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Recent advances in the valuation of ecosystem services and biodiversity

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Abstract Recent interest in the economics of biodiversity and wider ecosystem services has been given empirical expression through a focus upon economic valuation. This emphasis has been prompted by a growing recognition that the benefits and opportunity costs associated with such services are frequently given cursory consideration in policy analyses or even completely ignored. The valuation of biodiversity and ecosystem services is therefore increasingly seen as a crucial element of robust decision making and this has been reflected in a growing body of related research. We provide a critical review of some of this research, considering the valuation methods applied to date and focussing upon their limitations in respect to certain categories of ecosystem services (particularly cultural services) and the applicability of the extant literature to new settings. Substantial questions also remain at the interface of natural science and economics and we consider the potential contribution of the conceptualization of ecosystems as assets as a response to this challenge. As part of this review we also highlight the role which large scale 'ecosystem assessments' have played as an impetus to extending the valuation evidence base and the way in which frameworks and assessments of how ecosystems contribute to human wellbeing might be translated into policy thinking and decision analyses.

Key words: valuation, ecosystem services, biodiversity

JEL classification: Q51, Q57

I. Introduction

From humble origins in the early post-war period (e.g. Hotelling, 1949), the literature regarding the valuation of preferences for non-market costs and benefits has grown, initially slowly, but more recently at an almost exponential rate. Nowhere is this more true than in the field of environmental and resource economics, where the focus of empirical work is on public goods for which market prices are either poor reflections

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of underlying values or entirely missing. However, the application of economic valuation techniques to the complexities of the natural environment raises a number of significant challenges. Perhaps most fundamental is the need to ensure that such applications are based upon a sound foundation of natural science and, indeed, there is a highly cogent case to be made that all such applications necessarily require interdisciplinary collaboration between, at a minimum, the natural sciences and economics (arguably extending to a much wider fusion of disciplines). This requirement for interdisciplinarity is given a conceptual framework within the so-called 'ecosystem service' approach to decision-making. While typically characterized as emanating from the natural sciences, the approach is highly compatible with economic analysis, as it emphasizes the role of ecosystems and biodiversity in providing services which, in turn, either support production or are direct contributors to wellbeing. Ecosystem services are therefore defined as contributors to anthropocentric values and, while the natural sciences provide an understanding of the former, it is economics which is well placed to assess the latter. Economic valuation, in particular, becomes an essential element of the ecosystems-service approach to decision analysis.

While the term 'ecosystem services' is relatively recent, only being popularized in the wake of the Millennium Ecosystem Assessment (MA, 2005), environmental economists have been applying non-market valuation techniques to such services for many years (see, for example, Ruitenbeek, 1989; Adamowicz *et al*., 1994). Understanding the economic value of ecosystems and biodiversity is important for a number of reasons. One of these is undoubtedly the perceived persuasiveness of economic language. That is, conveying what it is that the natural world provides us with in monetary terms is seen as a powerful means of communicating the importance of conservation to a wider (and perhaps previously unreceptive) audience. For example, Bateman *et al*. (2011*b*) estimate that, in the United Kingdom, ecosystem services help contribute to 3 billion outdoor recreational visits annually, with the social value of the output created by these trips likely to be more than £10 billion. Gallai *et al*. (2009) calculate the global value of the services provided by insect pollinators to be about $190 billion (in 2005), just in terms of the benefits arising from pollination of crops for (direct) human consumption.

But beneath the rhetoric there is genuine substance in that these data can also be used to guide policy thinking and decisions. In the case, for example, of the recreational value of UK ecosystems, Bateman *et al*. (2011*b*) also show how location (of these sites) matters. A specific and moderate-sized nature recreation site, for example, might generate values of between £1,000 and £65,000 per annum, depending solely on where it is located. The critical determinant of this range is, perhaps not surprisingly, proximity to significant conurbations. Put another way, woodlands in the 'right' place (i.e. relatively close to potential visiting populations) are likely to give rise to higher social values (other things being equal), an insight of particular importance if policy-makers are contemplating new investments in these nature sites.

More generally, the key insight in explicitly placing a value on nature is that it redresses a fundamental imbalance whereby this value is—all too frequently—grossly misjudged or just plain ignored in private and (much of) social decision-making. And while debates about the intrinsic value of nature remain relevant, demonstrating that nature has significant instrumental value for human livelihoods or human wellbeing more broadly is then a crucial practical step in developing policy actions that address current and projected rates of ecosystem destruction and biodiversity loss. One much cited example, in this respect, is Barbier (2007). That study estimates the ecological value of mangroves in

Thailand—in terms of providing fuelwood, a habitat that supplies fisheries, and storm water attenuation (which reduces the risks of coastal flooding)—in order to compare those findings with the returns from the competing land-use activity of shrimp farming. Thus, private profits under these two different uses are $584 and $1,220 per hectare, respectively, giving, on the face of it, a clear (financial) case for mangrove conversion. However, social cost–benefit analysis reveals another story in that a representative hectare of conserved mangrove is shown to generate a social value of $12,392.[1]

These benefits that nature provides might even spill over to human populations living in countries other than where, say, an ecosystem is sited. In a study of Costa Rica's tropical forests, Bulte *et al*. (2002) conclude that the optimal area of forest land is more than twice as large as the actual (1998) area once the value of domestic externalities provided by this forest is taken into account. Bringing the value of global externalities (accruing to those outside of the country but provided by Costa Rica's forests) into this reckoning, results in the optimal forest cover being calculated to increase by a further 20 per cent. In monetary terms, the authors estimate that the present value of the loss of wellbeing arising from Costa Rica's forest cover falling short of this (global) optimum amounts to about $1.2 billion. Of course, the economic approach may not always provide us with the answer that ecosystems or biodiversity should be protected (and thus indicates the pitfall for those who see only the rhetorical worth in economic arguments). Nevertheless, and however the question is posed, determining how much of nature needs to be conserved is likely to require a significant effort to understand its value in economic terms as well as the (opportunity) costs of its conservation.

Any paper that seeks now to take stock of recent efforts to value ecosystem services and biodiversity has the advantage of following a number of comprehensive reviews such as Kumar (2010), Bateman *et al*. (2011*b*), Ten Brink (2011) and specific reviews of, for example, forests and coastal/marine ecosystems (see, respectively, Ferraro *et al*., 2012, and Barbier, 2012). In the current paper, while we will inevitably draw on these important contributions, we also hope also to add to insights about the direction of future endeavour in this field. In the section that immediately follows we briefly review possible classifications of ecosystem services but discuss in addition the more recent—but hugely important—development that traces further links to the underlying ecological assets that give rise to these services in the first place as well as the role of biodiversity. Section III outlines the key valuation methods and considers, in particular, gaps in the empirical record and the scope for filling these gaps. Section IV sets this consideration of economic valuation in relation to the evidence base needed to inform broader ecosystem assessments and policy decisions. Section V concludes.

II. A framework for valuing ecosystem services and biodiversity

In the past few years, interest in the problem of ecosystem and biodiversity decline has grown dramatically, among academics and policy-makers alike. Much of this recent

[1] It is, however, important to note that this is the value of a 'representative' hectare rather than *all* or the marginal unit. There is likely to be significant heterogeneity in terms of the services provided by mangroves at different locations (Barbier *et al*., 2008).

attention can be traced to the MA (2005) which made clear the scale of the challenge at hand in its identification of persistent and growing threats to ecosystems around the world. In addition, the focal valuation message in the Stern Review on Climate Change (Stern, 2007) appears not to have been lost on decision-makers within the domain of conservation policy. Assessments including the G-8/EU initiated 'TEEB Review' (The Economics of Ecosystems and Biodiversity, TEEB, 2010) and the UK National Ecosystem Assessment (NEA, 2011)[2] can be viewed as an attempt to generate a correspondingly increased awareness and strong policy response for biodiversity and ecosystem services, as well as a concerted effort to build on the momentum and insights generated by the MA.

Importantly, the MA had the effect of broadening the focus of concern from biodiversity loss to cover, in addition, the loss of ecosystem services, with the critical emphasis of the latter on 'the benefits people obtain from ecosystems' (MA, 2005, p. 53). From an economic perspective, ecosystem services are simply those contributions of the natural world which generate goods which people value. The term 'goods' is, as elsewhere in environmental economics, construed widely to mean physical products and less tangible outputs. This includes services which generate use values and non-use goods which are valued purely for their continued existence.

This now conventional understanding (within environmental economics) of the total economic value of some 'good' has been intertwined with a more nuanced understanding of the specific services that ecosystems provide. There are a number of variations on these classifications. Common to almost all is a distinction between: provisioning services; cultural services; and regulating services. The former two services nicely capture some elements of the previous distinction between use and non-use. Provisioning services, for example, are typically physical products such as food and natural materials provided by nature. Cultural services, by contrast, describe the experiences that people enjoy as a result of interactions with nature (e.g. recreation), as well as more intangible pleasures arising from knowledge about the existence of nature or its spiritual value.

Further classifications of ecosystem services do exist. Kumar (2010), for example, adds habitat services in recognition of the role that ecosystems provide in protecting 'gene pools' as well as crucial sets of interlinking habitats for migratory species. MA (2005) also emphasized the supporting services of ecosystems as the natural processes that underpin those services of provision, culture, and regulation. These services, such as nutrient cycling, thus provide a further intermediate tier to ecological production and, indeed, it is has since become more common to see these functions subsumed under the 'regulating services' heading (e.g. Kumar, 2010). Other classifications such as Heal *et al*. (2005) and de Groot *et al*. (2002) have focused more specifically on habitat services and regulating services. While this emphasis is partial, it encapsulates a key distinctive element of the effort to understand the economics of ecosystems. This likens the enjoyment of (final) ecosystem services to a process of (natural) production whereby critical inputs are, for example, regulating services. As an illustration, it is these

[2] The UK NEA involved a team of over 160 natural scientists assembled to quantify the status of ecosystem processes and the final ecosystem services they generate across the UK, looking at individual habitats classifications (e.g. wetlands and woodlands) as well as ecosystems services across these classifications. In addition, an economics team complemented this work and its structure with the added emphasis on the value of habitats and ecosystems services under investigation.

services—by, for example, regulating water flow (and the quality of that water) and the supply of insect pollinators—that contribute ultimately to the production of agricultural provisioning services (Goulder and Kennedy, 2011). Valuing ecosystem services has often focused on the end output by asking what is the final service that ultimately benefits people. Clearly, knowledge of what ecosystems provide as final goods and services that we consume is important. Yet it is equally crucial that we understand the way in which intermediate tiers of production contribute to this final output.

In many ecosystem classifications (including those which have been expanded to conceptualize ecosystems as assets), there appears to be no explicit place for the value of *biodiversity*. Indeed, a significant anxiety about recent ecosystem assessments is that the emphasis upon ecosystem services might ironically lead to the omission of the vital role which biodiversity plays in both the delivery of those services and as a source of value in itself. Mace *et al*. (2012) provide clarification of the issue, noting that biodiversity appears at three distinct points within the ecosystem service framework.

First, as discussed in detail by Elmqvist *et al*. (2010), biodiversity acts as a supporting service underpinning the delivery of what Fisher *et al*. (2009) term final ecosystem services. So, for example, soil biodiversity enhances farmland fertility, which in turn determines production of a good (here food). In fact, such functions provided by biodiversity have been likened by, for example, Pascual *et al*. (2010) to a form of insurance (following from earlier contributions such as Gren *et al*., 1994). On this view, a more diverse (ecosystem asset) portfolio has a distinct value in terms of maintaining resilience: that is, the capacity of a system to persist, in some state, in the face of shocks and stresses that it might experience (Perrings, 2006; Mäler *et al*., 2009).

Second, biodiversity acts as a final ecosystem service itself. For example, pollinator biodiversity directly enhances agricultural production. Third, certain aspects of biodiversity, such as the continued existence of iconic species such as the polar bear, itself constitutes a good (i.e. a direct source of wellbeing). These diverse roles suggest that attempts to value biodiversity will be challenging. It is to these challenges, and those entailed in valuing ecosystem services to which we now turn.

As reflected in our discussion thus far, much of the existing terminology in ecosystem valuation and biodiversity conservation has focused on *services*: that is, some flow of a benefit arising perhaps from the consumption of a good or broader amenity. Of course, policy interventions such as investments in ecosystem protection (or enhancement) will typically boost the flow of these services over time, thereby introducing a dynamic element into any economic analysis. Moreover, when ecosystems are perturbed by some change (be it a shift in land use or a degradation in state) the effect on wellbeing will similarly have an intertemporal dimension (e.g. Mäler, 2008; Dasgupta, 2009). Put this way, what we need to think about is the underlying ecosystem or biodiversity asset and, in particular, the changes in asset value that occur as a result of human interventions (be these positive or negative, deliberate or otherwise). Broadly speaking, what needs to be assessed here is the potential change in our *future* prospects, given what is happening to ecosystems and biodiversity now. Thinking about ecosystems as assets (as opposed to emphasizing only current services) is in its relative infancy but is becoming more prominent. In the view of Heal (2007), this brings the study of the economics of the natural world into line with other areas of the discipline. Barbier (2009) has shown how this extension of the ecosystems analytical framework results in a more explicit conceptual understanding of ecosystems as complex assets giving rise to multidimensional services.

Thinking explicitly about ecosystems or biodiversity as assets, thus, opens up a further range of valuation issues. That is, given that a change in asset value is equal to the difference in the present value of future services before and after the change, we need to consider how these future services are to be valued and, moreover, discounted. Clearly, neither of the implied measurement challenges is unique to valuing ecosystem assets. Questions about asset valuation (as well as answers to those questions) pervade many other areas of economics. On-going efforts to measure ecosystem and biological assets can usefully learn much from these existing insights. For example, the debate that has ensued since the Stern Review (e.g. Weitzman, 2007; Dietz and Stern, 2008) has thrown new light on the choice of the *social* discount rate in the context of climate change. A recent review by Gowdy *et al.* (2010) in the context of ecosystems and biodiversity illustrates that the issues there are likely to be no less controversial given the long-term characteristic of services provided by nature. However, to date this has received far less attention in this context (see, for example, Mäler *et al.*, 2009, for a brief discussion in the context of ecosystem accounting).

Discussion of ecosystems and biodiversity has also focused on the ability of valuation methods, for practical purposes, to deliver on addressing concerns about the complexity of ecosystems and the empirical relationship between asset stocks, the flow of services, and the way in which these services are valued at different stock levels (Pascual *et al.*, 2010). This is a point that can be traced back as least as far as Krutilla and Fisher (1974), but has been made more recently, and often with ecological wealth in mind, for the case of assets for which there are limited substitution possibilities (in terms of the wellbeing that they ultimately provide). That is, if the (marginal) value of the service (i.e. its relative price) is likely to increase all the more rapidly as the asset is increasingly degraded or converted.

Gerlagh and van der Zwann (2002), for example, consider the case where these substitution possibilities are a function of the asset stock itself. That is, when a resource such as an ecosystem is relatively abundant, losses in that asset 'do not matter' in the sense that this source of wellbeing could be easily replaced with something else and people essentially would be no worse off. However, after some threshold, substitution possibilities diminish rapidly. In other words, continued loss of the natural asset—beyond this critical point—increasingly cannot be compensated and, on the contrary, increases the prospect of significantly raised adverse impacts on future wellbeing. Hoel and Sterner (2007) and Sterner and Persson (2008) have indicated some initial steps towards a practical exposition of this thinking (in the context of valuing the damage arising from climate change). However, this empirical progress requires that a number of assumptions be made: most notably, a judgement needs to be arrived at about the 'elasticity of substitution' (between some natural asset and other productive stocks). Further investigation of these issues, within the ecosystem context, is urgently needed (although see Barbier (2009) for a discussion on modelling the likelihood of collapse of ecosystem assets and Farley (2008) on the broad principles that might guide future thinking about valuation as ecosystem assets become increasingly scarce and, in some cases, stocks approach critical levels).

Finally, it also worth noting that one particular approach to thinking about ecosystems as assets addresses a possibly critical issue with regards to diversity (discussed initially above) by treating 'ecological resilience' as a stock (Mäler, 2008; Mäler *et al.*, 2009). In other words, the ability of an ecosystem to withstand stresses and shocks (and to continue to provide services) has a distinct asset value which can be degraded (or enhanced) over time. Walker *et al*. (2010) look at the value of this resilience to agriculture in south-east Australia in terms of maintaining a saline-free water table (salinity problems are caused mainly through farmers cutting down trees to expand agriculture). Here agricultural expansion represents a driver depleting the stock of non-salinated soils (measured as the depth of soils for which saline intrusion is not a problem). As this depletion driver is increased, so the stock of ecological resilience falls. As the depleting process itself may generate benefits (here agricultural produce), there is a trade-off to be assessed between the benefits of depletion and the fact that losses of resilience may need to be reversed if stocks fall below some threshold level. Valuing this stock, unfortunately, is a relatively complex business and extending this approach beyond largely illustrative examples is in its infancy at best. Indeed, Walker *et al*. are themselves extremely guarded about using their empirical example in the 'real world', owing largely to apparent uncertainties about the scientific and economic data. Nevertheless, such developments represent an important addition to existing ecosystem service valuation work. It is to this existing body of evidence that we now turn.

III. Valuing ecosystem services: lessons and directions

The process of uncovering the true value of goods and using these data to ensure that decisions contribute to improving human wellbeing is a defining rationale for economic analysis. A number of recent comprehensive reviews make clear the proliferation of methods—and applications of those methods—to assess the value of ecosystem services and biodiversity (see, for example, US SAB, 2009; Pascual *et al*., 2010; Bateman *et al*., 2011*b*; Kareiva *et al*., 2011). These assessments have been important for revealing, on the one hand, what is known about ecosystem and biodiversity valuation and, on the other hand, in identifying what we still need to learn. In what follows, we can only hope to provide a (partial) synopsis of these developments but, in doing so, we alight on a number of issues that strike us as noteworthy.

(i) Economic valuation methods: a synopsis

There are many comprehensive reviews of economic valuation methods more generally (e.g. Champ *et al*., 2003; Freeman, 2003; Pearce *et al*., 2006; Hanley and Barbier, 2009). Table 1 provides a brief overview of the key approaches. What is important to note here is that *all* of these methods have been used in the ecosystems context. In large part this breadth of methods reflects, in turn, the diversity of services that practitioners have sought to value, rather than variety for its own sake.

The starting point for thinking about the valuation of ecosystem services is that such assessments rely upon standard economic theory but with an underpinning of the natural sciences (Daily, 1997; Pagiola *et al*., 2004; MA, 2005; Heal *et al*., 2005; Barbier, 2007; Sukhdev, 2008). Whether this valuation can be based on market prices or whether we must look to evidence from non-market behaviour (be this actual or intended) depends on the characteristics of the ecosystem good or service in question. In some cases, valuation might begin with market prices. For example, provisioning

Table 1: Summary of economic valuation methods used in ecosystem service valuation

Valuation method	Description	Typical applications to ecosystem services and biodiversity
Adjusted market prices	Using market prices adjusted for any distortions (e.g. taxes, subsidies, non-competitive practices)	Crops, livestock, woodland
Production function methods	Estimation of an ecological production function where the ecosystem service is modelled as an input to the production process and is valued through its effect on the output	Maintenance of beneficial species, maintenance of agricultural productivity, flood protection
Revealed preference methods	Examining actual expenditures made on market goods related to ecosystem services. When market goods are substitutes, avertive behaviour or mitigating expenditure approaches can be used (e.g. expenditures to avoid damage, such as buying bottled water or installing double glazing). Travel cost methods can be used when market goods are complements (e.g. travel costs for recreation). When the ecosystem service is a characteristic of the market good, hedonic price methods can be used (e.g. looking at the impact of noise or amount of green space on property prices)	Water quality, peace and quiet, recreation, amenity benefits
Stated preference methods	Using surveys to elicit willingness to pay for an environmental change (contingent valuation) or to ask individuals to make choices between different levels of environmental goods at different prices to reveal their willingness to pay (choice modelling)	Water quality, species conservation, air quality, non-use values

services are frequently market goods or near-market goods with close (market) substitutes. It follows, therefore, that market-based valuation has been prominent in such contexts, although perhaps these observed prices have needed to be adjusted for distortions (Table 1). However, the provisioning service is itself typically determined by some underlying service provided by an ecosystem process. Thus while the valuation of this final output is relatively straightforward, the analytical heavy-lifting is often done through the specification and estimation of an ecological production function. In other words, ecosystem services are frequently valued as a productive input (see Freeman, 2003; Barbier, 2007; and Hanley and Barbier, 2009). In this approach, an attempt must be made to isolate and uncover the value of ecosystems services from the perspective of their effect on some observed level of output (Table 1). This approach can be applied to a range of market (consumption) goods but has also been used for valuing regulating and 'protection' goods (where examples of the latter include flooding and extreme weather protection).

In other cases, however, the value that people place on ecosystem services is not adequately reflected in market prices, if at all. In such cases, non-market valuation techniques must be employed and applied to some ecological end-point which itself may have been estimated following some application of a production function. Revealed preference methods value non-market environmental goods by examining the consumption of related market-priced private goods. A number of variants of the revealed preference approach exist, depending on whether the environmental good and the related market good are complements, substitutes, or one is an attribute of the other (Table 1). In the first case, economists make use of the 'weak complementarity' concept introduced by Mäler (1974) to examine how much individuals are prepared to spend on a private good in order to enjoy the environmental good, thereby revealing the value of the latter. For example, the travel cost method examines the expenditure and time that individuals are prepared to give up to visit natural areas for recreation. In cases of substitutability between goods, approaches such as avertive behaviour or mitigating expenditures to avoid damages can be used, such as buying bottled water to avoid drinking contaminated water. Finally, the hedonic property price method assumes that we can look at the housing market to infer the implicit value of the underlying characteristics of domestic properties, be these structural, locational/with regard to accessibility, in terms of neighbourhood, or environmental (Rosen, 1974). It can be used, for example, to examine the premium which people are prepared to pay in order to purchase houses in areas with greater proximity to green spaces or habitat types (Gibbons *et al.*, 2011).

While revealed preference (RP) methods estimate original values by looking at *actual* behaviour, eliciting values by looking at *intended* behaviour is the province of stated preference (SP) methods. This is an umbrella term for a range of survey-based methods that use constructed or hypothetical markets to elicit preferences for specified changes in provision of environmental services (Table 1). By far the most widely applied SP technique is the contingent valuation method (see, for example, Alberini and Kahn, 2006).[3] However, in recent years, choice modelling has become increasingly popular. In this variant, respondents are required to choose their most preferred out of a (possibly relatively large) set of alternative policy or provision options offered at different prices and their willingness to pay is revealed indirectly through their choices (see, for example, Hanley *et al.*, 2001; Kanninen, 2007).[4]

In theory, SP approaches should be applicable to a wide range of ecosystem services and can be used to measure future/predicted changes in those goods. Importantly, such methods are thought to be the only option available for estimating those services which are valued for 'non-use' purposes. In practice, SP methods are mostly 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, then elicited values may not provide a sound basis for decision analysis. Such problems are most likely to occur for goods of which individuals have little experience and poor understanding (Bateman *et al.*, 2008*a*,*b*; 2010). Therefore, while stated preferences may provide sound valuations for high experience, use-value goods, the further we move to consider indirect use and pure non-use values, the more likely we are to encounter problems. Paradoxically, then, where SP techniques are most useful is also where they have the potential to be less effective.

A number of solutions have been proposed for the problem of valuing low experience goods. Christie *et al.* (2006) have proposed the use of intensive valuation workshops where participants learn about the environmental services being valued. However, the

[3] See, mostly for a summary, Carson's (2011) bibliography of published and unpublished CV studies from around the world.

[4] A number of studies combine RP and SP approaches in order to enhance the respective strengths of these data and minimize limitations (see, for example, Adamowicz *et al.*, 1994).

techniques involved are almost inevitably prone to reliance upon small unrepresentative samples which, after such intensive experiences, cannot be taken as reflecting general preferences. So while offering useful insights about overcoming the low experience problem, it must be asked whether the cure is worse than the disease. Others have proposed and implemented extensions of conventional, individual based SP applications. Bateman *et al*. (2009), for example, use virtual reality software to convey images of landscape goods. This avoids the difficulties of conveying attributes of goods, such as landscape, in unfamiliar units, such as hectares. Results show a significant reduction in the rate of preference inconsistencies through the application of such techniques.

While significant strides can be made in filling out the ecosystem valuation matrix without recourse to what might be judged by some to be more 'problematic methods', crucial gaps remain in the empirical record. This issue seems particularly acute in the case of many types of cultural ecosystem services. As stated by Chan *et al.* (2011, p. 206), 'few classes of value have been more difficult to identify and measure than those concerned with the cultural and non-use dimensions of ecosystems'. Cultural ecosystem services include use-related values such as leisure and recreation, aesthetic and inspirational benefits, spiritual and religious benefits, community benefits, education and ecological knowledge, and physical and mental health. Difficulties arise as some of these cultural services may be bound up by non-use motivations[5] such as altruistic, bequest, and existence values (Krutilla, 1967). Moreover, some of these benefits are also difficult to identify separately. As things stand, there appears to be a generalized lack of knowledge and a specific dearth of monetary information about the contribution of cultural ecosystem services to wellbeing. In the following sections, therefore, we discuss some of the challenges with regards to the 'health' and 'non-use' values of ecosystems in particular.

(ii) Health values

Despite increased recognition that ecosystem services can have substantial effects on human health, both directly and indirectly (e.g. Ulrich, 1984; Takano *et al*., 2002; de Vries *et al*., 2003; Hartig *et al*., 2003; Bird, 2007; Mitchell and Popham, 2008; Myers and Patz, 2009; Osman, 2005), our knowledge on the complex relationships linking the biophysical attributes of ecosystems with the many aspects of human health remains limited (Kareiva *et al*., 2011).

Environmental quality and proximity to natural amenities is increasingly recognized as having substantial effects on physical and mental health, both directly and indirectly. Broadly, this could arise in a number of ways. Ecosystems provide many services that sustain human health (such as nutrition, regulation of vector-borne disease, or water purification). Also, natural settings could act as a catalyst for healthy behaviour,

[5] An existence value can be derived from the simple knowledge of the existence of the good or the service. In the context of the environment, individuals may place a value on the mere existence of species, natural environments, and other ecosystems. If an individual derives wellbeing from the knowledge that other people are benefiting from a particular environmental good or service, this can be termed altruistic value. Such values accrue during an individual's lifetime, but vicarious valuation can also occur intergenerationally. The effect on wellbeing of knowing that one's offspring, or other future generations, may enjoy an environmental good or service into the future, such as by a biodiversity-rich forest being conserved, is termed bequest value.

leading for example to increases in physical exercise, which affect both physical and mental health (Pretty *et al*., 2007; Barton and Pretty, 2010). Finally, 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).

While there is a large literature on health valuation, a crucial gap is in relation to the contribution of ecosystems to these improvements. Moreover, the statistical evidence for the health–ecosystem link is still to be established unequivocally. For example, on the link between physical exercise and availability of green spaces, the suspicion is that even if the physical health link can be more firmly established the value is possibly likely to be small given the availability of substitutes for this physical exercise. Hence, it is more likely to be the mental health benefit that is plausibly the more substantial of these two (bundled) health outcomes. Less is known with regards to valuation here although it might be the case that life satisfaction approaches linked to monetary valuation is a promising path to explore further (see, for example, MacKerron and Mourato, 2011). A final but no less important challenge is to know what values are for *changes in* ecosystem provision, whereas most work to date has examined the possible health benefits associated with *current* provision.

(iii) Non-use values

Environmental non-use values are often thought to be substantial. Critically, however, when and where these arise remains the subject of some discussion. Due to their intangible nature and disconnect from actual uses, the valuation of non-use benefits is complex. As a result, there appears to be no systematic body of evidence about non-use values and, importantly, little consensus about how the empirical record (such as it is) can be used for practical assessment in the context of project (and policy) appraisals or broader national-level ecosystem assessments. In the former, a particular concern might relate to whether a (change in a) non-use value relates to a specific and discrete proposal (or the provision of a service more generally). In the latter, a concern might be double-counting or erroneously assuming that the same (per household or individual) non-use value estimate applies to all of the parts rather than something more broadly resembling the whole. Put another way, the physical ‘unit’ to which these non-use values applies is, on reflection, not at all obvious. Yet given the possible importance of non-use value in certain ecosystem contexts, this issue surely merits further investigation.

One significant obstacle to addressing this challenge is that, as noted above, SP methods are often thought to be the only economic valuation techniques capable of measuring non-use values and so any doubts about the application of those methods or the accuracy of such valuations will loom especially large in this context. Challenges in the application of SP methods to non-use values are readily identified. Lack of experience and familiarity is likely to be important when respondents, for example, are asked about their preferences for non-use biodiversity species which might well be located in distant lands. Related to this is the lack of adequate testing for preference consistency exhibited in many such studies (although, see for an exception, Morse-Jones *et al*. (2012), discussed in further detail below).

Other avenues for non-use valuation remain to be explored. For example, legacies can be argued to represent a pure non-use value. That is, individuals leaving a charitable bequest to an environmental organization in a will, for the purposes of supporting conservation activities, will clearly not experience the benefits of this work. Atkinson *et al.* (2009) estimate that while (in 2007) only 6 per cent of all deaths in Britain resulted in a charitable bequest, their value remained substantial. And while legacies to environmental charities will be a relatively small proportion of this total, Mourato *et al*. (2010), for example, have estimated that this amounts to more than £200m in the (financial) year 2008/9. Of course, legacies reflect only non-use in the marketplace at the time of death. Moreover, data on charitable giving to recipient organizations or according to demographic characteristics of donors is not easily accessible, particularly for analysis over time. This is indicative of a wider problem. No approach appears to offer a general panacea for the challenges inherent in valuing non-use.

Related to the notion of 'non-use' is current interest in what has been termed 'shared values' (see, for example, Fish *et al*., 2011). For some this appears to be unfinished business arising from earlier discussions about how people value environmental policy changes, more generally, as individuals or citizens (Sagoff, 1988). However, the concept has also been a way of conveying that there might be something extra to the value of an ecosystem, over and above adding up different elements of its total economic value.[6] The emphasis on shared values traces this missing element of value to the way in which ecosystems have collective meaning and significance for communities of people related perhaps to 'non-use' or perceptions about ecosystem aesthetics.

There is less obvious evidence to add empirical substance to these insights. However, the handful of studies that have sought to use deliberative monetary valuation approaches provide some practical understanding of the individual or collective value of certain proposed environmental changes in a group context (e.g. Macmillan *et al*., 2002; Alvarez-Farizo *et al*., 2007), although our aforementioned comments about the representativeness of such findings still stand. Investigating this notion of shared values for ecosystems through wider-scale testing than has been possible thus far is a possibly rich topic for further research.

As an indication of the direction in which such reasoning might proceed, one reinterpretation of the 'shared values' argument is that it is a confusion between the individual making decisions on their own behalf and the same individual acting as social planner. In both cases the economic model applies directly, but the beneficiary and hence the objective changes. Such a perspective is inherent in the contrast between the personal utility maximization problem faced by the individual (or profit maximization by a firm) and the optimization of net present value within a social cost–benefit analysis. A further source of confusion can arise from the observation that individual preferences are highly likely to be, at least in part, social constructs. Put another way, social context moulds individual values.[7] Under such an interpretation, the necessity of inventing new

[6] Arrow *et al*. (2000) have made an analogous point in the context of the physical processes, that the value of some system as a whole may be more than the value of the sum of its parts, perhaps because of complex ecological interactions.

[7] In much in the same way, that is, as a move across locations, and consequent environments, will alter the value of any given resource: e.g. water in the desert has a much higher marginal value than in areas of high rainfall.

ways to measure apparently elusive 'social values' evaporates to be replaced by a recognition that (i) the value of goods to an individual (who, for example, may bear only a fraction of any associated externality) may differ radically from the value of the same good from a societal perspective, and (ii) even those former individual values are highly likely to be in part the product of social (and other) contexts. None of this undermines the usefulness of social knowledge in the valuation process. Rather, it provides a framework for the incorporation of such understanding within the decision system (uniting natural science, economics, and social science), and shows that such knowledge is vitally important if we are to understand the meaning and decision relevance of values and how they may alter between contexts; an issue to which we now turn.

(iv) Value transfer and spatial variability

Complex valuation processes, such as many of those involving ecosystem services and biodiversity, can involve significant costs. It is therefore not surprising that a considerable literature has now evolved around the transferral of value estimates for environmental resources (Brouwer, 2000; Boyle *et al*., 2010) as a proxy for original primary valuation.[8] Although all value-transfer techniques involve the extrapolation of information from one context to another, Navrud and Ready (2007) identify two general approaches.[9] The simplest of these is to transfer mean values from some pre-assessed 'study' to the 'policy' context in question (see, for example, Muthke and Holm-Mueller, 2004). Such univariate transfers are frequently used in practical decision-making, but their validity depends crucially upon the significance of differences between the study and policy contexts, which should be small for transfer errors to be minimized. Clearly at some level all sites are dissimilar (for example, the ecosystem habitats or the spatial pattern of substitutes around a site are unique). However, it is the degree to which this dissimilarity affects values which will determine the appropriateness of such 'univariate transfer' techniques.

The principal alternative to the univariate approach is to use statistical analyses to estimate value functions from study context data and to transfer those functions to policy contexts. This approach implicitly assumes that the variables determining the value of a good in one context will be the same as those affecting value in another context. Furthermore, it assumes that the relationships between variables and values will hold constant (i.e. in an estimated value-transfer function the list of explanatory variables and their coefficients are assumed to stay constant across the study and policy contexts). However, while parameters are kept constant, the values of the explanatory variables to which they apply are allowed to vary in line with the conditions characterizing

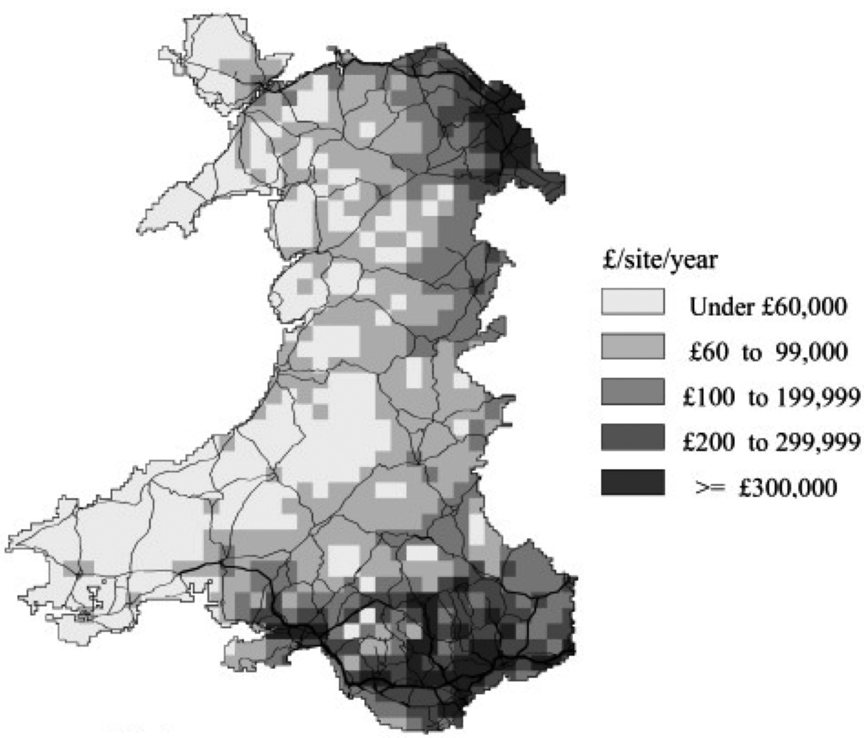
[8] The bulk of this literature concerns the transfer of valuation estimates for improving some environmental resource. As such actions generate positive values the literature is often labelled under the general heading of 'benefit transfers'. Such terminology is confusing as such techniques are also valid for the estimation of costs associated with resource losses. A more accurate and general term is therefore to refer to 'value transfers'.

[9] The development of such approaches can be traced through Desvousges *et al*. (1992), Bergland *et al*. (1995), Brouwer and Spaninks (1999), Zandersen *et al*. (2007), and Johnston and Duke (2009). Other variants include meta-analysis (e.g. Bateman and Jones, 2003; Lindhjem and Navrud, 2008) and Bayesian approaches to modelling value functions (e.g. Moeltner *et al*., 2007; Leon-Gonzalez and Scarpa, 2008).

each context. The value function transfer approach does not, therefore, look for similarity. Instead, it looks for heterogeneity so as to capture the variety of factors which determines values. Differences between sites become prime drivers of consequent variations in estimated values.

One of the largest ecosystem service value-transfer exercises conducted to date forms the core of the economic analysis underpinning the UK National Ecosystem Assessment (NEA, 2011). Here, value functions were estimated for multiple ecosystem services, including the provisioning value of agricultural food production, the regulating services of the environment as a store for greenhouse gases, and the so-called cultural services of both rural and urban recreation (including urban greenspace benefits). Following Bateman *et al.* (2011*c*), the functions were simplified to focus upon the main—theoretically expected—drivers of value, thereby avoiding the transfer of factors which only apply in a given context and are not general. The functions were also built in an integrated manner which linked the levels of each to the other. So, for example, if provisioning values are increased as a result of agricultural intensification, that same intensification feeds into an increase in greenhouse gas emissions and deterioration of rural recreation resources which result in a fall in both of these latter values. An example of the output obtained from such analyses, Figure 1 illustrates findings from the UK NEA analysis of rural recreation benefits arising from a change of land

Figure 1: Recreational values arising from a change in land use from farming to multi-purpose open access woodland in Wales



Source: Adapted from NEA (2011).

use from conventional farming towards multipurpose, open-access, woodland (as discussed earlier in the introduction to this paper).[10] The distribution obtained by transferring a recreational value function across the whole of Wales reflects various factors, including the distribution of population (this being highest in south-western Wales and in the areas of England neighbouring the north-east) and the availability and quality of the road network. Such spatially disaggregated outputs clearly allow decision-makers to target resources in the most efficient manner; an ability that is clearly of great importance during times of austerity.

Basing these integrated value-transfer exercises upon highly disaggregated, spatially sensitive, large observation databases, provides decision-makers with a rich and more holistic picture of the overall consequences of any given policy option. The advantages of such an approach were quickly realized by UK policy-makers, and the lessons of the UK NEA were explicitly incorporated in the UK Natural Environment White Paper (Defra, 2011), published in the immediate aftermath of the former report. Such academic and policy developments suggest that prospects for the incorporation of value-transfer techniques within institutional decision frameworks show promise. Notwithstanding this interim conclusion, there remains a need for tools capable of translating valuation information into policy action. We discuss this further in the next section.

IV. From values to ecosystem assessments and policy implementation

(i) Ecosystem valuation in the aggregate

The recent emphasis on large-scale ecosystem assessments—such as TEEB and the UK NEA—indicates some interest in searching for clues about the overall scale, in economic terms, of what has been lost (and what is likely to be lost in the future) as a result of the continued destruction of the natural world. While this is not a substitute for more detailed policy analysis, knowledge about these trends might be important for framing policy thinking. In addition, such information might throw light on whether ecosystem and biodiversity decline is a development problem as, for example, Stern (2007) demonstrated in the case of climate change.

One relatively long-standing insight here is that *particular* groups appear to be vulnerable to the loss of ecosystem services. Specifically, a number of studies have highlighted the dependence (of at least some portion) of the rural poor in the developing world on services provided by nature (Ten Brink, 2011; Jodha, 1986; Vedeld *et al*., 2004). These studies have been important in conveying the value of ecosystems and biodiversity to certain communities, which is otherwise only partially reflected in official statistics if at all. Less is known more generally, in either a developing or developed country context, about the way in which *aggregate* trends in, for example, ecosystem services and assets influence development (and development prospects). On the face of it, this is perhaps surprising and certainly contrasts markedly to the use of valuation in the climate

[10] This in turn builds on Bateman *et al*. (2003).

change context which has been, if anything, almost too exclusively concerned with global impacts. It seems worth asking why the hesitancy to aggregate has been so marked in the ecosystem and biodiversity context and also whether this matters.

With regards to the 'why', inevitably it must be mentioned that Costanza *et al.* (1997) have cast a long shadow over the thinking of the economics community in respect of this aggregation issue. Specifically, Costanza *et al.* sought to provide estimates of the global value of ecosystem services from (in effect) the entire stock of all ecosystem assets. In doing so, their study famously calculated that the value of services or the 'output' provided by the natural world, in 1994, was in the region of $33 trillion (i.e. substantially in excess of gross world income at that time).[11] Not surprisingly, substantial debate was generated in the wake of this striking result. And perhaps most vocal among the critics were environmental economists (Pearce, 1998; Bockstael *et al.*, 2000; Heal *et al.*, 2005). On the face of it, economists might be thought natural bedfellows of efforts to boost the profile of valuation practice. Yet, this issue of valuing the wellbeing provided by the entirety of the global flow of ecosystem services struck at the heart of the basic premise of economic valuation. Put another way, valuing total services assumes that our baseline is (in essence) the loss of all ecosystems and is a task that is unlikely to be adequately completed using methods that, instead, tell us something about the marginal value of a change in the stock of ecosystem assets.[12] Although it does not explain entirely the current (apparent) reticence to aggregate, unease about 'repeating the Costanza *et al.* error' cannot be ruled out altogether as a contributory factor.

In reflecting critically, in this vein, on the Costanza *et al.* contribution, Bateman *et al.* (2011*a*) note the paradox of the positive impact that this paper has had, more generally, in raising awareness of the economic value of the natural world. It seems worth asking, therefore, what has been lost by not answering these aggregate questions. Two recent studies have sought to revisit these issues but do so by calculating losses in natural assets likely to occur according to possible policy scenarios (and hence, in principle, ask a more defensible question than that about the totality of the current service flow). Hussain *et al.* (2012) estimate the losses arising from recent past and projected future loss of the world's aquatic ecosystems (specifically wetlands, mangrove, and coral reefs). The present value of this loss over the period 2000–50 (using a discount rate of 4 per cent) is reckoned in excess of $2 trillion (in 2007 US$) (with two-thirds of this accounted for by wetlands). The annualized value of this total change is just under $100 billion (that is, the value of the loss of these ecosystem assets each year is estimated to be of this magnitude) which, for example in 2007, was just 0.2 per cent of global gross income. Chiabai *et al.* (2011) conclude not entirely dissimilarly for the case of the loss of global forests over the same time period.

Needless to say, such global estimates of ecosystem loss require some heroic assumptions and generalizations. Indeed, for some critics, a search for a global value is a flawed project because of this. However, given the above findings, a tentative conclusion is that the pragmatic demands (for more highly aggregated indicators of trends) and principled

[11] This point estimate is calculated to lie within a possible range of $16 trillion to $54 trillion (in 1994 dollars).

[12] Only if the value of a marginal unit is constant is it then straightforward to go from valuing a single unit to valuing whatever number of units a given policy will create or destroy.

concerns (about the validity of such numbers) both point away from an emphasis on global trends. Greater practical significance, however, is to be found at the regional or country level. In the case of forests, for Brazil, estimated losses in natural wealth are found by Chiabai *et al.* (2011) to be substantial (as a percentage of the country's gross national income or GNI). Hussain *et al.* (2012) find that for aquatic ecosystems, for the South Asia region and for Indonesia, however, these annual losses in natural wealth were respectively 1.7 per cent and 4.0 per cent of GNI (in 2007).

These are magnitudes worth knowing more about. It would necessitate still closer scrutiny about the robustness of such estimates. The basic problem of accounting for the value of ecosystems can be put simply. It entails identifying a price or (unit) value and a quantity of (some change in) the provision of, for example, ecosystem services (Boyd and Banzhaf, 2007). An immediate challenge, however, lies in identifying the likely limits on how the available empirical record on ecosystem 'prices' and 'quantities' can be pulled and stretched over the assorted ecosystem areas needed to make robust aggregate generalizations. The issue of spatial variability here is central. This includes properly accounting for variation in the supply characteristics—the type and extent of functions—of ecosystems, as well as the demand characteristics—of the human population that consumes services that these functions give rise to. All this requires relatively sophisticated mapping and is demanding in information terms. However, it might be that at this national level (or sub-national levels) these issues become a little more tractable (see, for example, Kareiva *et al.*, 2011).

There are clear signs of growing interest in this question. An example of this is the linkages being made between (recent and on-going) ecosystem assessments and efforts to understand the way in which changes in natural wealth influence the sustainability of development through greening of national accounts (see, for example, World Bank, 2010; Arrow *et al.* forthcoming). The on-going World Bank-led consortium 'WAVES' project (Global Partnership for Wealth Accounting for the Value of Ecosystem Services) represents a practical application of this work to a number of proposed countries.[13]

Of course, much of what we currently term 'ecosystem services' may already be reflected in our national accounts. This is a point made recently in World Bank (2010). Examples of this might include the natural pollination services that (in effect) are capitalized in the value of agricultural land, or the recreational opportunities that are (implicitly and in part) provided by natural areas. On this view, ecosystems support market activity in a number of important (but indirect) ways and the accounting challenge is to re correctly -attribute the service value to the (ecosystem) asset which gave rise to it (Nordhaus, 2006). As a starting point, an emphasis on identifying what is already (somewhere) in the accounts has merit. In particular, given the traditional opposition by the national accountants to non-market valuation in relation to the accounts (Hecht, 2005), this approach has a strategic benefit.

(ii) Valuation and policy

While economics can contribute greatly to guiding the valuation of ecosystem and biodiversity services, it can also shape thinking about the implementation of policies aimed

[13] http://wwwr.worldbank.org/programs/waves (accessed February 2012).

at delivering such values. Unfortunately, at present, many of the policies employed to deliver ecosystem services fail to heed either evidence regarding the way in which values can vary over different patches of ecosystems, or the lessons of basic economic theory regarding incentives for actors to reveal truthfully their valuation of the services that they might provide. An example is provided by the UK Entry Level Stewardship (ELS) scheme (Natural England, 2010) which offers a flat-rate payment to all farmers, irrespective of their location.[14] Such schemes fail to target payments on those areas which yield the highest values and provide no incentive for farmers to provide anything other than the basic level of land management consistent with the scheme. Similar approaches characterize much of the increasingly substantial payments made under Pillar Two of the EU Common Agricultural Policy.

Thus economic valuation of itself is insufficient to improve the efficient delivery of ecosystem services. A simple example illustrates the problem and how economic intuition can help. Suppose that policy-makers seek to reduce diffuse water pollution from farms through a payment for ecosystem services (PES) scheme. A first requirement is to undertake a valuation exercise identifying those river catchments (and areas within those catchments) where reductions of pollution are likely to generate the largest net benefits. This might identify, for example, farms in locations above the inlet to water supply reservoirs as those most important to target. Now our focus must switch to the efficient implementation of such policies. One rather naïve approach might be simply to ask farmers to state the levels of compensation they require to move towards modes of production which avoid diffuse pollution. Of course, farmers have an incentive strategically to overstate their compensation requirements. However, the economic theory of auctions suggests that even relatively simple approaches can significantly improve implementation efficiency (Vickrey, 1961; Clarke, 1971; Groves, 1973; Groves and Ledyard, 1977). For example, switching to a simple sealed-bid contracting system might reduce the potential for strategic responses and improve incentive compatibility. This could be the case if farmers are told that contracts will be awarded according to the combination of pollution reduction and cost.

In certain circumstances even greater efficiency gains can be obtained. For example, where the delivery of ecosystem services can be readily measured (for example in policies seeking the provision of certain habitats) then land owners will be those best able to judge whether their land is particularly suitable for providing such goods (or faces the lowest opportunity costs). Such actors can outbid competitors by offering better outputs (or lower costs) than their rivals. To date practical examples of such agreements are, at least in the UK, generally confined to the experimental laboratory. However, proposals have been made by a number of policy-makers that the development of such implementation tools should be a major focus of the next phase of work under the UK NEA. The above example indicates that valuation, while typically necessary for good decision-making, it is not in itself sufficient.

One further point is that valuing ecosystems and biodiversity valuation is a complex endeavour and often at the frontier of valuation knowledge. This suggests good reason, in certain contexts, to be circumspect about the role that valuation might play

[14] An exception here is the minority of farms located above the 'Moorland Line' (English Nature, 2010) where a lower, but again flat-rate, payment is available.

in informing decisions about conservation. Decision-making in such situations where values are unknown—or where values cannot be established to any degree of validity—has generated much debate. In such cases, however, 'caution' (given what might be lost) might be a sensible watchword. Possible responses include the adoption of ecological standards, sometimes termed 'safe minimum standards', to ensure the sustainability of resources which are not amenable to valuation (Farmer and Randall, 1998) or compensating offsetting compensatory projects validated for their ecological suitability (Federal Register, 1995). In such cases, the role for valuation might be a greater emphasis on cost-effectiveness in meeting specified targets.

An illustration of this challenge in determining how exactly valuation should guide social decision-making is provided by the example of valuing biodiversity. Weitzman (1993)—using the example of the world's remaining species of cranes—defines biological importance of each species in terms of their taxonomic distinctiveness (e.g. of the whooping crane compared to other crane species)[15] and the likelihood of extinction (of a given species). Assuming that maximizing (expected) diversity is our objective, species conservation becomes a problem of cost-effectively distributing the marginal (available) unit of money from conservation funds to where it achieves the highest pay-off. Typically, this will be where there is some combination of high diversity and low survival probabilities.

Ideally, it would be useful to extend such insights with reference to the preferences that people might have for diversity. Somewhat reassuringly, Morse-Jones *et al*. (2012), for example, find that stated preference responses reveal expected substitution patterns across ecologically similar species—e.g. different small amphibians. However, preferences need not always conform to what is ecologically feasible or sustainable. Thus, in the Morse-Jones *et al*. study, respondents had a massively stronger preference for iconic, physically large, and especially furry animals which dwarfs concerns regarding ecologically crucial issues such as extinction threat. So, for example, willingness to pay to conserve lions, even where these animals are not threatened by extinction, hugely outweighs stated values for say a species of frog, even when it is on the brink of extinction. Another example is provided by Bateman *et al*. (2009). That study observes that while respondents had strongly positive preferences for enlarging an area of freshwater marshland suitable for visiting and viewing bird populations, they had negative values for an adjoining area of tidal mudflats, even though these were a major source of food attracting those birds to the area. In many respects, these findings are not surprising. However, what it does raise is a deeper question about whether the extent to which economic values can be a guide for decision-making, or whether ecological constraints need to be considered. Clearly, the claim that human preferences are (almost always) 'right' or 'wrong' is overly simplistic at either extreme. However, where to draw this line is far from obvious and—given changing knowledge—is anyhow likely to be a shifting target. Nevertheless, while recognizing the importance of economic values for thinking about the importance of ecosystems and guiding policy thinking, we need to be mindful of the complexities and uncertainties involved.

[15] Genetic distinctiveness is defined, by Weitzman (1993), as the evolutionary distance each existing species is from a common ancestor species.

V. Conclusions

The valuation of ecosystem services has become a crucial element (perhaps *the* crucial element) in quantifying the contribution of ecosystems and biodiversity to human wellbeing. A significant body of research has already begun to emerge and a number of recent national and international ecosystem assessments have helped provide further impetus to such efforts. Needless to say, significant challenges remain. Hence, while the evidence base is broad and deep—at least for some services—reflections on this literature in a variety of existing reviews have identified a large number of issues. These include: a need for greater understanding of ecological production, especially as it relates to spatial variability and complexities in the way that services are produced; the size and significance of inevitable gaps in the empirical record, as well as the ability to fill these gaps by judiciously transferring values; and, the scope and limits in using this evidence base to inform practical decision-making, both generally and in relation to concerns about whether the valuations that we find in this literature genuinely tell us about the importance of ecosystem assets and biodiversity.

In the current paper, we have sought to highlight some of these issues, although unavoidably our discussion has not been exhaustive. Much of our focus has been on valuation methods and, particularly, the challenges inherent in seeking to value non-market costs and benefits. Some of these challenges involve general considerations, although other issues are specific to valuing ecosystems and biodiversity, or at least seem particularly acute in that context.

Such challenges need to be viewed in context. The recent UK NEA (NEA, 2011) has shown how the empirical record can be put to use in an informative and policy relevant way. Thus, there are encouraging signs that value transfer methods (i.e. transferring the empirical record to new policy contexts and questions) can be used in an increasingly effective manner. If so, concerns about whether we can adequately measure the way in which ecosystem values vary across space (because of geographical variability in the way that services are supplied by nature and valued by people) might be addressed. These developments could be crucial in translating valuations into meaningful policy analysis. It may also offer some hope for shedding light on the value of what is lost when and if ecosystems and biodiversity are degraded and destroyed in more highly aggregated assessments. This is not just an issue of only identifying aggregate trends (for which policy uses would be limited apart from perhaps raising the profile of conservation issues generally). There are fruitful linkages to be made about the way in which what is happening to (natural) wealth influences development paths.

Thinking about ecosystems as assets also helps identify some critically important issues arguably neglected in most of the valuation literature as it has been applied to ecosystem services. This relates to the way in which future services are valued when an ecosystem asset undergoes some change. While such questions are commonplace elsewhere, in the ecosystem context these have only begun to be asked, although related issues of valuing ecosystem complexity have a longer standing. Progress on these matters, both in theory and practice, is surely only a matter of time. Nevertheless, it seems unavoidable that uncertainties will remain. That is, while we can conclude positively on the rapidly evolving scope for ecosystem and biodiversity valuation to contribute to a profound understanding of suitable policy responses, there remains room for debate about whether valuation is in itself enough to ensure effective policies as well as how to conduct decision analyses in those contexts where valuation and understanding of the natural world is likely to remain relatively uncertain.

References

Adamowicz, W. J., Louviere, J., and Williams, M. (1994), 'Combining Revealed and Stated Preference Methods for Valuing Environmental Amenities', *Journal of Environmental Economics and Management*, **26**, 271–92.

Alberini, A., and Kahn, J. (eds) (2006), *Handbook on Contingent Valuation*, Cheltenham, Edward Elgar.

Alvarez-Farizo B., Hanley, N., Barberan, R., Lazaro, A. (2007), 'Choice Modeling at the "Market Stall": Individual Versus Collective Interest in Environmental Valuation', *Ecological Economics*, **60**, 743–51.

Arrow, K., Daily, G. C., Dasgupta, P., Levin, S., Mäler, K.-G., *et al.* (2000), 'Managing Ecosystem Resources', *Environmental Science and Technology*, **34**, 1401–6.

— Dasgupta, P., Goulder, L. H., Mumford, K. J., and Oleson, K. (forthcoming), 'Sustainability and the Measurement of Wealth', *Environment and Development Economics*.

Atkinson, A., Backus, P., and Micklewright, J. (2009), '*Charitable Bequests and Wealth at Death in Great Britain*', Working Paper A09/03, University of Southampton.

Barbier, E. B. (2007), 'Valuing Ecosystem Services as Productive Inputs', *Economic Policy*, **22**(49), 177–229.

— (2009), 'Ecosystems as Natural Assets', *Foundations and Trends in Microeconomics*, **4**(8), 611–81.

— (2012), 'Progress and Challenges in Valuing Coastal and Marine Ecosystem Services', *Review of Environmental Economics and Policy*, **6**(1), 1–19.

— Koch, E. W., Silliman, B. R., Hacker, S. D., Wolanski, E., Primavera, J., Granek, E. F., Polasky, S., Aswani, S., Cramer, L. A., Stoms, D. M., Kennedy, C. J., Bael, D., Kappel, C. V., Perillo, G. M., and Reed, D. J. (2008), 'Coastal Ecosystem-based Management with Nonlinear Ecological Functions and Values', *Science*, **319**, 321–3.

Barton, J., and Pretty, J. (2010), 'What is the Best Dose of Nature and Green Exercise for Mental Health? A Meta-study Analysis', *Environmental Science and Technology*, **44**(10), 3947–55.

Bateman, I. J., and Jones, A. P. (2003), 'Contrasting Conventional with Multi-level Modelling Approaches to Meta-analysis: An Illustration Using UK Woodland Recreation Values', *Land Economics*, **79**(2), 235–58.

— Lovett, A. A., and Brainard, J. S. (2003), *Applied Environmental Economics: a GIS Approach to Cost–Benefit Analysis*, Cambridge, Cambridge University Press.

— Burgess, D., Hutchinson, W. G., and Matthews, D. I. (2008*a*), 'Contrasting NOAA Guidelines with Learning Design Contingent Valuation (LDCV): Preference Learning Versus Coherent Arbitrariness', *Journal of Environmental Economics and Management*, **55**, 127–41.

— Munro, A., and Poe, G. L. (2008*b*), 'Asymmetric Dominance Effects in Choice Experiments and Contingent Valuation', *Land Economics*, **84**, 115–27.

— Day, B. H., Jones A. P., and Jude, S. (2009), 'Reducing Gains/Loss Asymmetry: A Virtual Reality Choice Experiment (VRCE) Valuing Land Use Change', *Journal of Environmental Economics and Management*, **58**, 106–18.

— Fisher B., Fitzherbert E., Glew, D., and Naidoo, R. (2010), 'Tigers, Markets and Palm Oil: Market Potential for Conservation', *Oryx*, **44**(2), 230–4.

— Mace, G. M., Fezzi, C., Atkinson, G., and Turner, R. K. (2011*a*), 'Economic Analysis for Ecosystems Assessments', *Environmental and Resource Economics*, **48**, 177–218.

— Abson, D., Beaumont, N., Darnell, A., Fezzi, C., Hanley, N., Kontoleon, A., Maddison, D., Morling, P., Morris, J., Mourato, S., Pascual, U., Perino, G., Sen, A., Tinch, D., Turner, R. K., Valatin, G., *et al.*. (2011*b*), 'Economic Values from Ecosystems', *in The UK National Ecosystem*

Assessment Technical Report, UK National Ecosystem Assessment, UNEP-WCMC, Cambridge, *also available from* http://uknea.unep-wcmc.org/.

Bateman, I. J., Brouwer, R., Ferrini, S., Schaafsma, M., Barton, D. N., Dubgaard, A., Hasler, B., Hime, S., Liekens, I., Navrud, S., De Nocker, L., Ščeponavičiūtė, R., and Semėnienė, D. (2011*c*), 'Making Benefit Transfers Work: Deriving and Testing Principles for Value Transfers for Similar and Dissimilar Sites Using a Case Study of the Non-market Benefits of Water Quality Improvements across Europe', *Environmental and Resource Economics*, **50**(3), 356–87.

Bergland, O., Magnussen, K., and Navrud, S. (1995), '*Benefit Transfer: Testing for Accuracy and Reliability*', Discussion Paper D-03/1995, Department of Economics and Social Sciences, Agricultural University of Norway.

Bird, W. (2007), '*Natural Fit: Can Green Space and Biodiversity Increase Levels of Physical Activity?*', Sandy, Royal Society for the Protection of Birds.

Bockstael, N. E., Freeman, A. M. III, Kopp, R. J., Portney, P. R., and Smith, V. K. (2000), 'On Measuring Economic Values for Nature', Environmental Science and Technology, **34**, 1384–9.

Boyd, J., and Banzhaf, S. (2007), 'What are Ecosystem Services? The Need for Standardized Environmental Accounting Units', *Ecological Economics*, **63**(2–3), 616–26.

Boyle, K. J., Kuminoff, N. V., Parmeter, C. F., and Pope, J. C. (2010), 'The Benefit-transfer Challenges', *Annual Review of Resource Economics*, **2**, 161–82.

Brouwer, R. (2000), 'Environmental Value Transfer: State of the Art and Future Prospects', *Ecological Economics*, **32**, 137–52.

— Spaninks, F. A. (1999), 'The Validity of Environmental Benefits Transfer: Further Empirical Testing', *Environmental and Resource Economics*, **14**(1), 95–117.

Bulte, E., van Soest, D., van Kooten, C., and Schipper, R. (2002), 'Forest Conservation in Costa Rica when Non-use Benefits are Uncertain and Rising', *American Journal of Agricultural Economics*, **84**(1), 150–60.

Carson, R. T. (2011), *Contingent Valuation: A Comprehensive Bibliography and History*, Cheltenham, Edward Elgar.

Champ, P. A. Boyle, K. J., and Brown, T. C. (eds) (2003), *A Primer on Nonmarket Valuation*, Dordrecht, Kluwer.

Chan, K. M. A., Goldstein, J., Satterfield, T., Hannahs, N., Kikiloi, K., Naidoo, R., Vadeboncoeur, N., and Woodside, U. (2011), 'Cultural Services and Non-use Values', in P. Kareiva *et al.*. (eds), *Natural Capital: Theory and Practice of Mapping Ecosystem Services*, Oxford, Oxford University Press.

Chiabai, A., Travisi, C. M., Markandya, A., Ding, H., and Nunes, P. A. I. D. (2011), 'Economic Assessment of Forest Ecosystem Services Losses: Costs of Policy Inaction', *Environmental and Resource Economics*, **50**, 405–45.

Christie, M., Hanley, N., Warren, J., Murphy, K., Wright, R., and Hyde, T. (2006), 'Valuing the Diversity of Biodiversity', *Ecological Economics*, **58**, 304–17.

Clarke, E. (1971), 'Multipart Pricing of Public Goods', *Public Choice*, **8**, 19–33.

Costanza, R., d'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R. V., Paruelo, J., Raskin, R. G., Sutton, P., and van den Belt, M. (1997), 'The Value of the World's Ecosystem Services and Natural Capital', *Nature*, **387**, 253–60.

Daily, G. C. (1997), *Nature's Services: Societal Dependence on Natural Ecosystems*, Washington, DC, Island Press.

Dasgupta, P. M. (2009), 'The Welfare Economic Theory of Green National Accounts', *Environment and Resource Economics*, **42**, 3–38.

Defra (2011), '*The Natural Choice: Securing the Value of Nature*', Cm 8082, Department for Environment, Food and Rural Affairs, London, The Stationery Office.

De Groot, R. S., Wilson, M. A., and Boumans, R. M. J. (2002), 'A Typology for the Classification, Description and Valuation of Ecosystem Functions, Goods and Services', *Ecological Economics*, **41**, 393–408.

Desvousges, W. H., Naughton, M. C., and Parsons, G. R. (1992), 'Benefit Transfer: Conceptual Problems in Estimating Water Quality Benefits Using Existing Studies', *Water Resources Research*, **28**(3), 675–83.

De Vries, S., Verheij, R. A., Groenewegen, P. P., and Spreeuwenberg, P. (2003), 'Natural Environments—Healthy Environments? An Exploratory Analysis of the Relationship between Greenspace and Health', *Environment and Planning A*, **35**, 1717–31.

Dietz, S., and Stern, N. (2008), 'Why Economic Analysis Supports Strong Action on Climate Change, A Response to the Stern Review's Critics', *Review of Environmental Economics and Policy*, **2**(1), 94–113.

Elmqvist, T. *et al.*. (2010), 'Biodiversity, Ecosystems and Ecosystem Services', in P. Kumar (ed.), *The Economics of Ecosystems and Biodiversity: Ecological and Economic Foundations*, London, Earthscan.

Farley, J. (2008), 'Valuing Natural Capital: The Limits of Marginal Valuation in Complex Systems', *Conservation Biology*, **22**(6), 1399–1408.

Farmer, M. C., and Randall, A. (1998), 'The Rationality of a Safe Minimum Standard', *Land Economics*, **74**, 287–302.

Federal Register (1995), 'Federal Guidance for the Establishment, Use and Operation of Mitigation Banks', *Federal Register*, **60**(228), 58605–14.

Ferraro, P. J., Lawlor, K., Mullan, K. L., and Pattanayak, S. K. (2012), 'Forest Figures: Ecosystem Services Valuation and Policy Evaluation in Developing Countries', *Review of Environmental Economics and Policy*, **6**(1), 20–44.

Fish, R., *et al.*. (2011), 'Chapter 24: Shared Values for the Contributions Ecosystem Services Make to Human Well-being', in *The UK National Ecosystem Assessment Technical Report*, UK National Ecosystem Assessment, UNEP-WCMC, Cambridge.

Fisher, B., Turner, R. K., and Morling, P. (2009), 'Defining and Classifying Ecosystem Services for Decision Making', *Ecological Economics*, **68**(3), 643–53.

Freeman, A. M., III (2003), *The Measurement of Environmental and Resource Values: Theory and Methods*, 2nd edn, Washington, DC, Resources for the Future.

Gallai, N., Salles, J. M., Settele, J., and Vaissiere, B. E. (2009), 'Economic Valuation of the Vulnerability of World Agriculture Confronted with Pollinator Decline', *Ecological Economics*, **68**, 810–21.

Gerlagh, R., and van der Zwaan, B. C. C. (2002), 'Long-term Substitutability between Environmental and Man-made Goods', *Journal of Environmental Economics and Management*, **44**, 329–45.

Gibbons, S., Mourato, S., and Resende, G. (2011), '*The Amenity Value of English Nature: A Hedonic Price Approach*', Spatial Economics Research Centre, DP0074, London School of Economics.

Goulder, L., and Kennedy, D. (2011), 'Interpreting and Estimating the Value of Ecosystem Services', in P. Kareiva *et al.*. (eds), *Natural Capital: Theory and Practice of Mapping Ecosystem Services*, Oxford, Oxford University Press.

Gowdy, J., *et al.*. (2010), 'Discounting, Ethics and Options for Maintaining Biodiversity and Ecosystem Integrity', in P. Kumar (ed.), *The Economics of Ecosystems and Biodiversity: Ecological and Economic Foundations*, London, Earthscan.

Gren, I.-M., Folke, C., Turner, R. K., and Bateman, I. (1994), 'Primary and Secondary Values of Wetland Ecosystems', *Environmental and Resource Economics*, **4**, 55–74.

Groves, T. (1973), 'Incentives in Teams', *Econometrica*, **41**, 617–31.

— Ledyard, J. (1977), 'Some Limitations on Demand Revealing Processes, *Public Choice*, **29**(2), 107–24.

Hanley, N. D., and Barbier, E. B. (2009), *Pricing Nature: Cost–Benefit Analysis and Environmental Policy*, Cheltenham, Edward Elgar.

— Mourato, S., and Wright, R. E. (2001), 'Choice Modelling Approaches: A Superior Alternative for Environmental Valuation?', *Journal of Economic Surveys*, **15**(3), 435–62.

Hartig, T., Evans, G., Jamner, L. D., Davis, D. S., and Garling, T. (2003), 'Tracking Restoration in Natural and Urban Field Settings', *Journal of Environmental Psychology*, **23**, 109–23.

Heal, G. M. (2007), 'A Celebration of Environmental and Resource Economics', *Review of Environmental Economics and Policy*, **1**(1), 7–25.

— Barbier, E. B., Boyle, K. J., Covich, A. P., Gloss, S. P., Hershner, C. H., Hoehn, J. P., Pringle, C. M., Polasky, S., Segerson, K., Shrader-Frechette, K. (2005), *Valuing Ecosystem Services: Toward Better Environmental Decision Making*, Washington, DC, National Academies Press.

Hecht, J. (2005), *National Environmental Accounting*, Washington, DC, Resources for the Future.

Hoel, M., and Sterner, T. (2007), 'Discounting and Relative Prices', *Climatic Change*, **84**, 265–80.
Hotelling, H. (1949), 'Letter to the Director of the National Park Service', in R. A. Prewitt (ed.), *The Economics of Public Recreation*. *The Prewitt Report*, Washington, DC, Department of the Interior.
Hussain, S. S., McVittie, A., Brander, L. M., Vardakoulias, O., Wagtendonk, A., and Verburg, P. (2012), *The Economics of Ecosystems and Biodiversity: The Quantitative Assessment*, Nairobi, United Nations Environment Programme.
Jodha, N. S. (1986), 'Common Property Resources and the Rural Poor in Dry Regions of India', *Economic and Political Weekly*, **21**(27), 1169–81.
Johnston, R. J., and Duke, J. M. (2009), 'Willingness to Pay for Land Preservation across States and Jurisdictional Scale: Implications for Benefit Transfer', *Land Economics*, **85**(2), 217–37.
Kanninen, B. J. (ed.), (2007), *Valuing Environmental Amenities Using Stated Choice Studies*, Dordrecht, Springer.
Kareiva, P., Tallis, H., Ricketts, T. H., Daily, G. C., and Polasky, S. (eds) (2011), *Natural Capital: Theory and Practice of Mapping Ecosystem Services*, Oxford, Oxford University Press.
Krutilla, J. (1967), 'Conservation Reconsidered', *American Economic Review*, **57**, 777–86.
— Fisher, A. C. (1974), *The Economics of Natural Environments*, Baltimore, MD, Johns Hopkins University Press.
Kumar, P. (ed.) (2010), *The Economics of Ecosystems and Biodiversity: Ecological and Economic Foundations*, London, Earthscan.
Leon-Gonzalez, R., and Scarpa, R. (2008), 'Improving Multi-site Benefit Functions via Bayesian Model Averaging: A New Approach to Benefit Transfer', *Journal of Environmental Economics and Management*, **56**(1), 50–68.
Lindhjem, H. L., and Navrud, S. (2008), 'How Reliable are Meta-analyses for International Benefit Transfers?', *Ecological Economics*, **66**, 425–35.
MA (2005), *Ecosystems and Human Well-being: A Framework for Assessment*, Millennium Ecosystem Assessment, Washington, DC, Island Press.
Mace, G. M., Norris, K., and Fitter, A. H. (2012), 'Biodiversity and Ecosystem Services: A Multilayered Relationship', *Trends in Ecology and Evolution*, **27**(1), 19–26.
MacKerron, G., and Mourato, S. (2011), '*Mappiness: Quantifying Wellbeing in Relation to Environment Across Space and Time'*, paper presented at European Association of Environmental and Resource Economists, 18th Annual Conference, 29 June–2 July, Rome.
Macmillan D., Philip, L., Hanley, N., and Alvarez-Farizo, B. (2002), 'Valuing the Non-market Benefits of Wild Goose Conservation: A Comparison of Interview and Group-Based Approaches', *Ecological Economics*, **43**, 49–59.
Mäler, K.-G. (1974), *Environmental Economics: A Theoretical Inquiry*, Baltimore, MD, Resources for the Future.
— (2008), 'Sustainable Development and Resilience in Ecosystems'. *Environmental and Resource Economics*, **39**(1), 17–24.
— Aniyar, S., and Jansson, Å. (2009), 'Accounting for Ecosystems', *Environmental and Resource Economics*, **42**, 39–51.
Mitchell, R., and Popham, F. (2008), 'Effect of Exposure to Natural Environment on Health Inequalities: An Observational Population Study', *Lancet*, **372**, 1655–60.
Moeltner, K., Boyle, K. J., and Paterson, R. W. (2007), 'Meta-analysis and Benefit Transfer for Resource Valuation—Addressing Classical Challenges with Bayesian Modeling', *Journal of Environmental Economics and Management*, **53**, 250–69.
Morse-Jones, S., Bateman, I. J., Kontoleon, A., Ferrini, S., Burgess, N., and Turner, R. K. (2012), 'Stated Preferences for Tropical Wildlife Conservation amongst Distant Beneficiaries: Charisma, Endemism, Scope and Substitution Effects', *Ecological Economics*, forthcoming.
Mourato, S., Atkinson, G., Collins, M., Gibbons, S. MacKerron, G., and Resende, G. (2010), *Economic Analysis of Cultural Services*, UK NEA Economic Analysis Report.
Muthke, T., and Holm-Mueller, K. (2004), 'National and International Benefit Transfer Testing with a Rigorous Test Procedure', *Environmental and Resource Economics*, **29**, 323–36.
Myers, S. S., and Patz, J. A. (2009), 'Emerging Threats to Human Health from Global Environmental Change', *Annual Review of Environment and Resources*, **34**, 223–52.

Natural England (2010), *Entry Level Stewardship: Environmental Stewardship Handbook*, 3rd edn, Natural England.
Navrud, S., and Ready, R. (2007), 'Review of methods for Value Transfer', in S. Navrud and R. Ready (eds), *Environmental Value Transfer: Issues and Methods*, Dordrecht, Springer.
NEA (2011), *National Ecosystem Assessment Technical Report*, UK National Ecosystem Assessment, UNEP–WCMC, Cambridge.
Nordhaus, W. D. (2006), 'Principles of National Accounting for Non-market Accounts', in D. Jorgenson, S. Landefeld, and W. D. Nordhaus (eds), *A New Architecture for the National Accounts*, Chicago, IL, National Bureau of Economic Research/University of Chicago Press.
Osman, L. (2005), 'Chapter 2: Greenspace and its Benefits for Health', in CJC Consulting (ed.), *Economic Benefits of Accessible Green Spaces for Physical and Mental Health: Scoping Study*, Final Report for the Forestry Commission.
Pagiola, S., Ritter, K., and Bishop, J. T. (2004), *How Much is an Ecosystem Worth? Assessing the Economic Value of Conservation*, Washington, DC, The World Bank.
Pascual, U., Muradian, R., Gómez-Baggethun, E., Armsworth, P., Brander, L., Cornelissen, H., Farley, J., Loomes, J., Martinez-López, B., Pearson, L., Perrings, C., Polasky, S., and Verma, M. (2010), 'Valuation of Ecosystems Services: Methodology and Challenges', in P. Kumar (ed.), *The Economics of Ecosystems and Biodiversity: Ecological and Economic Foundations*, London, Earthscan.
Pearce, D. W. (1998), 'Auditing the Earth', *Environment*, **40**(2), 22–8.
— Atkinson, G., and Mourato, S. (2006), *Cost–Benefit Analysis and the Environment: Recent Developments*, Paris, Organization for Economic Cooperation and Development.
Perrings, C. (2006), 'Resilience and Sustainable Development', *Environment and Development Economics*, **11**, 417–27.
Pretty, J., Peacock, J., Sellens, M., and Griffin, M. (2005), 'The Mental and Physical Health Outcomes of Green Exercise', *International Journal of Environmental Health Research*, **15**, 319–37.
— Hine, R., Sellens, M., South, N., and Griffin, M. (2007), 'Green Exercise in the UK Countryside: Effects on Health and Psychological Well-being', *Journal of Environmental Planning and Management*, **50**(2), 211–31.
Rosen, S. (1974), 'Hedonic Prices and Implicit Markets: Product Differentiation in Pure Competition', *Journal of Political Economy*, **82**(1), 34–55.
Ruitenbeek, J. (1989), *Social Cost–Benefit Analysis of the Korup Project, Cameroon*, Report for the World Wildlife Fund and Republic of Cameroon.
Sagoff, M. (1988), *The Economy of the Earth*, Cambridge, Cambridge University Press.
Stern, N. (2007), *The Economics of Climate Change: The Stern Review*, Cambridge, Cambridge University Press.
Sterner, T., and Persson, U. M. (2008), 'An Even Sterner Review: Introducing Relative Prices into the Discounting Debate', *Review of Environmental Economics and Policy*, **2**(1), 61–76.
Sukhdev, P. (2008), *The Economics of Ecosystems and Biodiversity: An Interim Report*, Brussels, European Communities.
Takano, T., Nakamura, K., and Watanabe, M. (2002), 'Urban Residential Environments and Senior Citizens' Longevity in Megacity Areas: The Importance of Walkable Green Spaces', *Journal of Epidemiology and Community Health*, **56**, 913–18.
TEEB (2010), *The TEEB Synthesis Report Mainstreaming the Economics of Nature, A Synthesis of the Approach, Conclusions and Recommendations of TEEB*, The Economics of Ecosystems and Biodiversity, European Commission/UNEP/BMU-Germany.
Ten Brink, P. (ed.) (2011), *The Economics of Ecosystems and Biodiversity in National and International Policy Making*, London, Earthscan.
Ulrich, R S. (1984), 'View Through a Window may Influence Recovery from Surgery', *Science*, **224**(4647), 420–1.
US SAB (2009), *Valuing the Protection of Ecological Systems and Services*, Washington, DC, US Science Advisory Board.

Vedeld, P., Angelsen, A., Sjaastad, E., and Berg, G. K. (2004), *Counting on the Environment: Forest Incomes and the Rural Poor*, Environmental Economics Series No. 98, Washington, DC, World Bank.

Vickrey, W. (1961), 'Counterspeculation, Auctions, and Competitive Sealed Tenders', *Journal of Finance*, **16**, 8–37.

Walker, B., Pearson, L., Harris, M., Mäler, K.-G., Li, C.-Z., Biggs, R., and Baynes, T. (2010), 'Incorporating Resilience in the Assessment of Inclusive Wealth: An Example from South East Australia', *Environmental and Resource Economics*, **45**(2), 183–202.

Weitzman, M. L. (1993), 'What to Preserve? An Application of Diversity Theory to Crane Conservation?', *Quarterly Journal of Economics*, **108**, 157–83.

— (2007), 'A Review of the Stern Review on the Economics of Climate Change', *Journal of Economic Literature*, **45**, 703–24.

World Bank (2010), *The Changing Wealth of Nations*, Washington, DC, World Bank.

Zandersen M., Termansen, M., Jensen, F. S. (2007), 'Testing Benefits Transfer of Forest Recreation Values over a Twenty-year Time Horizon', *Land Economics*, **83**(3), 412–40.