VALUING BIODIVERSITY

Discussion Paper

For Department for Environment Food and Rural Affairs (Defra)

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SUMMARY

S.1 Introduction

This paper has been prepared as part of the project ‘Development of ‘look-up’ environmental value estimates for initial appraisal within cost-benefit analysis’. It has two main aims:

1. To present a broad discussion on the challenging issue of valuing biodiversity in the policy appraisal and cost-benefit analysis (CBA) context; and

2. To provide guidance on the potential for including ‘look-up’ biodiversity values within the ‘Environmental Value Look-Up Tool’ (EVL Tool) developed as part of the project.

The paper reviews the relative merits of different approaches to valuing biodiversity and associated impacts. A theoretically-correct approach for valuation for CBA purpose requires the use of willingness to pay (WTP) (demand)-based metrics. However, the empirical difficulties of estimating robust WTP values for some aspects of biodiversity, in particular non-use values, may justify the use of proxy measures, notably estimates of the opportunity costs of delivering externally determined biodiversity targets, even though these are conceptually quite distinct from ‘true’ welfare values.

S.2 Role for biodiversity look-up values in policy appraisal

Biodiversity is defined by the Convention on Biological Diversity (CBD) as ‘the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part, this includes diversity within species, between species and ecosystems’ (CBD, Article 2). It can be thought of as a kind of natural resource that supports the provision of ecosystem goods and services to human populations.

Biodiversity performs many functional roles in ecosystems, underpinning ecosystem service provision, where both the level and the stability of ecosystem services tend to improve with increasing biodiversity. Past and present loss of biodiversity, coupled with increasing recognition of its importance in underpinning multiple ecosystem functions and benefits, make biodiversity and ecosystem conservation, restoration and management important public policy priorities. Various initiatives have sought to highlight the need for improved understanding and accounting for biodiversity in policy analyses; for example the Millennium Assessment (MA), The Economics of Ecosystems and Biodiversity (TEEB), the UK National Ecosystem Assessment (UK NEA).

Developing look-up values for biodiversity could make assessments easier and quicker and may contribute in some way to ensuring that such impacts are represented in appraisal. Making look-up values accessible to policy makers and appraisers in all sectors, along with clear guidance on their use, could therefore make a contribution to the ‘mainstreaming’ agenda. However, there are also dangers in this approach, since the valuation evidence available is far from perfect. ‘Entry-point’ look-up values in particular will not reflect in any detail the spatial factors underpinning the full value of biodiversity. The values may be useful if they draw attention to biodiversity impacts and encourage thinking about ways of reducing the damages and enhancing the benefits of biodiversity, but not if they are used as a form of ‘box-ticking’ at the expense of setting biodiversity concerns to one side.
S.3 Conceptual basis for valuing biodiversity

Specific aspects of biodiversity are treated in many different ways by different analysts, and in various ecosystem services frameworks. Biodiversity can be viewed technically in terms of variability, with multiple levels from genetic to ecosystem, and indicators such as richness and turnover. Often, however, it is more loosely equated with ecosystems or the natural world generally, or more narrowly with species richness of particular ecosystems. There does appear, however, to be an emerging consensus that biodiversity is important at all levels in ecosystem services, playing the role of supporting, intermediate and final service (e.g. MA, TEEB, IPBES, MAES, UK NEA, CICES, ‘EGS-CS’). Whilst this is still a live topic of academic research and debate, some general lessons can be extracted for the purposes of valuation:

- There are clear concerns regarding the potential for double-counting when considering biodiversity values, since biodiversity plays a role at all levels in ecosystem services frameworks. Generally, the best approach is to value biodiversity only where it impacts human well-being through final utility or production functions; i.e. the benefits derived from final ecosystem goods and services.

- Valuation of final good and services is not sufficient, however, and proper consideration must be given to the importance of biodiversity and ecosystem protection and resilience, in order to understand the links from biodiversity to final goods and services. The approach of valuing only the final services does not imply that supporting services should not be considered in policy analyses.

- Valuation can focus on current flows of final goods and services if the assumption that biodiversity ‘stocks’ are sustainably managed and resilient to the combination of exploitation and external shocks is valid. If biodiversity is in fact declining or under threat, these potential future losses must be taken into account, requiring an understanding of the condition of the underlying biodiversity stock and what its deterioration implies for flows of final goods and services in the future.

In practical terms, one way to ensure that appropriate assumptions are made as to the applicability of valuation methods and their limits is to recognise the existence of biodiversity that is “critical natural capital” that cannot be traded off. Identification of critical natural capital requires informed judgement, based on evidence, and is dependent on views on the extent to which substitution across capital types can be allowed. “Weak” sustainability seeks to preserve capital overall, and allows natural capital to be substituted for by produced capital, whereas “strong” sustainability seeks conservation of each type of capital separately. In principle, it could be estimated (roughly) where capital starts to become critical/essential, though detailed discussion of how this might be done is beyond the scope of this discussion paper.

It should be stressed that any proposal that threatens critical natural capital is beyond the realms of using a ‘look-up’ value approach to valuation in appraisal, and that both science and economic assessment need to be more detailed to ensure an appropriate level of analysis.

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1 See Section 2.3 of the discussion paper for the treatment of biodiversity in these various initiatives.
S.4 Scope for biodiversity valuation

Building on the conceptual basis for understanding how biodiversity contributes to human welfare, the scope for valuation can be clearly established to focus on final goods and services. In particular:

- The (direct and indirect) use values associated with biodiversity are captured within provisional, regulating and cultural services, contributing to the values of the associated final goods and services (e.g. agricultural crops, climate regulation, recreation and aesthetic values for watching wildlife, etc.);

- Valuing intermediate or supporting services associated with biodiversity risks double-counting since these values are already embedded in the final values;

- Non-use value for biodiversity represents a legitimate additional category of value that can be added to direct and indirect use values for final goods and services.

As noted above, this only holds if the biodiversity necessary to maintain these values is sustainably managed and resilient to shocks, so that it will remain present at the necessary levels in the future. If this is the case, it is then reasonable to value service flows without considering the condition of stocks. This also means that the ‘resilience’ of biodiversity does not represent an additional value, since it is implicit within the assumption that values are maintained into the future. Therefore, whilst the importance of resilient biodiversity for resilient services should not be under-estimated, including a separate biodiversity ‘resilience value’ would also result in double-counting.

S.5 Valuation methods

Several approaches to valuation are well-suited to the reductionist perspective of separately identifying and valuing final ecosystem goods and services. This includes most market prices and revealed preference methods that can isolate the contribution of particular services to well-being through relationships with market goods (e.g. hedonic pricing and travel cost approaches). The extent to which the value of biodiversity is captured within these methods is, though, dependent on the degree to which variation in biodiversity influences demand for the market good. For the most part, the contribution of biodiversity to the provision of cultural services and final goods such as recreation and amenity is clear, but separate identification of the biodiversity component is not normally attempted. In many cases the specific role of biodiversity might be rather minor; the clearest examples where biodiversity plays a major role would be valuation of outdoor recreation that is explicitly concerned with nature watching, and in particular charismatic species (e.g. whale watching).

Other valuation methods are better suited to holistic valuations of ecosystems, or at least general environmental quality, in particular contingent valuation, especially where ‘embedding’ is an issue; i.e. where people are able to consider changes in the whole system, but find it hard to separate out specific service components because in reality they are delivered as part of an overall ‘package’.

Stated preference methods, such as contingent valuation and choice experiments, are the only way in which all the use and non-use components of biodiversity can be estimated. However, robust valuation of biodiversity using stated preference methods is challenging. The values estimated are in principle suitable for inclusion in CBA, though in practice many commentators are wary of using non-use values. This is partly because of their own concerns about the validity of these values, and partly because decision-makers may not see them as sufficiently robust. There may even be a risk...
of ‘contagion’ for other values, in the sense that if decision-makers think the non-use values are not robust, that might then lead to suspicion of valuation studies more generally, and/or a weakening of the overall conclusions.

Cost-based proxies (restoration costs, replacement costs) are also relevant but these are not consistent with the underpinning economic theory. The issue is that valuing biodiversity requires measurement of the ‘demand’ for biodiversity, while cost-based methods report the costs that would be associated with a particular action, and these have no relationship with demand. This is illustrated by the potential circularity of appraisal based on cost-proxies at the individual project level: if a habitat or services is ‘valued’ at the cost of replacing it, then a cost-benefit analysis of a decision regarding replacement of the habitat would be pointless. The estimated benefit would be identical to the cost, giving a meaningless benefit-cost ratio of 1. Regardless, cost-based proxies are quite commonly used in the absence of other evidence, since generally the services valued are a subset of those assessed, so complete circularity is avoided. Hence, a cost-based proxy may often be regarded as a better estimate than no value at all.

Another form of cost-based measure rests on the costs of meeting some pre-determined policy target. In the UK, this is used in particular for valuing greenhouse gas emissions, based on the marginal abatement costs of meeting international commitments. The rationale for this approach is twofold: firstly, the benefits of abatement are so long-term, complex (covering all ecosystems and services globally), uncertain and dependent on assumptions (such as the discount rate). Therefore, attempting to value them is an almost insurmountable challenge. Secondly, irrespective of the emissions target and the damage values, achieving efficient emissions reduction requires equalising the marginal abatement cost across all sources. In effect, the approach puts optimality concerns to one side, assumes that the policy targets have been derived in an appropriate fashion, and focuses instead on achieving these targets in a least-cost manner. A similar approach could be adopted for biodiversity, based, for example, on a ‘No Net Loss’ target or restoration targets. The method would require the estimation of the least cost solution for delivering the agreed level, taking account of any impacts on other costs and benefits (for example, impact on climate regulation services). A concern remains though, that this approach would not directly account for spatial aspects of biodiversity values since it does not reflect the demand for biodiversity.

A spatially explicit approach is demonstrated by integrated land-use models (e.g. The Integrated Model (TIM) developed in the UK NEA FO). These models explicitly account for location, allowing spatial interactions (such as availability of substitute sites) to be taken into account and allow marginal values to vary from cell to cell in the model. Here, biodiversity can be represented by various models/indicators (e.g. farmland bird index, red list species, red/amber list species, woodland species, and total number of species) that are included as spatial constraints for the optimisation of land use based on the values associated with final goods and services (e.g. various provisioning, regulating and cultural services). This approach does not directly use a biodiversity valuation, but rather, as a consequence of the analysis, it is possible to work out the (opportunity) cost of imposing the biodiversity constraint in the optimisation process; i.e. the possible gain in other services from marginally relaxing the constraint. This model output represents a ‘shadow price’ for biodiversity that could input to policy analyses, or look-up values. With this type of model it would be possible to test the opportunity costs of many different biodiversity constraints, though for look-up value purposes the relevant exercise would be to focus on the opportunity costs of current biodiversity policy targets.
S.6 Conclusions

Significant challenges are faced in valuing biodiversity for policy analyses. These are not insurmountable and, conceptually, the approach to be developed is clear and there are a number of avenues of valuation research that can be explored. These include improved estimates for non-use values by testing and further developing stated preference methods. There is also substantial potential for further developing large-scale land use optimisation models such as TIM, leading to better integration of spatial interactions in deriving values. The present challenge lies more in the limited nature of the currently available UK-focused evidence which makes reliable valuation difficult, and hence does not readily fit with the needs of look-up values. Therefore, the conclusion in this regard is that it is not appropriate to use either available welfare-based values or cost-based values as look-up values at present. More considered analysis is required when valuing impacts on biodiversity, which at the very minimum implies that formal value transfer approaches should be applied, and even these may not be sufficient if available evidence does not avoid double-counting with other valuations in an appraisal or permit for appropriate assessment of spatial factors.

Overall, further research effort is required to integrate the understanding of the role of biodiversity in contributing to human well-being into a workable approach for practical policy analyses. More effort is also required to produce robust biodiversity valuation evidence for use in appraisals, particularly via value transfer analysis. Any new evidence that is developed will need to be reconciled with existing evidence for other non-biodiversity impacts to disentangle impacts and ensure that double-counting of biodiversity values is avoided. Whether this evidence can adapted for use as look-up values or as a separate database in the future should also be explored.

It should also be further stressed that look-up values are intended for use in first-cut assessments, and for valuing secondary or incidental impacts for appraisals and assessments that might otherwise overlook environmental impacts. This approach should not be taken for full analysis where there are significant impacts on biodiversity and ecosystems - in such cases, both the scientific and economic assessments need to be more detailed, likely entailing specifically commissioned research.
1. **INTRODUCTION**

1.1 **Purpose**

This discussion paper has been prepared for the Department for Environment Food and Rural Affairs (Defra) as part of the project ‘Development of ‘look-up’ environmental value estimates for initial appraisal within cost-benefit analysis’\(^2\). The paper has two main aims:

1. To present a broad discussion on valuing biodiversity in the policy appraisal context; and

2. To provide guidance on the potential for including biodiversity values within the ‘Environmental Value Look-Up Tool’ (EVL Tool) developed as part of the project.

Valuation of biodiversity is a complex field. A first step in understanding the issues is to review how the term ‘biodiversity’ is defined and used in various contexts. A technical definition focuses on the variety of life at some scale, but often the term is used to refer to ‘the natural world’ more generally. Both conservation and valuation efforts may focus on entire systems, specific species, or other features such as ‘landscapes’ or particular services related to living organisms and systems. Similarly, biodiversity is treated in a variety of ways in ecosystems services frameworks, so care is required to avoid double- or under-counting when combining with other service values (an issue that is important in developing look-up tables). This paper therefore reviews the definition of biodiversity, its integration in ecosystem services frameworks and its inclusion as a natural capital asset in accounting frameworks, as essential background to discussion of valuation itself.

The relative merits of different approaches to valuing biodiversity and associated impacts are considered. A theoretically-correct approach for valuation for cost-benefit analysis (CBA) purposes requires the use of willingness to pay (WTP) (demand)-based metrics. However, the empirical difficulties of estimating robust WTP values for some aspects of biodiversity - in particular non-use values - may justify the use of proxy measures, notably estimates of the opportunity costs of delivering externally determined biodiversity targets, even though these are conceptually quite distinct from ‘true’ welfare values. The paper outlines the alternative options, highlights current discourse and examples (biodiversity-related and non-biodiversity where no current examples exist, such as carbon valuation) and identifies opportunities for future work, including developing an approach that fits well with other policy measures (e.g. ‘No Net Loss’). Though theoretically less consistent, in certain circumstances, target-based methods might be more intuitive and/or acceptable to a wider audience\(^3\).

The paper also suggests an approach in the interim which can inform the development of the look-up values. Two main caveats should be stressed: Firstly, this may be thought of as a ‘stop-gap’ approach, to be used if it can improve decision processes and motivate greater consideration for biodiversity impacts, with the intention that more refined, spatially-detailed and welfare-based biodiversity values will be available in the future. Secondly, this is for look-up and ‘entry-level’

\(^2\) Refer to the Technical Report for further details of project objectives and scope.

\(^3\) Carbon values based on marginal costs of achieving predefined targets provide a precedent, though for the very different situation of a uniformly mixing global pollutant, where the local details associated with emission are largely irrelevant. By contrast, biodiversity values are strongly influenced by spatial factors, though this is an issue for any biodiversity valuation, not just the use of target-based proxies.
valuations only; projects and policies with important impacts on biodiversity will call for a more rigorous consideration of the possible values at stake.

1.2 Structure

The paper is structured as follows:

- **Section 2**: reviews various definitions and understandings of biodiversity - as an ecosystem service, a function, and a fundamental natural capital asset - in order to determine what this implies for its valuation;

- **Section 3**: surveys the range of economic welfare and cost-based methods for assigning monetary valuations to biodiversity;

- **Section 4**: overviews the UK-relevant evidence base for biodiversity valuation; and

- **Section 5**: provides conclusions, including on the potential for developing look-up values for inclusion in the EVL Tool.

The accompanying annex to this paper provides a brief tabular summary of selected UK biodiversity valuation studies. This supports the evidence review that underpins the EVL Tool (see Technical Report).
2. RATIONALE FOR VALUING BIODIVERSITY

2.1 Overview

Biodiversity is defined by the Convention on Biological Diversity (CBD) as “the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part, this includes diversity within species, between species and ecosystems” (CBD, Article 2). It can be thought of as a kind of natural resource that supports the provision of ecosystem goods and services to human populations. Biodiversity is, however, often damaged by human activities and has been declining globally for some time. As with any resource, effective decision-making regarding the conservation and sustainable use of biodiversity requires consideration of the values it supports as well as the costs of conserving it.

The challenge here is to ensure that the important role of biodiversity is recognised and accounted for, while avoiding the pitfall of double-counting the same benefits. In effect, the value of biodiversity is embedded in final service values, so any separate value of biodiversity is generally not additive with these final values. The resolution of this challenge is not independent of the purpose of the assessment, its boundaries, and the valuation methods used.

The situation is further complicated by the fact that biodiversity is a natural renewable resource that can be subject to over-exploitation and decline, or to recovery and growth. Valuation and appraisal often focus on valuing flows of ecosystem services, but this can be misleading if natural capital stocks are declining or under threat due to thresholds. In such cases, it is important to consider not only the service flows, but also changes in the value of the natural capital stock.

Understanding the requirements for valuing biodiversity therefore calls for an understanding of what biodiversity is, how it fits in to ecosystem services frameworks, and the policy and appraisal contexts within which values can be useful. This section of the paper covers these background issues, prior to turning to the question of the valuation methods that can be applied to value biodiversity.

2.2 Policy and appraisal context

Biodiversity performs many functional roles in ecosystems, and can be said to underpin all ecosystem services, with evidence suggesting that both the level and the stability of ecosystem services tend to improve with increasing biodiversity (Norris et al., 2011). Past and present loss of biodiversity, coupled with increasing recognition of its importance in underpinning multiple ecosystem functions and benefits, make biodiversity and ecosystem conservation, restoration and management important public policy priorities.

However, the challenges relating to effective conservation are complex and widespread. The European Commission’s post-mortem on the failure of the 2010 target to halt biodiversity loss (Fournier et al., 2010) argued that, despite the instruments used, and the 2006 Biodiversity Strategy and Action Plan, there was: (a) incomplete implementation of existing legislation; (b) insufficient funding; (c) limited awareness about biodiversity; (d) inadequate governance and administrative capacity; and (e) gaps in skills and knowledge.

All five problems combine in hampering efforts to ‘mainstream’ biodiversity concerns across all policy sectors. This is recognised as vital for closing the implementation gap in biodiversity policy
targets (notably, see CBD 2010: Aichi Strategic Goal A). Often, multiple, apparently minor impacts in the context of wider policy areas and economic activities add up to major pressures on ecosystems. Such pressures are difficult to address, and may be completely or largely overlooked in appraisals, perhaps because taken case-by-case the impacts do not appear significant enough for appraisal effort to be proportionate; only in a cumulative overview does the significance become clear.

But, inadequate attention to biodiversity and its role in supporting ecosystem services is a more general problem. Part of this is due to lack of data and understanding, and the Millennium Ecosystem Assessment (MA: Biodiversity Synthesis) argued that, “Improved capability to predict the consequences of changes in drivers for biodiversity, ecosystem functioning, and ecosystem services, together with improved measures of biodiversity, would aid decision-making at all levels.” However, there are also problems associated with mainstreaming and recognising the non-marketed benefits of biodiversity. Atkinson et al. (2012) cite, “growing recognition that the benefits and opportunity costs associated with such services are frequently given cursory consideration in policy analyses or even completely ignored.”

This all suggests that developing look-up values for biodiversity could make their assessment easier and quicker and may contribute in some way to ensuring that these impacts are represented in appraisal. Making look-up values accessible to policy makers and appraisers in all sectors, along with clear guidance on their use, could therefore contribute to the ‘mainstreaming’ agenda. However, there are also dangers in this approach, since the valuation evidence available is far from perfect. Entry-level look-up values in particular will not reflect in any detail the spatial factors underpinning the full value of biodiversity. The values may be useful if they draw attention to biodiversity impacts and encourage thinking about ways of reducing the damages and enhancing the benefits of biodiversity, but not if they are used as a form of ‘box-ticking’ at the expense of setting biodiversity concerns to one side.

### 2.3 Biodiversity in ecosystem services classifications

Specific aspects of biodiversity are treated in many different ways by different analysts, and in various ecosystem services frameworks. Biodiversity can be defined technically in terms of variability, with multiple levels from genetic to ecosystem, and indicators such as richness and turnover. Often, however, it is more loosely equated with ecosystems or the natural world generally, or more narrowly with species richness of particular ecosystems. Biodiversity plays the role of a boundary object that is “both adaptable to different viewpoints and robust enough to maintain identity across them” (Star and Griesemer, 1989). Essentially it is agreed that biodiversity is generally ‘a good thing’ that needs to be conserved, while working with different specific realisations of the concept. This lack of precision has, however, created challenges for economic valuation. In particular, there has been no clear agreement on how biodiversity should feature in ecosystem service classifications, meaning that appraisals and assessments built on an ecosystem services framework face uncertainty regarding how biodiversity should be addressed (if at all).

There does appear to be an emerging consensus that biodiversity is important at all levels in ecosystem services, playing the role of supporting, intermediate and final service (Mace et al. 2012) but that it should only be valued for appraisal purposes at the point where it enters utility
functions\(^4\) (Landers and Nahlik, 2013). In this context, it should also be noted that what people value about biodiversity may be very different from how an ecologist might consider the contribution of biodiversity to healthy natural systems. There is evidence that people express higher values for iconic species with particular characteristics (Morse-Jones et al., 2012) and that values may not be coherent across different levels of analysis. For example, people have been found to express negative values for estuarine habitats but positive values for the birds they support (Bateman et al., 2009). This means that it is unlikely that there will be clear mapping between any technical definition of biodiversity and the constructs which people value. In turn, this emphasises the importance of following through ecosystem service analysis to find the final services valued by people and supported by biodiversity.

### 2.3.1 Millennium Ecosystem Assessment

Although it adopted the CBD definition of biodiversity with its focus on ‘variability’, the Millennium Ecosystem Assessment (MA: Biodiversity Synthesis) in effect envisaged biodiversity as synonymous with ‘life on Earth’ and as the container supporting all ecosystem services (see Figure 2.1), stating that “Biodiversity is the foundation of ecosystem services to which human well-being is intimately linked” and “Biodiversity includes all ecosystems — managed or unmanaged”. In particular, Finding #2 of the Millennium Ecosystem Assessment notes that “Biodiversity contributes directly (through provisioning, regulating, and cultural ecosystem services) and indirectly (through supporting ecosystem services) to many constituents of human well-being”. Finding #3 notes the importance of improved valuation techniques and data, in particular demonstrating that the social costs of biodiversity loss often outweigh the private gains from the activities damaging biodiversity.

\(^4\) See Section 3.1.
The Millennium Ecosystem Assessment notes that biodiversity has cultural values in its own right “because many people ascribe intrinsic value to biodiversity, and because it represents unexplored options for the future (option values)”, but this does not lead to direct representation of biodiversity as an ecosystem service in the Millennium Ecosystem Assessment framework.

2.3.2 The Economics of Ecosystems and Biodiversity

Following the Millennium Ecosystem Assessment, biodiversity does not appear in The Economics of Ecosystems and Biodiversity (TEEB) publications as a distinct ecosystem service. Habitat services of ‘lifecycle maintenance’ and ‘gene pool protection (conservation)’ are listed, along with provisioning service ‘genetic resources’, and cultural services that do not specifically reference biodiversity (de Groot et al., TEEB D0 AppC). In effect, biodiversity is treated on a level with ecosystems as the overarching natural capital from which services derive (Figure 2.2).
2.3.3 Intergovernmental Platform on Biodiversity and Ecosystem Services

The Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) considers a similar ‘biodiversity and ecosystems’ in the overarching ‘Nature’ category, anterior to any specific derived services (Figure 2.3).
2.3.4 Mapping and Assessment of Ecosystems and their Services

The (rather more tractable) EU ecosystem assessment framework (Figure 2.4) instead locates ‘biodiversity’ as the central feature of ecosystems, and then breaks it down further to a number of component parts which support ecosystem functioning and in some cases (including species and genetic diversity - the right wing of butterfly in Figure 2.4) also provide direct ecosystem services.
2.3.5 UK National Ecosystem Assessment

The UK National Ecosystem Assessment (NEA) reaches a similar end point. It adopts the Convention on Biological Diversity (CBD) definition of biodiversity, recognising that it is quite broad and encompasses many more specific interpretations. One implication is that in moving to an ecosystem services framework, biodiversity features in several places (Figure 2.5), with a separation between underpinning natural processes and the role of biodiversity as part of cultural heritage:

- As a component of fundamental ecosystem processes that underpin final ecosystem services. In the UK NEA, ‘wild species diversity’ appears alongside related concepts, in particular evolutionary processes and ecological interactions;
- Biological diversity at the level of genes and species appears as a provisioning service, specifically for bio-prospecting and medicinal plants; and
- Species diversity contributes directly to welfare as a good in itself, and appears as a cultural service, in particular for recreation.

In terms of valuation, there are risks of double-counting if biodiversity is valued separately in all its roles. To avoid this, valuation needs to focus on the final services that enter utility and production functions. Atkinson et al. (2012) argue that final services can be considered as arising through a ‘natural production function’ in which ecosystem characteristics such as biodiversity and regulating functions are important inputs. In many cases, manufactured capital is also required to generate final benefits (for example boats and angling equipment for recreational fishing). Valuation generally focuses on the final good - e.g. recreation - and it becomes necessary to unpick the production function to determine what values are attributable to the natural system, and how final values change with changes in inputs (such as biodiversity).
2.3.6 Common International Classification of Ecosystem Services

The Common International Classification of Ecosystem Services (CICES) goes further, not only excluding biodiversity but also supporting services. The report (Haines-Young and Potschin, 2013) stresses a clear distinction between final ecosystem services and ecosystem goods or products:

- Human well-being arises from adequate access to the basic materials, freedom of choice and action, health, good social relations and security. This is partly dependent on access to ecosystem goods and benefits.
- Ecosystem goods and benefits are created or derived from final ecosystem services by humans. These products and experiences “are no longer functionally connected to the systems from which they were derived.”
- Final ecosystem services, in contrast, retain a direct connection to the underlying ecosystem functions, processes and structures that generate them. They are ‘final’ as the outputs of ecosystems that most directly affect human well-being. CICES is a classification at this level; i.e. services, not benefits.
- Intermediate and supporting services are functions and processes that underpin the final services. They are not directly included in CICES because they are only indirectly consumed or used, and may simultaneously facilitate the output of many ‘final outputs’.

The exclusion of supporting services from CICES is not intended to suggest that they are unimportant. Rather, the rationale is directly connected to accounting: “if ecosystem and economic accounts are to be linked, then an essential step is to identify and describe the ‘final outputs’ from ecosystems that people use and value, so as to avoid the problem of double-counting” (Haines-Young and Potschin, 2013: 8). Though at the same time “there is no reason why fully developed environmental and economic accounts cannot also record changes in underlying ecological structures, processes and functions, and systems like CICES may well be extended to cover them” (ibid: 8) - but in physical terms, not monetary, to avoid double-counting. As such, CICES is intended to provide a framework focused on final services within which information about supporting or intermediate services can be nested and referenced. Haines-Young and Potschin (ibid)

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5 See CICES V4.3; 17/3/15 (http://cices.eu/).
argue that such treatment may be especially useful for mapping ecosystem services and propose that “CICES should be explored through the development of experimental accounts, especially in the context of using accounts to check the integrity of underlying ecological assets” (ibid, p8).

2.3.7 Final Ecosystem Goods and Services Classification System

The FEGS-CS (Final Ecosystem Goods and Services Classification System) likewise focuses on final goods and services, but with an emphasis on classifying both service and beneficiary together. Figure 2.6 illustrates how a six-digit classification code is built up. Specifically on biodiversity (§4.4.1), it is stated that there is no dispute that biodiversity is important or an ecosystem service, but it is stressed that it is not a final service “because humans do not directly interact with biodiversity” - the public “does not directly use, consume, or enjoy biodiversity - if anything, they use, consume, and enjoy the present species of flora, fauna, and fungi, and the number of species present (i.e. biodiversity) may indicate the extent to which the FEGS is provided” (Landers and Nahlik, 2013).

Figure 2.6: Final Ecosystem Goods and Services Classification System (FEGS) basic structure

Overall, there is some disagreement over whether biodiversity should be treated as an ecosystem service *per se*, as a key component of ecosystems with one or more manifestations as final ecosystem services, or synonymous with the natural system as a whole and underpinning all services. In effect, different aspects of what we term ‘biodiversity’ can be thought of as playing roles at all of these levels. Approaches that conflate biodiversity with ‘life on earth’ move away from a strict definition of biodiversity as the *variability* in life, though this holistic definition matches with the popular understanding of the term. Even a strict interpretation as variability,
however, does not meet criteria for a final ecosystem service. Ecosystem functions maintain variability (biodiversity), and that variability in turn provides resilience and supports other services; but the impacts on human welfare are all indirect and can be represented through other ecosystem services. This includes cultural services associated with recreation, heritage and other services that can be closely dependent on species and other manifestations of biodiversity.

### 2.3.8 Summary

In effect, a major contribution of the Millennium Ecosystem Assessment was to shift attention from biodiversity loss (the underlying problem) to resulting loss of services (the instrumental consequences), providing an easier route to communicating with headline policy concerns and mainstreaming across sectors. However this leads to the problem of possible double-counting if including values for biodiversity in appraisal and analysis. Biodiversity's major role in ecosystem service terms is the functioning and resilience of the system that supports all final services. Furthermore, the focus on service flows risks taking attention away from the question of whether the biodiversity 'stock' is being managed sustainably. Valuation limited to current flows is only appropriate if stocks are stable and resilient.

In the reductionist setting of an ecosystem services framework, the ambition is often to value as many final services as possible individually. There will usually be some services that cannot be valued in monetary terms, however, due to lack of physical/ecological or economic data, lack of appropriately robust valuation methods, or more fundamental ethical concerns or refusals to express certain values in monetary metrics. There are also double-counting concerns, in particular where biodiversity is concerned, since the supporting and intermediate services biodiversity provides are reflected in final services and should not be valued additionally. Where the valuation context is focused on the value of natural capital assets such as the stock of biodiversity - for example to take account of declining stocks, or for accounting purposes - more holistic approaches to whole system valuation may be appropriate. In practice, the valuation techniques may be similar, but the values are treated differently.

Some techniques are well-suited to reductionist valuations, including most market prices, and non-market techniques that can econometrically isolate particular services via the weak complementarity assumption\(^6\) in revealed preference or choice experiments. Some methods for deriving proxy values may also be focused on a specific service; for example, replacement/avoidance cost estimates, or some payments for ecosystem services (PES)\(^7\). It should be stressed that these proxy methods do not produce values in the economic sense and they are not theoretically compatible with welfare values and cost-benefit analysis: their use is therefore a second-best solution to consider where welfare values are not available (or not robust) and it is

\(^6\) The assumption of weak complementarity relies on a private good that is consumed with the non-market environmental good - for example, travel to a recreation site. The environmental good can then be thought of as the (unpriced) ‘quality’ of the private good. The private good is assumed to be non-essential, so there is some ‘choke price’ at which the consumption of the private good falls to zero. Willingness to pay for the improvement in the environmental good (the “quality”) can be estimated as the change in the compensating variation the consumer receives from the consumption of the private good.

\(^7\) The acceptability of PES as proxy values depends on how they are elicited. If they are fixed via competitive auctions, then there is some theoretical justification for arguing that they represent minimum Willingness to Accept (WTA) values from providers. More generally, PES are fixed following political decisions, which may or may not have been informed by welfare-based valuation, but are often based more on cost estimates than on assessment of benefits. So, in effect, using PES as proxy values is often tantamount to assuming that policy in question is ‘optimal’. This is generally questionable, but the approach at least has the merit of making appraisals consistent with the policy.
considered that including a proxy value is likely to lead to better decisions than including no value at all. Other valuation methods are better suited to holistic valuations of ecosystems, or at least general environmental quality, in particular contingent valuation, especially where embedding is an issue; i.e. where people are able to consider changes in the whole system, but find it hard to separate out specific service components (because in reality they are delivered as part of an overall ‘package’). Proxy methods are also relevant, including most restoration cost methods, and PES for general environmental quality or management/habitat improvements, though again it should be stressed that these methods do not produce welfare-compatible values.

The physical boundaries of assessment are also highly relevant in determining whether supporting services (and some regulating services) should be separately valued or whether to do so would involve double-counting. In principle, all supporting and intermediate services feed through to final services and therefore, should not be valued alongside final services in appraisal (they should of course be considered, just not expressed in monetary terms for summing up). However, if the final services arise outside the boundaries of the assessment it may be necessary to reflect the contribution to their value from supporting services within the assessment. This may often arise for example where the assessment is limited to a marine ecosystem that supports final services on land, or vice versa. In principle, the point is that analyses should be extended to include all value streams. In practice, this may be intractable, but the extent to which further impacts arise outside the boundaries of an assessment should always be discussed.

2.4 Biodiversity as a natural capital asset

2.4.1 Natural capital assets

The Millennium Ecosystem Assessment (Biodiversity Synthesis) notes that “A country’s ecosystems and its ecosystem services represent a capital asset, but the benefits that could be attained through better management of this asset are poorly reflected in conventional economic indicators.” As emphasised by the ecosystem service frameworks classifying biodiversity as a feature of ecosystem functioning, or synonymous with the living system, biodiversity is a fundamental part of natural capital. In many ways, it is easier to consider biodiversity as an asset (rather than as a service or process) that influences ecosystem functions, resilience and the ability to provide services over time. Resilience can be viewed as a characteristic of the stock which may affect future ecosystem service flows. Even here, there is disagreement on details, with some suggestions that resilience can be considered as a separate ‘stock’ or characteristic of ecosystems in particular configurations (Maler et al., 2009; Walker et al., 2010), somewhat separately from biodiversity as defined in the CBD, though putting this approach into practice is challenging.

Considering biodiversity as a holistic asset, rather than through its contribution to multiple individual services, also leads to slightly different valuation issues. This could be necessary where the context is one of accounting rather than CBA of a policy change or project, or where the available valuation evidence or techniques do not allow a reductionist approach. Asset values can be assessed as the net present value of flows of future benefits, though it is generally more relevant and more practical to consider changes in asset value (due to projects or policies), estimated as the difference in the present value of future services before and after the change.

In any event, whether attempting asset valuation or individual service valuation, resilience and sustainability concerns need to be considered, and the potential for both service flows and asset values to change over time should be taken into account. In principle, the same valuation approaches that apply to current ecosystem services apply to future services, though with the caveat that future generations cannot be surveyed directly and so values must be ‘transferred’
from current estimates. This assumes that the preferences of future generations will match current preferences, which is an increasingly strong assumption the longer the time horizon. Preferences are heavily dependent, *inter alia*, on cultural, technological, religious and traditional factors that can evolve in unpredictable ways. Secondly, incomes are likely to vary substantially across scenarios. Production function derived values and values based on WTP are constrained by individuals’ ability to pay, and therefore depend heavily on the distribution of incomes and property rights. Both levels and distributions could be very different from today’s, again in ways that are difficult to predict, and this would result in major differences in values. However, conserving biodiversity and other capital stocks will pass *opportunities* to future generations (i.e. preserving flexibility), which goes some way to defusing these concerns.

2.4.2 **Discounting future flows of value**

The determination of a discount rate with which to convert future flows to net present values is crucial to the valuation of natural capital. A recent US Environmental Protection Agency (EPA) expert panel of 12 economists (Arrow *et al.*, 2012) unanimously agreed that “the Ramsey formula provides a useful framework for thinking about intergenerational discounting.” However, they did not reach agreement on “how the parameters of the Ramsey formula might be determined empirically”, let alone on actual values. They explain this with reference to a long-running debate between a “descriptive” approach (based on behaviour observed in markets) and a “prescriptive” approach (focusing on ethical considerations to set parameters). So, despite its importance, there is no universally accepted way of calculating a discount rate, resulting in a multiplicity of estimates. Discount rates of a few percent, standard for short-term policy appraisal, result in substantial discounting of long-term impacts. Some authors advocate declining or hyperbolic discount rates (Kirby, 1997) to address this issue. Others use a low constant rate (e.g. the Stern Review (Stern, 2006) used 1.4%). This issue is not biodiversity-specific, and the UK has official (declining) discount rates for public sector appraisal, so this is not a central concern for look-up values, but needs to be kept in mind regarding the interpretation of estimates.

2.4.3 **Threshold effects, non-linearities and critical natural capital**

A more challenging issue lies in dealing with how the (marginal) values of biodiversity and the services it supports change as levels of biodiversity and other natural capital change. Non-linearities, threshold effects, and areas of highly inelastic demand / rapidly changing values all have consequences for valuation, both within individual studies, and in particular for attempts to transfer values across studies, for grossing-up across spatial scales, or to construct meta-analysis functions. If some part of biodiversity can be considered “critical natural capital” - i.e. a stock of essential natural capital, for which no substitutes exist, and without which human well-being faces large (non-marginal) losses - valuation can be very difficult or impossible, suggesting a need for precautionary policies and setting limits to the applicability of cost-benefit methods where catastrophic changes are plausible.

Critical natural capital is usually defined as that part of the natural environment that performs important and unique functions, and therefore ought to be maintained in any circumstances for present and future generations. The idea of critical natural capital reflects the view that there is some level of natural capital that is ‘essential’ and provides important ecosystem services that

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8 It is worth noting that market bond rates are currently significantly lower than the HM Treasury discount rate, which was introduced in 2003 when real and nominal market rates were much higher.
cannot be substituted by other forms of capital, such as human or social capital (Pearce and Turner, 1990; de Groot et al., 2003; Dietz and Neumayer, 2007). Depending on the scale, this could mean globally essential - for example continuing human life on the planet - to locally essential - e.g. a minimum level of accessible green space for mental well-being - and anything in between. Typical examples include essential ecosystem services, such as freshwater resources, climate regulation, fish stocks, wild pollinators, and fertile soils (e.g. Ekins et al., 2003).

In economic terms, critical natural capital can be conceptualised as an area of perfectly inelastic demand below a certain level of provision. It is a natural extension to consider gradually increasing demand elasticity above an absolute threshold (Figure 2.7; Farley, 2008).

**Figure 2.7: The demand curve for natural capital**

![Figure 2.7: The demand curve for natural capital](image)

Following Figure 2.7, there are limits to the use of economic methods where marginal values rise steeply, and the recognition that critical natural capital cannot be traded-off. Identifying critical natural capital is partly outside the remit of economics (a matter of biophysical science) but also partly socio-economic, depending on human demands/requirements as well as on ethical deliberation, and how minimum thresholds of acceptable outcomes are defined. The challenge here is to define both the physical and socio-economic thresholds regarding biodiversity stocks.

For example, it is possible to argue on cultural/ethical grounds, that particular sacred sites should be accorded critical status, and excluded from trade-offs, though this has nothing to do with ecology or natural functions. This can go some way to addressing the concerns relating to incommensurability of values, and/or the practical problems of estimating robust non-use values, by setting 'hands off' areas where trade-off is not permitted - for example the Natura 2000 network and Sites of Special Scientific Interest (SSSIs) - considered to fall in Region III of Figure 2.7. To the extent that it can be assumed that such critical capital will be protected by strong policy measures, from the perspective of developing look-up values it is likely to be sufficient to assume that changes are of a non-critical nature. In practice, there can also be requirements to demonstrate the benefits of strong protection policies, but for this purpose, with a clear focus on biodiversity values and high stakes, look-up values would not be sufficient in any case.
Assuming the system under investigation is in a state for which valuation is feasible and appropriate, ecosystem accounting principles imply that asset values would be measured as expected service flows, generally based on the current pattern of use (SEEA Para 2.40) unless there is strong evidence to think otherwise. Defra/ONS Principle 14.1 states that “Any departure from a constant service flow assumption would need to be justified and evidenced” (Defra and ONS, 2014). However, this appears to reverse the appropriate burden of proof in some cases (cf. the precautionary principle), in particular for exploitation of ecosystem goods and services that may not be sustainable. Where current flows are sustainable, the constant flow assumption is relatively unproblematic, although the capacity for enhanced future flows is ignored - but it is appropriate that any claim of increased future flows should be justified and evidenced.

But, current flows may not be sustainable (e.g. over-fishing, soil erosion and so on) and in these situations, the constant flow assumption could be incorrect. It would be appropriate, therefore, that the burden of proof be to demonstrate that the sustainability of a constant flow is a reasonable assumption. It may be preferable to use dynamic models of ecosystem service provision to account for possible changes and risks. Even though the level of uncertainty in these models is likely to be significantly greater than the uncertainty in current flow measurement, it does not necessarily follow that the assumption of constant flows is less uncertain, or justified.

2.5 Biodiversity supporting ecosystem services

Within the overarching ‘biodiversity’ category, different forms play different roles in supporting various services. Norris et al. (2011) report that the number of biodiversity groups playing an important role in the UK varies between ecosystem services, citing for example 3 out of 17 biodiversity groups for water quantity, 6/17 for socially valued landscapes and waterscapes, 11/17 for food production (“crops, livestock, fish”) and all 17 for wild species diversity. As Figure 2.8 (below) demonstrates, there is considerable variability in the amount of evidence, and the level of agreement within that evidence, regarding the importance of each contribution.

From a valuation perspective, two points in particular stand out: Firstly, many of the ‘higher’ biodiversity groups (fish, amphibians, reptiles, birds, mammals) are thought to make relatively little contribution to most of the regulating services, but are mostly very important for cultural services (listed as “meaningful places” and “socially valued land- and waterscapes” in Figure 2.8). These are the charismatic, visible and audible creatures that nature lovers seek out. “Cultural services” is something of a catch-all term, ranging from clearly defined and highly measureable outdoor recreation, through less clear-cut categories of aesthetic appeal and sense of place, to highly intangible discussions of the “spiritual values” of particular systems and locations. So, the robustness of evidence is highly variable, but the evidence presented in the NEA suggests that a good part of the value of these groups could be accessed via valuation of outdoor recreation, aesthetic values and non-use values for conservation. However, there is some evidence that these groups can have important ecosystem-level effects (for example, large predators controlling grazing animals) with consequent implications for some regulating services, and further investigation would be warranted.
Secondly, almost all of the groups are important for ‘wild species diversity’. This can be interpreted as contribution to the pool of genetic resources that supports and provides resilience for many ecosystem functions and services. Thus, with the possible exception of reptiles, no group can be considered of ‘low importance’ to biodiversity overall. Valuing this importance is very challenging, however, since it relates not so much to expected values of services as to their resilience and robustness under a range of future conditions - essentially a form of option value.

Measuring (changes in) biodiversity is an important step in any attempt at valuation. Mace et al. (2014) show that both extinction rate and species richness are weak metrics for assessing changes to biodiversity, that do not scale well from local to regional or global levels. This is not just a matter of incomplete knowledge about the role of biodiversity for ecosystem functioning at different scales, but a more fundamental issue that these metrics do not reflect the complex ways in which biodiversity underpins functions, services and resilience. They argue that a planetary boundary for biodiversity could be better reflected at the genetic level (“the genetic library of life”), at the functional level (via the diversity of functional types) and/or at the habitat level (biome condition and extent).
The CBD agreed on a list of provisional indicators for assessing progress towards the 2010 biodiversity target that can be implemented worldwide, or at national or regional scales at the 7th Conference of the Parties (COP7) in 2004:

- Trends in extent of selected ecosystems;
- Trends in abundance and distribution of selected species;
- Trends in status of threatened species; and
- Changes in genetic diversity.

However, indicators based on trends, well-suited to tracking progress towards targets, are not directly useful for valuation purposes. One alternative is to use composite indices, such as the Natural Capital Index (NCI) (ten Brink and Tekelenburg, 2002). This is defined as the product of the size of the remaining area (quantity) and the quality of the area, both expressed in terms of proportions of the ‘baseline’ state. Thus the NCI itself is a value between 0 and 1 (but generally expressed as 0%-100%). Quality is generally calculated by counting the average of the relative abundance of a representative cross-section of characteristic species for the ecosystem asset under consideration; alternatively, process and/or structure variables can also be used (NCI, 2002). Other indices include the GLOBIO Mean Species Abundance Index (Alkemade et al., 2009), the Living Planet Index (Loh et al., 2005), the Biodiversity Intactness Index (Scholes and Biggs, 2005) and the Norwegian Nature Index (Certain et al., 2011). These composite indicators are the result of a long tradition in ecology of expressing complex changes through indices.

From a (look-up) valuation perspective, it could be possible to derive values in terms of contributions to changes in indices, following primary valuation or value transfer to produce ‘benchmark’ values. Changes to condition and extent of biomes would be relatively easier to assess than genetic or functional impacts in almost all cases. The complexity lies rather in the relationship between biome condition/extent and the value and resilience of services supported. For UK purposes, the unit of assessment could be broad habitats, and the questions of interest would revolve around changes to the extent, integrity and condition of areas of these habitats.

A more spatially explicit approach is taken by Bateman et al (2014) in the UK NEA FO ‘TIM’ (The Integrated Model; Figure 2.9). TIM is an “integrated, modular, optimizing approach” to multiple land uses. This approach represents a major advance on more traditional partial equilibrium analysis and standard appraisal methods, that look at alternative uses of resources at a specific place, but do not fully account for substitute sites and other spatial interactions. In TIM, biodiversity is represented by various models/indicators of biodiversity (including the farmland bird index, red list species, red/amber list species, woodland species, and total number of species) that are implemented as spatial constraints on land-use in an optimisation model; i.e. the indices must not decline within each grid cell in the model. This does not directly use any valuation, however, as a consequence of the analysis, it is possible to work out the (opportunity) cost of imposing the biodiversity constraint in the optimisation process; i.e. the possible gain in other services from marginally relaxing the constraint, and this could be interpreted and used as a ‘value’.

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The existing TIM implementation has a limited set of biodiversity constraints, though these could be expanded. The important innovation in TIM is that it is spatially sensitive and demonstrates the consequences (and, potentially, values) of policies in each specific area – at present, for each 2km square across GB. Therefore, TIM has considerable potential as a basis for improved look-up values for biodiversity.

2.6 Summary

The above discussion reveals many conceptual issues surrounding the definition, measurement and valuation of biodiversity that are still live topics of academic research and debate. Nevertheless, some general lessons can be extracted for the purposes of developing look-up values:

- There are clear concerns regarding the potential for double-counting when considering biodiversity values, since biodiversity plays a role at all levels in ecosystem services frameworks. Generally, the best approach is to value biodiversity only where it enters final utility or production functions; i.e. final ecosystem services.

- At the same time, giving proper consideration to the importance of biodiversity and ecosystem protection and resilience makes it vital to understand the links from biodiversity to final services. The approach of monetising only the final services (to avoid double-counting) does not imply that the supporting services should not be considered.

- Valuation can focus on current flows only on the assumption that biodiversity ‘stocks’ are sustainably managed and resilient to the combination of exploitation and external shocks. If biodiversity is in fact declining or under threat, these potential future losses must be taken into account.

- Practically, one way to achieve this is to set limits on the applicability of valuation methods to recognise the existence of biodiversity that is “critical natural capital” that cannot be traded.
off. Of course, any proposal that threatens critical natural capital should not be dealt with using an entry-level / look-up value approach in any case, and accompanying guidance should clearly explain what these limits are.

For look-up value purposes, defining values based on indicators of ecosystem health or biodiversity is a promising avenue, provided care is taken to avoid double-counting. With appropriate benchmarking, this could be applied at the broad habitat level. However, a key challenge for biodiversity valuation is to recognise the dependence of values on specific spatial factors. Models such as TIM, that explicitly address spatial factors and allow biodiversity constraints and values to be locally specific, therefore have great potential as a mid-term solution.
3. VALUING BIODIVERSITY: METHODS

3.1 Role of economic valuation

Economics is founded on the principle that “scarcity implies choice” (Robbins, 1935); it is not possible to achieve all objectives simultaneously, so trade-off, whether implicit or explicit, is inevitable (Costanza et al., 2011). Economic valuation is concerned with assessing trade-offs in explicit terms (Farber et al., 2002). Burkhard et al. (2013) argue that, while multiple indicators can be used for assessing trade-offs across different ecosystem services, the analysis can be easier if these are standardised to a common indicator, generally economic value.

Most valuation effort is expended in the context of policy and project appraisal, with economic valuation methods used to estimate marginal values for changes in service provision in a static and partial equilibrium setting. There has been increasing interest in larger-scale mapping of ecosystem service values, with a wide variety of techniques (Crossman et al., 2013), and in analysing the macroeconomic implications of changes in ecosystem goods and services, including the UK NEA FO scoping study (Anger et al., 2014).

One reason for the use of many different approaches is that there are also different purposes and uses for valuation evidence, including ecosystem accounting, policy and project appraisal, awareness raising and so on. Each of these may call for different specific methods and coverage, and different requirements for accuracy and research expenditure commensurate with the context. A focus on trade-offs, comparisons of ‘states of the world’ and what may be lost or gained from decisions is more policy-relevant than absolutes estimates, which make for catchy headlines but “have no specific decision-making context” (Costanza et al., 2014). Taking account of relationships and feedbacks at broad scales can help to defuse the objection that multiple projects change prices and substitute sets in ways that conventional appraisals overlook (Hoehn and Randall, 1989). Increasingly, attention is turning also to environmental and ecosystem accounting, calling for different types of value, and many policy assessments consider economic impacts (contributions to gross value added and employment) as well as, or instead of, welfare-based estimates.

While monetary valuation has been controversial, this can be interpreted in the context of gradual progression in the framings of human-environment interactions. On a practical level, Mace (2014) recognises that most environmental decisions are made on the basis of economic arguments, arguing that refusing to engage with valuation risks further marginalisation of nature from decision-making: “If the benefits provided by nature are assigned no value, they are treated as having no value, and current trends in the decline and deterioration of natural systems will continue.” At the same time, strongly reductionist approaches to valuation are set in a ‘nature for people’ framing that is most likely to elicit rejection on principle. A softer ‘people and nature’ framing is more acceptable, and it is towards this that many initiatives (such as IPBES) are tending, but this represents a challenge for existing valuation methods.

The various groups of valuation methods are summarised in Table 3.1 and discussed in Section 3.2 and 3.3.
Table 3.1: Valuation methods

<table>
<thead>
<tr>
<th>Family and methods</th>
<th>Description</th>
<th>Suitability for valuing biodiversity</th>
</tr>
</thead>
<tbody>
<tr>
<td>Market-based techniques: <strong>Market prices</strong>  <strong>Production functions</strong></td>
<td>Market prices are rarely equal to economic values since they do not include consumer surplus but do include resource costs. Market information may require substantial analysis to deliver usable values; for example correcting for taxes and subsidies, or estimating how values change with quantity.</td>
<td>Capture extent to which biodiversity supports marketed services, but not necessarily resilience. Very limited use for other biodiversity values, though price premiums on some “green” products (e.g. Marine Stewardship Council (MSC) fish) could reflect non-use values for conservation.</td>
</tr>
<tr>
<td>Revealed preference</td>
<td>Methods based on values for environmental resources that are ‘revealed’ by behaviour in associated markets.</td>
<td>Applicable to use values for recreation and potentially aesthetic values, though again these are generally valued under those service categories without splitting out biodiversity.</td>
</tr>
<tr>
<td>Stated preference</td>
<td>Methods based on surveys in which people express preferences through responses to hypothetical payment questions or choices about alternative states of the world.</td>
<td>Applicable to any good or service, including biodiversity, and capable of capturing non-use values. However, double-counting is a risk, in particular due to embedding/part-whole bias.</td>
</tr>
<tr>
<td>Cost-based techniques</td>
<td>Proxies that do not assess economic value, but rather the costs that are avoided due to some ecosystem asset, or the costs that would be incurred to replace or restore the asset.</td>
<td>Widely applicable to restoration of ecosystems and potentially where targets for conservation and restoration exist. Risk of double-counting if these are combined with values of services supported by the systems.</td>
</tr>
<tr>
<td>Expenditure measures</td>
<td>Measure expenditure, not economic value; the bases of estimating regional economic impacts through input-output modelling and multipliers.</td>
<td>Commonly used in the case of nature-based recreation and tourism, though generally this is valued as such, without splitting out a ‘biodiversity’ component (e.g. from ‘landscape). Not commensurate with TEV (Total Economic Value) values but useful for other purposes.</td>
</tr>
<tr>
<td>Value transfer</td>
<td>Not a valuation method, but a means of allowing existing value evidence to be applied to new cases, with more or less sophisticated adjustments, avoiding the cost and time required for primary valuation studies.</td>
<td>Applicable but dependent on availability of suitable source studies from one or more of the above categories.</td>
</tr>
</tbody>
</table>
3.2 Welfare-based methods

The estimation of economic value is usually based on ‘willingness to pay’ (WTP), a monetary expression of how individuals are willing to trade-off across different goods and services. In practical cases, it is also necessary to aggregate these values across individuals, to provide a monetary measure of society’s preferences for alternative uses of its scarce resources. In effect, social preferences are taken as the aggregate of the individual preferences of members of society.

3.2.1 Total economic value

The concept of total economic value (TEV) (Figure 3.1) encompasses WTP for any item that enters an individual’s welfare (utility) function. The TEV framework is not inherently selfish, despite being based on preferences, since individuals often have altruistic preferences, and indeed express these through charitable donations and so on. Thus, non-use values of biodiversity, including existence and bequest values, are covered by the framework. Nevertheless, the TEV framework is inherently anthropocentric and is focused on individual preferences not social goals. Other ethical systems can give rise to different evaluation frameworks and decision rules.

Figure 3.1: Total Economic Value framework

Economic valuation methods seek to determine individuals’ preferences, whatever the individual’s tastes, motivations, status or knowledge - though in practice, most applications will use averages for a representative group of individuals, rather than identifying impacts for each individual. The strength of an individual’s preference is measured in terms of their WTP to secure some change (or their willingness to accept (WTA) compensation for giving something up). The basic idea is that the more positively (or negatively) individuals are affected by a change, the more of their finite income and wealth they will be willing to trade-off in order to secure (or prevent) the change. This approach to assigning value is relatively straightforward and can be applied to a very diverse range of goods and services. Resulting monetary valuations can be compared and aggregated across individuals.
3.2.2 Market-based techniques

Market prices

For most economic goods and services, preferences are revealed via the markets in which individuals purchase goods and services, and sell their labour. Prices and quantities are observable features of markets, but a distinction must be made between value and price. Price is what an individual actually pays for a good or service; value is the impact on that individual’s utility, and can be conceptualised as the maximum price that person would have been willing to pay. The locus of maximum willingness to pay (WTP) across all consumers constitutes the ‘demand curve’ for a good or service. Consumer surplus is the difference between WTP and price, and is a central concept in the calculation of TEV (but should, in principle, be omitted from exchange values used in environmental/ecosystem accounting\(^1\)).

If markets exist, and there are no market failures, prices observed in the market reflect WTP for marginal quantities. For larger quantities, it is necessary to estimate the demand curve, since WTP changes with quantity. This market approach to valuation works well for perfectly competitive markets. However, various ‘market failures’ mean that markets do not always reflect true values. These include, in particular:

- Externalities: economic activity creates pollutants or other external damages (or in some cases external benefits) that are not fully considered by the actors creating them;
- Imperfect competition: market power (e.g. monopoly) allows some market participants to manipulate prices for their own benefit;
- Imperfect information: choices are made without full information, leading to ‘mistakes’ in the sense of choices that do not reflect true values;
- Transaction costs: the administrative, mental, financial costs of carrying out a mutually beneficial trade are higher than the value of the trade, so it does not take place; and
- Missing markets: where property rights are not defined, there can be no market and, therefore, no forum for expression of values.

So, while market prices guide the behaviour and choices of individuals, they do not always reflect ‘true’ economic values. In appraisal, prices often need to be adjusted to correct for market failures in deriving better measures of social benefit. For example, non-market valuation techniques may be used to estimate values for impacts not (usually) traded in markets, such as regulating services.

Markets exist for many provisioning ecosystem services in particular (food, timber) and the contribution of biodiversity to these services may be reflected through these markets. There may be a need to correct for market failures (such as agricultural subsidies and pollution) and also to subtract costs of labour, marketing and manufactured capital but, for present purposes, this can be left to one side, to be dealt with under the look-up values for provisioning services. The relevant point here is that the direct contribution of biodiversity is reflected through these markets. However, there may be biodiversity values that are not reflected in this way. Market prices cover only direct use values and so do not account for any non-use values associated with biodiversity. Moreover, biodiversity brings resilience/robustness to the provisioning system. The value of this

\(^{11}\) See accompanying ‘Applying values in ecosystem accounting’ discussion paper prepared as part of this project’s outputs.
resilience, in effect an option value, is unlikely to be directly represented in most appraisals or modelling and is difficult to identify.

**Market price proxies**

Market proxies can be used for some goods, where there is no direct market but there is a market in a closely related good. For example, home grown, gathered or hunted food could be valued using market prices for equivalent goods, thereby capturing some of the value of biodiversity in underpinning these services; though in the UK, the main value of these services is likely to be recreation rather than sustenance. Again, these values only capture direct use value.

**Production function approach**

Production functions use statistical analysis to determine how changes in some ecosystem function affect production of another good or service which is a traded resource\(^\text{12}\), or which can be valued using another technique. The primary difficulty in this method is the availability of scientific knowledge and/or data, necessary to allow estimation of the production function. It may be important to account for non-linear relationships between value and area, for example in wetlands providing flood defence services. Models such as InVEST\(^\text{13}\) (Natural Capital Project\(^\text{14}\)) are based on production function modelling, linking spatially explicit maps of habitat types to specific ecosystem service outputs. Models such as ARIES\(^\text{15}\) rely instead on Bayesian belief networks linking specific land use-land cover categories to service provision, but do not directly model local ecosystem functions. The values captured by a production function approach depend on the valuation method used for the output of the production function - often, this is a marketed good or service (so only direct use values are captured) but production function approaches can also be used with non-market valuation for the outputs, in which case potentially all categories of TEV could be covered.

**Cost of illness**

Cost of illness methods are a particular class of production function where environmental services are linked to health measures, as part of estimating the health damage of pollution, or the health benefits of a clean environment. To give a monetary value, the health impacts need to be valued using additional methods, such as the avoided costs of treatment that is rendered unnecessary by the management intervention, and/or estimates of WTP to avoid illness. These methods could be important in valuing physical and mental health benefits associated with outdoor activities and clean environments, and biodiversity underpins these services. But, again, the biodiversity link is indirect. The recreation and health valuations will capture some part of biodiversity value, but there is generally no requirement to include a separate assessment of the biodiversity component of these values.

**3.2.3 Revealed preference methods**

Revealed preference methods are a formal approach for determining economic values from observed market behaviour. These exploit the relationships that exist between the demand for

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\(^\text{12}\) For example Fezzi et al. (2014) use a production function approach to examine and value the impact of a changing climate on UK agriculture.

\(^\text{13}\) InVEST: Integrated Valuation of Ecosystem Services and Tradeoffs.

\(^\text{14}\) \url{http://www.naturalcapitalproject.org/invest/}

\(^\text{15}\) ARIES: Artificial Intelligence for Ecosystem Services.
some market-priced goods and preferences for related non-market goods and services (e.g. environmental resources). Two fundamental requirements are that: (i) changes in the provision of the non-market good actually influence some observable customer behaviour (i.e. demand for the market good); and that (ii) changes in demand are at least partly (and to an observable extent) responses to changes in the provision of the non-market good. Without this variation it is not possible to infer the value of the non-market good. These caveats limit the scope for applying revealed preference methods. In practice there is only a distinct set of circumstances where they are satisfied such that expenditure on some market good of interest can be observed to vary with the provision of the non-market good (e.g. demand for property, demand for recreation, and demand for products that compensate for losses in the provision of some environmental services).

There are three basic relationships of interest, which centre on the benefit that an individual ordinarily derives from the provision of the non-market good:

- The market and non-market goods are substitutes: an individual can derive the same benefit from consumption of either the market good or the non-market good, which is the basis for avertive behaviour approaches;
- The market and non-market goods are complements: an individual requires the joint consumption of both the market good and the non-market good to derive the benefit. This is the underlying principle for the ‘traditional’ travel cost method, which examines demand for visits to recreation sites where time and money spent on travel to a site is complementary to recreation; or
- The non-market good is an attribute of the market good: the non-market good is a characteristic or an attribute of a marketed good. The classic examples are demand for housing, where along with the characteristics of properties, the attributes of the local neighbourhood and environment are key determinants of demand, and choice of recreation site to visit, where the characteristics of the site are key determinants of demand. This attribute relationship provides the basis for hedonic pricing approaches and random utility models (alternatively called ‘discrete choice models’), respectively.

The extent to which the value of biodiversity is captured within these methods is dependent on the degree to which variation in biodiversity influences demand for the market good. For the most part, the contribution of biodiversity to the provision of cultural services and final goods such as recreation and amenity is clear, but separate identification of the biodiversity component is not normally attempted. In many cases the specific role of biodiversity might be rather minor. Hedonic analysis of property markets, for example, could detect values associated with high environmental quality, but it would be difficult to split out biodiversity separately. The clearest examples where biodiversity plays a major role would be valuation of outdoor recreation that is explicitly concerned with nature watching, and in particular charismatic species (e.g. whale watching). Only the direct use value is captured, since revealed preference methods do not assess option or non-use values.

3.2.4 Stated preference methods

Stated preference methods obtain values for non-market goods by asking choice or direct valuation questions to individuals via survey interviews. The questionnaire or ‘survey instrument’ presents a hypothetical market to the respondent in which they can trade-off the provision of a non-market good(s) of interest with their household income, thus providing an estimate of WTP. The two most commonly applied techniques are contingent valuation and choice experiments. The former is characterised as an approach that obtains the respondent’s maximum WTP for a specified (discrete) change in the level of provision of the good, which could for example be a ‘bundle’ of final ecosystem goods, via a direct question. In the latter case, individuals are presented with alternative bundles or goods that are differentiated by a set of characteristics (attributes and their
levels), and are asked to choose their most preferred alternative. The choice experiment approach is equivalent to the random utility model in the revealed preference methods family, except that it relies on stated preference data (rather than observed market data).

The flexibility of stated preference methods, through which any conceived hypothetical market can be presented to individuals, conceptually at least implies that it is suited to all non-market goods or services including the valuation of biodiversity. In particular, it represents the only set of methods that can capture the non-use value component of biodiversity as a distinct final good. However, there are challenges in constructing credible valuation scenarios and ensuring that individuals understand the trade-offs that are presented and WTP values are valid/robust (see Section 4). In particular, problems of embedding can arise, whereby respondents find it difficult to focus on the specific items under consideration in the survey instrument. Often this is because they consider that the hypothetical changes postulated in the valuation scenario would inevitably come bundled with other impacts. For example, a survey may focus on protecting a specific charismatic species, but respondents may think in terms of protecting the habitat that is essential for the species to survive, and all the wider conservation and other benefits that that would entail.

Similar problems arise with scope insensitivity, where values expressed do not vary as expected with the scale of provision. For example, values expressed for protecting 10ha, 100ha and 1000ha of some rare habitat might be very similar. This can be related to ‘good cause’ motivations, with respondents expressing a willingness to contribute to the good cause rather than a clearly defined, preference-based trade of a certain amount of money for a specific change in biodiversity. Conversely, there can be ‘protest bids’ (either unrealistically high, or non-genuine zeros) associated with respondent rejection of the presumed property right, or refusal to make a trade-off about an environmental asset. There are also concerns about ‘hypothetical bias’, whereby respondents over-state WTP compared to what they would actually be willing to pay in a real situation.

These issues mean that it can be difficult to understand exactly what the stated preferences represent. However, careful study design and participant debriefing can go some way to addressing these concerns, and considerable effort has been expended in improving the methods. Loomis (2014) for example presents a number of ex-ante and ex-post strategies for overcoming hypothetical bias. Methods, biases and possible solutions are a wide subject, discussed for example by Ninan (ed) (2014).

In summary, stated preference methods are the only way in which all the components of TEV can be estimated for biodiversity, but robust valuation of biodiversity using stated preference methods is highly challenging with many pitfalls. The values estimated are in principle suitable for inclusion in CBA though in practice many authors are wary of using non-use values because of their own concerns about the validity of these values, and because decision-makers may not see them as sufficiently robust (with a risk of ‘contagion’ for other values). This may be relevant to the development of look-up values in that, the look-up tables are likely to be seen as a package, so including values seen as less robust under one category could impact users’ perceptions of the robustness and validity of the look-up values as a whole.
3.3 Cost-based methods

3.3.1 Replacement costs and avoided costs

Cost-based proxies are not consistent with the economic theory underpinning TEV. The issue is that TEV requires measurements of WTP, while cost-based methods report the costs that would be associated with a particular action, and these may be greater or less than the WTP. This is perhaps best illustrated by considering the potential circularity of appraisal based on cost proxies: if a habitat or service is ‘valued’ at the cost of replacing it, then a cost-benefit analysis of a decision regarding replacement of the habitat would be pointless. The estimated benefit would be identical to the cost, giving a meaningless benefit-cost ratio of 1. Nevertheless, these proxies are quite commonly used in the absence of other evidence. Generally the services valued are a subset of those assessed, so complete circularity is avoided, and on the grounds that including a cost-based proxy may often be a better estimate than the default value of zero.

Avoided cost methods (i.e. avertive expenditure) value an ecosystem service through the increase in costs that would be incurred if those services were no longer available/delivered. Generally this is a lower-bound estimate, since there may be additional levels of benefit beyond the costs avoided.

Replacement cost methods estimate a value based on the cost to replace an ecosystem function or service. This can be applied to entire ecosystems (for example, the cost of providing new habitats to compensate for habitat losses) or more often to replace specific ecological functions with human engineered alternatives; e.g. the cost of wastewater treatment plants instead of wastewater processing by natural systems such as saltmarshes. This is often a lower-bound estimate, if the service is essential and would have to be replaced, but can be an over-estimate where the costs of replacement are higher than the consequences of losing the services.

3.3.2 Target-based costs

Another form of cost-based measure rests on the costs of meeting some pre-determined target. In the UK, this is used in particular for valuing greenhouse gases, based on the marginal abatement costs of meeting international commitments16. The rationale for this approach is twofold. Firstly, the benefits of abatement are so long-term, complex (covering all ecosystems and services globally), uncertain and dependent on assumptions (such as the discount rate). Therefore, attempting to value them is an almost insurmountable challenge. Secondly, irrespective of the emissions target and the damage values, achieving efficient emissions reduction requires equalising the marginal abatement cost across all sources. In effect, the approach puts optimality concerns to one side, assumes that the policy targets have been derived in an appropriate fashion, and focuses instead on achieving these targets in a least-cost manner.

A similar approach could be adopted for biodiversity, based for example on a ‘No Net Loss’ target, or restoration targets. Visions, targets and indicators have been agreed at international, European and national levels (Table 3.2). Although in principle economic analysis should always seek to estimate values associated with a given change, this is not always feasible. If the theoretically correct welfare-based values are not available, or are insufficiently reliable for use in a CBA,

analysts may need to look at other indicators of value. If there are political agreements regarding some socially desirable level of provision (such as no net loss), then there is some justification for using costs associated with this level of provision in lieu of true values. The method would require the estimation of the least-cost solution for delivering the agreed level, taking account of any impacts on other costs and benefits (for example, impact on climate regulation services). This approach may not directly account for spatial aspects of biodiversity values. However these could be ‘bolted on’ by applying the target or restriction to each region (or other scale) individually.

Table 3.2: Visions, target and indicators for biodiversity at different scales

<table>
<thead>
<tr>
<th>Scale</th>
<th>2020</th>
<th>2050</th>
</tr>
</thead>
<tbody>
<tr>
<td>International: Aichi (CBD 2010)</td>
<td>Take effective and urgent action to halt the loss of biodiversity in order to ensure that by 2020 ecosystems are resilient and continue to provide essential services, thereby securing the planet’s variety of life, and contributing to human well-being, and poverty eradication.</td>
<td>By 2050, biodiversity is valued, conserved, restored and wisely used, maintaining ecosystem services, sustaining a healthy planet and delivering benefits essential for all people.</td>
</tr>
<tr>
<td>EU (European Commission 2011)</td>
<td>Halt the loss of biodiversity and the degradation of ecosystem services in the EU by 2020, and restore them insofar as is feasible, while stepping up the EU contribution to averting global biodiversity loss.</td>
<td>European Union biodiversity and the ecosystem services it provides - its natural capital - are protected, valued and appropriately restored for biodiversity’s intrinsic value and for their essential contribution to human well-being and economic prosperity, and so that catastrophic changes caused by the loss of biodiversity are avoided.</td>
</tr>
<tr>
<td>England (Defra 2011)</td>
<td>Halt overall biodiversity loss, support healthy well-functioning ecosystems and establish coherent ecological networks, with more and better places for nature for the benefit of wildlife and people.</td>
<td>By 2050 land and seas will be rich in wildlife, biodiversity will be valued, conserved, restored, managed sustainably and be more resilient and able to adapt to change, providing essential services and delivering benefits for everyone.</td>
</tr>
</tbody>
</table>

The vision statements set out in Table 3.2 generally support a ‘no net loss’ approach, but also a non-specific ambition to restore degraded systems and services. More detailed targets are set out in strategies. For England, for example, ‘Biodiversity 2020: A strategy for England’s wildlife and ecosystem services’ (Defra, 2011) sets targets for Outcome 1 (Habitats and ecosystems on land, including freshwater environments) calling for improvements by 2020 that will ensure:

- 1A. Better wildlife habitats with 90% of priority habitats in favourable or recovering condition and at least 50% of SSSIs (Site of Special Scientific Interest) in favourable condition, while maintaining at least 95% in favourable or recovering condition;

- 1B. More, bigger and less fragmented areas for wildlife, with no net loss of priority habitat and an increase in the overall extent of priority habitats by at least 200,000 ha;

- 1C. By 2020, at least 17% of land and inland water, especially areas of particular importance for biodiversity and ecosystem services, conserved through effective, integrated and joined up approaches to safeguard biodiversity and ecosystem services including through management of our existing systems of protected areas and the establishment of nature improvement areas; and
• 1D. Restoring at least 15% of degraded ecosystems as a contribution to climate change mitigation and adaptation.

These targets are numerically specific meaning that in principle the marginal cost of achieving them could be estimated and used as a proxy for the value of different improvements. The potential for using TIM (The Integrated Model) and similar models in this regard has been highlighted above (see Section 2.4). It should be noted, however, that these changes would result in multiple benefits, not only biodiversity-related. There is, therefore, a risk of double-counting in any assessment which also included estimates of improving these other services, though this should be possible to avoid in one of two ways: The simpler option is to exclude values for the additional benefits. However, this has the weakness that these benefits could well be greater than the costs (it would be hoped that expenditures would have benefit cost ratios greater than one), and it would seem preferable to include all benefits that can be monetised, reserving use of cost proxies for benefits that cannot be monetised. The second option is to break the cost estimates down across benefit categories, in order to isolate the marginal cost associated with biodiversity provision. The drawback here is that the breakdown of costs across categories will be somewhat arbitrary, but this should be viewed in the context of the use of proxy values and the modest level of ‘accuracy’ required. For look-up values, the important point is that guidance should be clear and unambiguous regarding what values can and cannot be combined in any given analysis.
4. VALUING BIODIVERSITY: EVIDENCE

Following from the preceding discussion, the task is to establish the different aspects of biodiversity that could be valued, and determine how to include these in assessments, while avoiding the potential for double-counting.

The Common International Classification of Ecosystem Services (CICES) focuses on avoiding double-counting, while simultaneously exploring ways of reflecting the supporting function role of biodiversity, which is clearly important from this perspective. There is a need for care and clarity regarding the objects of valuation, their measurement, and the valuation techniques used. The UK NEA, for example, includes both ‘environmental settings’ and ‘wild species diversity’ under cultural headings, citing recreation as an example for both. There would be a strong risk of double-counting in any attempt to assess these recreational values separately. Equally, there are non-use elements of cultural values that would not be captured in assessments of recreation alone.

Hence, the TEEB database (Van der Ploeg and de Groot, 2010) does not include ‘biodiversity’ as a service directly, but rather includes ‘genepool’ (biodiversity protection) service and ‘genetic resources’ service. Other manifestations of biodiversity are ‘hidden’ within other categories, such as cultural services. If it is accepted that the use values of biodiversity and wildlife are (in principle) captured under other categories - i.e. recreation and aesthetic values for uses involving watching wildlife, and provisioning or regulating services for other direct or indirect uses of biodiversity - then, as a supporting/intermediate service underpinning these values, it would be double-counting to include the same values again under a separate ‘biodiversity’ category. But other values, for example for non-use, could legitimately be included there.

It must be stressed, however, that this argument only holds if in fact the biodiversity necessary to maintain these services is sustainably managed and resilient to shocks, so that it will remain present at the necessary levels in the future (as discussed in Section 2.4). Then it is reasonable to value service flows without considering stock values. On the other hand, if it is expected that biodiversity will decline, it is necessary to reduce estimates of the future value of final services supported by biodiversity, reducing asset values (and increasing the marginal value of conservation) - although in practice scientific knowledge to do this may be lacking, and the complexities of attempting to value changes in stocks may call for simplified assumptions, as discussed in Bateman et al. (2011).

A related but slightly separate point concerns the role of biodiversity in providing resilience for final services. Resilience is not directly reflected in the expected value of service flows, unless it is modelled directly, though it is often indirectly reflected via the assumption that these flows can be maintained in future. It is moot, therefore, whether this value needs to be separately identified - it would be double-counting to assume that services are maintained (due to resilience) and also to value the resilience. The importance of biodiversity in providing resilience should, however, be recognised, and this returns to the point above, that if resilience declines, this should be reflected through reduced future service and asset values.

What remains, then, is the non-use component of existence, altruistic and bequest values associated with biodiversity conservation. This component can be difficult to measure in physical or monetary terms. Following from Section 3.2, non-use values generally need to be estimated via stated preference techniques, though the UK NEA also advanced the use of actual bequests as a proxy for bequest values (see for example Atkinson et al., 2012). Recent UK-based evidence is reviewed below.
4.1 Literature evidence

4.1.1 Valuing biodiversity through stated preference methods

The evidence review undertaken as part of the wider project to develop the ‘Environmental Value Look-Up Tool’ (EVL Tool) identified a number of valuation studies that examine the cultural services component of biodiversity, particularly the non-use aspect in terms of existence, altruistic and bequest values associated with conservation (see Annex for a summary of a selection of studies). The following focuses primarily on the UK evidence base in keeping with the remit of the EVL Tool. Whilst the larger international literature has much methodological relevance, a major review is beyond the scope of this paper.

The work of Willis et al. (2003) for the Forestry Commission provides a useful starting point for the review. This study draws on and extends the analysis of Garrod and Willis (1997), to estimate non-use values for improvements in UK forest biodiversity. They present estimates of £0.35 per household per year for enhanced biodiversity in each 12,000 ha (1%) of commercial Sitka spruce forest, £0.84 per household/year for a 12,000 ha increase in Lowland New Broadleaved Native forest, and £1.13 per household/year for a similar increase in Ancient Semi Natural Woodland. They note that these values are “particularly difficult to capture” through stated preference methods (the only option) since:

- Individuals can have widely different preferences for wildlife, leading to high variation in mean WTP;
- Variation in WTP is primarily driven by taste rather than income (WTP for biodiversity is a very small fraction of income), so the variation across individuals is difficult to explain (noting that this holds only within the expenditure range - it would not be expected to hold for higher expenditure levels); and,
- Biodiversity is a complex issue and individuals find it difficult to understand and trade-off across different components. In effect, preferences need to be ‘formed’ in order to complete valuation questionnaires, and this is difficult, casting doubt on the interpretation of results.

These conclusions largely set the tone for subsequent UK studies that have made similar attempts to value biodiversity, or ecosystem health generally. This includes a series of research studies for Defra that have directly sought or included aspects of biodiversity value in stated preference surveys (e.g. Christie et al. 2004; Christie et al. 2011; GHK Consulting, 2011).

Christie et al. (2004; 2006) test the use of stated preference methods to value biodiversity, with a particular focus on the issue of respondents’ understanding and familiarity with biodiversity concepts. A standard respondent survey approach (approx. 750 respondents) is compared to a workshop approach (approx. 50 respondents) which permits more time for respondents to become familiar with the explanatory material concerning biodiversity and to discuss their understanding with other participants. A range of policy scenarios concerning farmland in Cambridgeshire and Northumberland were examined. Using a choice experiment approach, biodiversity is described to respondents in terms of the following characteristics: ‘familiar species of wildlife’, ‘rare and unfamiliar species of wildlife’, ‘habitat quality’, and ‘ecosystem processes’. Valuations are provided for improved/enhanced levels of these characteristics against a status quo of ‘do nothing’ and implied degradation. Contingent valuation scenarios are also tested for specific policy

17 Refer to the Technical Report for further details on the evidence review.
scenarios: biodiversity enhancements associated with agri-environment measures; biodiversity enhancements associated with wildlife habitat re-creation; and biodiversity loss associated with land development, such as house building.

The results reported by Christie et al. (2004; 2006) generally conform to prior expectations. The choice experiment results indicate respondents’ preferences for improving habitats and ecosystem processes, and protecting familiar species. Preferences with respect to rare, unfamiliar species are less clear - ensuring recovery of these species is valued positively, but an intermediate step of slowing the rate of decline from the current level of degradation is not. Analysis of the workshop approach indicates a learning effect in respondents’ understanding of biodiversity, but the small sample size makes it difficult to draw comparisons to the conventional survey approach. WTP estimates from the contingent valuation exercise are in the range of £40 - £70 per household across the set of policy scenarios, for the Cambridgeshire and Northumberland sub-samples, and for the most part there is no statistically significant difference in the range of mean WTP estimates. The key conclusion that Christie et al. draw from this is that people do place a positive value on biodiversity enhancement, but the scope of the improvement and how this is achieved does not necessarily matter. Such a finding tends to imply presence of the embedding and scope insensitivity issues noted in Section 3.2.4.

Christie et al. (2011) estimate the value of changes in biodiversity and associated ecosystem services that are expected to result from the delivery of the UK Biodiversity Action Plan (UK BAP) over the period 2010 - 2020. The analysis attempts to capture the multiple dimensions through which people benefit from biodiversity, including via provisioning services (e.g. wild food), cultural services (the appreciation of wildlife and experiencing of the natural environment), and the role of biodiversity in supporting and regulating services (e.g. carbon storage and flood alleviation). A choice experiment approach was applied and implemented via respondent workshops (600 respondents). Biodiversity was described in terms of wild food, non-food products, climate regulation, water regulation, sense of place, charismatic species and non-charismatic species. Levels of provision ranged from ‘no further BAP funding’ to ‘present BAP implementation’ and ‘full BAP implementation’. The likely provision of ecosystem services across UK BAP habitats was then assessed via an expert-weighting exercise in order to use the study’s results to value the benefits of the UK BAP.

A range of valuation estimates are provided by Christie et al. (2011), both for biodiversity improvements within (the respondent’s) own region and the rest of the UK. Values for different treatments for the level of information provided to respondents are also presented. As an illustrative example of results, the willingness to pay for the full implementation of the BAP in native woodlands is estimated to be approximately £145 per household per year\(^\text{18}\). A particular challenge that Christie et al. highlight in their reporting is the gap in the underpinning natural science evidence base concerning the level of ecosystem service provision associated with BAP conservation activities. The expert-weighting exercise is presented as a pragmatic solution but is recognised to be subject to key caveats relating to its accuracy (being based on the judgement of select individuals). The study therefore provides an important contribution in demonstrating the need to better understand the underlying relationships between biodiversity and ecosystem services in relation to valuations to support the case for investment in conservation activities.

\(^{18}\) This is the estimated aggregate value ‘sum of within own region and rest of UK’ reported by Christie et al. (2011).
The Christie et al. (2011) methodology is further applied in relation to ecosystem service benefits supported by biodiversity in the context of Sites of Special Scientific Interest (SSSIs) (GHK, 2011; Christie and Rayment, 2012). The same set of biodiversity characteristics are incorporated with varying levels of provision from ‘maintain funding’, ‘increase funding’, and ‘remove funding’ via a workshop-based approach (approx. 150 respondents). Interestingly, the results indicate that the workshop-based approach and the opportunity to provide enhanced information via an iterative process to respondents does not necessarily impact valuations (i.e. the provision of additional information did not change results; although sample size may be an issue). Hence, it is difficult to draw conclusions on the extent to which methods have advanced since Willis et al. (2003) in overcoming individuals’ constrained understanding of biodiversity and consequently their ability to trade-off impacts in stated preference exercises.

This conclusion particularly reflects the issue of pushing scientific uncertainty into the valuation study. Without actually modelling the ecological relationships or consequences for final services, stated preference studies of WTP for conservation actions or results, in effect, assume that respondents have informed preferences about conservation outcomes, even though the science base is not available to explain the consequences of changes. Though respondents generally can give an answer or make a choice, the lack of scientific underpinning raises the question of what these answers represent, and how they should be interpreted. This is a particular concern not only for using results directly in policy analyses, but also subsequently via value transfer or even a look-up value approach. The ecosystem services framework seeks in part to overcome this type of uncertainty, by explicit modelling or estimation of service changes. Hence, returning to generalised stated preference studies to cover ‘everything else’ under a biodiversity/non-use heading could be argued to partly defeat the object.

Further recent UK studies also serve to highlight similar concerns. For example, in an assessment for Defra for the UK Marine Bill, McVittie and Moran (2008) carried out contingent valuation and choice experiment surveys for non-use values (associated primarily with marine biodiversity conservation), with national aggregate estimates of approximately £890m - £1,700m per annum depending on the scheme implemented. In the final cost-benefit analyses these (intended) non-use values are not treated as additional in order to avoid possible double-counting with final services. As noted above, this reflects the common concern that stated preference surveys used to assess non-use values may be detecting a substantial part of use values too.

There are also examples of UK studies that concentrate on the local level and site-specific benefits. For example, Luisetti et al. (2011a, 2011b) examine managed realignment on the Blackwater Estuary (Essex), using a combination of methods: market costs avoided for coastal defence work, market prices for fish production function, market prices adjusted for subsidies for the value of agricultural land, and carbon prices for greenhouse gas fluxes. A stated preference approach is used for a “composite environmental benefit” that is intended to cover a wide range of impacts without double-counting: recreation, aesthetics, water quality, and biodiversity. From a look-up value perspective, this approach indicates a route by which stated preference values can be derived in a way that corrects for potential embedding (of general environmental quality within values expressed for specific components or services) and avoids double-counting.

The aggregation method demonstrated by Luisetti et al. allows for distance-decay and a non-linear relationship with wetland area. Thus, the estimates for the composite environmental benefit show the diminishing marginal utility generated by provision of additional areas of high environmental quality: in the ‘Policy Targets’ scenario (81.6 ha wetlands) the value estimate is £6.3m per year of which £4.4m is use value; in the ‘Deep Green’ scenario, with 10 times more wetlands, the value is only a little higher at £7.7m per year, of which £5.8m is use value, while in the ‘Extended Deep Green’ scenario, with 30 times more wetland than ‘Policy Targets’, the value is £8.3m per year of
which £6.4m is use value. This strong non-linearity illustrates the potential problems of assuming constant linear (per hectare) look-up values. Coupled with other spatial factors, including the proximity of human populations and the availability of substitutes, this stresses the importance of considering look-up values as only an entry-point, first cut estimate of values that will require refinement in cases of significant policy impacts.

4.1.2 Valuing landscapes

Another branch of stated preference research has focused on valuing landscapes, either as whole landscapes/areas, or specific features. The Environmental Landscape Features (ELF) model (IERM/SAC 1999; Olgethorpe, 2005) represents a form of meta-analysis / value transfer for valuing landscape features in England. Values, based on contingent valuation studies, were included for rough grassland, heather moorland, salt marsh, woodland, wetland and hay meadow (1999) and hedgerows and field margins (2001). The estimates are intended only to account for values of residents, and to allow for diminishing marginal values of additional units of a feature, but aim to value the entirety of a given resource within an area. eftec (2006) used choice experiments to examine the value of environmental changes in Severely Disadvantaged Areas (SDAs) across England, for similar types of 'landscape feature'. More recently Boatman and Willis (2010) provide estimates of the benefits of the Environmental Stewardship scheme, where the biodiversity element is bundled within wildlife and landscape benefits.

Swanwick et al. (2007) conclude that “there are strong arguments for a whole landscape approach as representing more realistically the way that people view and value landscapes”, but temper this with the observation that the choice between whole landscape and component-based valuation can depend on the proposed use or policy application of the results. They further suggest that contingent valuation is more suited to whole landscape approaches, whilst choice experiments are more suited to landscape component (or feature) valuation. A general issue with all these valuations, though, is that they are very likely to contain elements of both use and non-use values. Individuals, and survey instruments, may not be able to distinguish clearly between values for viewing and experiencing a landscape in a particular configuration or quality, and non-use values associated with the same features. This is not a problem for assessing the total (use and non-use) value of a given area, but it does give concern regarding possible double-counting if supposedly non-use values for biodiversity and cultural heritage are drawn from these studies and included in assessments alongside values for recreation estimated separately. This applies a fortiori if landscape feature studies are used only for cultural values, with separate estimation of non-use values for biodiversity.

4.1.3 Meta-analysis

More recently, some meta-analyses have attempted to draw conclusions based on the body of evidence, although typically not UK-based examples. Juutinen (2008) presents meta-analysis of contingent valuation studies for biodiversity value of old-growth boreal forests in Finland, arriving at £200 per ha per year; a value that puts forest in the range between thresholds for delaying harvesting (£84/ha/yr) and permanent conservation (£398/ha/yr). This value is not, however, suitable for direct transfer to the UK. Lindhjem (2007) presents a meta-analysis of mean WTP for forest protection in multiple-use forestry, with a value of £120 per household per year (and standard deviation of £138/hh/yr). However, the value is scale-insensitive, and so it is difficult to derive per hectare measures.

Nunes et al. (2009) present a meta-analysis of studies for forest biodiversity values, covering 65 separate studies with 248 value estimates. They use a general conception of biodiversity as the supporting service underpinning all other values, so their data points are for the total values of
forest ecosystem services. Not all studies cover all values, so they include dummy variables for cultural services and for provisioning/regulating services, against the omitted category ‘all’ services. This can allow cultural services to be separated out. Applying the meta-analysis model to English forests gives the following approximate results:

- €640/ha/yr for all services;
- €45/ha/yr for cultural services only, and
- €20/ha/yr for provisioning/regulating services.

Considering the balance of €575 per ha per year to relate to non-use values would be naïve, however, due to the log-linear form of the value equation (which becomes multiplicative in non-log form). The problem is that the meta-analysis function is a good statistical fit for explaining the variability in the results of valuation studies, but does not give additive results for the different services.

Another common issue with these models is that the values per hectare are highly sensitive to the area under consideration - the values in the list above have been calculated for all English woodlands, but if instead a smaller 100,000 ha forest is considered, the function predicts over €2,000 per ha year for the service value. The relationship between area and service value is highly dependent on the service under consideration. Many provisioning and regulating services have broadly linear relationships (e.g. timber production and carbon sequestration in woodlands) while other services are strongly dependent on specific size and location of the resource and its substitutes relative to human populations (e.g. recreation).

For biodiversity, there is clear spatial dependence in terms of provision: small areas of conservation habitat generate lower per hectare levels of biodiversity than larger areas. The marginal relationship is roughly sigmoidal up to an equilibrium level after which a constant return to scale can be assumed (Bateman et al. 2006). Biodiversity supports a wide variety of values and these may in turn be spatially dependent. Recreational values of biodiversity will follow a similar pattern as that for recreation generally. Regulating services such as pollination and pest control will vary spatially with the value of relevant agriculture. The spatial pattern of non-use values is under-researched, though it is thought that the distance-decay of non-use values might be less pronounced than that for use values (Bateman et al. 2006), though over long distances there may be a ‘local identity’ effect where non-use values remain fairly stable across a region or nation, but fall rapidly outside the area (Bateman et al. 2005). Hence, in conclusion, further research is needed to better establish the ways in which biodiversity values are spatially dependent, and the extent to which simple look-up values can reflect these spatial factors.

### 4.1.4 Cost-based methods

Although cost-based methods do not give WTP / TEV estimates, they can be viewed as relatively straight-forward to apply in principle though, in practice the costs of specific interventions can vary greatly based on local conditions and opportunity costs. For example, payments for ecosystem services such as higher level stewardship payments for environmentally sensitive agricultural management can be viewed as proxies for the benefits of the interventions. However, as noted above, there is both circularity in this argument (defining benefits as equal to costs) and a double-counting concern (the payments do not distinguish between services). Hence, it should be stressed (again) that using such cost estimates in lieu of biodiversity benefit estimates cannot be recommended as a robust solution for providing values.
Similar costs can be estimated for habitat creation and restoration. GHK (2011) estimate the costs of biodiversity offsets based on:

- Habitat creation and restoration costs from the UK BAP costings;
- Land purchase costs based on current prices for agricultural land; and,
- An additional percentage to allow for administrative, transaction and regulatory costs.

They assess two different approaches: (i) a management agreement approach, which does not involve land purchase but does involve compensating landowners for lost income; (ii) and a land purchase approach that ignores lost incomes (Table 4.1). The lost incomes are estimated to be 70% of the total management agreement costs, with the remaining 30% being costs of management interventions. From an economics perspective, the relevant concept is opportunity cost and foregone production should be included as a cost. In principle, future production values should be capitalised into land values and should reflect opportunity costs, though in practice distortions (such as agricultural subsidies and tax breaks), uncertainties (e.g. about planning policy) and high private sector discount rates may drive a wedge between land values and social opportunity costs. In principle, then, the methods should give broadly similar results, except that the ‘land purchase’ approach ignores any returns that may still arise (to the new owners / the state; for example, created wetlands may still be grazed, and created woodlands may be harvested for timber); any surplus payment for the land price could be treated as a transfer payment rather than a real cost. Hence, the ‘management agreement’ approach figures may be more representative, though again it should be recognised that all of the costs can vary significantly according to location and other factors.

Table 4.1: Biodiversity offset costs under ‘management agreement’ approach (Present Value (PV), 100 years)

<table>
<thead>
<tr>
<th>Habitat</th>
<th>Restoration PV unit cost (£/ha)</th>
<th>Creation PV unit cost (£/ha)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Upland habitats</td>
<td>2,151</td>
<td>7,382</td>
</tr>
<tr>
<td>Lowland heathland</td>
<td>8,530</td>
<td>11,791</td>
</tr>
<tr>
<td>Lowland grassland</td>
<td>10,168</td>
<td>11,293</td>
</tr>
<tr>
<td>Woodland</td>
<td>7,776</td>
<td>7,436</td>
</tr>
<tr>
<td>Wetlands</td>
<td>9,435</td>
<td>11,072</td>
</tr>
<tr>
<td>Coastal</td>
<td>4,509</td>
<td>48,758</td>
</tr>
</tbody>
</table>


4.1.5 Economic activity-based methods

There is also evidence from various sources on the importance of biodiversity in supporting tourism, including data on associated tourist expenditures. Anger et al. (2014) cite several studies reporting gross value added (GVA) and employment impacts attributable to wildlife and nature tourism. Like the cost-based figures, measures of economic activity such as GVA and employment are not TEV-based, and are not additive with ecosystem service values in CBA, but are useful for other policy and analysis purposes. However, from the perspective of developing look-up values, the economic impact assessment approach is unlikely to be useful, both because values are highly locally-specific, and because it is difficult to isolate the components of these values that is attributable to biodiversity.
4.2 Valuing flows over time / valuing assets

Biodiversity and ecosystems are assets that may be capable of producing enhanced services in future or that may be suffering unsustainable exploitation, leading to unavoidable decline in flows. Issues related to irreversible depletion of natural capital, ecological thresholds, non-linear relationships and hysteresis may arise - though if they do, these should be treated in full and use of look-up values is not recommended.

For the purposes of look-up values, therefore, attention can be limited to cases where changes in extent and condition of ecosystems may occur, but these are either minor, or relatively linear and predictable, in the context of natural capital and services at broader scales. Under these specific conditions, valuation of the asset is essentially the same as estimating the net present value of constant future flows, using official UK discount rates. This will draw on all the individual services estimated in the ecosystem services framework, with the addition of non-use values for biodiversity (e.g. the types of studies highlighted in Section 4.1). These calculations could be used to develop a set of look-up asset values for particular types of habitat, in particular conditions and types of locations (reflecting for example distance from populations).

This does, however, beg the question of cumulative impacts and, here it is necessary to consider the longer term and possible unsustainable reduction in biodiversity. Valuation then becomes more challenging (Bateman et al., 2011); though this is not an issue for the use of look-up values, but rather for their derivation and regular updating. In any event, any form of biodiversity that is considered critical natural capital ought to be strongly protected by policy (either through strict protection or through a No Net Loss policy with measures for compensatory habitat provision to offset any damages). The problems of cumulative impacts potentially degrading critical natural capital would be too great to leave to look-up values, or any other entry-level tool.
5. CONCLUSIONS AND RECOMMENDATIONS

5.1 Summary

Operating within an ecosystem services framework, the focus of valuation should be on final goods and services. Valuing intermediate or supporting services risks double-counting since these values are already embedded in the value of final services. This is likely true even for resilience values, since valuation generally assumes resilience (although typically it is an implicit assumption) and assumes that the future flows of services can be maintained. Although the importance of resilient biodiversity for resilient services should not be under-estimated, this does not mean that it requires separate inclusion in appraisal, since it is already implicit within the constant flow assumption.

The main aspect of biodiversity that might require separate valuation is non-use value. It can be considered additional to the use values of recreation, aesthetic value and so on. This is not to suggest that it is necessarily easy to separate out from these values, but this has been attempted in some studies, and is an important step in avoiding double-counting. In practice, however, the evidence base is quite thin, reliant on stated preference methods as these are the only approach capable of capturing non-use values, and available estimates may not be viewed as very reliable. Indeed, some argue they are not currently reliable enough to use in CBA (e.g. Bateman et al., 2013). Further research is therefore required to test and develop valuation methods to improve the ways in which biodiversity non-use values are estimated, in order to give greater confidence in the resulting valuations.

5.2 Developing future look-up values for biodiversity

The summary above assumes that final services are being separately identified and valued. Where this is not the case (i.e. where data are lacking and/or it is not considered proportionate to develop such an assessment) the requirement for look-up values will relate not to biodiversity as a specific feature of ecosystems, but rather to changes in the habitat extent and/or quality overall.

The double-counting highlighted above would not occur, however, if the appraisal context calls for a broader value for the ecosystem as an asset overall - as might be the case for look-up values relating to small changes in habitats (e.g. land taken for development) - where separate assessment of the individual impacts on provisioning, regulating and cultural services might be considered a disproportionate use of appraisal resources. Hence, there could be scope for a dual approach to look-up values for biodiversity:

1. A value focusing just on the non-use and resilience components of biodiversity, for use where final services are directly and individually addressed in an appraisal; and

2. A habitat-based (per hectare) value representing all services, for use where individual assessment of final services is not feasible (or is not considered proportionate in the context of the appraisal).

Non-use valuations for (1) would need to be developed through future research studies, as the current evidence base is limited and does not lend itself to being interpreted and presented in the broadly generalised format that is required for look-up values. Future studies would need to be designed and analysed with the objective of using the resulting valuations in subsequent value transfer analyses.
The suggested ‘per hectare’ values for particular habitats under (2) might be thought of as look-up values for biodiversity though, in practice these would most likely be developed using ecosystem service assessments, via value transfer or meta-analysis, such that land-use/land-cover features could be associated with typical service patterns and values. This is essentially the approach adopted in the ARIES project (Villa et al., 2014). It should be noted, however, that many services/values are highly non-linear in respect to habitat extent, and also highly dependent on spatial features including the location of human (beneficiary) populations, and the availability of substitute resources. Therefore, a linear area-based assessment assuming constant per hectare values can only be a gross approximation and must be treated with appropriate caution. This uncertainty and the potential for error might be accommodated in ‘minor’ appraisal contexts (e.g. screening, initial assessments), which is the intended setting for the use of look-up values. More robust evidence would be required where the scale of impacts and decisions to be made are more significant.

Alternatively, values could be developed using large-scale land use optimisation modelling such as TIM (Bateman et al. 2014). This is not yet a viable proposition for look-up values, but developments in the short term (e.g. the next five years) could lead to the use of such models to develop regionally-specific sets of values for different kinds of biodiversity / land-use change (based on the opportunity costs of constraints in the models) that could also take account of some local features (such as beneficiary populations and substitute sites). The constraints would need to reflect actual policy decisions or targets (i.e. this approach does not ‘value’ biodiversity in welfare terms directly, but rather derives opportunity cost estimates that are consistent with the priorities enshrined in biodiversity policy). The opportunity cost estimates would require periodic updating, and values would need to be updated when biodiversity policy/targets changed. But, overall, the approach could allow relatively straightforward estimation of values of changes to ecosystem extent and condition based on actual UK land-use and the services modelled.

5.3 Cost-based biodiversity values

In the interim, perhaps the most basic approach for look-up values might be to estimate the costs of achieving biodiversity targets, for example drawing on GHK Consulting (2011). Conceivably, cost-based estimates could be gradually replaced with improved valuations as the approaches suggested above develop. However, it must be stressed that the relationship between cost-based and welfare-based values is not clear even though the former are intended to be proxies of the latter. This represents a significant conceptual barrier for the general use of cost-based values outside of a target-based policy. Moreover, a key concern is the insensitivity of cost-based approaches to spatial contexts at local scales and using cost estimates in lieu of biodiversity benefit estimates cannot be recommended as a robust solution for providing valuations.

5.4 Conclusions

Significant challenges are faced in valuing biodiversity for policy analyses. These are not insurmountable and, conceptually, the approach to be developed is clear and there are a number of avenues of valuation research that can be explored.

At present, the main difficulty lies in the limited nature of the currently available UK-focused evidence. This makes reliable value transfer a challenge and consequently means that the current evidence base does not readily fit with the needs of look-up values. The conclusion at this stage, therefore, is that it is not appropriate to use either available welfare-based values or cost-based
values as look-up values. However, insofar as the supporting function of biodiversity is reflected in provision of final goods and services, look-up values will provide partial coverage of the value of biodiversity. More considered analysis will likely be required when explicitly focusing on valuing impacts on biodiversity, including potential non-use values. At the very minimum this implies that formal value transfer approaches should be applied, and even these may not be sufficient if available evidence does not avoid double-counting with other valuations in an appraisal or permit for appropriate assessment of spatial factors.

Overall, further research effort is required to integrate the understanding of the role of biodiversity in contributing to human well-being into a workable approach for practical policy analyses. More effort is also required to produce robust biodiversity valuation evidence for use in appraisals, particularly via value transfer analysis. Any new evidence that is developed will need to be reconciled with existing evidence for other non-biodiversity impacts to disentangle impacts and ensure that double-counting of biodiversity values is avoided. Whether this evidence can adapted for use as look-up values or as a separate database in the future should also be explored.

It should also be further stressed that look-up values are intended for use in first-cut assessments, and for valuing relatively minor or incidental impacts for appraisals and assessments that might otherwise overlook environmental impacts. This approach should not be taken for full analysis where there are significant impacts on biodiversity and ecosystems. In such cases, both the scientific and economic assessments need to be more detailed, likely entailing specifically commissioned research.
REFERENCES


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IUCN (1994), The economic value of biodiversity


Spash, C.L. and N Hanley (1994), Preferences, information and biodiversity preservation, MPRA Paper No. 38351


Turpie, J. 2003. The existence value of biodiversity in South Africa: how interest, experience, knowledge, income and perceived level of threat influence local willingness to pay. Ecological Economics, 46, 199-216


## ANNEX: SUMMARY OF SELECTED BIODIVERSITY VALUATION STUDIES

This Annex provides a summary of selected UK valuation studies that estimate changes in the provision of biodiversity as an ecosystem service. The range of studies is not exhaustive and is provided to illustrate the coverage and approach across different valuation studies.

<table>
<thead>
<tr>
<th>Study</th>
<th>Study good context and methodology</th>
<th>Definition of the good</th>
<th>Study good site</th>
<th>Substitutes</th>
<th>WTP (as reported in study)</th>
<th>Value</th>
<th>Unit</th>
<th>Sample size</th>
</tr>
</thead>
</table>
| Christie et al. (2004; 2006) | Bundle of services provided by enclosed farmland, freshwaters, and woodlands (stated preference: choice experiment) | Aesthetic benefits, protection from hazards, and clean water provided as a result of changes in biodiversity | Cambridgeshire | The availability of substitutes is not accounted for. Other substitutes include other locations with farmland in the UK. | Cambridgeshire:  
- Protection of rare species: 35.65  
- Protection of rare and common species: 93.49  
- Recovery of unfamiliar species: 115.13  
- Habitat restoration: 34.40  
- Habitat creation: 61.36  
- Protect ecosystem services that have direct impact on human populations: 53.62  
Northumberland:  
- Protection of rare species: 90.59  
- Protection of rare and common species: 97.71  
- Recovery of unfamiliar species: 189.05  
- Habitat restoration: 71.15  
- Habitat creation: 74.00  
- Protect ecosystem services that have direct impact on human populations: 105.22 | £/hh/yr over 5 yr period | Cambridgeshire: n=341  
Northumberland: n=395 |
| Christie et al. (2004; 2006) | Aesthetic value provided by enclosed farmland, freshwaters, and woodlands | Aesthetic benefits, provided as a result of changes in biodiversity | Cambridgeshire | The availability of substitutes is not accounted for. Other substitutes include other | Cambridgeshire:  
- Conversion to arable stewardship: 74.27  
- Conversion to arable stewardship: 54.97  
- Loss of biodiversity on farmland due to urban development: 45.30 | £/hh/yr over 5 yr period | Cambridgeshire: n=341  
Northumberland: n=395 |
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<tr>
<th>Study</th>
<th>Study good context and methodology</th>
<th>Definition of the good</th>
<th>Study good site</th>
<th>Substitutes</th>
<th>WTP (as reported in study)</th>
<th>Sample size</th>
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</thead>
</table>
| GHK Consulting (2011)        | Bundle of services provided by coastal margins; mountains, moorlands and heaths; woodland; semi-natural grasslands; freshwaters; enclosed farmland (stated preference: choice experiment) | Food, fibre, recreation, aesthetics, wildlife, and protection from hazards provided by Sites of Special Scientific Interest (SSSIs) to prevent decline of the percentage of SSSIs that achieve favourable status from 65% (current) to 35% (potential). Non-charismatic species include all trees, flowering and non-flowering plants, insects and fungi. | All Site of Special Scientific Interest (SSSIs) across England (1.10 million hectares) and Wales (170 thousand hectares). | The availability of substitutes is not accounted for. Substitutes may include SSSIs in the rest of the UK. | Maintain funding scenario:  
- Provisioning services: 6.50  
- Climate regulation: 89.00  
- Water regulation: 66.30  
- Sense of experience: 29.92  
- Charismatic species: 136.95  
- Research and education: 68.00  
Increase funding scenario:  
- Provisioning services: 3.25  
- Climate regulation: 89.00  
- Water regulation: 66.30  
- Sense of experience: 24.68  
- Charismatic species: 49.80  
- Research and education: 56.10 | £/hh/yr   
| McVittie and Marine non-     | Non-use value                                                                                     | Network of The availability of SSSIs in England:  
- England: | £/hh/yr   |

Northumberland:  
- Habitat re-creation: 47.49  
- Loss of biodiversity on farmland due to urban development: 36.84

McVittie and Marine non-use value
Network of The availability of SSSIs in England: 

England:
<table>
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<tr>
<th>Study</th>
<th>Study good context and methodology</th>
<th>Definition of the good</th>
<th>Study good site</th>
<th>Substitutes</th>
<th>WTP (as reported in study)</th>
<th>Sample size</th>
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</thead>
</table>
| Moran (2010)                 | use value of biodiversity/wildlife (stated preference: choice experiment)                        | of biodiversity/wildlife from Marine Conservation Zones (MCZs)                           | MCZs in UK marine environment (150,000 km² of MCZ)                              | of substitutes is not accounted for. Substitutes can include other marine areas and other habitat types providing non-use value to beneficiaries | - Low: 36.11  
- Central: 69.49  
- High: 102.87  

Scotland:  
- Low: 7.70  
- Central: 20.92  
- High: 33.82  

Wales:  
- Low: 17.47  
- Central: 107.39  
- High: 189.11  

Northern Ireland:  
- Low: 17.45  
- Central: 33.90  
- High: 50.27 |
|                              |                                                                                                   |                                                                                        |                                                                                 |                                                                                                                                               | n=219  
- Scotland: n=144  
- Wales: n=143  
- Northern Ireland: n=143 |
| Luisetti et al. (2011b)      | Bundle of services provided by coastal margins (stated preference: choice experiment)           | Biodiversity; recreation and tourism; aesthetic value                                  | Essex, Norfolk, and Suffolk                                                     | Accounted for in the study (distance to the nearest site).  
Scenario 1: Policy targets (81.6 ha of intertidal habitat created):  
- Town 1 (8 miles from Abbott’s Hall - a known managed realignment site): use and non-use values: 771,120; use value: 546,764  
- Town 2 (15 miles from Abbott’s Hall - a known managed realignment site): use and non-use values: 4,106,770; use value: 2,874,739  
- Town 3 (23 miles from Abbott’s Hall - a known managed realignment site): use and non-use values: 1,109,206; use value: 764,194  
- Town 4 (32 miles from Abbott’s Hall - a known managed realignment site): use and non-use values: 369,906; use value: 281,704  

£/yr (aggregate values)       | 508                                                                                              |                                                                                                                                 |
<table>
<thead>
<tr>
<th>Study</th>
<th>Study good context and methodology</th>
<th>Definition of the good</th>
<th>Study good site</th>
<th>Substitutes</th>
<th>WTP (as reported in study)</th>
<th>Sample size</th>
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<td>Value</td>
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<td>realignment site): use and non-use values: 360,129; use value: 243,260</td>
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<td>Scenario 2: Deep green (816.5 ha of intertidal habitat created):</td>
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<td>• Town 1 (8 miles from Abbott’s Hall - a known managed realignment site): use and non-use values: 930,385; use value: 706,029</td>
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<td>• Town 2 (15 miles from Abbott’s Hall - a known managed realignment site): use and non-use values: 4,987,360; use value: 3,749,329</td>
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<td>• Town 3 (23 miles from Abbott’s Hall - a known managed realignment site): use and non-use values: 1,354,171; use value: 1,009,132</td>
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<td>• Town 4 (32 miles from Abbott’s Hall - a known managed realignment site): use and non-use values: 443,092; use value: 326,223</td>
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<td>Scenario 3: Extended deep green (2404.1 ha of intertidal habitat created):</td>
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<td>• Town 1 (8 miles from Abbott’s Hall - a known managed realignment site): use and non-use values: 1,005,170; use value: 780,814</td>
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<td>• Town 2 (15 miles from Abbott’s Hall - a known managed realignment site): use and non-use values: 1,005,170; use value: 780,814</td>
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Sample size
<table>
<thead>
<tr>
<th>Study</th>
<th>Study good context and methodology</th>
<th>Definition of the good</th>
<th>Study good site</th>
<th>Substitutes</th>
<th>WTP (as reported in study)</th>
<th>Sample size</th>
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<tbody>
<tr>
<td></td>
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<td></td>
<td>realignment site): use and non-use values: 5,392,037; use value: 4,160,006</td>
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<td>• Town 3 (23 miles from Abbott’s Hall - a known managed realignment site): use and non-use values: 1,469,184; use value: 1,124,145</td>
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<td>• Town 4 (32 miles from Abbott’s Hall - a known managed realignment site): use and non-use values: 482,048; use value: 355,179</td>
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<tr>
<td>Boatman and Willis (2010)</td>
<td>Bundle of ecosystem services provided by terrestrial habitats (stated preference: contingent valuation)</td>
<td>Aesthetic value and biodiversity provided by woodland, enclosed farmland; semi-natural grassland as a result of the Environmental Stewardship (ES) Scheme</td>
<td>England</td>
<td>Not accounted for in the study</td>
<td>• Definite yes to ES; payment card amount; distribution free: 26.09</td>
<td>£/hh/yr</td>
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<td></td>
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<td>• Definite yes to ES; payment card amount; theory constrained: 22.41</td>
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<td>• Definite yes to ES; midpoint: 35.08</td>
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<td>• Probably yes to ES; payment card amount: 32.26</td>
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<td>• Probably yes to ES; midpoint: 52.42</td>
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<tr>
<td>Christie et al. (2011)</td>
<td>Biodiversity in terrestrial BAP habitats (stated preference choice experiment)</td>
<td>Ecosystem services provided by BAP habitats: wild food, non-food related provisioning services, climate</td>
<td>UK</td>
<td>Not accounted for in the study</td>
<td>Increased spend scenario:</td>
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<td>• Scotland: 82.64</td>
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<td>• Wales: 22.17</td>
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<td>• Northern Ireland: 23.43</td>
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<td>• England: 50.52</td>
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<td>• Scotland: 161.28</td>
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<td>• Wales: 41.82</td>
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<td>Study</td>
<td>Study good context and methodology</td>
<td>Definition of the good</td>
<td>Study good site</td>
<td>Substitutes</td>
<td>WTP (as reported in study)</td>
<td>Sample size</td>
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<td>regulation, water regulation, sense of place, charismatic species, non-charismatic species</td>
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<tr>
<td>Willis et al. (2003)</td>
<td>Bundle of ecosystem services provided by woodlands (value transfer)</td>
<td>Recreation, landscape amenity, biodiversity, carbon sequestration, pollution absorption, water supply and quality, and protection of archaeological artefacts</td>
<td>Great Britain</td>
<td>Not accounted for in the study</td>
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<td>• Enhanced biodiversity for 12,000 ha increase of commercial Sitka spruce: 0.35 enhanced</td>
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<td>• Enhanced biodiversity for 12,000 ha increase in lowland broadleaved forest: 0.84</td>
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<td>• Enhanced biodiversity for 12,000 ha increase in ancient semi-natural woodland: 1.13</td>
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<td>Northern Ireland: 40.53</td>
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<td>England: 109.28</td>
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<td>Note that total values are presented here but a breakdown of ‘within own region’ and ‘rest of UK’ values is provided.</td>
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£/hh/yr
n/a - value transfer study