in greenspace. Mourato *et al.* (2010) analyse each in turn.

The only exercise that should be directly attributed to the provision of natural settings is what Willis (2005) calls 'created exercise', i.e. exercise which would not have occurred otherwise. Exercise which would have occurred anyway in another setting (e.g. the gym or urban pavements) should not be included in the calculations as it is not truly additional. It is, however, very difficult to identify created exercise. The following calculations follow the Willis (2005) approach and attempt to focus on created exercise under a scenario whereby changes in countryside and parks management lead to an additional reduction of 1 percentage point in the numbers of sedentary people in the UK. Reduction in sedentary life and increase in exercise lead to a number of proven health benefits which include reductions in mortality and morbidity due to: i) coronary heart disease (CHD); ii) colo-rectal cancer; iii) stroke; and iv) stress, anxiety and depression (morbidity only). We obtained up-to-date data on mortality and morbidity for CHD, colo-rectal cancer, stroke and depression. The change in excess cases of morbidity and mortality from these conditions associated with a 1 percentage point reduction in sedentary behaviour are then calculated. This is valued using the theoretically correct WTP approach (e.g. Krupnick 2004; Pearce *et al.* 2006), based on the trade-offs that individuals would make between health and wealth, to estimate the economic value of these health impacts. For mortality, government estimates of the value of a preventable fatality (VPF) of £1,589,800 (DfT 2007) are used; for morbidity the value used for CHD prevention is based on the Department for Transport's (DfT 2007) value for a slight injury (£13,769), while the stroke prevention value is based on its value for a serious injury (£178,640). The value for cancer prevention is taken from Hunt & Ferguson (2010) and reflects the existence of a 'dread' factor associated with diseases that are long and painful (£288,304). Finally, the value for reduction of mental illness is based on Morey *et al.*'s (2007) estimate of WTP to eliminate depression (£5,343).

Estimates of the value of health benefits arising from a 1 percentage point reduction in the sedentary population are discussed in detail by Mourato *et al.* (2010). These show that a change in natural habitats that causes a 1 percentage point reduction in sedentary behaviour would provide a

⁸⁰ This Section draws on Mourato *et al.* (2010).

⁸¹ For comparison, the health costs of obesity are estimated at £10 billion p.a. (pers comm., Mallika Ishwaran, Defra 2011).

⁸² Furthermore, tackling climate change can deliver significant synergies in terms of knock-on air quality benefits. Defra (2011) presents evidence showing that the synergy benefits of optimizing for climate change and local air quality objectives to 2050 have a net present value of around £24 billion.

⁸³ The cost of the levels of noise prevailing in 2010 is estimated at between £7billion–£9billion p.a.—an amenity cost of £3–5 billion, health cost of £2 billion and productivity losses of £2 billion (IGCB(N) 2010).

⁸⁴ Source states that, even if only amenity values are considered then benefit:cost ratio can reach as high as 8:1 in some areas.

total benefit of almost £2 billion p.a. (using WTP-based values), across the three physical conditions (CHD, colorectal cancer and stroke) and the mental health condition considered (stress, anxiety and depression). However, if all people over 75 years are excluded from the analysis (on the basis that they are less able or likely to be physically active), then the benefits fall to just over £750 million. Given this, the key question left to answer is: if a green living environment does indeed provide an incentive to be physically active, how much true additional exercise is created with the extra provision of greenspaces that would not have taken place otherwise? Unfortunately, there are large gaps in knowledge in this area, as environmental attributes appear to be among the least understood of the known influences on physical activity. There is a limited body of evidence that appears to suggest patterns of positive relationships between some environmental attributes and physical activity, such as walking or cycling. Reviews by Humpel *et al.* (2002), Owen *et al.* (2004) and Lee & Maheswaran (2010) show that the aesthetic nature of the local environment, the convenience of facilities (such as footpaths and trails) and accessibility of places to walk to (such as parks and beaches) are often associated with an increased likelihood of certain types of exercise orientated walking. However, several other studies found no link between recreational physical activity and greenspace provision. A recent large-scale study of nearly 5,000 Dutch people by Maas *et al.* (2008) found that the amount of greenspace in people's living environment has little influence on their level of physical activity. Given this, Mourato *et al.* (2010) find no conclusive evidence on the strength of the relationship between the amount of greenspace in the living environment and the level of physical activity. This would suggest that, at the present time, it is not possible to accurately value the health benefits of created exercise due to additional provision of greenspace. However, this is a rapidly developing field. For example, recent research by Coombes *et al.* (2010) shows that those who live within 500 m of accessible green space are 24 per cent more likely to meet recommended levels of physical activity. Figures from the Department of Health suggest that better access to open spaces could reduce healthcare costs by over £2 billion per year (pers comm., Mallika Ishwaran, Defra, 2011). While such cost savings cannot be taken as valid estimates of the benefit of such health improvements, nevertheless they serve to underline the substantial nature of likely values.

22.3.16.2 Valuing the health and well-being benefits of exposure to nature

There is now a substantial body of evidence suggesting the existence of a wide range of health and well-being benefits associated with greenspace over and above those induced by increased exercise. In a recent review, Lee & Maheswaran (2010) reports associations between contact with greenspace and a variety of psychological, emotional and mental health benefits, reduced stress and increased quality of life. This has

led to a recent linkage between the economics of happiness and environmental economics (Welsch 2009). Moreover, research spanning more than two decades suggests that mere views of nature, compared to most urban scenes lacking elements of the natural environment, appear to have positive influences on emotional and physiological states, providing restoration from stress and mental fatigue (Ulrich 1986; Kaplan 2001) and even improve recovery following operations in hospital (Ulrich 1984). These health benefits of non-exercise-related exposure to nature are likely to be substantial and pervasive, given the lack of substitutes and the size of the population potentially affected.

Mourato *et al.* (2010) use novel techniques, including a newly commissioned geo-located survey, to estimate the physical and mental health effects associated with UK ecosystem types, domestic gardens, managed areas and other natural amenities. Data were collected by a web survey during August 2010. A total of 1,851 respondents completed the survey. Measures of general and physical health were obtained,⁸⁵ including assessments of the impact of health upon personal utility (broadly speaking, the individual's well-being). These were then related to indicators of the local environmental characteristics such as the ecosystem types describing the physical land cover within a 1 km radius of the respondent's home location (such as woodland, freshwater, farmland or mountains) and direct questions regarding views of greenspaces and water from the respondent's home, frequency of use of domestic gardens, of open countryside, and of non-countryside greenspaces such as parks, recreation grounds and cemeteries, as well as distance to various natural and environmental amenities, such as coastline, rivers, National Parks and National Trust properties. A wide variety of further information was gathered to allow for differences between gender, age, qualifications, work status, religiosity and income as well as house prices and postcodes.

Analysis of these various data detected positive links between proximity of the home to specific habitat types and the health-related utility score, although such links were not observed between habitat types and simple aggregate physical and emotional health indicators. There appear to be strong, positive relationships between green views from the home and emotional well-being and health utility. Specifically, having a view of greenspace from one's house increases emotional well-being by 5% and the general health utility score by about 2%; regular use of gardens and greenspaces has a similar positive effect on well-being. Using a garden weekly, or more often, increases physical functioning and emotional well-being by around 3.6% and the health utility score by 2.7%; Similarly using non-countryside greenspace monthly, or more frequently, increases physical functioning and emotional well-being by 3.4% and 2.6% respectively, and the health utility score by 1.8%. Furthermore, an increase in 1% of the area of freshwater, farmland and broadleaved woodland within a 1 km radius of the home increases health utility by 0.3%,

⁸⁵ The RAND SF-36 Health Survey was employed (see Brazier *et al.* 2002). This is the leading general health measure, comprising 36 survey items, with standardised administration and item scoring to produce several validated sub-scales. The 'physical functioning' and 'emotional well-being' subscales were used as outcome variables.

0.1% and 0.1% respectively. **Table 22.16** summarises these effects. However, it is important to note once again that the associations we have estimated cannot be interpreted as causal effects. There may be variables omitted from the models that cause changes in both the dependent and explanatory variables, and/or the dependent variable may itself be a cause of some explanatory variables.

The final column in **Table 22.16** reports tentative values of the health changes estimated above. The general health measure used by Mourato *et al.* (2010) is capable of detecting changes in health in a general population (Hemmingway *et al.* 1997). As such, it may be possible to use our survey results to tentatively estimate the monetary value of the health benefits associated with increasing the number of people making monthly visits to greenspaces and having views of grass, or associated with increasing particular types of land cover. To achieve this, Mourato *et al.* (2010) first relate the health index used in their survey to Quality Adjusted Life Year (QALY) measures associated with the environmental changes of interest. Quality Adjusted Life Years are measures of health benefits that combine length of life with quality of life, where quality of life is assessed on a scale where 0 typically represents death and 1 represents full health (Drummond *et al.* 1997). There is an emerging literature attempting to empirically estimate the value of QALYs (e.g. Jones-Lee *et al.* 2007; Mason *et al.* 2009; Tillig *et al.* 2009; Baker *et al.* 2010). Although there is currently no consensus about what the monetary value of a QALY is or how to calculate it (Willis 2005; Tilling *et al.* 2009; Donaldson *et al.* 2011), one approach involves deriving a ‘value of a life year’ from existing empirical estimates of the VPF (Jones-Lee *et al.* 2007). Of particular interest to us is a special case of this approach proposed very recently by Mason *et al.* (2009), that consists of estimating the value of a QALY based only on quality of life changes. The Mason *et al.* (2009) study is based on UK figures and provides a value

for the prevention of a non-fatal injury, from which they in turn estimate monetary values of a QALY ranging from £6,414 to £21,519. Given that the environmental changes being considered are likely to have impacts mostly on quality of life (rather than on life expectancy), these seem to be the most appropriate values to use.

The last column of **Table 22.16** contains the very tentative results of the calculation outlined above. It shows the estimated annual health benefits associated with having a view of nature, using the garden often, visiting greenspaces regularly and increasing the proportion of broadleaved woodland, freshwater and farmland cover. We note that these figures are indicative only, are subject to many assumptions as described above, and should therefore be treated with caution.

22.3.16.3 Direct impacts of climate change upon health⁸⁶

In its report on the health impacts of climate change, the Department of Health estimates that under a medium to high scenario, climate change might, by the 2050s, reduce the number of cold-related deaths by up to 20,000 whilst increasing the number of heat-related deaths by 2,800 (POST 2004; Department of Health 2008). We can value these net avoided deaths by applying the UK official value of statistical life (£1.1 million in 2008 prices) to obtain a benefit estimate of £18.9 billion. However, such estimates need to be treated with some caution. First, many of those whose deaths would be averted would be elderly, with a short remaining life expectancy because of pre-existing conditions. Second, some studies present evidence to support the use of declining values for preventing fatalities in such circumstances.⁸⁷ Finally, climate change is likely to be far more erratic than just a simple increase in temperature, and this variability requires a sophisticated treatment of likely impacts which calls for extensions beyond the present work.

Table 22.16 Health changes and contact with nature: summary findings.* Source: Mourato *et al.* (2010).

Explanatory variable	Difference in explanatory variable	Associated health differences			
		Physical functioning (%)	Emotional well-being (%)	Health utility score (%)	Tentative annual value per person (£)
Having a view over greenspace from your house	No view → any view	–	+5.0	+2.1	135–452
Use of own garden	Less than weekly → weekly or more	+3.5	+3.7	+2.7	171–575
Use of non-countryside greenspace	Less than monthly → monthly or more	+3.4	+2.6	+1.8	112–377
Local freshwater, wetlands and floodplains	+1% within 1 km of the home (+3.14 out of 314 ha)	–	–	+0.3	20–68
Local enclosed farmland	+1% within 1 km of the home (+3.14 out of 314 ha)	–	–	+0.1	4–12
Local broadleaved/mixed woodland	+1% within 1 km of the home (+3.14 out of 314 ha)	–	–	+0.1	8–27

* Based on analyses of data for England and Wales.

⁸⁶ This Section draws on Maddison (2010).

⁸⁷ Aldy & Viscusi (2007) find an inverted U-shaped relationship between age and the value of statistical life.

22.3.17 Agricultural Food Production

22.3.17.1 Introduction and overview

The natural environment clearly plays a major role in agricultural food production. However, when undertaking an economic analysis of agricultural ecosystem services we need to control for the contribution to food values which also comes from other inputs such as machinery, labour and chemical fertilisers. To ignore the latter would be to implicitly assume that ecosystem services are the only inputs to agricultural food production and so significantly overestimate the value of those services and undermine the validity of our analysis.⁸⁸ One way to avoid this problem is to examine the change in value of agricultural output when we vary a given ecosystem service by some marginal (unit) amount, holding all other inputs constant.⁸⁹ Within reason,⁹⁰ we can then use these findings to estimate the impacts of whatever multiple unit change in ecosystem services is of policy interest. Again within limits, we can extend this approach to also consider cases where more than one ecosystem service changes at the same time.⁹¹

Given this methodology, an obvious initial question concerns which ecosystem services might be of interest to decision makers. Obviously, even the most self confident of policy makers will not be interested in the impacts upon agriculture of changing the elevation of an area. Indeed, there are a number of ecosystem services which are likely to stay fairly constant into the future and are therefore of limited policy interest. However, one obvious source of potentially significant variation in pertinent ecosystem services is climate change.⁹² We examine two of the key effects of climate change upon agriculturally significant ecosystems services: variations in temperature and precipitation.

Our analysis draws upon the newly compiled, highly spatially disaggregated datasets embracing temporal variation across a long time series. Economic theory is drawn upon to construct new behavioural models of land use decision making. These predict how farm land use varies, not only because of factors such as the prices of goods, costs of manufactured inputs and changes in agricultural policy but also with the farm's environmental characteristics, including temperature and precipitation.

The model is validated through standard comparisons of actual versus predicted measures. Here we deliberately omit

some of the data available to us, for example by dropping observations on land use in the most recent years of our dataset. We feed the remaining data into our analysis and produce a model of the factors determining land use. We then use that model to predict land use in the omitted years. These predictions are compared with the actual land use in those omitted years. The error between our predictions and what actually happened gives us a very robust insight into the reliability of our model. If, as we show later is indeed the case, we find our model to be highly reliable, we can use all of the available data to improve it even more and feel justified in using that model to examine what would happen if circumstances changed—within reason. This latter caveat is important. Any analysis that draws upon data from the past cannot be reliably applied to totally different future conditions, that is we cannot push the model too far outside the range of prior observations. However, our ambition of using it to examine the impacts of predicted climate change has a good claim to being robust in this respect. Because we build our model using data from right across the full extent of GB (including the generally warm and dry South East and the typically colder and wetter North West) and across many years (including both warm and cold periods), then this information embraces much (although not all) of the range of climatic conditions predicted for at least the first half of the present century.⁹³

One caveat that we do acknowledge is that, due to time constraints, the analysis presented here does not adjust from market prices to underlying values. To do so requires allowances to be made to remove market imperfections such as those brought about by subsidies and other interventions. This is likely to reduce the size of estimates reported here, an issue which should be kept in consideration throughout this analysis and that conducted for agricultural values within Chapter 26.

22.3.17.2 The CSERGE land use model

Recent research within the SEER project based at the Centre for Social and Economic Research on the Global Environment (CSERGE), University of East Anglia, develops a new model of agricultural land use which is particularly suitable for the ecosystem service assessment conducted under the UK NEA. Below we briefly overview the model specification and the data used for estimation, and summarise the main

88 These manufactured and human capital inputs could be reassigned to other uses. Therefore the loss of some portion of ecosystem services could in part be compensated for by reassigning the former capital to other ends. Note, however, that this offsetting compensation should ideally be valued by examining the 'opportunity cost' value (i.e. the value that this non-ecosystem capital could generate if applied to the next best alternative use).

89 Of course, basic microeconomics shows that if one factor of production changes (e.g. the level of some ecosystem service) then it is likely that it will be cost-beneficial to alter other inputs. However, ignoring this substitution gives us an insight into the value of that initial input.

90 Recall from our methodological summary that marginal values are typically robust for some ranges but can change substantially if we consider very large alterations in circumstances (e.g. the marginal value of a 1% reduction in water availability might provide a perfectly good basis for valuing more substantial losses up to a point, but eventually a further change in water availability starts to have a very different impact on food production). This is why marginal values cannot be used to estimate the total value of ecosystem services.

91 Similarly, any combined change in ecosystem and man-made inputs can be assessed. However, it should be pointed out that there are real limits to the state of natural science understanding regarding what may happen when large numbers of ecosystem services all change simultaneously, particularly in the context of an overarching stressor such as climate change.

92 Elsewhere in this chapter we consider several of the indirect effects which agriculture has upon other ecosystem services such as carbon storage, water quality and biodiversity. An issue which is flagged for future consideration is the problem of soil erosion.

93 This is particularly true for most areas of the country where climate predictions are that conditions in, say, the north and west will become more similar to those of the south and east. Obviously the region which will most noticeably move into new climatic territory (i.e. not captured in UK data from the past) will be the South East and so arguably our model is less robust there.

results. For a more detailed discussion of the methodology, see Fezzi & Bateman (in press).

Theoretical basis and statistical modelling. The model is rooted in basic economic theory (Chambers & Just 1989), which is used to link profit-seeking behaviour by farmers to their consequent land use.⁹⁴ The model considers the full range of possible outputs which GB farmers have produced to date, the prices of those outputs,⁹⁵ the cost of inputs and the existing policy regime including incentives, disincentives and constraints. The model also incorporates detailed descriptions of the physical environmental characteristics of each farm. All of these data are collected at a very detailed spatial resolution with information on outputs being held at a 2 km grid square and other data held at the finest resolution available. The analysis then seeks to examine how changes in these factors across space (all of GB) and time (dating back to the late 1960s) result in farms allocating different shares of their available land to different activities. Care is taken to allow for the likelihood that many of the relationships underpinning farm land use decisions are interrelated and non-linear. The model building procedure uses statistical estimation techniques which allow for this complex set of relationships and the nature of the underlying data. It also estimates all land use decisions simultaneously, to mimic the decision process of the farmer who considers all farm land and all output options together when determining the land use for the farm.

Data sources. The data used for this analysis integrates multiple sources of information dating back to the late 1960s to assess the financial, policy and environmental drivers of land use change. Collected on a 2 km grid square (400 ha) basis, these data cover the entirety of England, Wales and Scotland, and encompass, for the past 40 years: i) the share of each land use and the numbers of livestock, ii) environmental and climatic characteristics, and iii) policy and other drivers. However, data on yields and profits are not available at the disaggregated level required by this analysis. While we could overcome this problem by moving the analysis to a less disaggregated level at which such information is available (and indeed do so in other work—see Fezzi *et al.* 2010b), this would reduce the accuracy with which we could understand the impact of variations in ecosystem services upon agricultural land use. Given that this is the main purpose of the present exercise, we retain the highly detailed spatial basis of this land use analysis and use secondary data to assign money values to these uses.

Data on agricultural land use and livestock numbers for each 2 km (400 ha) grid square for the whole of GB were taken for 17 unevenly spaced years between 1969 and 2006.⁹⁶ This yields roughly 60,000 grid-square records each year, giving about 1 million records in total. This allows us to explicitly model six of the agricultural land use types as defined in the Agricultural Census: cereals (including wheat,

barley and oats); oilseed rape; root crops (potatoes and sugar beet); temporary grassland (grass being sown every 3 to 5 years and typically part of an arable crop rotation); permanent grassland (grassland maintained perpetually without reseeding); and rough grazing. Together, these account for more than 88% of the total agricultural land in GB. We include the remaining area in an 'other' land category encompassing horticulture, other arable crops, woodland on the farm, set-aside, bare, fallow and all other land (e.g. ponds, paths). In addition to the above, the model also allows us to estimate three rates of livestock intensity for dairy and beef herds and flocks of sheep.

For each 2 km grid square we consider a detailed specification of the environmental factors influencing farmers' decision making. For each grid square, we represent climate through Met Office data on i) average temperature in the growing season (April–September) and ii) accumulated rainfall during the growing season. Other data on environmental characteristics included soil depth to rock, volume of stones, various categories of soil texture (fine, medium fine, medium, coarse, peaty), mean altitude, and a measure of slope for the agricultural land in the square. Met Office values are taken from 5 km grid square climatic averages for the period 1961–1990 as calculated from the monthly data available from the Met Office website (www.metoffice.gov.uk) and interpolated to 2 km to match with our land use data. This is the same baseline used by the UKCIP09 (www.ukcip.org.uk) to derive climate change scenarios. Soil characteristics are derived from the 1 km raster library of the European Soil Database (van Liedekerke *et al.* 2006), which we aggregate at a 2 km level. Altitude and high slope (greater than 6 degrees) were both derived via geographical information system (GIS) analysis of the Ordnance Survey, Digital Terrain Model.

Policy determinants which in some way alter agricultural prices or costs were directly incorporated into the model. Area designations such as Nitrate Sensitive Areas (NSAs), Environmentally Sensitive Areas (ESAs) and National Parks were incorporated by denoting the area in each 2 km square under each scheme.

It should be noted that NSAs are voluntary, being established in 1990 and extended in subsequent years. These were introduced in order to test the effects of farming practices on nitrate levels in aquifers, as well as to reduce nitrate levels in selected groundwaters used for public water supply. ESAs were introduced in 1987 and have undergone various extensions in subsequent years. They were launched to safeguard and enhance areas of particularly high landscape, wildlife or historic value. Participation in ESA schemes is also voluntary, and farmers receive monetary compensation for engaging in environmentally friendly farming practices, such as converting arable land to permanent grassland, and establishing hedgerows. Many National Parks were established in the 1950s with some

94 We freely acknowledge the evidence of non-profit motivated behaviour discussed by Pike (2008) and others. However, our model draws upon a long time series of data from across the country. Analysis shows that, over such long time periods and after allowing for lags in response times, a profit maximising assumption provides a strong fit to the data on observed behaviour and land use choices.

95 Prices obviously have a highly significant impact on agricultural land use. For an insight into how these have changed in real terms from the middle of the last century see Tinch *et al.* (2010).

96 These data are derived from the June Agricultural Census as available online from EDINA (www.edina.ac.uk). Note that in 2005 and 2006 only Welsh data are available. No more recent data were available. Descriptive statistics for each land use and livestock numbers are given by region and nationally in Fezzi *et al.* (2011).

extensions in the 1980s. Farms located within the boundaries of National Parks can benefit from direct payments if they manage their land by environmental planning and undertake low-intensity activities.

Results. The analysis provides a set of equations describing the share of each of the six land uses (plus the 'other' agricultural land) and the number of each of the three types of livestock in each 2 km square. Details of these equations are given in Fezzi & Bateman (in press), but in summary these show that both land use and livestock numbers are determined by agricultural prices, input costs, a variety of policy measures, and a large number of physical environmental conditions on farms, including those temperature and precipitation variables affected by climate change.

Validation and extension to all of the UK. Our analysis is tested using a comparison of predicted with actual values, as outlined previously. This is undertaken for all land use types and livestock numbers. Formal statistical testing shows that the model performance is highly satisfactory. **Figure 22.6** illustrates two of these comparisons, showing actual and predicted shares of cereals and rough grazing in 2004. Even though some minor differences can be seen (e.g. the model somewhat overpredicts cereals in the English Midlands and underpredicts cereals in Eastern Scotland) the two comparisons show essentially the same spatial patterns of land use. It should be noted that the actual data is somewhat 'blocky', with abrupt changes in recorded cereal between grid squares. This is due to data being gathered at parish level and subsequently allocated to grid squares. The predicted values avoid this problem.

Therefore, we now have a model which provides robust estimates of land use change based upon observations from the past 40 years and across the entirety of GB. Note, however, that data were not available on farm performance in Northern Ireland. Nevertheless, the range of environments and circumstances in our model encompass those observed within Northern Ireland and therefore, we can extend our

analysis to the whole of the UK by applying the relationships estimated for GB to data detailing the physical environment of Northern Ireland.

22.3.17.3 Valuing ecosystem services: the impact of climate change

As outlined above, by examining those agricultural ecosystem services most likely to be altered by climate change we estimate how farm outputs will vary and hence assess the value of those services.⁹⁷ The UK Climate Impacts Programme (UKCIP) (www.ukcip.org.uk) provides the most up-to-date predictions regarding future climate in the UK. Importantly, the most recent UK Climate Projections (UKCP09) are spatially explicit, being presented at a 25 km grid square resolution. Such data is inherently compatible with our spatially explicit model of agricultural land use.

For the purposes of valuing ecosystem services, we examine the impacts on the value of agricultural production of the UKCIP09-predicted changes in monthly average minimum temperature, maximum temperature and precipitation in the growing season (from April to September). Predictions are taken up to the end point of the UK NEA analysis in 2060 and show temperatures increasing and growing season precipitation falling over this period. For further sensitivity, we consider predictions calculated under both the low and high GHG emission scenarios set out by the IPCC.⁹⁸ Obviously, trends are somewhat more extreme under the higher emission scenario.

As an illustration of the UKCP09 trends, **Figure 22.7** shows precipitation in the growing season in 2004 and 2040. Similarly, **Figure 22.8** repeats this analysis for temperature (measured as growing season average). Inspection of these figures shows that rainfall is reduced over time, particularly in the eastern and central parts of England. In contrast, temperatures increase noticeably over time in all areas.

Land use change predictions. By feeding the UKCP09 climate predictions into our model, we obtain predictions of the change in land use in each 2 km grid square across

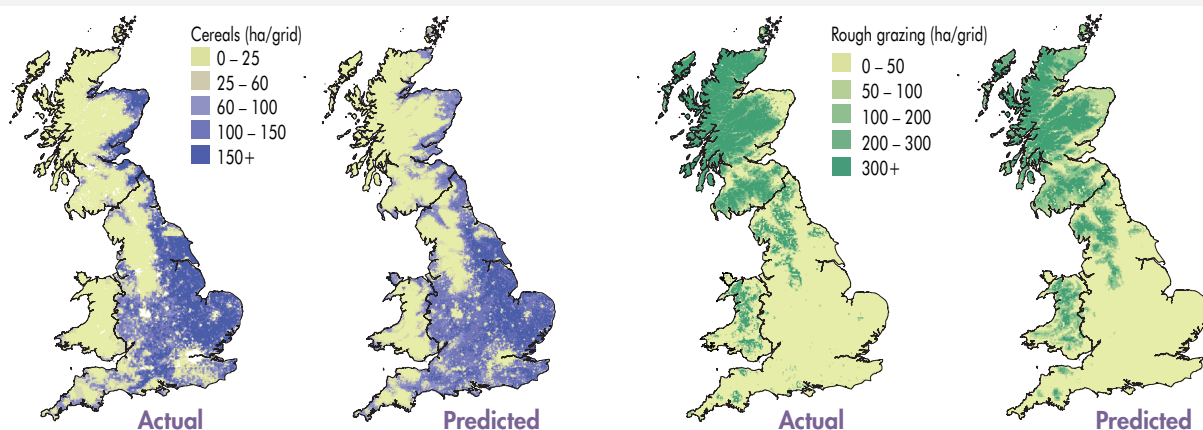


Figure 22.6 Comparisons actual and predicted land use for cereals and rough grazing in 2004. Source: SEER (2011).

⁹⁷ A point of debate here concerns precisely which service we are valuing here. Fezzi *et al.* (2010b) argue that climate services are under assessment, with their value being reflected in the induced variation in food production. Another reviewer has argued that pure provisioning service (food production) is being valued.

⁹⁸ These correspond respectively to the SRES B1 and the SRES A1FI in the IPCC Special Report on Emissions Scenarios (Nakicenovic *et al.* 2000).

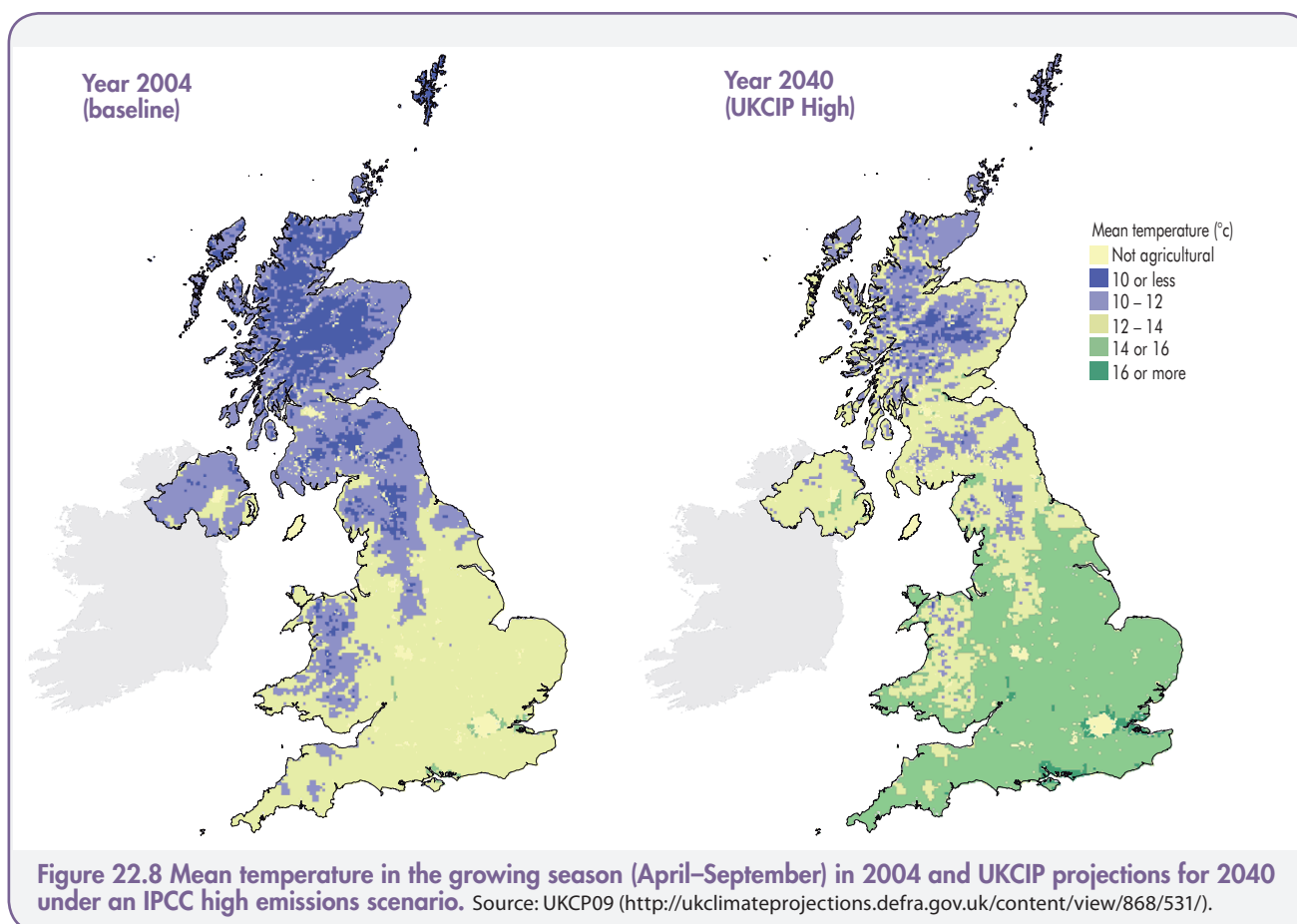
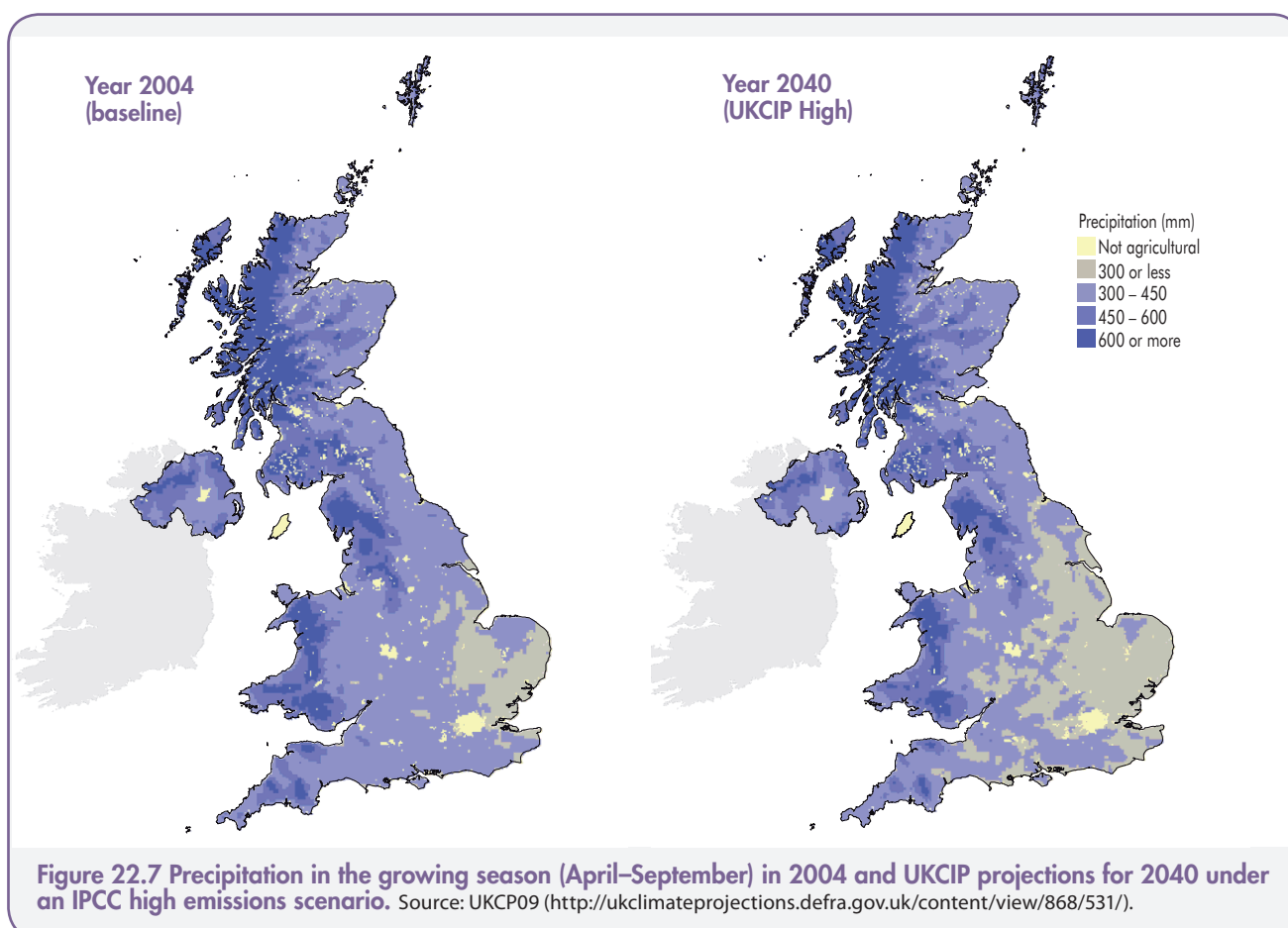


Table 22.17 Average predicted land uses and livestock intensities in Great Britain (2004–2060) under both low and high emission climate change scenarios. *Land use cells:* upper value is average hectare per 2 km grid square (400 ha); lower value (in parentheses) is the percentage of the square. *Livestock cells:* average number of head per 2 km grid square (400 ha). Source: SEER (2011).

Year	Land use						Livestock		
	Cereals	Oilseed rape	Root crops	Temporary grass	Permanent grass	Rough Grazing	Dairy	Beef	Sheep
Low emission scenario									
2004	61.1 (15.3)	7.4 (1.9)	0.7 (0.2)	19.4 (4.9)	85.1 (21.3)	98.2 (24.6)	28.7	90.8	535.8
2020	47.9 (12.0)	4.4 (1.1)	1.0 (0.3)	21.0 (5.3)	110.6 (27.7)	74.3 (18.6)	49.4	84.3	524.5
2040	41.2 (10.3)	3.4 (0.9)	1.1 (0.3)	22.3 (5.6)	113.5 (28.4)	71.0 (17.8)	55.2	75.6	498.4
2060	36.8 (9.2)	2.8 (0.7)	1.3 (0.3)	22.8 (5.7)	110.4 (27.6)	72.7 (18.2)	57.2	67.3	473.8
High emission scenario									
2004	61.1 (15.3)	7.4 (1.9)	0.7 (0.2)	19.4 (4.9)	85.1 (21.3)	98.2 (24.6)	28.7	90.8	535.8
2020	48.5 (12.1)	4.5 (1.1)	1.0 (0.3)	21.0 (5.3)	110.8 (27.7)	74.9 (18.7)	48.8	86.1	530.0
2040	37.8 (9.5)	2.9 (0.7)	1.2 (0.3)	22.8 (5.7)	113.6 (28.4)	72.4 (18.1)	57.3	72.4	488.5
2060	21.7 (5.4)	1.3 (0.3)	1.4 (0.4)	26.1 (6.5)	107.0 (26.8)	84.3 (21.1)	65.7	55.6	431.8

the UK. Taking the UK as a whole, descriptive statistics for predicted levels of the different land uses and livestock intensities are reported in **Table 22.17**. These figures suggest declines over time in some farmland uses, most notably amongst cereals and rough grazing, the latter being of some concern regarding associated biodiversity values. Interestingly, the area of a broad category of ‘other farmland’, which encompasses farm woodland, vegetables and other arable crops, is foreseen to increase more or less steadily over time. This may reflect the creation of climatic conditions suitable for the cultivation of new or currently marginal crops. Turning to consider livestock, the marked increase in permanent grassland is accompanied by a rise in numbers of dairy cows but a decline in beef livestock, although the reduction in rough grazing sees a fall in sheep

numbers. Considering the various grassland types together implies a substantial increase in dairy stock intensities, a substantial decline in beef stocking and a more modest decline in sheep stocking densities. While the changes in the dairy sector would appear counter-intuitive considering the recent trends in the livestock sector (with dairy, beef and sheep stocks falling by as much as 25% in the past 10 years) we must recall that these scenarios describe the impact of climate change *ceteris paribus*. In other words, trends in husbandry practices, technology and other economic and social factors are not taken into account. It may well be that if current trends do persist into the future then these may overpower the impacts of climate change.

The relative trends in these UK level predictions are summarised in **Figure 22.9**, which describes the percentage of total UK agricultural land allocated to each land use type under each climate scenario. As can be seen, notable trends include a decline in cereals offset by an increase in permanent grassland.

Although national figures are of obvious importance, they disguise a number of marked regional trends in which a given activity will increase in prevalence in one area while declining in another. The highly disaggregated and spatially explicit nature of our model is ideally suited to such analysis. **Figure 22.10** details the spatial distribution of changes in our main agricultural land uses over time. For simplicity, we map results just for the high emissions scenario, reporting these for changes from our base year of 2004 to 2020, 2040 and 2060. Maps are coloured such that purple tones indicate reductions in the land use shown and green tones indicate increases, with yellow indicating relatively little change. Note that each map has a different range relating to predicted changes in that activity. However, as these ranges differ between activities, a given shade of colour for one activity does not refer to the same amount in a different activity.⁹⁹

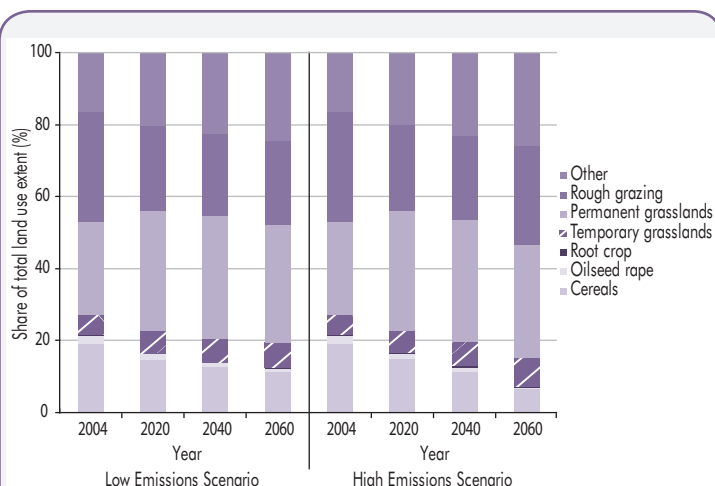


Figure 22.9 Predicted percentage shares of UK agricultural land use under two climate scenarios. Baseline year is 2004. Source: SEER (2011).

99 It is effectively impossible to determine a single colour scale which works for all activities yet still highlights the sensitivity of changes in each individual activity. To see this, contrast the maps for rough grazing, which embrace a range of ± 100 ha, with that for temporary grassland, which ranges from -5 ha to $+20$ ha, figures which barely span two of the categories for the former activity.

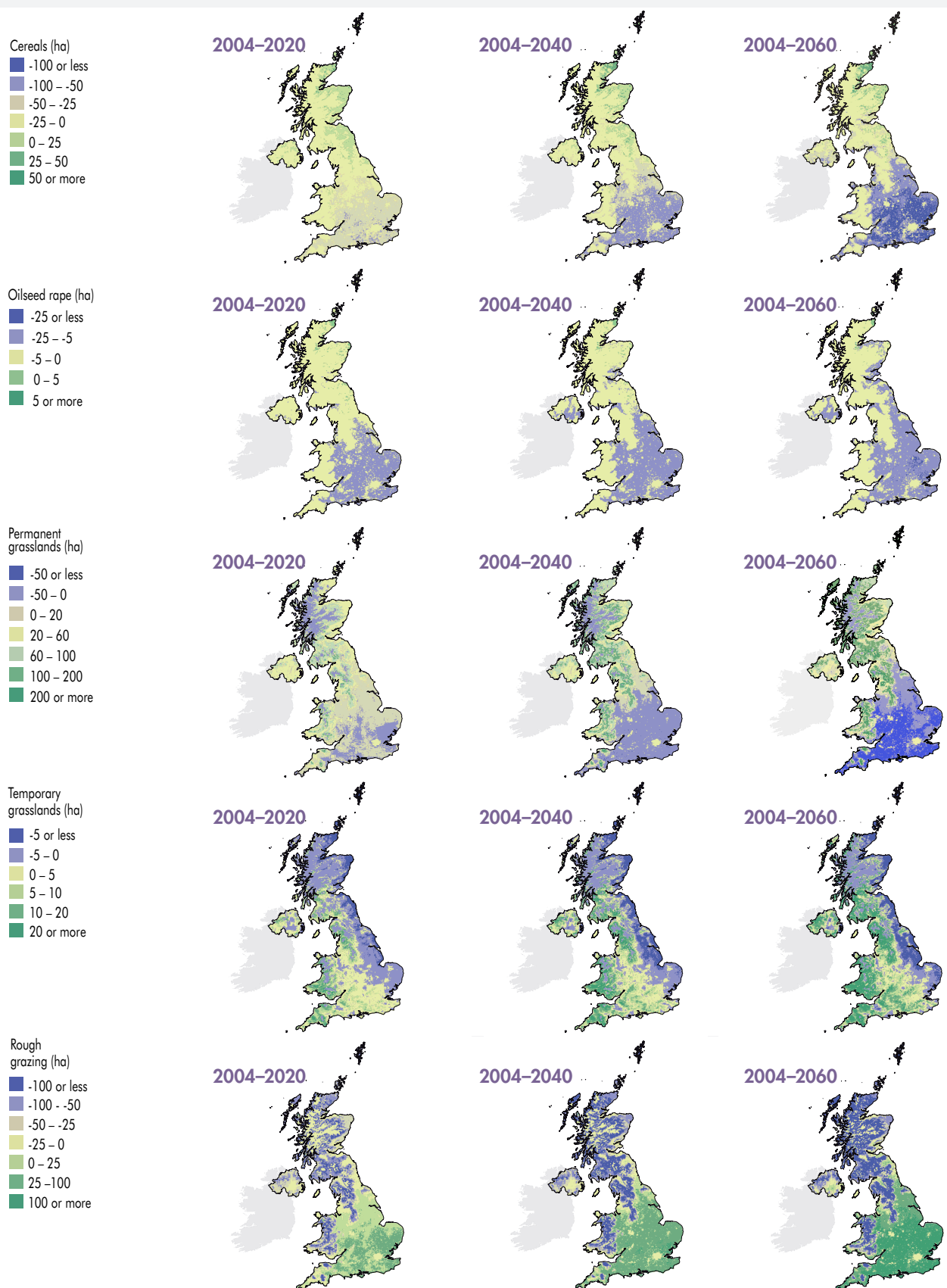


Figure 22.10 Predicted changes in land use from the base year 2004 to three future dates (2020, 2040 and 2060) under the UKCIP high emission climate change scenario (changes shown as the number of hectares (ha) per 2 km grid square). For each of the maps, the colour scheme ranges from dark purple, indicating the largest reductions, to dark green, indicating the largest increases, with yellow indicating relatively little change. Note that each map has a different upper and lower bound indicating the absolute changes but, as these ranges differ for each activity, the same shade of colour means different things across maps. Source: SEER (2011).

When interpreting **Figure 22.10** it is important to note that each land use type is mapped using its own category scale. This is necessary, as a single scale could not capture the quite diverse absolute differences in changes between land use types. However, this does mean that any given colour for one land use does not have the same meaning for another. Nevertheless, within each land use we can readily observe trends in losses and gains across different areas. Considering the first row of maps, we see a marked reduction in cereals in south and east England as climate change brings with it problems of droughtiness in this area. However, this is somewhat offset by an increase in cereals in eastern Scotland as the same processes reduce problems of cold and waterlogging in that area. Another interesting trend is provided by the contrast of changes in temporary and permanent grasslands (third and fourth rows) and rough grazing (final row). Here we see a marked switch from rough grazing to permanent grassland in Wales, north-western England and Scotland (with temporary grassland also increasing in the former two areas). As discussed in more detail subsequently, trends such as the predicted increase in rough grazing in the south-east of England should be treated with caution as they correspond to the area of the country where predicted climates rise most above historical trends and hence out of the range of data.

Figure 22.11 shows the changes in predicted livestock numbers in England and Wales in 2020, 2040 and 2060 compared to the base year (2004). Echoing the rise of grasslands shown previously, the overall number of dairy cows is expected increase substantially, particularly in Northern Ireland, England and Wales and lowland areas of Scotland. Beef cattle and sheep are predicted to generally increase in less extreme upland areas such as Wales and the Borders, but to decline across most of England as lands become more suitable for more profitable undertakings.

When combined, the results for land use and livestock intensity mapped in **Figure 22.10** and **Figure 22.11** predict the profile of farm activities across the period to 2060. This in turn allows us to calculate the implied changes in value induced by these changes in ecosystem services through that period. Ideally, we would use prices adjusted for all subsidies and interventions. However, if we assume that these are relatively marginal shifts, an approximation to that value can readily be obtained by applying the farm gross margin (FGM) value of each output, where FGM is simply the difference between per unit farm revenue and associated variable costs for a given activity.¹⁰⁰ While gross margins are heavily influenced by subsidy levels (see Tinch *et al.* 2010; Bateman *et al.* 2003), examining changes in those margins (i.e. holding subsidies constant) should provide

some indication of underlying shifts in values. **Figure 22.12** illustrates changes in FGM across the UK as evaluated using baseline (2004) prices and (for contrast) the low emissions scenario.¹⁰¹

Figure 22.12 shows some interesting trends in FGM.¹⁰² In particular, there is a clear north-south trend, with strong increases in the north and small decreases in the driest areas of the south, which progressively become more and more significant with the warming and the drop in precipitation. However, assumptions concerning the response of farmers to these circumstances mean that we have some doubts that the forecast loss for the south east of England will arise to the extent predicted, if at all. We now turn to consider these and related caveats.

22.3.17.4 Caveats

Several caveats need to be taken into account when considering the results produced by this analysis. Firstly, the model scenarios are not predictions of the future, but rather represent the impact of climate change *ceteris paribus*, i.e. keeping all other drivers of land use and agricultural production fixed to their baseline levels (year 2004). Therefore, for example, market prices and government involvement (subsidies, levies, milk quotas) are assumed to stay constant. However, changes in both prices and agricultural policies can be expected to take place in the future. For example, global warming could cause major shifts in the supply of all the main agricultural products, while the growth of developing economies such as China and India could have significant implications for demand. Also, UK policies are likely to change, in accordance with the ongoing reforms of the CAP.

Considering our measure of financial impacts, FGM, two important limitations need to be acknowledged. Firstly, since FGM is defined as the difference between revenues and variable costs, all farm fixed costs (e.g. machinery, buildings, rent) are not included in the analysis. Secondly, conversion costs are also not included. In other words, all changes in land use and FGM refer to equilibrium conditions, but do not take into account possible costs encountered in order to reach these new equilibriums.

It is important to note that the UKCP09 scenarios, particularly those relating to periods furthest into the future, include climatic conditions for some areas of the country (notably the South East and south coast of England) which are considerably above those experienced for any length of time over the past 40 years. Therefore these conditions lie outside the range of data used to estimate the model. For this reason, the results have to be interpreted cautiously. In particular, since the model uses farmers' past behaviour to predict their future response, it cannot include the impact of

¹⁰⁰ While FGM is a very widely applied measure within the field of agricultural economics, it lacks the simple link to welfare of a measure such as profit. However, as noted earlier, farm profit data are not available on the disaggregated regular grid of the agricultural census data used for this analysis. The CSERGE SEER project is currently examining possibilities for supplementing this analysis with data from the Farm Business Survey which would address this problem. A further issue concerns the extent to which these shifts are marginal. Again, this is a topic of ongoing research.

¹⁰¹ FGM forecasts for 2004 are taken from Fezzi *et al.* (2010a) as follows: cereals = £290/ha, root crops = £2,425/ha, oilseed rape = £310/ha, dairy = £576/head, beef = £69/head, sheep = £9.3/head. Appendix 1 of Fezzi *et al.* (2011) provides an analysis of the variation in these estimates induced by changes in agricultural prices for different outputs. This shows that such variation can alter absolute FGM values considerably, although the overall spatial pattern in changes remains the same as that illustrated in Figure 22.11.

¹⁰² Note that the trends here are significantly different to those shown in various of the valuations of scenarios reported in Chapter 26. In the analysis reported in the present chapter the only driver of change is shifts in the climate, yielding the patterns illustrated in Figure 22.12. However, in Chapter 26 multiple drivers of change are acting simultaneously producing, in many cases, quite different patterns of response.

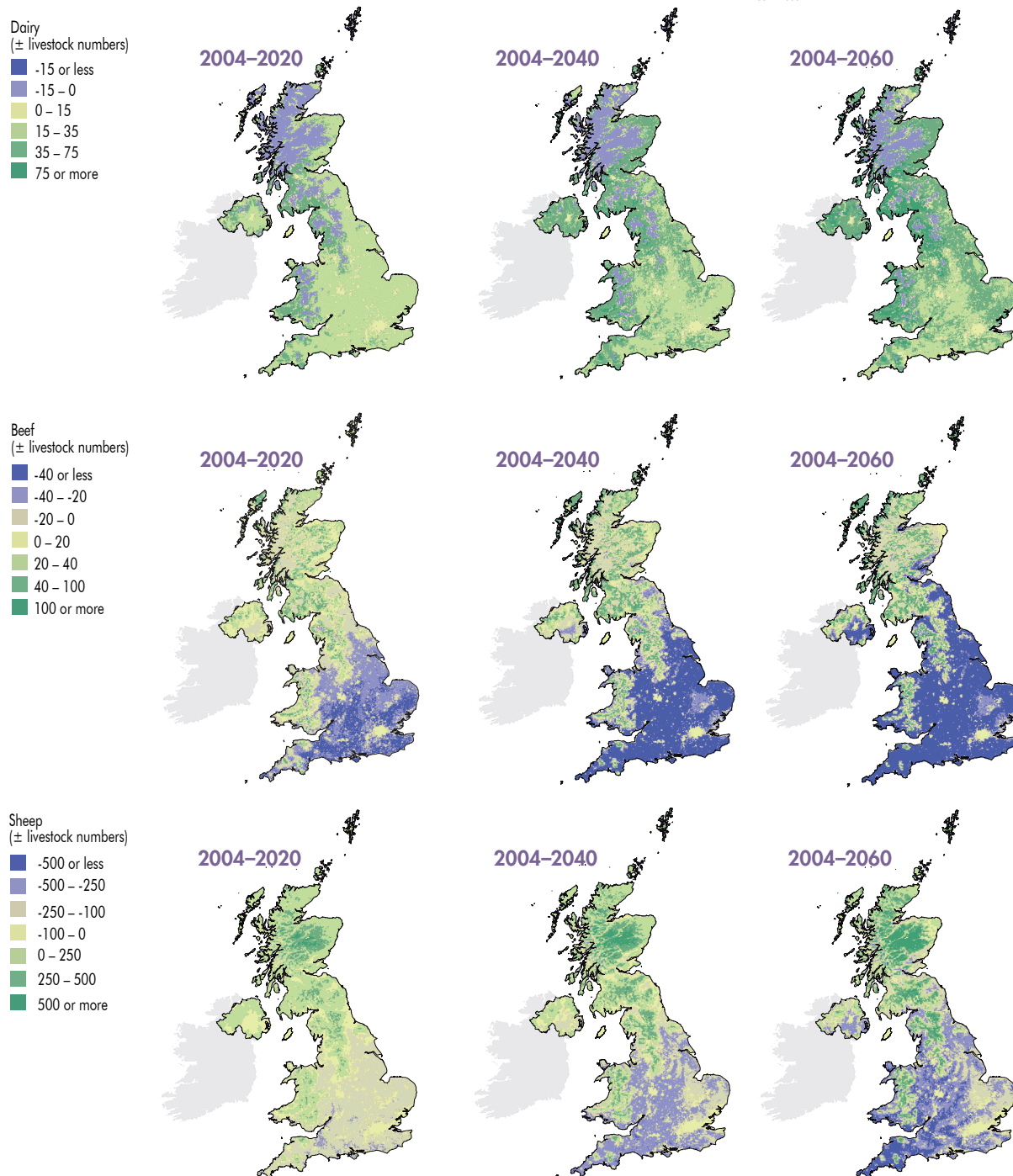


Figure 22.11 Predicted changes in livestock numbers from the base year 2004 to three future dates (2020, 2040 and 2060) under the UKCIP high emission climate change scenario (changes shown as the number of head of livestock per 2 km grid square). For each of the maps, the colour scheme ranges from dark purple indicating the largest reductions, to dark green indicating the largest increases, with yellow indicating relatively little change. Note that each map has a different upper and lower bound indicating the absolute changes but, as these ranges differ for each activity, the same shade of colour means different things across maps. Source: SEER (2011).

introducing new crop types which have not been significantly present in UK farmland in the past (e.g. outdoor tomatoes, vineyards). This relates to a further caveat concerning technological innovation. Although our model includes a time trend which provides some indication of technical progress, this is not assumed to change. Taking these factors together, the predictions for the warmest areas are subject to the highest degree of uncertainty and the results for the

most extreme scenarios (e.g. 2060 high emissions) for these areas should be interpreted cautiously. Conversely, however, the results for the north of England, Wales, Scotland and Northern Ireland should be more robust.

Our analysis focuses on the impact of changes in temperature and precipitation, and not on other things that might be affected by climate change. For example, Mendelsohn & Dinar (2009) and others suggest that

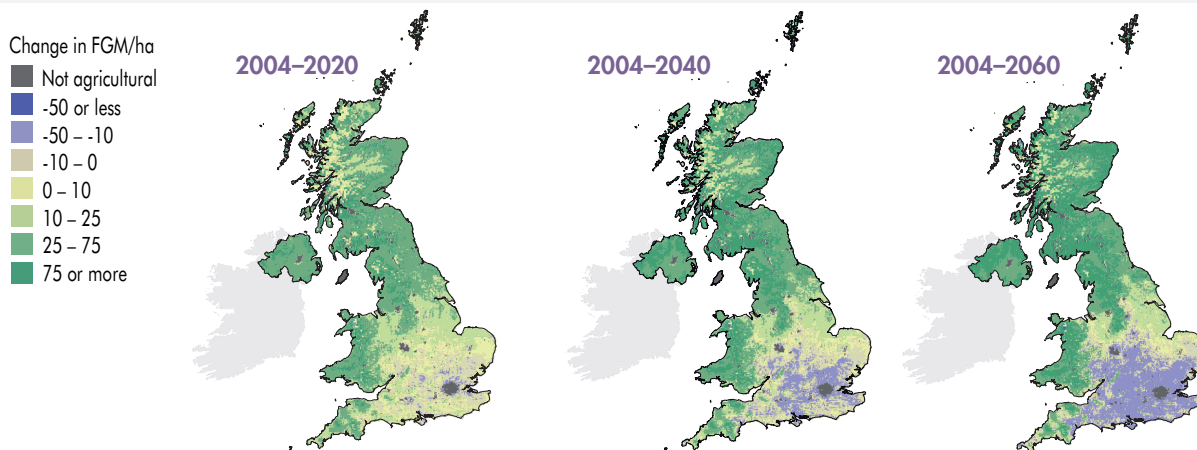


Figure 22.12 Valuation of climate ecosystem services: the impact of climate change (UKCIP low emission scenario) upon UK farming calculated as the induced change in annual farm gross margin (FGM) per hectare (ha) compared to its level in 2004. Source: SEER (2011).

increased carbon dioxide fertilisation may increase crop yields. However, there may be a trade-off between quantity and quality, as the projected increase in crop growth is offset by a decline in nutritional value (Jablonski *et al.* 2002). Another factor which is likely to change in the future is pollination. Current research (e.g. Potts *et al.* 2010) indicates a significant decline in pollination ecosystem services in recent years. Among the most important drivers are land use change, with the consequent loss and fragmentation of habitats increasing, pesticide application, environmental pollution and climate change. This could have a significant impact on yields. Furthermore, there is a growing body of evidence to suggest that climate change may modify (and generally exacerbate) crop disease patterns in ways that are, to date, still poorly understood (Harvell *et al.* 2002).

22.3.17.5 Agricultural food production: conclusions

The analysis develops a novel, spatially explicit model for estimating changes in agricultural land use as a result of changes in any combination of policy, price or environmental drivers. A detailed spatially and temporally variable dataset is compiled and applied to this model to yield estimates of farm land use under analyst-controlled scenarios. The UKCIP09 climate change predictions are applied to this model, and land use change impacts are estimated. These are in turn employed to calculate farm gross margin estimates of the value of changes in ecosystem provisioning services.

Our analysis remains incomplete, yet findings to date suggest that changes in ecosystem inputs induced by climate change will have a substantial influence upon the gross margins generated by farm food production. Interestingly, climate change seems likely to generate both positive and negative impacts across different part of the UK. These patterns include a new north–south divide, reversing the characteristic direction of that inequality, with the winners in this case being in northern areas and losers being in areas of the south of England.

22.3.18 Carbon Storage and Annual Greenhouse Gas Emissions: Terrestrial¹⁰³

22.3.18.1 Introduction

Regulation of the carbon cycle and emissions of GHGs has become an increasingly important element of contemporary land use decision making, both in the UK and globally. The inclusion of land use choices and land management activities as an integral part of assessments of climate regulation services is important for several reasons. Climate is a key determinant of land use, and climate change would be expected to result in regional shifts in land use. Different land uses are, in turn, associated with varying regulation capacity, and land use change might therefore itself lead to increases or decreases in GHG emissions. Finally, land management can be tailored to differing land use so as to manipulate the potential for GHG mitigation. Given that agriculture accounts for 10–12% of the total global anthropogenic emissions of GHG (Smith *et al.* 2007), the potential for such mitigation is clearly substantial.

An interesting example of how land use change can affect carbon storage is illustrated by the case of forestry. Estimates show total net carbon sequestration by UK woodlands planted after 1921 rising from 2.4 million tonnes of carbon dioxide (Mt CO₂) in 1945 to a peak of 16.3 Mt CO₂ in 2004, subsequently falling to 12.9 Mt CO₂ in 2009 (Valatin & Starling 2010). Over the period 2001–2009 these estimates imply annual mean net sequestration rates of around 5.2 tCO₂/ha across all UK woodlands (with an additional 0.3 tCO₂/ha net increase in carbon storage in harvested wood products). If assumed permanent (e.g. because of future woodland expansion) and valued at the DECC (2009) central social value of carbon estimate of £53/tCO₂ in 2009, the estimates suggest that the total value of net carbon sequestered annually by UK woodlands increased five-fold from £124 million in 1945 to £680 million in 2009 (at 2010

¹⁰³ This Section draws in part from Abson *et al.* (2010) and Valatin & Starling (2010).

prices). It would also imply a mean value per hectare of the carbon sequestered annually by UK woodlands (£239/ha) of more than triple the mean value for softwood production (£66/ha) and of the order of 10 times the value of hardwood production (£7–£25/ha) in 2009. Forecasts of net carbon sequestration based upon the continuation of current rates of woodland creation indicate a drop of more than half in net carbon sequestration by woodlands from 2010 to 2028. When combined with changes in carbon storage in harvested wood products, the forecasts show combined net sequestration falling from 14.5 Mt CO₂ in 2010 to a minimum of 2.5 Mt CO₂ in 2034, before gradually rising to 3.3 Mt CO₂ by 2050 (Valatin & Starling 2010). However, recent analysis for the Read Report (Matthews & Broadmeadow 2009) suggests an additional 12 Mt CO₂ to 15 Mt CO₂/yr could be sequestered in 2060, were a programme of enhanced afforestation of an additional 23,200 ha a year adopted. Although apparently much more valuable than the timber produced by UK woodlands, carbon sequestration nevertheless remains a largely non-market value, with little incentive at present for private landowners to increase provision of this ecosystem service (or to maintain existing carbon storage).¹⁰⁴ This may in part be addressed through the Woodland Carbon Code which is currently being developed by the Forestry Commission to help stimulate emerging markets for carbon sequestration in the UK.

In addition to forest, peatland is an interesting example of a land use which provides climate regulating services. Of course, while carbon stocks held by peatland may be significant, where land use remains stable the flow values may be negligible. So, for example, around 40% of UK soil carbon is found within Mountain, Moorland and Heath habitats (Tinch *et al.* 2010), much of which are peatlands, while Natural England (NE 2010a) estimate that some 6,700 ha of peatland stores around 584 Mt C, equivalent to about 2.14 billion tonnes of carbon dioxide equivalents (CO₂e). However, most of this is stored in stable conditions. That said, estimated emissions from peatlands are currently about 2.48 Mt CO₂e/yr (Tinch *et al.* 2010). This is equivalent to about £130 million/yr at DECC's 2010 price for carbon dioxide.

So where land use changes, so does the level, and hence the value, of its climate regulation service. The analysis presented here considers how these climate regulation services will alter as climate change induces shifts in UK agricultural land use. The analysis outlines the assumptions made to estimate this value, based on the predicted climate change associated with the UKCIP low and high GHG emission scenarios (UKCIP 2009) for the years 2004, 2020, 2040 and 2060. These are the same predictions used in the agricultural analysis presented earlier in this chapter, and the changes in land uses are drawn from the outputs of the CSERGE agricultural land use model (Fezzi & Bateman in press). This has an important benefit in that it allows us to simultaneously assess both the agricultural and carbon

storage values associated with the UKCIP climate change predictions.

The analysis includes both estimates of changes in potential equilibrium carbon stocks (i.e. that level of carbon that can feasibly be stored) and changes in the annual flow of GHGs associated with the shifts in modelled land use patterns in the Enclosed Farmland Broad Habitat. The stock estimates for the modelled Enclosed Farmland land use patterns are based on i) the carbon stored in, above and below ground vegetation and ii) the potential equilibrium soil organic carbon (SOC) levels of the soils under those land use patterns. The flow estimates are based on the annual GHG emissions from farm activities (including energy usage, emissions from fertilisers and livestock) and the annual SOC emissions or accumulations resulting from changes in land use. All impacts are converted to carbon dioxide equivalents.

Land uses in the Enclosed Farmland Broad Habitat include cereals, oilseed rape, root crops (sugar beet and potatoes), temporary grassland, permanent grassland, rough grazing, on-farm woodland¹⁰⁵ and other agricultural land uses (including horticulture, and bare/fallow land). This document first presents an analysis relating to changes in the capacity to store carbon, then an analysis of the changes in annual emissions resulting from changes in land use and associated land management. Finally, we provide an economic valuation of the changes in climate regulation given specified climate change scenarios. Uncertainty in the valuation estimates is assessed by comparing results using the two main approaches to carbon pricing: social cost of carbon and the marginal abatement costs of carbon.

22.3.18.2 Changes in the UK terrestrial capacity to store carbon

An analysis was conducted of the change in carbon stocks, including changes in SOC and vegetative carbon stocks (full details in Abson *et al.* 2010). It is important to note that the analysis provides information about potential long-term equilibrium estimates, while in reality carbon stocks are dynamic as they are subject to changes in growth and decomposition rates driven by climate and land management. The results of the analysis give a total UK estimate of vegetative carbon stocks for the baseline year (2004) of 134 Mt C, of which 77% is stored in woodland. This compares with Milne *et al.*'s (2001) estimate of 113.8 ±25.6 Mt C for GB of which 80% was estimated to be stored in woodland. Vegetative carbon stocks are relatively evenly spread across the UK, with the highest stocks in wooded areas such as Thetford forest and southern Scotland.

While the vegetative stock of carbon is substantial, it is dwarfed by that in soils (Bradley *et al.* 2005). The analysis suggests that 50% of the carbon stocks in the UK's terrestrial ecosystems are found in Scotland (2,365 Mt C), with a further 37% (1,755 Mt C) in England, 7% (338 Mt C) in Wales and 6% (292 Mt C) in Northern Ireland. The highest stocks are found

¹⁰⁴ This depends upon the permanence assumption and whether the carbon substitution benefits of using the wood harvested are also included in the comparison.

¹⁰⁵ On-farm woodland is subsumed within an 'other land' category in the Fezzi & Bateman (in press) model. However, because of the importance of woodland in regulating climate, this land use is separated out in the present analysis, but it is assumed that its extent remains unchanged within the UK NEA Scenario timelines (see Chapter 25).

in the upland peat areas of northern England, Northern Ireland and Scotland.

Next we model how the land use change predicted under the UKCIP low and high GHG emission scenarios will affect the equilibrium carbon stock for the UK. Here it should be noted that SOC may take many years to reach new equilibrium levels after land use change (particularly in organic soils), therefore the potential equilibrium stock estimates do not represent the actual stocks in the analysis year, but rather they indicate the potential equilibrium stocks associated with the modelled land use configuration for that analysis year. Under these scenarios, only the Fens in the East of England and small areas of the north-east Scottish Highlands show a consistent increase in carbon stocks, this being due to a reduction in intensive cereal production on organic soils. There are significant reductions in potential equilibrium carbon stocks in the lowland agricultural regions of southern England in both the low and high emissions scenarios, these losses being most pronounced in the high emissions scenario towards the end of our analysis period (2060). Conversely, the largest reductions in carbon stocks occur in the SOC stored in peatland and upland areas of the UK. Overall patterns are broadly similar across the high and low emissions scenarios, although potential equilibrium stocks decline more rapidly in southern regions and under the high emissions scenario. Land use change in Scotland is predicted to most dramatically reduce the potential equilibrium carbon stocks, with a decrease in stocks (relative to the present day) of approximately 37% (113 tonnes C/ha) for the 2060 land use configuration in both scenarios. This change in stock is due to increases in arable and improved grassland activities on peat and other soils with a high SOC. The total reduction in potential UK equilibrium carbon storage from the baseline year to 2060 is 1,381 Mt C for the low emissions scenario and 1,560 Mt C for the high emissions scenario; this would equate to total carbon dioxide emissions of approximately 5,064 Mt CO₂e and 5,719 Mt CO₂e respectively. For comparison, the total UK emissions of GHGs in 2008 has been estimated as 628.5 Mt CO₂e (DECC 2008).

22.3.18.3 Changes in UK land-based greenhouse gas emission flows

Four major sources of GHG emissions were considered when estimating changes in annual GHG emission flows:

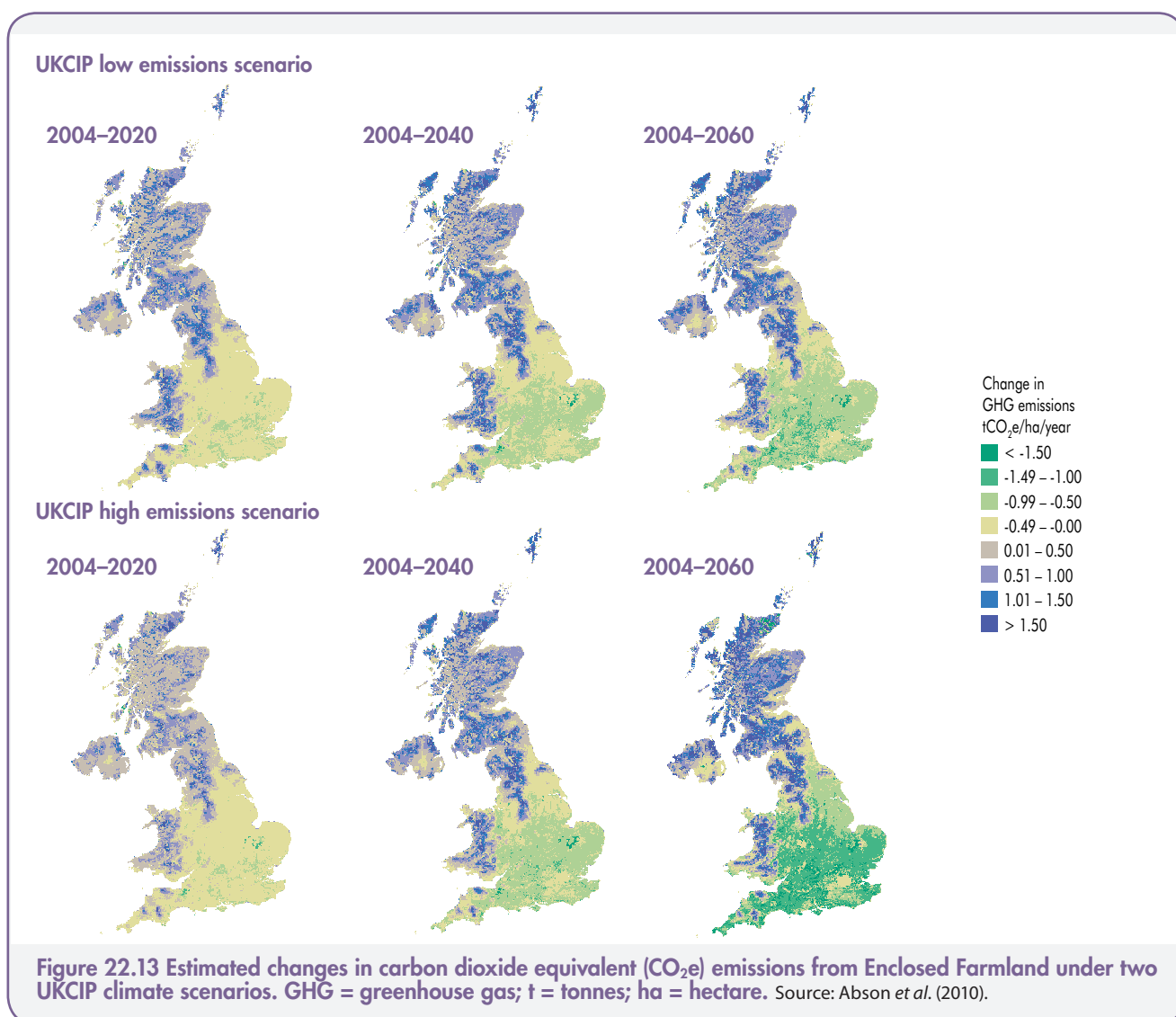
- i) The indirect emissions due to energy use from farmland activities such as tillage, sowing, spraying, harvesting and the production, storage and transport of fertilisers and pesticides. Per hectare estimates of GHG emissions for typical farming practices were applied to each type of land use in order to map these emissions across the UK.
- ii) Emissions of nitrous oxide and methane from livestock, including beef cattle, dairy cows and sheep through the production of manure and enteric fermentation.
- iii) Direct emissions of nitrous oxide emissions from artificial fertilisers.
- iv) Annual flows of carbon from soils due to land use changes. For example, permanent grassland converted

from arable farming will be accumulating SOC, while permanent grassland on land that was previously under rough grazing may be losing SOC. For the baseline year (2004) annual flows of SOC were only estimated for organic (peat) soils as there is insufficient data on land use change prior to the baseline to accurately model changes in SOC in non-organic soils. In the analyses of subsequent years (2020, 2040 and 2060), SOC flows due to land use change in both organic and non-organic soils are included in the annual GHG emission estimates. In both UKCIP low/high emissions scenarios there are considerable changes in annual emissions.

We estimate that the annual GHG emissions from Enclosed Farmland for the baseline year (2004) to be 48 Mt CO₂e (approximately 9% of UK net GHG emissions for that year) with emissions from enteric fermentation and the direct release of nitrous oxide from both artificial fertilisers and the application of farmyard manure representing the biggest sources of GHG emissions from Enclosed Farmland in the UK.¹⁰⁶ **Figure 22.13** maps the distribution of changes in farmland emissions across the UK for three time periods under two climate change scenarios. In general, results suggest that emissions will fall in the lowland areas of England and increase in more upland areas. These trends echo the shifts in land use expected for those areas, with the latter areas seeing increases in livestock numbers and in arable and horticultural production, leading to increased emissions of methane and nitrous oxide. These trends are exacerbated by the potentially large rise in GHG emissions from the conversion of peatland from rough grazing and semi natural grassland into improved grassland.

Considering **Figure 22.13**, while spatial patterns are more pronounced under elevated climate change, overall predicted emissions from agriculture are similar for both scenarios, with aggregate emissions differing by only 4% on average between the two scenarios. In the UK, GHG emissions from Enclosed Farmland are estimated to increase from 2.14 tonnes CO₂e/ha/yr in 2004 to 2.33 tonnes CO₂e/ha/yr in 2060 under the UKCIP low emissions scenario, and to rise to 2.21 tonnes CO₂e/ha/yr in 2060 under the high emissions scenario. There are effectively two opposing land use dynamics identified by the model, increasing carbon intensities (per hectare) of agriculture in the north and decreasing carbon intensities in the south. Aggregate UK GHG emissions under the high emissions scenario are slightly lower than for the low emissions scenario, due to greater carbon reductions in the south by 2060 under the high emissions scenario. These land use changes equate to an aggregate increase in UK GHG emission from agriculture of approximately 11% between 2004 and 2020 under both emissions scenarios. Trends then peak with annual changes in GHG emissions remaining relatively stable between 2020 and 2040 for both scenarios (with an approximate increase in emissions of 10% from the baseline year). Towards the end of the analysis period trends begin to improve somewhat, with 2060 annual GHG emissions being 8.9% higher than the baseline for the low emissions scenario

¹⁰⁶ Official estimates for the GHG emissions from UK agriculture for 2004 range from 44.5 Mt CO₂e (Defra 2007) to 51.7 Mt CO₂e (DECC 2008), with the differences in the two estimates in part due to different definitions of what represents a GHG emission from agriculture.



and 3.2% higher than the baseline year for the high emissions scenario. The reduction in aggregate GHG emissions between 2040 and 2060 is largely driven by a climate-induced switch to less intensive land uses, resulting in reductions in fertiliser usage and consequent nitrous oxide emissions, with concurrent increases in SOC due to a reduction in tillage on non-organic soils. All in all, while most southern regions see significant drops in GHG emissions, northern regions see increasing emissions due to increased livestock numbers and a shift to more intensive land uses (primarily improved grassland and arable production) as the climate makes these new land uses economically viable. Additionally, it is worth noting that while in the baseline year net GHG emissions from UK peat soils are estimated at 3.76 Mt CO₂e/yr, these increase to 7.67 Mt CO₂e/yr by 2060 (high emissions scenario), with Scotland accounting for almost half of these emissions, due mainly to land use changes from rough grazing to permanent grasslands.

Contrasting the spatial pattern of changes in carbon flux with the distribution of shifts in agricultural values presented in Section 22.3.17.3 (see **Figure 22.11** and **Figure 22.12**), we can see the duality of effects which climate change is expected to bring in these respects. The increases in temperature and shifts in rainfall patterns

brought about by climate change will result in a shift towards relatively more intensive agriculture in upland Britain (**Figure 22.11**). While this will generate increases in farm income in parts of upland Britain (**Figure 22.12**), the present analysis shows that this will also be synonymous with increased emissions in such areas (**Figure 22.13**).

22.3.18.4 The value of agricultural climate regulation

Providing estimates on the value of non-market GHG emissions is problematic (particularly when the estimates are for future emissions) for two main reasons. First, climate science is complex and we do not yet have a definitive relation between emissions and climate change. Moreover, there is considerable uncertainty regarding the relationship between climate change and its impacts on the economy, dependent as those impacts are on socio-technological responses to changes in the climate. Second, when forecasting carbon values, the societal cost associated with the emission of an additional tonne of carbon is dependent on how many tonnes of carbon have previously been emitted (and abated), the eventual concentrations at which carbon dioxide is stabilised in the atmosphere, and the emissions trajectory adopted to achieve this stabilisation (DEFRA 2007). As such, future carbon prices

are depend upon the emission and climate scenarios upon which they are based. The issues of carbon pricing are further complicated by the choice of methodology used to construct these prices. There are two main approaches to carbon pricing: the social cost of carbon (SCC); and the marginal abatement cost of carbon (MACC). We apply two separate price functions to investigate the sensitivity of results to the choice of carbon value. The UK government's official non-market MACC prices from DECC (2009) are applied to both climate scenarios. However, for comparison we also apply an endogenous SCC price derived from Stern (2007). Stern's (2007) business as usual price is applied to the UKCIP high emissions scenario and the atmospheric concentration of 550 parts per million CO₂e price is applied to the UKCIP low emissions scenario. For the DECC prices, the carbon prices for each point in the scenarios are based on a linear interpolation of the prices provided by DECC (2009).¹⁰⁷ **Table 22.18** details the prices arising from these various strategies. All prices are in 2009 values.

Table 22.18 Greenhouse gas pricing for non-market greenhouse gas in the UKCIP scenarios. Source: Abson *et al.* (2010).

Year	DECC* (£/tCO ₂ e†)	STERN 550 ppm‡ (£/tCO ₂ e†)	STERN BAU§ (£/tCO ₂ e†)
2004	44.00	25.47	88.38
2020	60.00	34.96	121.32
2040	135.00	51.95	180.28
2060	265.00	77.20	267.89

* Department of Energy and Climate Change.

† Tonnes of carbon dioxide equivalents.

‡ Parts per million.

§ Business As Usual.

Table 22.19 Estimated total annual costs of UK agricultural greenhouse gas emissions. Source Abson *et al.* (2010).

Carbon price function	2004 (£ million)	2020 (£ million)	2040 (£ million)	2060 (£ million)
DECC* low emissions scenario	2,134	3,261	7,334	14,000
Stern low emissions scenario	1,235	1,900	2,822	4,078
DECC* high emissions scenario	2,134	3,141	7,121	13,265
Stern high emissions scenario	4,286	6,352	9,509	13,409

* Department of Energy and Climate Change.

The prices provided in **Table 22.18** are used in **Table 22.19** to estimate the total annual cost of GHG emissions from UK agriculture for the predicted land uses under the two UKCIP climate scenarios. Annual costs of carbon emissions from agriculture are predicted to increase from £2.1 billion p.a. in 2004 to £14.0 billion in 2060 under the UKCIP low emissions scenario, based on the DECC price function and to £4 billion under Stern's price function. While some of this steep increase in costs is due to the predicted 8.8% increase in GHG emissions from agriculture, it is largely driven by the increase in the predicted price of carbon.

By calculating the difference between the estimated cost of emissions for the baseline year (2004) and those for the modelled land uses in 2020, 2040 and 2060 we identify the impact of predicted future land use change on the value of carbon regulating service provided by UK agriculture. **Figure 22.14** presents a regional analysis of the relative change in annual carbon costs (per hectare) of climate-driven land use change in the UK. This is achieved by comparing the carbon costs associated with the baseline and predicted land uses for a given year (2020, 2040, 2060) at that year's DECC carbon price. While agriculture remains a net emitter of GHGs for all regions of the UK, land use changes are predicted to results in relative decreases in costs per hectare of emissions in southern regions of the UK (compared to the emissions associated with the baseline land uses) and relative increases in costs in northern regions. For example, in 2060 the average cost of GHG emissions from agriculture in the East of England are predicted to be approximately £300/ha lower than if the baseline land use patterns had been retained, while in Scotland the cost of carbon for agriculture in Scotland is predicted to be £250/ha higher due to changing land uses.

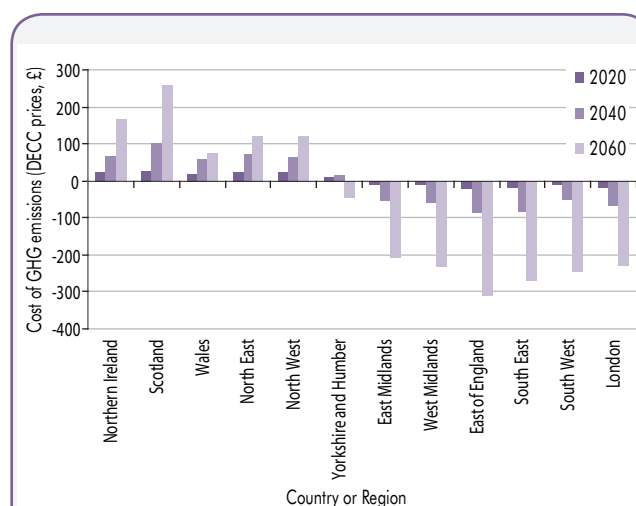


Figure 22.14 Predicted impact of land use change on the cost of greenhouse gas (GHG) emissions (Department of Energy and Climate Change (DECC) prices) from agriculture in the UK under the UKCIP high emissions scenario, compared to estimated costs in the baseline year (2004). Source: Abson *et al.* (2010).

¹⁰⁷ Stern's (2007) prices were converted from US dollars using the long-term exchange rate (\$/£) of 1.61 and assumed to increase by 2%/yr in real terms. All prices are in 2009 values, calculated using the treasury gross domestic product (GDP) deflator (HM Treasury 2010). Where £/tonne of carbon (£/tC) were reported, a standard conversion ratio of 44/12 was used to convert to CO₂e.

Table 22.20 Regional analysis of cost from agricultural greenhouse gas emissions per hectare (ha) (based on Department of Energy and Climate Change (DECC) prices). Source: Abson *et al.* (2010).

	Baseline 2004 (£/ha/yr)	UKCIP low emissions scenario			UKCIP high emissions scenario		
		2020 (£/ha/yr)	2040 (£/ha/yr)	2060 (£/ha/yr)	2020 (£/ha/yr)	2040 (£/ha/yr)	2060 (£/ha/yr)
Northern Ireland	140	217	501	980	213	497	1,007
Scotland	86	154	363	735	144	361	774
Wales	89	155	355	660	142	335	615
North East	102	167	385	758	163	384	737
North West	129	204	470	907	197	459	895
Yorkshire Humber	98	146	325	614	144	317	547
East Midlands	85	107	219	385	107	206	305
West Midlands	91	116	238	414	116	224	319
East of England	90	101	203	356	101	191	233
South East	74	80	158	261	83	144	175
South West	108	143	302	523	139	279	404
London	54	54	111	179	59	101	100
UK total	94	144	324	618	139	314	585

Table 22.20 presents a regional analysis of the total cost of annual per hectare emissions of GHG from Enclosed Farmland based on the DECC (2009)-MACC price function for the two UKCIP emissions scenarios. Whereas **Figure 22.14** identifies the relative carbon costs of changing land uses (the change from the baseline in carbon emissions multiplied by the carbon price for a given year), **Table 22.20** presents absolute costs (i.e. those based on the total emissions in a given year multiplied by the price in that year). Therefore **Table 22.20** differs from **Figure 22.14** in that it considers the value of a particular set of emissions at a particular point in time. For example, under the high emissions scenario Scotland is predicted to see a nine-fold increase in the cost of agricultural GHG emissions, rising from £86/ha/yr in 2004 to £774/ha/yr in 2060, yet Scottish agricultural GHG emissions are predicted to increase by around 50%. The majority of the nine-fold increase in absolute carbon costs is driven by a six-fold increase in predicted GHG prices between 2004 and 2060. Using the DECC price function under the high emissions UKCIP scenario, the highest cost from carbon in Enclosed Farmland will be in Northern Ireland (£1,007/ha/yr) and the lowest (excluding London) will be in the south-east of England (£175/ha/yr). On average, the cost of carbon emissions from Enclosed Farmland in the UK is predicted to increase by £491/ha/yr from 2004 to 2060.

22.3.19 The Non-use Value of Biodiversity: Towards Cost-effective Provision of Sustainable Populations

We have highlighted a variety of caveats regarding both the use of stated preferences and legacies as estimates of biodiversity non-use values. Furthermore, we recognise

that certain non-use motivations such as ethical or spiritual concerns may not transfer well into a monetary valuation paradigm (see discussions in Chapter 16). These uncertainties add to the challenges facing natural science models of biodiversity relationships within and across species and habitats under a context of general climate change and anthropocentric pressures. Given this, a risk averse strategy might be to allow policy in this area to be guided by precautionary standards for biodiversity conservation, with economic assessment being focused upon the cost-effective provision of those standards (Bateman *et al.* 2009b). The UK NEA analysis seeks to provide an initial indication of such a strategy. The SEER project undertook two complementary studies of bird diversity (taken as an indicator of biodiversity as per HMG 2007),¹⁰⁸ both of which are presented below. In Chapter 26, these biodiversity assessment models are applied to a number of different scenarios for the future of Great Britain. A range of economic values are also assessed for each of these scenarios. By contrasting these values with the biodiversity implications of each scenario, the decision maker can observe the costs of attaining different levels of biodiversity. This cost-effectiveness approach provides a useful guide for decision making in situations where the full monetary benefits of a value stream (here biodiversity) cannot be reliably established.

Preparing for cost-effectiveness analysis 1: Modelling breeding bird diversity as a function of land cover.¹⁰⁹ Birds have the highest public profile amongst the UK's biodiversity and are high in the food chain, so are widely considered to be good indicators of wider ecosystem health. They are also better monitored than any other group in the UK. It has been demonstrated that birds can be sensitive to land use change;

¹⁰⁸ Note that the Public Service Agreement referred to here (HMG 2007) is currently suspended pending replacement under the forthcoming White Paper on the Environment.

¹⁰⁹ This Section draws from the work of Hulme & Siriwardena (2010).

indeed, changes in farming practices have contributed to a 53% decrease in the England farmland bird index between 1966 and 2009 (Defra 2010c). Most of this decline occurred during the 1970s and 1980s; since then numbers have continued to fall, but at a more modest pace. Birds are, therefore, the best available means by which to assess the biodiversity implications of land use change, including that envisioned through scenarios developed under the UK NEA.

UK land use information derived from the CEH Land Cover Map 2000 (Fuller *et al.* 2002) was matched with bird data from the British Trust for Ornithology (BTO) Breeding Bird Survey (BBS) which assesses widespread, terrestrial bird species at the 1 km Ordnance Survey grid square level (further details being given in Risely *et al.* 2010). For this analysis, BBS annual data from 1995 to 2006 provided a large database centred upon the date of the land use data, although note there is no census data available for 2001 due to access restrictions arising from the foot-and-mouth outbreak. Species recorded on an average of fewer than 40 squares/yr were omitted,

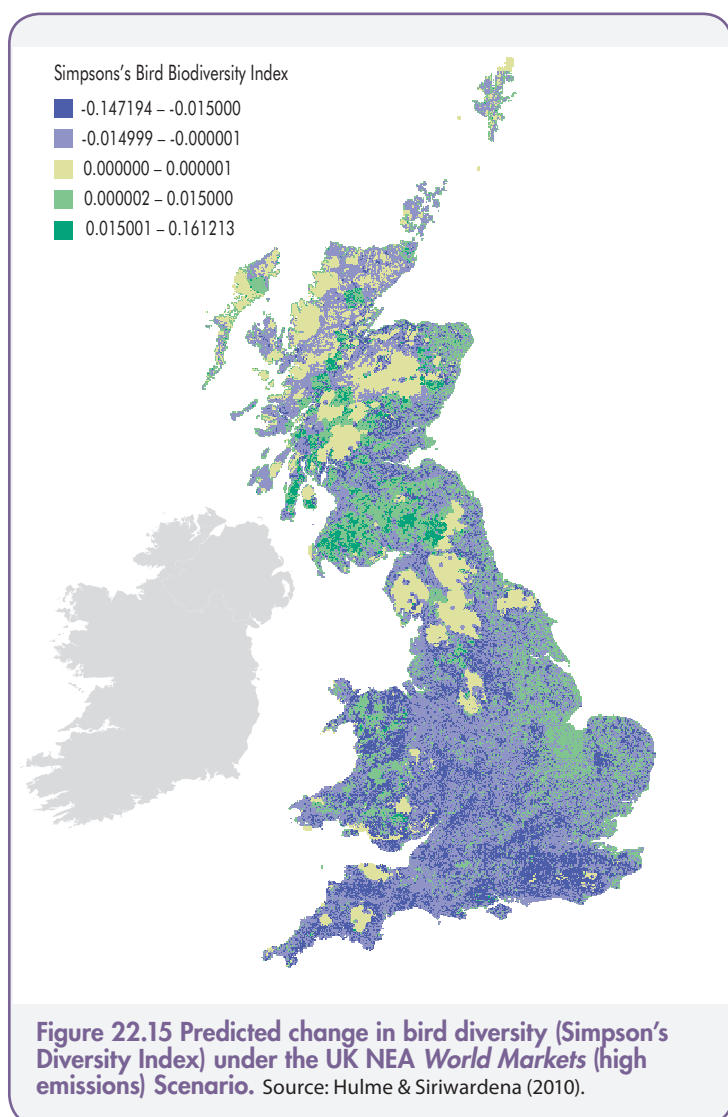
leaving 96 bird species recorded across a sample of 3,468 1 km grid squares across Britain.¹¹⁰

The composition of the bird community represented by the presence and abundance of the bird species in each survey square was summarised using Simpson's Diversity Index (see Hulme & Siriwardena 2010), calculated for each square across all years within the study period in which that square was surveyed. This index provides a simple summary of diversity which has high values where many species are present and are equally abundant. The variation in diversity was analysed with respect to land use (from the CEH Land Cover Map 2000) using standard techniques (generalised linear models and model averaging) to produce statistically sound results. The land cover classes used were chosen to match those included within the UK NEA Scenarios (Chapter 25): Coastal Margins, Freshwaters – Openwaters, Wetlands and Floodplains, arable and horticultural land, improved grassland, Semi-natural Grasslands, broadleaved woodland, coniferous woodland, upland habitats and Urban habitats.

Diversity across the whole of the UK was predicted at the 1 km grid square level from the land use predictions for each of the UK NEA Scenarios. While results indicated that there is significant unexplained variation in bird diversity and that, at a UK scale, it is regional geographic drivers such as altitude which provide the strongest determinants of bird viability, nevertheless all land cover variables except for coastal habitats and inland water cover were shown to have significant effects on diversity. Given that, obviously, influences such as altitude are constant across time, it is this strong relationship between diversity and land use which is of greatest importance to policy makers.

For illustrative purposes, the changes in diversity predicted under the UK NEA *World Markets* (high emissions) Scenario for the whole of Britain is presented in **Figure 22.15** (details of all scenarios are given in Chapter 26 and in Hulme & Siriwardena 2010). As can be seen, the *World Markets High* scenario is predicted to have significant positive and negative impacts in the absolute diversity index, which, depending on the area of the country under consideration, decreases by as much as 0.131 (e.g. south-east England) and increases by up to 0.040 (e.g. Scottish Borders area).¹¹¹ As a general guide, changes of this magnitude represent the loss or gain of around one locally scarce species from a low diversity (e.g. upland) square or a change in abundance of approximately 5–20% of a common species in a higher diversity square (e.g. lowland, with a matrix of woodland, farmland and gardens). It is important to note, therefore, that all the variation shown in **Figure 22.15** represents only minor changes in species number and abundance, not wholesale changes in communities.

Notwithstanding the limited extent to which *absolute* diversity is predicated to change, there are clear local differences in relative effects. These will each reflect the influences of changes in the areas of habitats associated with particular bird communities, as well as variation in the

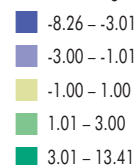


110 This will ignore the occurrence of some rare, conservation-priority birds, but both reflects the species range that is monitored adequately by the survey and will produce an index that better reflects broad ecosystem health, rather than unrelated factors that often influence the distribution of rare species.

111 The numbers mapped in Figure 22.15 are changes in the absolute Simpson's diversity index between the baseline model for Land Cover in the year 2000 and the predictions under the World Market scenario.

a) Go with the Flow 2060 Low

Predicted change in guild richness



b) Go with the Flow 2060 High

Predicted change in guild richness

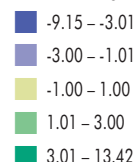


Figure 22.16 Changes in predicted guild richness occurring by 2060 under the UK NEA *Go with the Flow* Scenario for a) low emissions and b) high emissions. Source: Dugdale (2010).

presence or abundance of species that tend to benefit from the juxtaposition of multiple habitats within a landscape. Interpretation of the detail of any given scenario requires close examination of the habitat changes predicted for specific local areas. Overall, however, the greater the land use change, the larger the impact (both positive and negative) upon bird diversity. However, patterns of change are patchy, reflecting the highly heterogeneous British countryside and the highly uneven distributions of birds revealed by bird atlas surveys (e.g. Gibbons *et al.* 1993).

The changes in land cover (linked to an increased overseas ecological footprint) in the *World Markets High* Scenario lead to expectations of decreases in diversity, especially in the southern half of the UK. Importantly, overall changes in diversity may also mask important impacts on individual species. For example, increases in deciduous upland woodlands are likely to impact adversely on the species currently located in the areas where such planting occurs, but increase the representation of currently common lowland species. Note, however, that all the changes in diversity per se are still predicted to be small in absolute terms.

There are a number of important limitations to the data and model interpretations which are discussed in detail by Hulme & Siriwardena (2010). However, in principle, such a modelling approach is well suited to analysis of economic cost-effectiveness. That said, the analysis reported above uses a wide focus across most UK bird species, whereas it is farmland birds which have exhibited the sharpest declines over the past 40 years (Chapter 4). Therefore we complement

the above analysis with a focused consideration of just the latter group.

Preparing for cost-effectiveness analysis 2: habitat association modelling for farmland birds.¹¹²

Chapter 4 of the UK NEA highlights the plight of UK farmland birds as the group which has exhibited the sharpest falls in population numbers over recent decades, declining some 47% between 1970 and 2008. A focused analysis of such birds was undertaken using a methodology which was completely compatible with that used to predict agricultural land use developed by Fezzi & Bateman (in press) as discussed in 22.3.17.2. Such compatibility ensures that any land use change scenario can be simultaneously assessed in terms of both its agricultural impact (including measures of associated values) and its consequences for farmland birds. This compatibility allows the decision maker to investigate a wide variety of policy options from multiple perspectives. For example, we can use this joint modelling approach to provide cost-effectiveness analyses of land use change measures in terms of both financial and bird biodiversity impacts.

The present analysis of farmland birds proceeded by developing habitat association models for 19 bird species that belonged to the same guild (i.e. set of species with similar dietary requirements as assessed via consumption of seeds and invertebrates during the breeding season). The predicted change in guild richness in 10 km squares in England and Wales was calculated using a baseline richness taken from *The New Atlas of Breeding Birds in Britain and*

¹¹² This Section draws from the work of Dugdale (2010).

Ireland (Gibbons *et al.* 1993). Spatially referenced data on agricultural land use were then obtained from the 1988 Agricultural Census (the same data source as used in the Fezzi & Bateman (in press) farm land use model). Other land uses such as woodland were derived from the CEH Land Cover Map 1990, while urban outlines were obtained from standard Ordnance Survey sources.

Statistical regression analyses (detailed in Dugdale 2010) confirmed that the percentage of each 10 km square utilised for cereals, temporary grassland, coniferous woodland and urban use along with the mean altitude were all found to be highly significant predictors ($p < 0.001$) of guild richness. The resulting models allow us to examine the consequences for guild richness under any desired land use scenario. **Figure 22.16** maps the changes in bird guild richness under the UK NEA *Go with the Flow* low and high emissions Scenarios. Patterns are broadly similar across the two scenarios, confirming our previous results that show that, under the *Go with the Flow* Scenario, the switch from low to high emissions makes relatively little difference. Analysis of summary statistics indicates that on average, both scenarios predict an overall decline in guild richness, with this being marginally more severe under the high emission case. However, the maps highlight that the main effect is in terms of spatial heterogeneity, with upland areas generally seeing an increase in farmland birds and the English Midlands and Welsh borders suffering the most significant declines.

Preparing for cost-effectiveness analysis 3: summary.

The contrast in findings between analyses 1 and 2 underlines the importance of considering more than one measure of biodiversity when considering policy in this area. While analysis 1 suggested that upland areas would see a decline in overall bird diversity, analysis 2 shows that the reverse holds for farmland birds, with guild richness in upland England and Wales increasing.

Both analyses are constructed to be entirely compatible with the agricultural land use and valuation modelling undertaken by Fezzi & Bateman (in press) and reported earlier in this chapter (Section 22.3.17.2). In Chapter 26 we contrast the biodiversity impacts of a variety of future scenarios with monetary values for key ecosystem service-related goods. By comparing across scenarios, the decision making can observe the trade-off between economic values and biodiversity offered under each scenario. Such a cost-effectiveness analysis is a significant aid to decision making in the absence of full reliable monetary values for all benefit streams.

22.3.20 Recreation and Tourism

22.3.20.1 Outdoor informal recreational day trips¹¹³

Introduction. Outdoor recreation forms one of the major leisure activities for the majority of the population. According to the most recent figures (NE 2010b) even just focusing upon

English recreational behaviour, there are some 2,858 million visits made p.a., with direct expenditure of some £20.4 billion p.a. This suggests that the true value of these visits is substantially higher than this sum. Considering the location of these visits, research undertaken for the England Leisure Visits Survey (ELVS 2006) report shows that, “during a 12 month period 64% of adults had visited a town/city with 62% visiting a seaside town/city, 59% visited the countryside and 37% had visited the seaside coast. Across England as a whole, 40% had visited a wood/forest in the past year. A quarter (25%) of people had visited a stretch of inland ‘water with boats’ whilst just under one-fifth (18%) had taken a trip to ‘water without boats’” (p.8). Clearly, outdoor visits generate substantial value and it is likely that changes to the natural environment would affect those values in ways which should be considered within policy- and decision-making institutions.

While the majority of outdoor recreation involves informal activities such as walking, nature watching and picnicking, some more distinct activities deserve mention. For example, angling is a major pastime, with about 1 million licensed anglers in England and Wales, although an estimated 2.6 million people go fishing each year. Other notable distinct activities include inland waterway recreation (O’Gorman *et al.* 2010) and bird watching. Licensed anglers fished a total 30 million days during 2005, about 26 million for coarse fishing and 4 million for game (salmon and trout) fishing (EA 2009c). Recreational fishing involves estimated expenditures of about £1,000 million/yr¹¹⁴ in England and Wales, associated with the equivalent of 37,000 full-time jobs. The economic gross value added from an extra 1,000 days of coarse fishing is estimated at £15,000–19,000, varying according to region (EA 2009c).

The CSERGE SEER model. While specific activities are clearly important, it is general, informal activities which form the bulk of ecosystem service-related recreation. Here one of the major problems facing assessment is that the outdoor recreation values generated by any given resource are likely to vary substantially, depending upon spatial context. Put simply, the same resource located in different areas will generate very different numbers of visits and values. This means that any attempt to simplify the recreation decision-making process to the level of assuming a set value for a resource, irrespective of its location, is unlikely to be reliable.

In order to overcome this difficulty and provide the foundations of a general tool for recreation planning and decision making, as well as generating valuations for the UK NEA, the CSERGE SEER project developed and implemented a novel methodology for combining the spatial analytic capabilities of a GIS with new data and econometric analyses to model how the distribution of natural environment and urban resources interact with population distribution to determine recreational visit flows. This new methodology was applied to the Monitor of the Engagement with the Natural Environment (MENE) which was recently released

113 This Section draws principally from Sen *et al.* (2010) and the CSERGE SEER project, although many of the supporting documents prepared for the UK NEA economic chapters (Chapter 22 and Chapter 26) discuss recreation issues. We would like to thank the Monitor of the Engagement with the Natural Environment (MENE) teams at Natural England, Defra and the Forestry Commission, Luke Brander at IVM Amsterdam, the UK NEA Economics group members, Natural England and their contractor, TNS, for sharing their valuable data with us.

114 To clarify; this statement refers to expenditure, not to net economic value in terms of WTP.

by Natural England, Defra and the Forestry Commission. This is a major new database intended to provide baseline and trend information on how people use the natural environment in England. It provides an unrivalled source of data and our present analysis is, as far as we are aware, the first major empirical use of MENE.

The methodology developed for this analysis consists of three elements:

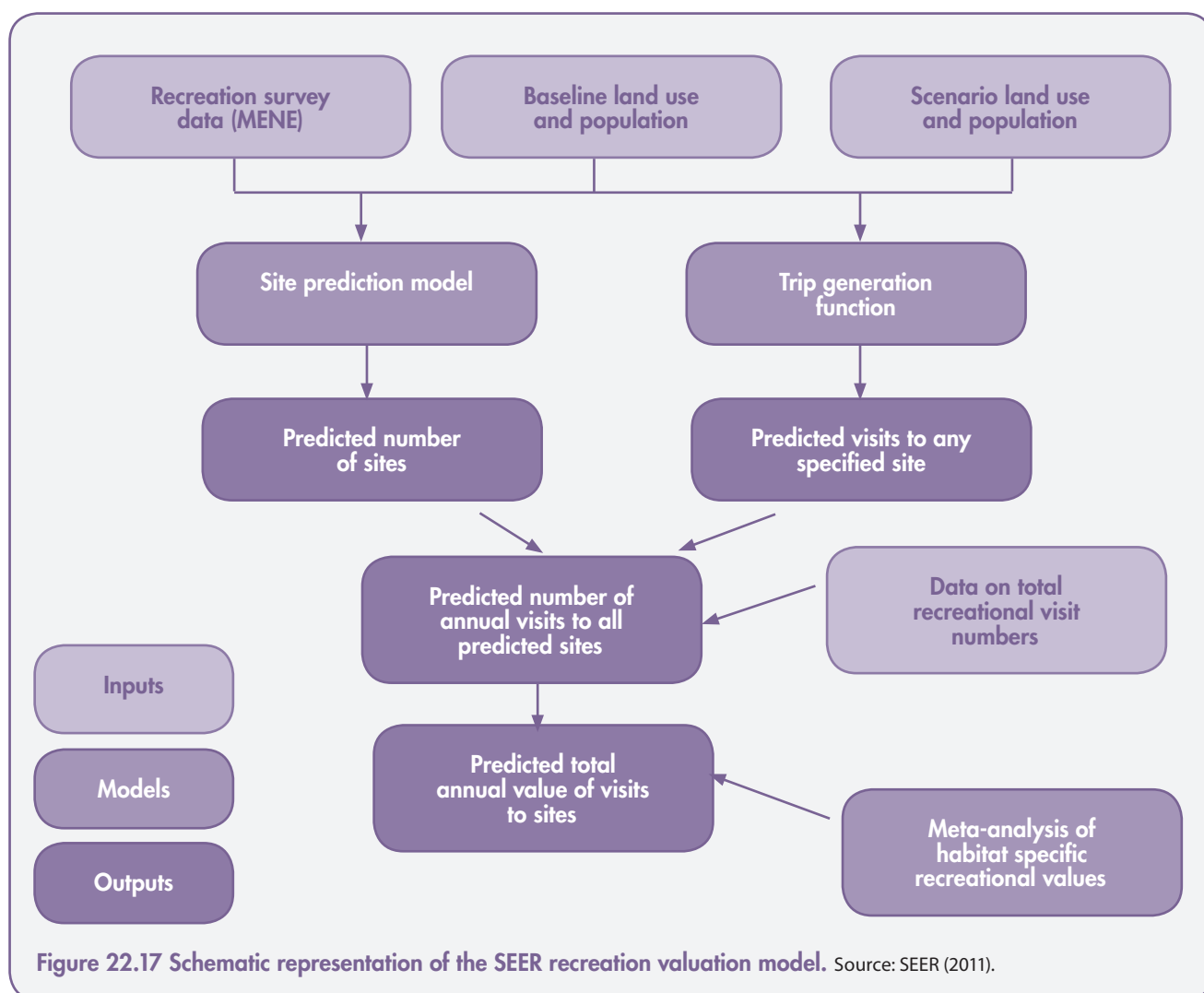
- i) A site prediction model (SPM): Normally, the location of existing and proposed recreation sites is known. However, the economic analysis of the UK NEA Scenarios described in Chapter 26 extends to future worlds where such locations are not known. To address this problem, for the scenario analysis alone we need a way of predicting the likely location of recreational sites in new variations of the world. The SPM achieves this by taking information from MENE on the location of outdoor recreational sites and examining how these are related to: the type of natural resources at that site, the distribution of the population around that site, and travel times from that population to the site. While the location of sites is known for England via MENE, this model also allows us to predict the location of sites for the rest of the UK. This avoids reliance upon secondary sources which are liable to omit informal recreation sites which are not officially recorded as such, but may generate a large proportion of overall trip numbers.
- ii) A trip generation function (TGF): This models the factors determining the number of visits from each UK Census Lower Super Output Area (LSOA) to any given recreational site.¹¹⁵ The analysis takes information on the location of both LSOAs and sites. We incorporate measures of the environmental characteristics of sites (which could be taken either directly from MENE or from the predictions of the previous model) and their surroundings so as to assess their attractiveness to potential visitors. We also examine the accessibility of environmental characteristics within and around LSOA outset locations, so assessing the availability of substitutes which may divert potential visitors away from any given site. Allowance is also made for the population of each LSOA and its socioeconomic and demographic make-up, as this may affect people's propensity to undertake visits.
- iii) A valuation meta-analysis (VMA): Once we know where sites are located and the number of visits to each of those sites, we now seek to value those visits. This stage in the study re-analyses nearly 200 previous estimates of the value of a recreational visit, examining the influence of the environmental characteristics of visited sites and differences in the methods used to generate those value estimates.

Once the SPM has been estimated using data for England taken from MENE, it is then used to generate a predicted

number of potential recreational sites in each 5 km square cell. This model is then extrapolated to all of Great Britain allowing for variation in transport infrastructure, population distribution and habitat type. The only assumption made in this extrapolation is that, allowing for those factors, it is assumed that attitudes towards issues such as distance are roughly consistent across the country. The TGF is then used to estimate the predicted number of visits per week to a site in each of the 5 km cells. By weighting that estimate by the number of sites per cell (as predicted by the SPM) we begin to get a sense of the spatial distribution of visits. However, adjustments have to be made for the sampling strategy of the MENE survey. The survey is well designed for extrapolation purposes, with households from all areas of the country being sampled at all periods across the year, thus avoiding spatial and temporal biases. However, of course only a subset of households can be interviewed, and even these are just asked about the trips they make during a 1-week period, with just one of these being selected at random for detailed study including outset and destination data. Any extrapolation process therefore has to make allowance for all of these sampling characteristics. As the potential for grossing up errors is substantial in such exercises (Jones *et al.* 2002), adjustments were calibrated by official estimates of the total annual number of outdoor visits to all sites. Once this adjustment is made, we obtain our estimate of the predicted number of visits to each 5 km cell allowing for both the number of sites and number of visits to those sites.

The final step of our assessment is to value these predicted visits. Our meta-analysis allows the value of a visit to vary according to the habitat type characteristics of the visited site. We assume that these characteristics can be proxied by information on the physical environment of the 5 km cell into which a site falls. This allows us to generate a site-specific value per person per visit for each trip. Multiplying this by the predicted number of trips to each site in that cell allows us to estimate its annual recreational value. This obviously varies according to the natural environment of the area, the availability of substitutes, the transport infrastructures and the distribution and characteristics of the population in and around that area. The resulting recreational value is therefore highly spatially explicit, reflecting variation in all of these factors. This provides a useful input to decision making, allowing the efficient allocation of scarce resources, which is particularly necessary in times of austerity. Furthermore, these values can be aggregated up across any desired spatial unit up to and including country level to provide an estimate of total annual recreational value under a given scenario. Analyses of policy change or future scenarios can then be undertaken by applying our SPM, TGF and VMA models to the various land use and population distributions envisioned under those policies or scenarios. **Figure 22.17** provides a schematic overview of the methodology developed in this analysis.

¹¹⁵ LSOAs are small areas of around 400–600 households which, particularly in urban areas, means that the influence of location upon visits can be accurately modelled. We used population-weighted LSOA centroids as the outset point for our analysis. Further details regarding LSOAs are available at: www.neighbourhood.statistics.gov.uk/dissemination/Intro.do?page=aboutneighbourhood/geography/superoutputareas/soa-intro.htm. For our modelling of Scottish outset areas we used the Census Data Zone (DZ) unit. A preliminary analysis using Northern Ireland Super Output Area (SOA) data was undertaken, but as this would not have been ready for when the UK NEA went to print, it was not completed.



The analysis allows us to estimate where recreational sites are located, how many visits they will generate and the value of those visits. Importantly, for decision-making purposes, the models allow us to vary policy-relevant elements of the analysis to examine their impacts on recreational values. So, for example, we can examine how new land use scenarios would alter the environmental characteristics of potential sites, making them more or less attractive to visitors and enhancing or degrading the value of any visits made. Furthermore, because of the spatially explicit nature of this analysis, models can readily be linked to other grid-referenced data or analyses so that, for example, we can investigate how changes in the CAP might alter farm incomes and land use (as discussed in Section 22.3.1) and then feed these outputs into the present analysis to examine consequent impacts upon recreational behaviour and values. Further linkages to elements such as water pollution and biodiversity indicators (e.g. bird populations) are an inherent part of the SEER programme of research.

In the present chapter we describe the full SPM, TGF and VMA models. We illustrate their operation through a case study of just a single (although substantial and highly heterogeneous) area. In Chapter 26 this remit is extended to consider all of GB under the full range of population and land use change Scenarios developed by the UK NEA.

Initial data preparation. The intention of this analysis was to produce a decision analysis tool which would not require perpetual reanalysis or updating and should, once constructed, be relatively easy to query by decision makers. However, the model construction phase of the analysis is necessarily data intensive so as to incorporate the complexity of the real world and those locational factors which determine the ways in which recreational values vary across space.

The most crucial and novel source of spatially explicit data used in the analysis was MENE. The data for MENE were provided by a year-long, in-house, face-to-face survey. Respondents were asked about the number of visits that they had made seven days prior to the day of their interview. One of these trips was then randomly selected by the interviewer and the respondent was asked to give detailed information regarding this visit, including the location of the destination. This was then recorded alongside the outset location, providing the vital information required for this analysis. Survey results from MENE were published in September 2010 (NE 2010b) and have been used for economic analysis for the first time in this report.

The methodology developed for this study was applied not only to England where the survey data were gathered, but throughout Great Britain.¹¹⁶ A description of the methodology underlining the GIS-based calculation of locational and travel

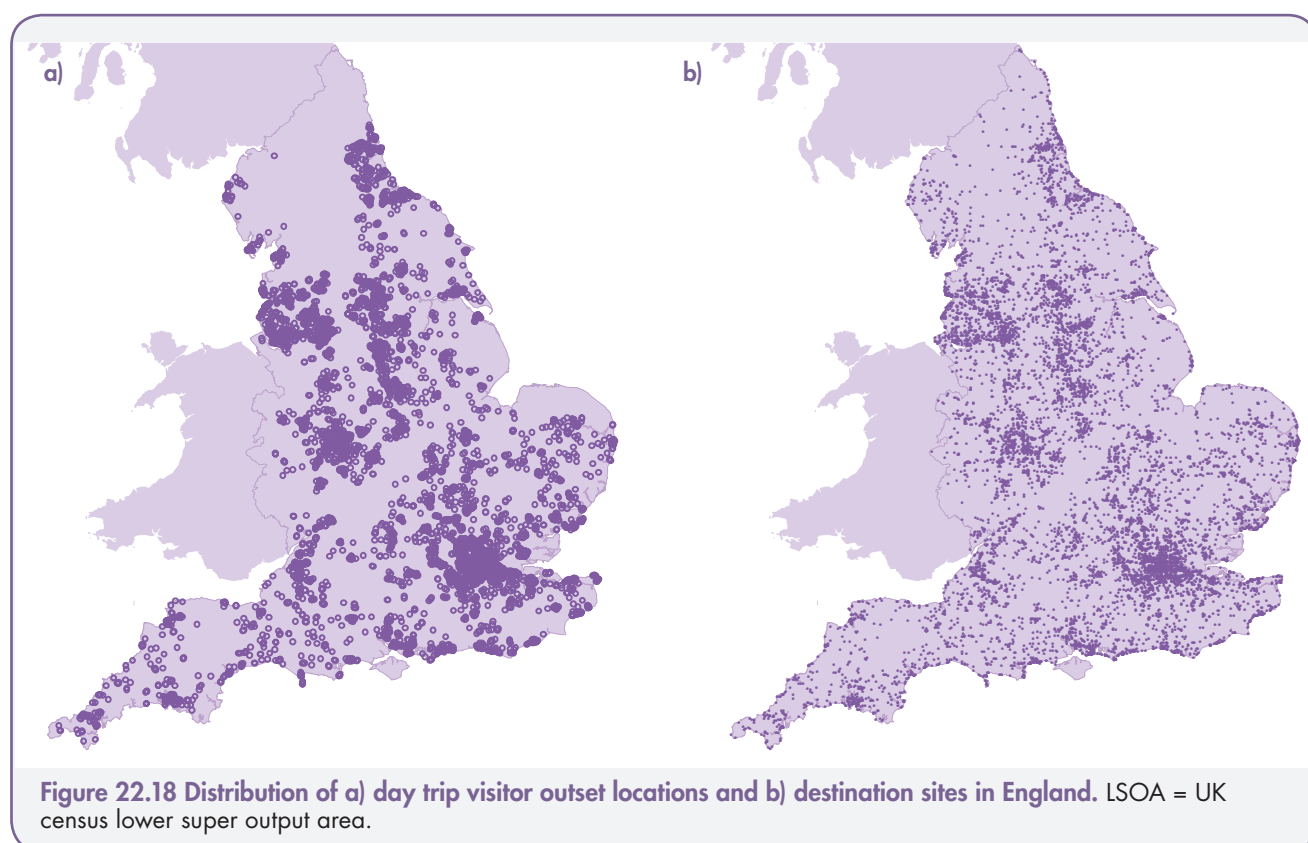
time variables is provided in Sen *et al.* (2010). In summary this entailed the following steps:

- Respondent home and visited site locations were obtained.
- The environmental characteristics for both the visited site and its surroundings were defined.
- A GIS was used to calculate travel times via the entire road network between all potential outset points (LSOAs) and both potential and actual destination sites.
- Potential substitute sites were defined, including measures of the density of different land use and habitat types around each potential outset point.
- Socioeconomic and demographic variables describing each LSOA were obtained.

From an original dataset of 48,514 respondents, 5,305 were omitted due to incomplete locational information and a further 751 were omitted as they were on holiday during the interview period (only day trippers were considered in our analysis) leaving a final sample size of 42,458 respondents.¹¹⁷

An analysis of potential ‘edge-effects’ was undertaken, to examine whether those who live on the land borders of England appeared to have lower than expected visit rates due to visits to locations outside England being truncated. This analysis indicated that a small number of respondents (approximately 150 people) were affected in this way and these were omitted from further analysis. Of the remainder, some 27,593 did not take a visit during the seven days preceding the survey, although these were retained within our subsequent analysis to adjust model estimates for these valid zero visit observations. From the MENE survey, 8,292 distinct destination sites were identified, each having a 1 km square grid reference. **Figure 22.18** maps the location of LSOA outset areas and destination sites.

The environmental characteristics of sites were defined by linking their 1 km square grid cell locations to habitat proportions derived from the 25 m resolution UK-wide Land Cover Map 2000 data (Fuller *et al.* 2002).¹¹⁸ This dataset was used for its coverage and availability. Habitat categories here were: i) broadleaved woodland; ii) coniferous woodland; iii)



¹¹⁶ There is an implicit assumption here that the preferences of English respondents can be generalised across the UK. While we see no clear cultural case against this assumption, one concern is whether the environmental characteristics of England embrace the diversity of the UK. Generally this is not thought to be a problem. Perhaps the weakest element of this assumption is in regard to mountains. England contains a considerably lower density of such environments and does not contain any of the high peaks of Wales and none of the major mountains of Scotland. Obviously it would be ideal to have comparable data from all UK nations. However, perhaps surprisingly, information on both outset and destination location is not collected in surveys other than MENE. Note that while our application considers all of GB, it could readily be applied throughout the UK or further afield, provided that sufficient data are available.

¹¹⁷ Subsequent investigations further restricted our analysis to the more than 90% of day trip journeys with a one-way duration of 60 minutes or less. This restriction was imposed to avoid the very large number of zero visit outset locations imputed when we permit our analysis to allow day trip visits from any outset to any destination across the entire country.

¹¹⁸ LCM2000 is provided by the Centre for Ecology and Hydrology (CEH), Wallingford, UK. The procedure used by the SEER project employs a substantially greater degree of spatial accuracy than that used in the UK NEA Scenarios. As a result of this, the SPM and TGF models reported in the present chapter had to be re-estimated using the simplified land use map employed by the UK NEA Scenarios team before they could be applied to value those scenarios (see Chapter 26). However, the models reported in this chapter are based upon our preferred, high spatial accuracy, approach.

coast (littoral and supra littoral); iv) Enclosed Farmland; v) freshwater body; vi) Mountains, Moorlands and Heaths; vii) estuary (sublittoral); viii) Semi-natural Grassland; and ix) urban and suburban (see details in Sen *et al.* 2010). Percentages of each habitat type in each 1 km square cell were calculated and used to define sites for the SPM estimation. For prediction across GB, habitat proportions were calculated at a 5 km grid square resolution.

Travel times between outset and destination locations were calculated for all of GB, predominantly using the Ordnance Survey Meridian road network. Average road speeds were taken from Jones *et al.* (2010). These discriminate between road types (motorway, A-road, B-road and minor road), as well as between urban and rural contexts. The road network was converted into a regular grid of 100×100 metre cells, with each cell contained a value corresponding to travel-time-per-unit distance. Allowances for locations off the regular road grid were made using adjustments for walking speed (Jones *et al.* 2002). The resultant travel time map was used to calculate the minimum travel time between any outset location and any destination site.¹¹⁹ An example of the resulting travel time surface for just one destination is given in **Figure 22.19**.

The number of visits to a specific site from a given outset location will be lower when that outset area is well served by

other local substitute sites. Ignoring the impact of substitutes is likely to inflate the attractiveness of more distant sites. To allow for this, the availability of substitute resources around each potential outset location across the country was assessed. This was achieved by defining circular areas around each LSOA and calculating the percentage of each land use and habitat type in that area.¹²⁰ This measure of substitute availability was then included within the TGF. The radius of these circles was varied and the analysis repeated to identify the optimal size of the surrounding area for capturing this substitution effect.¹²¹

Previous research suggests that visit rates vary across LSOAs, depending in part upon the socioeconomic and demographic characteristics of those areas (Jones *et al.* 2010). To allow for this possibility, such characteristic data were obtained for all LSOAs from the UK Census with income variables being obtained from Experian data.¹²² Comparable statistics for the rest of Great Britain were also obtained for predictive purposes.

As noted above, we expect that the probability of recreational sites being located in an area is in part a function of the size and distribution of the local population. To include this factor within the SPM, a spatially weighted measure of the population around any point was calculated

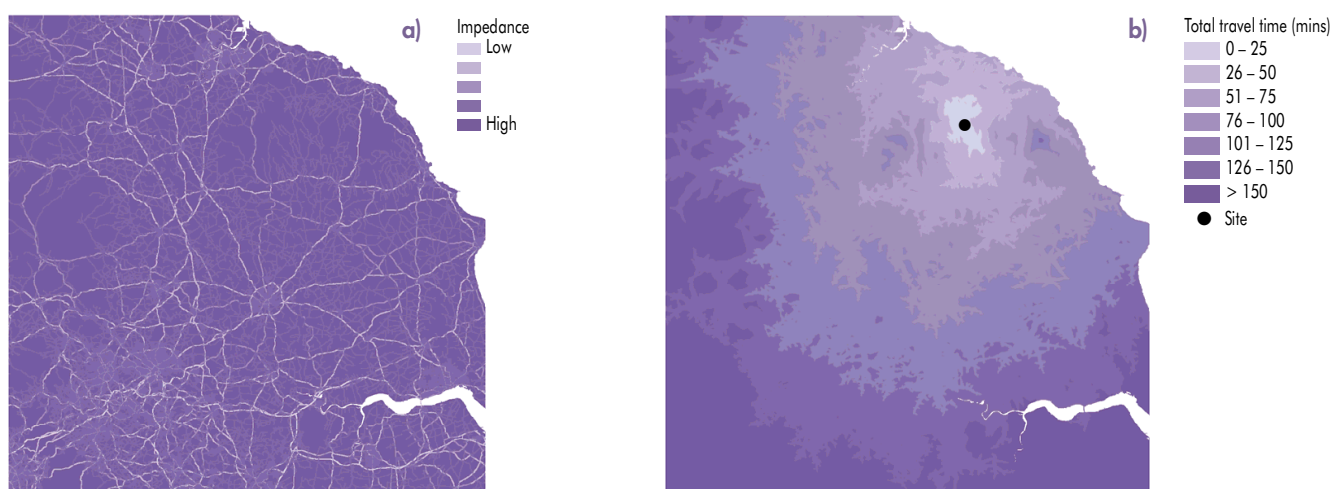


Figure 22.19 Impedance surface (a) and estimated travel time bands (b) for potential outset locations around a single recreational visit site near to Pickering in the North York Moors. Source: SEER (2011).

119 Essential simplifications for the SPM analysis were that all visitors are assumed to start their journey from the population-weighted centroid of their home LSOA and to travel using the shortest time route to their chosen destination site, the location of which is taken to be the geometric centroid of the 1 km grid square containing that site. A similar approach was used for the TGF analysis although here, 5 km grid square centroids were used for the location of destination sites. Bateman *et al.* (1996, 1999) show that actual and GIS predicted routes are highly correlated and the latter provides a strong predictor of the former for modelling purposes. The calculations needed for this analysis were undertaken using the 'Cost Distance' (impedance surface) command in ESRI ArcGIS.

120 Zonal Statistics ++, a module of the 'Hawth's Tools' plug-in for ArcGIS (Beyer 2004), was used to count the cells entirely within the search radius that were of a particular substitute type. These were converted into percentages of the total circle area (25 m cells entirely within the search radius).

121 Radii of 1, 2.5, 5 and 10 km were used for defining substitution availability measures around outset locations. Resultant measures were used within a variety of model specifications including travel time from the population-weighted centroid of each LSOA to the nearest substitute site and interactions between travel time and the proportion of the above circles taken up by substitutes. An AIC criterion (Akaike 1974) comparison of different models indicated that a measure of the density of each land use/habitat type within a 10 km radius of the LSOA population-weighted centroids provided the best fit to the MENE visitation data.

122 This of course assumes that LSOA statistics can be used as valid estimates for the households interviewed in the MENE survey. Note that UK Census 2001 data were used for all socio-demographic variables but that the 2009 Experian data on income was employed. Experian data is held at MIMAS, University of Manchester.

by first taking a 1 km grid square map of population and aggregating this up to the 5 km grid used by that model. Population from outside any 'focal' 5 km square are likely to have a non-zero but diminishing probability of visiting a site in that cell. As there is no theoretical guidance regarding the form of this relationship, it can be determined through purely empirical means. To investigate this we first define a population weight (w) as the following inverse power function:

$$w = \frac{1}{d^y}$$

Where w = population weight
 d = distance from focal cell¹²³
 y = empirically determined exponent

As can be seen, w is defined so that populations at a greater distance from a given location site have a diminishing impact on the probability of that location being a recreational site. The larger the value of the exponent (y) then the faster this diminishment occurs. Empirical analysis suggested that a good fit to the data on actual site locations could be found by an SPM containing two versions of this weight, the first with $y=1$ and the second with $y=2$. This was improved by constraining values of w lower than 0.125 to be zero. **Figure 22.20** illustrates the resultant weight functions.

Analysis 1: the site prediction model (SPM). The first element of our analysis seeks to predict the likely location of recreational sites. While this is not needed where the location of existing or planned recreational sites are known, this is required both for extrapolation of our analysis beyond the base-data area of England, and to apply the model to the new worlds envisioned within the UK NEA Scenarios.

Two broad factors were postulated as determinants of recreational site location:

- the nature of any potential destination site (e.g. its environmental and land use characteristics); and
- the availability of population around that site.

The data drawn from across the entirety of England provide a good deal of variation in both of these dimensions (see details in Sen *et al.* 2010). Analysis of competing model specifications resulted in our best-fitting SPM as reported in **Table 22.21**. This takes Enclosed Farmland as the base land use category such that the coefficients on other land uses show their influence relative to that base case.

Because of the (negative binomial) form of the model the coefficients cannot be directly interpreted as marginal effects. However, their signs do allow simple interpretation of the direction of their effects. To interpret the coefficients on the land use variables we need to recall that these show the differences in effect from the baseline which is set as Enclosed Farmland. Given this, a positive coefficient shows a land use or habitat which is more likely to yield recreational sites than does Enclosed Farmland (and the opposite applies for negative coefficients). This means that coastal, freshwater, Semi-natural Grassland, estuary, broadleaved woodland and even Urban areas yield a higher number of recreation sites than Enclosed Farmland. One

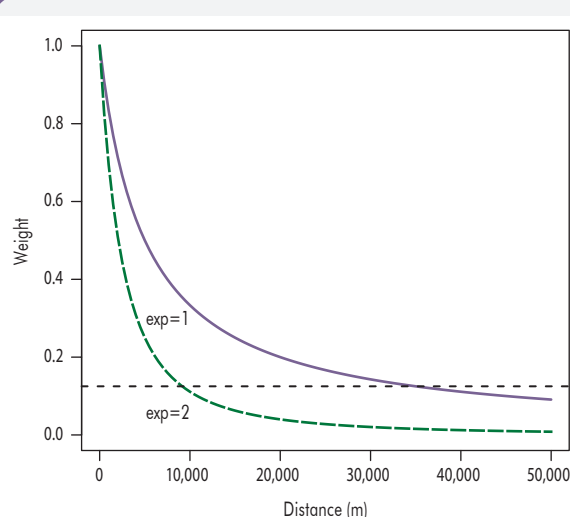


Figure 22.20 Weight function relating population to the probability of recreational sites over increasing distance to that potential site. Exponent (exp) values of 1 and 2 and dotted line indicating cut-off value of 0.125 are empirically determined.

Table 22.21 Site probability model: predicting the number of recreation sites visited in England in each 5 km square using a negative binomial model with robust standard errors. Base category land use is Enclosed Farmland. Statistically significant results are indicated by: * $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$. Source: Sen *et al.* (2010) and the SEER (2011) project.

Dependent Variable [†]	Coefficients	t-statistic	p-value
% of coast in cell	0.00769**	2.603	0.009
% of freshwater in cell	0.0651***	6.128	<0.0001
% of semi-natural grass in cell	0.00545**	3.151	0.002
% of mountains & heath in cell	-0.0149***	-4.949	<0.0001
% of estuary and ocean in cell	0.0134***	12.27	<0.0001
% of urban area in cell	0.0543***	32.07	<0.0001
% of coniferous forests in cell	-0.00631	-1.461	0.144
% of broadleaved forests in cell	0.0267***	10.24	<0.0001
weighted population density ($y=1$) [‡]	0.000000417***	5.541	<0.0001
weighted population density ($y=2$) [‡]	-0.00000486***	-9.103	<0.0001
Constant	-0.805***	-20.62	
Log alpha	-0.644***	-12.22	
Observations [§]	5,497		

[†] Dependent variable is number of visited MENE sites in a 5 km cell.

[‡] The variables 'weighted population density ($y=1$)' and 'weighted population density ($y=2$)' refer to transformations of the weight function (w) described previously with exponent (y) values of 1 and 2 respectively.

[§] The number of observations refers to the number of 5 km square grid cells in England on which the estimation was based. This is less than the number of sites in the MENE dataset due to multiple sites falling within the same grid square.

¹²³ Distance (d) was defined as $d = (\text{centroid distance from focal cell centroid (in metres)} + 5,000) / 5,000$ giving a maximum weighting of 1 for the population of the focal cell.

clear exception is Mountains, Moorlands and Heaths. Again this is in line with expectations as, while such habitats yield high quality recreational experiences (as evidenced in our subsequent TGF and VMA analyses), they are characterised by few access points relative to their size. Interestingly, coniferous forests were insignificantly different from Enclosed Farmland in terms of site probability, a result in stark contrast to the positive and significant effects found for broadleaved woodland. The weighted population density variables indicate a positive and significant but marginally diminishing impact on the expected count of recreational sites.¹²⁴

The estimated site prediction model described above is used to generate a predicted count of potential recreational sites in each 5 km square cell of GB. This count is then divided by the total predicted count of sites for GB to generate a weight for each cell. This figure can then be used in conjunction with the output from our TGF to estimate the number of visits to each cell.

Analysis 2: the trip generation function (TGF). The combination of large numbers of potential outset points and visit sites generates a dataset of more than 4 million observations for analysis within our TGF. The function predicts the number of visits made from each outset location (defined as each LSOA within 60 minutes (one-way travel) of a potential site), to any given recreational site (whether observed or predicted from the SPM) as a function of: the travel time to the site; the accessibility of other potential substitute recreational areas near to outset locations; socioeconomic and demographic characteristics of population in the outset area; and the land use and habitat characteristics of the potential destination site (see Sen *et al.* 2010 for summary statistics on these variables). **Table 22.22** reports our best-fitting TGF.

Examining the relationships captured in the TGF we see that by far the most powerful predictor of visits from an outset area to a potential visit site is the travel time involved. Here the highly significant negative coefficient shows that as travel time increases, so the number of visits falls. This is an important finding as it underlines the vital importance of space in optimal decision making; location is a major driver of value. The impact of the availability of substitutes is also strongly in line with prior expectations, with all substitutes working to reduce visits to more distant sites with the exception of mountains where (as discussed previously) access to sites is limited by the available road infrastructure relative to the size of such areas.¹²⁵

A set of variables is included in the TGF to describe the attractiveness of land use and habitat type across different

potential visit sites. By specifying all site habitat variables to contrast with a baseline of Enclosed Farmland we see that most of the habitat types exert a positive impact upon visits (i.e. they are considered more attractive than enclosed farmlands). Mountains, coasts, freshwater sites and Woodlands exert significant positive effects in attracting visitors. Notice that while mountainous outset locations are associated with a low substitute availability effect, nevertheless they have a positive effect as destinations for visits from other areas.

A set of socioeconomic and demographic variables pertaining to the population in the outset area are also included in the TGF. We observe significantly higher levels of engagement in recreation from retired and richer populations and lower engagement amongst ethnic groups. This latter result highlights the importance of government initiatives to broaden the engagement of ethnic groups in recreational activities.

The estimated TGF allows us to predict the number of visitors who would arrive at a site located in any given 5 km square cell of GB. However, as we have already seen from our SPM analysis, the distribution of sites across the country is far from uniform. Therefore, by multiplying the predictions of visit counts in a given cell (from the TGF) by the expected number of sites in that cell (from the SPM analysis) we obtain an estimate of the total number of visits in each grid square which is fully adjusted for the characteristics and location of that cell. The resulting spatial distribution of predicted visits can readily be mapped for decision support purposes or aggregated up to any desired area including country or GB level. However, we now need to allow for the fact that the characteristics of sites may influence the value of any predicted visits. For this we turn to our VMA model.

Analysis 3: the valuation meta-analysis (VMA). The literature on the valuation of outdoor recreation activities is substantial and a review revealed some 193 value estimates within 98 relevant studies. A meta-analysis of these findings related valuations to both the resources they were concerned with and to various variables describing the type of studies and populations used to provide those estimates. To improve comparability across studies, all the value estimates from non-UK studies were adjusted using purchasing power parity data and all estimates were converted to common pound sterling (2009) prices. Sen *et al.* (2010) detail summary statistics for all variables and **Table 22.23** presents the estimated VMA model.¹²⁶

The estimated model detailed in **Table 22.23** conforms well to prior expectations. Most of the methodological variables are statistically insignificant, which suggests that the framing issues observed in many individual studies may

124 In detail, the implications of the specified weighting function are as follows: i) In centres of high population the value for the inverse square weighted population ($y=2$) is greater than for the inverse linear weighting ($y=1$) such that the site probability is reduced; ii) In areas outside but near to high population centres the inverse linear weighted population ($y=1$) is greater than the inverse square weighted population ($y=2$) such that the site probability is increased; iii) In areas away from high population centres the inverse linear weighted population is slightly greater than inverse square weighted population such that the site probability weight is slightly increased. However, the weight has to be considered in conjunction with the distribution of population before its combined impact upon site probability can be determined. This combined effect is best demonstrated in Chapter 26 where we see that, as expected, the probability of sites declines markedly away from areas of higher population densities.

125 Note that the 'other marine' category does not include coast and generally picks up the effect of less accessible marine areas. But this is insignificantly different from the Enclosed Farmland base category.

126 The mode in Table 22.23 is estimated using OLS with Huber-White-standard errors to adjust for the presence of significant heteroskedasticity. This was insignificantly different from a cluster robust model allowing for the fact that the meta-analysis dataset consist of some studies which report multiple value estimates.

be less of a problem when studies are pooled within a meta-analysis. Interestingly, although the SPM highlighted that mountain areas provided a lower density of recreational sites (a finding reflecting the TGF's low substitution availability offered in mountainous outset areas), the VMA model suggests that those visits that are made to such areas yield relatively high per visit values (a result which chimes with the TGF's attractiveness of mountains as destinations).

Case study. The methodology developed here is flexible and readily applied to a variety of policy questions. In Chapter 26 we apply the method to valuing the variety of changes envisioned in the UK NEA Scenarios. However, the approach can also be applied to more commonplace decision contexts such as the simple question of how to optimise the recreation value generated by a limited budget. Such a question is addressed here so as to demonstrate the versatility of the methodology.

Our illustration considers a simple scenario in which a policy maker has the funds to convert a single area of farmland into recreational forest and wants to know where best to locate that forest. For this simple illustration we bypass the site prediction model (SPM), which is mainly of use when we seek to transfer findings outside England to the rest of the UK (a stage considered in Chapter 26). Therefore we omit this stage and pass straight on to applying our TGF.

The estimated TGF reported in **Table 22.22** shows that Woodland is significantly more attractive to recreational visitors than Enclosed Farmland (the base case for that model). However, the strong influence of travel time shows that both land uses become relatively less attractive the further away a site is from an outset location. This is illustrated in **Figure 22.21**, which shows the predicted visitor rates for each of these land uses at different travel times.

Figure 22.21 demonstrates not only that Woodlands attract more visitors than Enclosed Farmland, but also that there is a strong distance decay in these visit rates. This means that the location of sites will significantly determine the number of visitors they attract. We apply our methodology to examine how the recreational values created by converting Enclosed Farmland to Woodland will vary depending upon the location of that conversion. For simplicity we illustrate this by considering the consequences of placing our new forest in ten randomly chosen locations across the North Humberside area illustrated in **Figure 22.22**. If we were undertaking a formal review of such a scheme then this process would be iterated for all potential sites across the entire area (a process which is rapid and straightforward given modern computing speeds) so as to identify the optimal location for such a scheme.

For each of the randomly chosen land use conversion sites we calculate the various substitution measures needed for the TGF. These are added to data on site characteristics and the socioeconomic and demographic variables included in that model. Applying our TGF visit rates to each location, first under its present agricultural land use and then under Woodland, we can estimate the change in visit numbers generated by the land conversion policy. The final stage of our analysis is to use our VMA model to value predicted visits to each site under first Enclosed Farmland and then Woodland.

Table 22.22 Trip generation function: predicting visit numbers from an outset location to a site destination estimated using a Multilevel Poisson regression model. Statistically significant results are indicated by: * $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$. Enclosed Farmland is taken as the base case for both the 'substitute availability' and 'site' characteristic variables. Source: Sen *et al.* (2010) and the SEER (2011) project.

Variable	Coefficient	t-statistic
Travel time from a LSOA/DZ to a site [†]	-0.0594***	-106.3
Coast substitute availability [‡]	-0.0115***	-4.156
Urban substitute availability [‡]	-0.0211***	-32.99
Freshwater substitute availability [‡]	-0.0633***	-5.109
Grassland substitute availability [‡]	-0.0225***	-10.16
Woodland substitute availability [‡]	-0.0166***	-8.446
Other marine substitute availability [‡]	0.000710	0.738
Mountain substitute availability [‡]	0.0148***	3.725
% of coast in site [†]	0.00940***	6.504
% urban in site [†]	-0.00219***	-4.464
% of freshwater in site [†]	0.0102***	4.220
% of grasslands in site [†]	0.00158	1.343
% of woodlands in site [†]	0.00286**	2.948
% of estuary and ocean in site [†]	-0.0156***	-11.89
% of mountain and heath in site [†]	0.0226***	10.54
% non-white ethnicity [†]	-0.00580***	-6.537
% retired [†]	0.00642***	3.678
Median household Income [†]	0.00000874***	9.414
Total population of outset area [†]	0.000225***	5.899
Constant	-3.195***	-37.84
Insig2u[†]		
Constant	-0.737***	-21.76
Observations	4,141,089	

[†] The site characteristic variables are the number of visits from a specified small area Census unit (Lower Super Output Area (LSOA) in England and Wales; Census Data Zone (DZ) in Scotland) to a specified site.

[‡] The substitute availability variables are calculated as the percentage of a specified land use type within a 10 km radius of the outset point.

[†] log sigma2u = natural logarithm of the variance of the random intercept term in the multilevel model. The random intercept term captures the unobserved heterogeneity between the different sites.

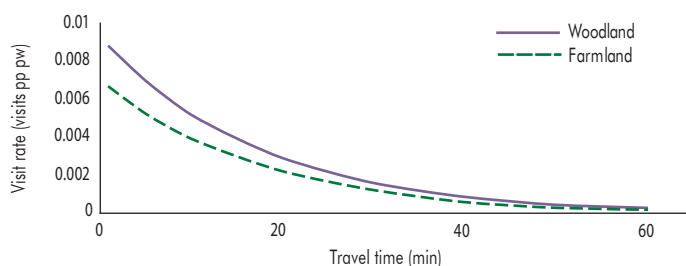


Figure 22.21 Trip Generation Function predictions: travel time impacts on visit rate for woodland and farmland sites. Visit rate calculated per person per week (pp pw). Source: Sen *et al.* (2010) and SEER (2011).

Table 22.23 Valuation meta-analysis (VMA) model of recreational value estimates (£, 2009 prices). Model estimated using Ordinary Least Squares with Huber-White standard errors to adjust for heteroskedasticity[†]. R² (adjusted) = 0.75; Statistically significant results are indicated by: *p<0.10, **p<0.05, *p<0.01, ****p<0.001**

Variable [‡]	Variable definition	Coefficient	t-statistic
Good characteristics[§]			
Mountains & heathlands	1 if recreational site valued is mountain or heath; 0 otherwise	1.771*	1.834
Grasslands, farm & woods	1 if recreational site valued is grasslands, farm and woodlands; 0 otherwise	0.579*	1.886
Freshwater, marine & coastal	1 if recreational site valued is freshwater, marine & coastal; 0 otherwise	0.222	0.763
Designated site	1 if recreational site holds some official designation; 0 otherwise	0.0225	0.121
Study characteristics[§]			
Published	1 if study published in peer-reviewed journal or book; 0 otherwise	0.133	0.468
Survey year	Discrete variable: 1 = published in 1975, to 29 = published in 2008	0.0360	1.364
Log sample size	Logarithm of sample size	-0.493**	-2.143
In-person interview	1 if survey mode is in-person; 0 otherwise	0.130	0.469
Use value only	1 if use value study; 0 otherwise	0.372*	1.787
Substitutes considered	1 if substitute sites included in the valuation study; 0 otherwise	-0.117	-0.570
Valuation unit[§]			
Per household per year	1 if value in terms of per household per year; 0 otherwise	2.825****	8.583
Per person per year	1 if value in terms of per person per year; 0 otherwise	2.090****	6.251
Other valuation unit	1 if value in terms of per household/ person, per day/ month; 0 otherwise	2.101****	4.648
Valuation method^{††}			
RPM & mixed valuation	1 = revealed preference or mixed valuation methods; 0 otherwise	1.494**	2.335
Open-ended format	1 = stated preference using open-ended WTP elicitation format; 0 otherwise	-0.363*	-1.838
Payment vehicle tax	1 = payment vehicle is a tax; 0 otherwise	0.351	1.316
Study country characteristics			
Log of population density	Population density of state/country in which the site is located	0.360	1.206
Non-UK countries ^{††}	1 = study conducted overseas; 0 otherwise (UK)	1.193***	3.215
Constant		-0.110	-0.123
Observations		193	

[†] This was insignificantly different from a cluster robust model allowing for the fact that the meta-analysis dataset consist of some studies which report multiple value estimates.

[‡] Dependent variable is logarithm of recreational value (willingness to pay or consumer surplus) (£, 2009).

[§] Omitted land use base case = urban environments.

[§] Base case for valuation units is per person per visit.

^{††} Base case for valuation method is close-ended stated preference methods.

^{††} Non-UK countries considered: North America, Western Europe, Australia and New Zealand.

Table 22.24 presents results from this illustrative analysis. As can be seen, in each of the ten locations considered, the number of visits increases when the land is converted into Woodland. However, the magnitude of this change and the value they generate varies very substantially across locations. Site P9 yields the highest increase in value from this change in land use while site P4 provides the lowest value. Clearly, the incorporation of spatial variation into decision making is a vital aid to efficient resource allocation, particularly in a time of austerity.

Under a cost-effectiveness analysis this would conclude our assessment. However, a full economic cost-benefit analysis would supplement this recreational value with the other market and non-market benefits generated and set this against the costs of each scheme in each location. Because costs such as the loss of agricultural output values will also

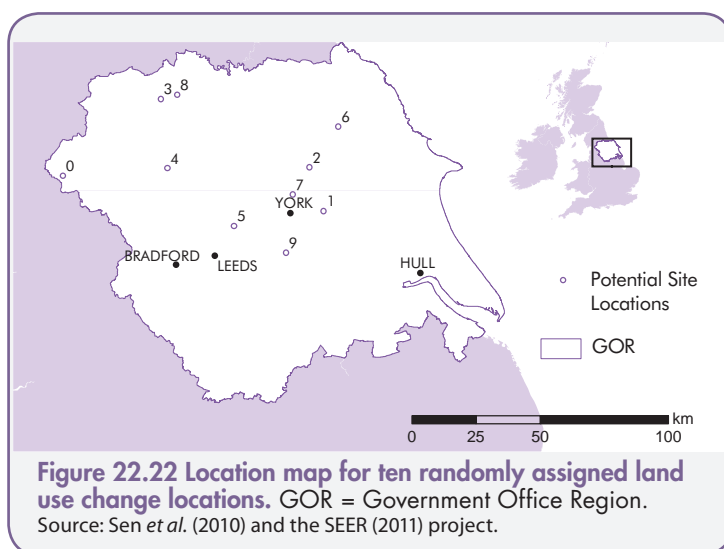


Table 22.24 Predicted increase in recreational visits and valuations at alternative sites following conversion from farmland to woodland (£/yr, 2010 prices). Source: Sen *et al.* (2010) and the SEER project (2011).

Site No. →	P0		P1		P2		P3		P4	
Description	Very remote site but near to rural A-road		Close to York		A little remote and with substitutes like P7 nearer to York		Slightly further from Middlesbrough than site P8		Very remote and with no nearby major roads	
Travel bands (min)	Extra visits (p.a.)	Value of extra visits (£ p.a.)	Extra visits (p.a.)	Value of extra visits (£ p.a.)	Extra visits (p.a.)	Value of extra visits (£ p.a.)	Extra visits (p.a.)	Value of extra visits (£ p.a.)	Extra visits (p.a.)	Value of extra visits (£ p.a.)
1	0	0	0	0	0	0	0	0	0	0
5	0	0	0	0	0	0	0	0	0	0
10	0	0	195	643	0	0	0	0	0	0
20	162	537	541	1,788	0	0	595	1,967	0	0
30	241	796	3,159	10,444	90	298	451	1,492	0	0
40	201	664	602	1,991	251	830	1,004	3,318	33	111
50	931	3,076	1,042	3,445	875	2,891	958	3,168	28	92
60	709	2,342	1,671	5,523	290	957	822	2,719	67	222
Total	2,243	7,414	7,210	23,834	1,506	4,977	3,831	12,664	129	425
Site No. →	P5		P6		P7		P8		P9	
Description	Midway between York & Leeds with good road links		Remote site but near to rural A road		Very close to York		Quite near Middlesbrough but no main road link		Midway between York & Leeds with excellent motorway links	
Travel bands (min)	Extra visits (p.a.)	Value of extra visits (£ p.a.)	Extra visits (p.a.)	Value of extra visits (£ p.a.)	Extra visits (p.a.)	Value of extra visits (£ p.a.)	Extra visits (p.a.)	Value of extra visits (£ p.a.)	Extra visits (p.a.)	Value of extra visits (£ p.a.)
1	0	0	0	0	0	0	0	0	165	545
5	0	0	261	862	0	0	0	0	130	431
10	0	0	292	965	584	1,930	0	0	0	0
20	1,028	3,398	271	894	2,705	8,942	866	2,862	1,948	6,438
30	3,581	11,836	361	1,194	2,046	6,764	301	995	4,574	15,119
40	4,601	15,209	402	1,327	719	2,378	1,389	4,590	4,852	16,039
50	4,643	15,349	996	3,291	1,740	5,752	968	3,199	4,718	15,595
60	2,183	7,215	590	1,949	1,914	6,326	1,091	3,608	3,372	11,148
Total	16,036	53,008	3,171	10,483	9,708	32,092	4,614	15,253	19,759	65,315

vary spatially, it is not necessarily the case that the site which generates the highest recreational value is necessarily the optimal location for such land use conversion. Nevertheless, given the prevailing shadow value of agriculture it seems very likely that many of these sites, if chosen, would pass benefit-cost tests (although note that there is a substitution effect here; once one new site is created this forms a substitute for, and lowers the value of, any other proposed site in the vicinity—our methodology can readily be automated to permit the capture of such effects within the decision analysis system). Why then do such sites not presently exist? This is, in part, a reflection of market failure; at present land users are not compensated for the recreational and other non-market benefits they provide and hence such services are, from a social optimality perspective, under-supplied. It is the task of government to address such market failures through incentives or other mechanisms (including the removal of market imperfections and distortions which, perversely, often reinforce the problems of missing markets for environmental goods).

22.3.20.2 Outdoor tourism

The UK tourism industry is a major contributor to the economy, yielding a direct value of £52 billion p.a. (roughly 4.0% of gross domestic product (GDP)) in terms of businesses providing tourism-related goods and services, with a substantially larger sum being claimed as an indirect contribution through supporting businesses in the supply chain (VB 2010). Estimates of the number of visits by overseas residents to the UK vary from 20 million p.a. (ONS 2010) to 30 million p.a. (Visit Britain 2010) although there is closer agreement on their related spending at about £16 billion (ONS 2010; VB 2010).

It is unclear to what extent these sums might be attributed to ecosystem services, or to what extent variation in those services might change this expenditure and what the underlying economic values might be.¹²⁷ Nevertheless, given the size of expenditures involved and the likelihood of ecosystem service contributions to such values, this would appear to be an area worthy of further investigation.

22.3.21 Urban Greenspace Amenity¹²⁸

22.3.21.1 Introduction and overview

While the natural science assessments of the UK NEA consider Broad Habitats and the ecosystem services they provide (Chapters 4–16), the economic analysis focuses upon the goods and values that those services offer. However,

a problem arises when we consider habitats which yield sets of goods, the amounts of which are highly correlated together. This is the case for urban greenspace which yields multiple ecosystem related goods such as recreation, visual aesthetics, reductions of air and noise pollution, all of which tend to be highly correlated (i.e. larger parks generally provide more opportunities for recreation, more visual amenity and better levels of noise and pollution reduction than do smaller parks).¹²⁹ In such situations it is very difficult to separate out the effect of any one individual good upon people's well-being, and hence individual valuation becomes problematic.¹³⁰ In such cases, we are instead forced to value the collective bundle of correlated goods and tend to refer to this bundle through the shorthand of the habitat name. However, we should not forget that it is these goods, rather than the habitat from which they are derived, that we are valuing. That said, one of the convenient features of the urban greenspace amenity bundle of goods is that, within reason, it does not duplicate values estimated elsewhere. For example, it excludes the benefits of private gardens and the values of rural recreation, both of which we consider in Section 22.3.14. Indeed rather than resulting in a net over-estimate of values, a lack of data meant that our analysis is liable to underestimate values, as we omit items such as the impact of urban greenspace on the reduction of downstream flooding risks. The values presented should therefore be treated as lower bound estimates. Double counting should, therefore, not be a major issue here. Any possibility of overlap between the three categories of greenspace designation, and which might lead to some overstatement of values are discussed by Perino *et al.* (2010).

The analysis developed the following methodology:

- i) A meta-analysis of previous urban greenspace valuation studies was undertaken, with particular emphasis being placed upon the spatial location of study households in relation to various categories of urban greenspace including city parks, the urban/rural fringe and informal greenspace. The meta-analysis provides value functions, quantifying how values vary with proximity to the former two types of urban greenspace and the percentage cover in a 1 km square of the latter.
- ii) A set of UK urban centres, ranging from relatively small cities like Norwich to major conurbations like Glasgow, were spatially analysed using GIS techniques. This analysis provided information on the proximity of each household in the city to urban greenspaces in that city and the percentage cover of informal greenspace in the household's vicinity.¹³¹

¹²⁷ Some habitat specific estimates are available. For example, Beaumont *et al.* (2010) report that UK seaside tourism is valued at £17 billion annually. However, such a value appears at odds with other estimates. (ONS 2005, 2006). Other habitats generate more modest expenditures, such as the £3 million spent annually upon skiing in Scotland (Tinch *et al.* 2010), although here the link with ecosystem services such as climate are clearly easier to demonstrate and, reflecting this, visitor numbers have fluctuated with the weather (Tinch *et al.* 2010). While there is clearly a dearth of detailed research into this issue, the size of sums involved suggests that this might be a fitting subject for further investigation.

¹²⁸ This Section draws on Perino *et al.* (2010). We thank Olena Talavera for excellent research assistance.

¹²⁹ Of course there are exceptions and as we show subsequently in this section, there is no reason to suppose a linear relationship between the size of a park and the benefits it offers.

¹³⁰ This problem is not always insurmountable. For example, Day *et al.* (2007) manage to collect enough information to generate separate valuations for different sources of noise. However, this typically takes very substantial amounts of data (in the latter case more than 10,000 observations were used) and this level of information was not available in the case of urban parks.

¹³¹ Note that proximity measurements were taken from the centroid of each full postcode, although as these typically contain just 20 households, any error induced by this assumption should be minor. This caveat applies throughout this Section.

- iii) These data were then fed into the value functions obtained from our meta-analysis to estimate values for each greenspace category. Summing these together gave a total urban greenspace value for the status quo configuration of urban greenspace.
- iv) Changes in urban greenspace were then obtained from the UK NEA Scenarios team (Chapter 25). By inputting these scenarios into the value functions and contrasting findings with those for the status quo, we can estimate the change in values induced under each of these scenarios. We consider two of these scenarios in detail within this chapter and the full set of UK NEA Scenarios are considered in Chapter 26.
- v) By considering the characteristics of those cities assessed in detail, and comparing these to all Urban areas across Great Britain, value estimates were obtained for all cities. These were summed to obtain a national estimate of the value of changes in urban greenspace under each scenario.

22.3.21.2 Meta-analysis of urban greenspace valuation studies

A meta-analysis is essentially a study of studies through which prior research is assessed together, typically using quantitative methods. A review of the relevant literature (see Perino *et al.* 2010) produced a set of five studies that value benefits associated with urban greenspace in UK cities, from which 61 marginal valuations were extracted¹³². These studies embraced three different valuation methods, namely hedonic pricing (two studies giving 37 values), contingent valuation (two studies providing six values) and expert interviews (one study yielding 18 values). Analysis showed that these studies covered a wide variety of circumstances, including areas both close to and distant from both small and large areas of urban greenspace.

Meta-analysis (reported in full by Perino *et al.* 2010) of the valuations gleaned from the literature showed that the value of urban greenspace declined with increasing distance from the valuing household and increased with the size of the greenspace in question.¹³³ Both of these are marginally diminishing effects such that, for example, as the size of greenspace doubles so its value increases, but by less than double. This reflects a basic finding characteristic of virtually all goods, whether related to ecosystem services or not.

The valuation functions estimated from the meta-analysis were then applied to estimates of distance to, and size of, urban greenspaces for the set of UK cities subjected to spatial analysis, and estimates of resultant values for the status quo were obtained.

22.3.21.3 Scenario analysis methods

The UK NEA Scenario team provided a variety of future visions of UK cities. We use two of these: *Go with the Flow* and *Green and Pleasant Land* to illustrate the method developed for valuing urban greenspace amenity. Values are assessed by comparing outcomes under each scenario with present day situations. **Table 22.25** presents relevant aspects of these scenarios as specified by the UK NEA Scenario team.

The changes envisioned in these scenarios were implemented for our sample of UK cities using a set of simple assumptions. Changes in urban area were assumed to occur evenly around the perimeter of a city and a similar procedure was adopted for changes to the size (and hence location) of existing greenspaces. Increases in population were allocated so as to preserve the relative densities observed at present. The scenario descriptions supplied specify the state of the world in 2060 but do not provide any details about the period in between. Therefore the assumption was made that changes are spread evenly across the 50 years considered.

Under these assumptions, each of the scenarios was applied to each of the cities considered within our spatial analysis. This alters the size of each urban greenspace and its distance to each household. Feeding this data into the meta-analysis model allows us to calculate the change in value generated under each scenario for each household p.a. As these values are spread over a 50-year time horizon we apply standard HM Treasury (2003) discounting rules to obtain their present value.¹³⁴ Values for an example city are presented in Perino *et al.* (2010). However, in the present chapter we focus upon the implications of this analysis at the national level.

22.3.21.4 Scenario values for Great Britain¹³⁵

Given that the smallest city considered in our analysis was Norwich, we are wary of over-extrapolating our values for urban greenspace, and hence restrict ourselves to considering urban centres with populations in excess of 50,000. The

Table 22.25 Urban dimensions of two UK NEA Scenarios: *Go with the Flow* and *Green and Pleasant Land*.

Scenario name	Change in urban area (%)	Change in urban population (%)	Change in formal urban greenspace area (%)	Change in informal urban greenspace area (%)
<i>Go with the Flow</i>	3.0	32.2	36.2	0.0
<i>Green and Pleasant Land</i>	0.0	21.7	38.9	5.4

¹³² While there is a wider international literature, this introduces problems associated with translations across economic and cultural contexts from countries which may have very different availability of such greenspace. All of these factors will influence values, making the pooling of estimates problematic unless a large number of observations are available to control for these various influences.

¹³³ Ideally we would have wished to base these analyses upon travel times rather than distances. Indeed this is the approach taken in the valuation of recreation work where both outset and destination locations were available. However, such information was not available for the urban greenspace analysis.

¹³⁴ Note that there is a degree of inconsistency here in that the HM Treasury discounting rules are based on the assumption of a 2% average growth rate of the UK economy while the UK NEA Scenarios adopt growth rate assumptions varying from 0.5% (Local Stewardship scenario) to 3% (Nature@Work scenario).

¹³⁵ We restrict our analyses to GB, as comparable data for Northern Ireland were not available.

general reasoning behind this restriction is that for smaller towns, urban greenspace plays a lesser role in the provision of many related ecosystem services as, by their very nature, most households live relatively close to rural areas.

As the analysis did not have access to data allowing the measurement of distance from all urban households to all greenspaces in each city across Britain, a simpler extrapolation was undertaken. This sought to characterise each small census area (lower super output area or LSOA) in each of the cities in our spatial analysis in terms of their local area income and population density as well as larger scale measures of the size of city in which they were based. A simple regression model then related the median urban greenspace value in each LSOA under each scenario to these characteristics. These characteristics are known for all LSOAs in every urban area across Britain and so the model allows us to produce an estimate of how each scenario will change the value of urban greenspace (relative to the present day situation) as experienced in every urban LSOA. Summing across all these areas gives

us our national level estimate of the value changes induced under each scenario.

Figure 22.23 details the spatial distribution of changes in the discounted value of urban greenspace across GB under the *Go with the Flow* Scenario (**Figure 22.23a**) and the *Green and Pleasant Land* Scenario (**Figure 22.23b**). The maps illustrate that per household changes in benefits are highest in the centres of large conurbations. However, what is more important is the nature and scale of these changes. The *Go with the Flow* scenario leads to a worrying reduction in urban greenspace amenity values as large increases in urban extent and population and static informal greenspace overwhelm the relatively modest increases in formal city park areas. In contrast, under the *Green and Pleasant Land* Scenario urban greenspace values increase as more modest changes in urban population and extent are complemented by relatively large increases in both formal and informal areas of greenspace.

By summing the values estimated for each urban LSOA we can obtain GB-level estimates of the change in urban greenspace amenity values under each scenario.¹³⁶ These are

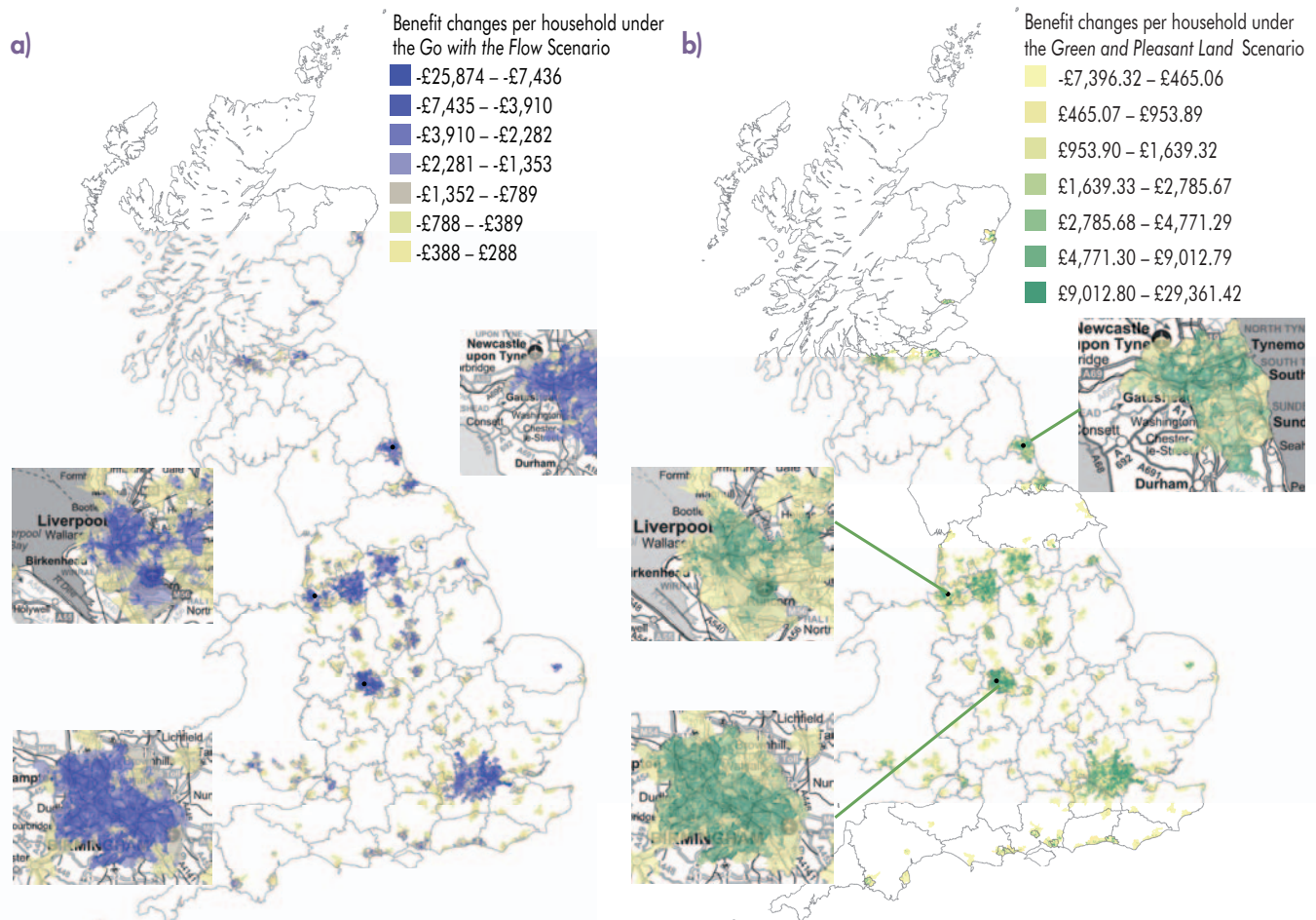


Figure 22.23 The spatial distribution of changes in the per household net present value of urban greenspace changes across Great Britain under a) the *Go with the Flow* Scenario and b) the *Green and Pleasant Land* Scenario. Source: Perino *et al.* (2010) and the SEER (2011) project. © Crown Copyright/database right 2010. This work is based on data provided through EDINA UKBORDERS with the support of the ESRC and JISC and uses boundary material which is copyright of the Crown and the Post Office.

¹³⁶ These are calculated for the 15.2 million urban households living in the areas included in the extrapolation. This is, of course, an underestimate of total values, as those living outside these areas may well also hold values for improvements in urban parks (reflecting their actual or potential use of those parks and any non-use values).

Table 22.26 Changes in the value of urban greenspace in Great Britain and per household under two UK NEA Scenarios, each compared with the present day situation. Source: Perino *et al.* (2010) and the SEER (2011) project.

Assessment unit	Assessment period	<i>Go with the Flow Scenario</i>	<i>Green and Pleasant Land Scenario</i>
Great Britain	2010–2060 (discounted value)	£55 billion	£66 billion
Great Britain	Per annum (annualised equivalent)	£1.94 billion	£2.32 billion
Average urban household	2010–2060 (discounted value)	£3,600	£4,400
Average urban household	Per annum (annualised equivalent)	£128	£152

detailed in **Table 22.26** as discounted present values for the entire period (2010–2060) and their annualised equivalents. Average values for each urban household considered in the analysis are also reported.¹³⁷

Table 22.26 summarises the findings of **Figure 22.23** showing us the magnitude of losses under *Go with the Flow* and the potential gains under *Green and Pleasant Land*. Average annual impacts upon household welfare are a loss of nearly £130 under the former scenario and a gain of just over £150 p.a. under the latter. While such changes might appear rather modest, when aggregated across the majority of British households that live in urban areas, they generate substantial welfare changes of the order of roughly £2 billion p.a.

22.4 Summary and Conclusions

This chapter provides a summary of findings from the detailed economic reports compiled for the UK NEA (see Section 22.1).

The chapter opened with a summary of the methodology underpinning economic analyses of ecosystem services (further details of which are found in Bateman *et al.* 2011a). This clarified that the main focus of the UK NEA economic analysis was to examine the value of ecosystem service flows. This is a substantial advance upon conventional financial analyses which focuses upon market-priced goods to the exclusion of the many non-market values generated by ecosystem services. Nevertheless, an early

caveat concerned the recognition that there is inadequate understanding of the sustainability of many ecosystem services and that awareness of potential thresholds beyond which our use of natural resources is unsustainable is a priority for future research. It is clear from the evidence presented that ecosystems provide a very substantial stock of economic value and that ecosystem services represent a significant flow of economic value at the national level.

The methodological summary introduced what we hope will be seen as a simple terminology to help common understanding of the ecosystem service concept across economists and other social scientists, all areas of the natural sciences (not just the biological sciences which have traditionally dominated ecosystem concepts) and decision makers. Because of the potential for error and double counting, if we try to value all of the interlocking relationships which make up the complexity of the natural world, the economic focus is upon those ‘final ecosystem services’ which are the last link in the chain of natural processes which contribute to human well-being by inputting to the production of ‘goods’. Our use of the term ‘goods’ goes well beyond the common conception of market-priced items to include non-market contributors to well-being, be they physical or non-physical (pure experiential) objects. While some of these goods come straight from the natural world without the intervention of humans (e.g. the visual amenity of beautiful natural landscapes), others require some inputs of manufactured or other human capital (e.g. intensive food production). We also discussed the need to adjust our assessment of ecosystem service values for these other capital inputs and the fact that while economics can value most goods, non-monetary methods are an important complement for assessing those which are not amenable to economic appraisal.

The majority of our methodological summary considered the transition from goods to their value. We made the distinction between prices and values and noted that the latter can arise in both use and non-use contexts. Our summary reviewed the variety of economic valuation methods which have been developed, showing the differing situations in which each is most appropriate. Reference has also been made (in this chapter and a number of the natural science chapters—e.g. Chapter 12) to various financial value estimates that exist, for example tourism day visit expenditure, specific recreation expenditure and employment creation, related to ecosystem services. While these data are useful in order to signify the importance of such services, they are not economic values and cannot be aggregated with the latter.

Another area which we also emphasised was the key distinction between the total and marginal value of a resource. While total values are arguably of importance for highlighting the overall contribution and importance of ecosystem services to human well-being, they are of little help in the decision-making process, which is very rarely concerned with, say, the total loss of a resource, but rather focuses upon the trade-offs involved in alternative options.

¹³⁷ Per household rather than per hectare values are reported, because the value of a hectare of urban greenspace is highly dependent on its location (driven, for example, by the number of households living close by). Furthermore, the extrapolation procedure is based on household information, since data on urban greenspace are not available at sufficient detail at the level of Great Britain.

For these latter decisions, what is needed is an assessment of how an increase or loss of a unit of the resource will affect well-being. Such unit or 'marginal' values can then be used within analyses of trade-offs to allow the decision maker to determine the best use of available resources¹³⁸. This led us to a simple prescription for such analyses: that they should i) understand the change in provision of the good under consideration; ii) know its marginal value; and iii) understand how ii) might alter as i) changes.

Our methodological summary then continued via a brief discussion of decision making for delayed costs and benefits through the process of discounting before a case study illustrated the four guiding principles of economic analysis for ecosystem service assessment:

- i) Integration of natural sciences and economic assessments of the relationships determining the provision of ecosystem service;
- ii) Valuation of the benefits of all welfare-bearing goods, including those either directly or indirectly provided by ecosystem services;¹³⁹
- iii) Efficient use of resources; and
- iv) Distributionally aware decision making.

The rest of the chapter applied these tools and principles across the wide gamut of goods which ecosystem services either directly or indirectly provide. The detailed ecosystem service valuations presented in the main body of this chapter can be broadly categorised into those that assess past trends and those that consider likely future scenarios.

Considering the first category, there has been relatively little work which has adjusted for the value of manufactured and human capital in ecosystem service-related output values. This means that many of the estimates in this category are liable to overstate the contribution of ecosystem services to resultant values. Nevertheless, ecosystem inputs are often vital to the production of such goods and accepting this caveat, **Table 22.27** gives a summary of the variety of value estimates provided by this chapter.

22.4.1 Integrated Valuations

A number of the economic valuation exercises undertaken for the UK NEA were designed to integrate together so that policy makers could readily examine the impact of a given impetus for change across multiple impacts. An example of this is given in the integration of work undertaken by the CSERGE SEER project at the University of East Anglia, the British Trust for Ornithology and the University of Leeds. Here the CSERGE SEER Land Use Model was used to estimate the impact of combinations of market forces, policy shifts and environmental change (especially the UKCIP climate change scenarios). The resulting shifts in land use were used both directly to produce valuation estimates in terms of farm gross margin changes, and indirectly as the basis for predicting consequent changes in bird diversity (as an indicator of biodiversity) and carbon storage. Ongoing work under the SEER project will add in further integrations to examine linked issues such as the impact of this land use change upon recreation.

Table 22.27 Summary of UK NEA ecosystem service valuations.

Section	Good	Valuation method*	Valuations
22.3.1.1	Marine food production	Market prices [†]	■ The value of UK fish landings is around £596 million p.a., while that of aquaculture (fish and shellfish farming) is around £350 million annually. However, there is insufficient data to isolate ecosystem contribution from manufactured capital inputs.
22.3.1.2	Woodland-related food production	Market prices	■ Venison valued at over £24 million p.a.
22.3.2.1	Pollination services	Production function method	■ £430 million p.a.
	Maintaining genetic diversity	Production function method	■ No values currently available
	Bioprospecting	Production function method	■ No values currently available
22.3.3.1	Biodiversity: non-use values	Stated preference [‡]	■ Terrestrial biodiversity: £540 million to £1,262 million p.a. (mid-range estimate £845 million p.a.) ■ Inland wetlands: £273 million p.a. (marginal value = £304/ha p.a.) ■ Coastal wetlands: £1,275 million p.a. (marginal value = £1,866/ha p.a.) ■ Marine biodiversity: £1,714 million p.a.
22.3.3.2	Biodiversity: non-use values	Revealed preferences (legacy values)	■ £89.7 million p.a. [§]

¹³⁸ Of course, such analyses have to be aware of the danger of incremental losses—hence our stress on the need for understanding of thresholds and their consequences for resource sustainability. However, it is also true that an economic marginal analysis which ensures no net loss of environmental stocks must de facto be sustainable.

¹³⁹ As noted before, there is a role for non-monetary assessment of those goods which cannot be robustly valued through economic analyses.

Table 22.27 continued. Summary of UK NEA ecosystem service valuations.

22.3.4	Timber production	Market prices	<ul style="list-style-type: none"> ■ 8 million green tonnes p.a. @ £12/tonne = £96 million p.a. ■ Softwood production = £66/ha; hardwood production = £7 to £25/ha. No allowance made for manufactured capital inputs.
22.3.5.1	Carbon storage and GHG flux: Marine and Coastal Margins	DECC values	<ul style="list-style-type: none"> ■ Marginal (and total) values for coastal margin carbon storage (sand dune marginal sequestration value = £32 to £241/ha p.a.; saltmarsh marginal sequestration value = £61 to £622/ha p.a.). UK emissions from all lost coastal margins rise by £82 million/year by 2060 (mainly due to increase in DECC carbon storage value).
22.3.5.2	Carbon storage and GHG in Marine and Coastal Margins	DECC values	<ul style="list-style-type: none"> ■ Carbon storage in marine habitats potentially substantial but unquantified.
22.3.6	Water quality and quantity	Market prices, cost savings and stated preferences	<ul style="list-style-type: none"> ■ Water quality improvements would lead to some cost reductions in the costs of potable water supplies although commercial confidentiality means that the scale of these benefits is unclear. ■ The costs associated with changing agricultural land use to reduce nutrient loadings into rivers are substantially smaller than the benefits which such changes would bring. However, the former costs are concentrated within rural communities while benefits are distributed across a mainly urban society. ■ Water quality benefits of inland wetlands approximately £290/ha p.a.; coastal wetlands approximately £1,790/ha p.a. Total value up to £1.5 billion p.a. ■ Potential benefits of improvements to river water quality up to £1.1 billion p.a. Average benefits are £15.6/km, £18.6/km and £34.2/km for improvements that lift water quality from low to medium, from medium to high and from low to high respectively. ■ Climate change losses upon UK water availability are estimated at £350–490 million p.a.
22.3.7	Flood protection: inland	Market priced cost savings	<ul style="list-style-type: none"> ■ Climate change induced increases in flooding costs range up to £23 billion p.a. depending upon strategy. ■ Marginal value of flood defence from wetlands = £407/ha p.a.
22.3.8	Flood protection: coastal	Stated preference	<ul style="list-style-type: none"> ■ Marginal value of coastal flood protection by wetlands £2,498/ha p.a. Total value up to £1.5 billion p.a.
22.3.9	Pollution remediation	n/a	<ul style="list-style-type: none"> ■ No valuations currently available.
22.3.10	Energy and raw materials	Market prices	<ul style="list-style-type: none"> ■ Fossil fuels currently meet 90% of UK energy demand. Market price £112 billion p.a. (of which £35 billion tax and duties). Renewables meet 3% of UK energy demand and 7% of electricity generation (nuclear power = 17%). ■ Marine-based biotic raw materials = £95 million p.a. UK aggregates industry worth £4.8 billion p.a. of which up to £114 million p.a. comes from the marine environment.
22.3.11	Employment	n/a	<ul style="list-style-type: none"> ■ Economic benefits unquantified. Potentially substantial cultural and social cohesion benefits.
22.3.12	Game and associated landscape values	Market prices	<ul style="list-style-type: none"> ■ Woodland game revenues up to £3/ha p.a. Thought to be higher for Scottish sporting estates.
Section	Good	Valuation method*	Valuations
22.3.13	Amenity value of the climate	Revealed preference and life satisfaction	<ul style="list-style-type: none"> ■ £21 billion p.a. to £69 billion p.a.
22.3.14	Amenity value of nature	Hedonic pricing, stated preference	<ul style="list-style-type: none"> ■ Significant positive effects on house prices from increases in local greenspace, rivers and freshwater, wetlands, woodland, farmland, National Parks, National Trust sites. High environmental amenity valued at around £2,000 p.a. per household. Geographical distribution of environmental values mapped for England. ■ Marginal amenity value of inland wetlands = £230/ha/yr; coastal wetlands = £1,400/ha p.a. Total wetland amenity value up to £1.3 billion p.a.

Table 22.27 continued. Summary of UK NEA ecosystem service valuations.

22.3.15	Education and environmental knowledge	Wage rate assessments, travel and time cost valuations	<ul style="list-style-type: none"> ■ Environmental knowledge embodied in higher qualifications valued at £2.1 billion p.a. ■ School trips to just 50 nature reserves valued at £1.3 million p.a.
22.3.16	Health	Stated preference	<ul style="list-style-type: none"> ■ Value of health benefits of green exercise not quantified. Tentative assessments of health changes arising from a variety of contacts with nature provided, ranging from around £10/person p.a. for a marginal increase in woodland within 1 km of the person's home to around £300/person p.a. for views of greenspace from the person's home. ■ Climate change is likely to have health impacts and on balance, the direct effects are likely to be positive (the reduction in cold-related deaths outweighs the increase in heat-related deaths). This ignores the indirect effects arising due to climate-induced global economic change.
22.3.17	Agricultural food production	Production function method	<ul style="list-style-type: none"> ■ Land use model developed from data from the 1960s to the present day. Relates land use and farm gross margin (£/ha) to a variety of ecosystem services and manufactured inputs. Distributions of marginal values mapped at a 2 km square resolution (see discussions in Section 22.4.1 of integrated valuations). Example valuations examine changes in climate ecosystem services induced by climate change from the present day to 2060. Most values vary from (mainly lowland) losses of £50/ha p.a. to (mainly upland) gains of £75/ha p.a.
22.3.18	Carbon storage and annual GHG emissions: terrestrial	Department of Energy Climate Change (DECC) and Stern report values	<ul style="list-style-type: none"> ■ Mapped distributions of the marginal value (£/ha p.a.) of changes in carbon dioxide equivalent (CO₂e) emissions under each of the agricultural land use change scenarios (from Section 22.3.1). Emissions rise in uplands and fall in lowland areas. Monetised using DECC and Stern carbon storage valuations. ■ UK-wide valuations for agricultural greenhouse gas (GHG) emissions (i.e. costs) estimated for all of the UK ranging from £4,286 million p.a. in 2004 to £13,409 million p.a. in 2060 (both calculated using Stern values for the UKCIP high emissions scenario). ■ Specific examples: Within the above costs, emissions from peatlands are estimated at £130 million p.a. Total value of net carbon sequestered (i.e. benefits) annually by UK woodlands = £680 million (marginal value = £239/ha p.a.)
22.3.19	Biodiversity: non-use values	Cost-effective provision of biodiversity indicator species [§]	<ul style="list-style-type: none"> ■ Maps of the change in bird diversity under each of the agricultural land use change scenarios.
22.3.20	Recreation and tourism	Travel and time cost valuations, stated preferences, meta-analysis	<ul style="list-style-type: none"> ■ English recreation: 2,858 million visits p.a. with direct expenditure of £20.4 billion p.a. (UK-wide values may exceed £30 billion p.a. In addition, foreign visitors spend £16 billion p.a. in the UK). Economic valuation shows that physically identical nature recreation sites can generate values of between £1,000 p.a. and £65,000 p.a. depending upon location.
22.3.21	Urban greenspace amenity	Meta-analysis of hedonic pricing, stated preference and expert assessments	<ul style="list-style-type: none"> ■ Valuations vary from losses of £1.9 billion p.a. to gains of £2.3 billion p.a. depending on policy scenario.

* Where no studies are currently available, this column refers to potentially applicable methods.

† See caveats and cited texts in the methodological summary regarding caveats surrounding the use of market prices in economic analyses.

‡ See Section 22.3.2.1 for caveats and reservations regarding stated preference estimates of non-use biodiversity values.

§ Based upon leagues to just the top five environmental charities: The National Trust; the Royal Society for the Protection of Birds (RSPB); WWF UK; The Woodland Trust; National Trust for Scotland. See caveats in Section 22.3.3.2; this is very much a lower bound estimate of non-use value.

§ As discussed in the Section 22.3.19, this is not a valuation method. Rather it provides estimates of the cost of efficient provision of desirable biodiversity outcomes.

22.4.2 Final Conclusions

We do not pretend that the list of goods assessed in this chapter is complete or that those assessments themselves are definitive. Furthermore, time constraints have precluded more than a cursory consideration of the uncertainties surrounding many of the assessments presented here. However, we would suggest that the economic analysis presented here provides at least a useful initial step for better informing the way in which decisions are made in the UK (and indeed internationally). We believe that the principles and direction which the present analysis adopts are a contribution to the longer term aim of ensuring the sustainability of human society through a recognition of the need to live within our means and work with, rather than against, nature. Given the very large financial and economic values (stock and flows) that are provided by healthy functioning ecosystems, future economic development can best be sustained through policy directed at the safeguarding of the natural capital that ecosystems represent. Proper long-term management of ecosystems can lay the foundations for a thriving 'green' economy and an improving level of general well-being in society as social capital stocks are nurtured in parallel.

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Appendix 22.1 The Economic Case for the Sustainable Management and Use of Natural Capital

Natural capital and the various services it provides contribute to economic activity and human welfare in two general ways:

- Directly as an input to production; for example, through the provision of fossil fuels, minerals and the contribution of sectors such as farming, forestry and fishing to economic activity.
- Indirectly through its productivity-enhancing effects on other factors of production; for example, through better human health outcomes from improved local air quality and provision of greenspaces; by providing a sink for waste generated as a by-product of economic activity; and the mitigation of some of the risks posed by climate change such as from flooding.

Some of the contributions of natural capital have a market value, and are at least partly reflected in measures of economic activity such as Gross Domestic Product (GDP). However, while GDP and similar measures reflect the value of goods and services provided through the market, they exclude others that are not provided through the market, but nevertheless facilitate economic activity and contribute to overall human welfare.

Natural capital also contributes to wider societal well-being; for example, through the non-material benefits people obtain from ecosystems such as from aesthetic enjoyment and recreational services. Well-being is a multidimensional concept including a range of objective and subjective factors. For example, the Commission on the Measurement of Economic Performance and Social Progress (Stiglitz *et al.* 2009) identified the key dimensions of well-being to include:

- i) material living standards (income, consumption and wealth);
- ii) health;
- iii) education;
- iv) personal activities including work;
- v) political voice and governance;
- vi) social connections and relationships;
- vii) environment (present and future conditions);
- viii) and insecurity, of an economic as well as a physical nature.

The Commission notes that “all these dimensions shape people’s well-being, and yet many of them are missed by conventional income measures.” (Stiglitz *et al.* 2009; p15). Thus, the contribution of the natural environment to society’s overall well-being needs to be considered alongside its contribution to economic growth and welfare.

Securing future economic prosperity and well-being requires ensuring the availability of natural capital, and the services it provides, into the future. However, markets alone will be unable to deliver sustainable management and use of natural capital. The value of the goods and services provided by natural capital are, at best, imperfectly priced into economic decisions to produce and consume. This

leads to overuse of such resources and their depletion and/or degradation beyond economically efficient levels. For example, without policy intervention, a firm releasing pollutants into the atmosphere does not pay the full cost to society of the negative health effects resulting from its actions. This leads to higher levels of production (and pollution) than if the firm faced the full higher cost of such resource use.

Correcting for this ‘market failure’ will improve the overall (allocative) efficiency of the economy and ensure that environmental goods and services are not consumed beyond their economically efficient level. However, the sustainable management and use of natural capital requires consideration of some additional attributes unique to natural capital, namely:

- Finite limits or critical thresholds beyond which non-linear and/or irreversible changes may occur; for example, ‘source limits’ in fish stocks and top soil where breaching the threshold could lead to a change or collapse in the ecosystem.
- Services provided by natural capital may not be readily substitutable by other types of capital; for example, technology and produced capital could not easily substitute for the ecosystem services provided by the ozone layer.

Declining levels of some natural assets can be consistent with environmentally sustainable growth as long as adequate investments are made in other types of capital. However, to the extent that the services provided by natural assets have critical thresholds, or cannot be substituted for by other goods and services, maintaining a minimum stock of these assets needs to be considered.

Policy action is required in order to ensure that natural capital is managed and used sustainably, both in terms of ensuring the efficient level of natural capital and of protecting key natural assets. Environmental policy of this nature should achieve its environmental objectives without significant adverse macroeconomic impacts (acknowledging that some sectors could disproportionately benefit or lose out in the process), particularly if implemented through cost-effective interventions (or package of interventions) and using market instruments wherever possible.

Indeed, fears surrounding the macroeconomic impacts resulting from a strengthening of environmental policy are not borne out by the economics literature. A large number of studies have examined the impact of environmental regulation costs on different aspects of industrial competitiveness (for instance, on trade and foreign direct investment patterns and on productivity and employment levels) for a range of economies including the UK (Gray & Shadbegian 2003; Ederington *et al.* 2005; Cole & Elliott 2007; Cole *et al.* 2010; Ekins *et al.* in press). These studies have generally found no, or only very limited, evidence that environmental regulation costs adversely influence

industrial competitiveness. While these studies tend to focus on the impact on competitiveness of pollution abatement costs, rather than the costs of correctly pricing natural capital more broadly, their findings provide some indication of the potential impacts of these latter costs.

Possible reasons for the lack of evidence of macroeconomic impacts associated with environmental regulations include: the fact that the most pollution-intensive firms tend to be physically capital intensive and, hence, less suited to relocation to (or displacement by) low regulation, labour-intensive economies; the fact that pollution regulation costs form only a small proportion of a firm's total costs even within pollution intensive firms; and the possibility that a strengthening of environmental regulations can actually stimulate innovation in firms, which may at least partially offset the cost of complying with these regulations (the so-called Porter hypothesis; Porter & van der Linde 1995). Related to this latter point, a recent report

by the United Nations Environment Programme (UNEP) provides further evidence of the potential economic benefits of 'green investment' by indicating that investment of this nature can enhance economic growth by stimulating certain industries and, crucially, by reducing environmental risks (UNEP 2011). It is obviously vital that the costs of failing to reduce such risks are taken into account when quantifying the overall economic costs of environmental policy. If natural capital is not adequately protected its depletion and degradation is likely to have negative effects on growth and welfare through the loss of inputs to production (such as reduced availability of clean water), the loss of assets which contribute to resilience of business and communities (such as reduced flood risk management), or through negative effects on well-being (such as the loss of biodiversity and associated recreation services). The management of environmental risks seem likely to be key to maximising both well-being and economic growth over the long-term.