Appendix Biodiversity

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Executive Summary

The NEA adopts the CBD definition of biodiversity which incorporates the attribute of diversity – a measure of variation between genes, species and ecosystems and the composition and relative abundance of living things. This section focuses on the potential additional values associated with diversity.

We identify broadly three areas where diversity per se may be responsible for values beyond those manifest in the ecosystem services biodiversity supports.

The role of diversity in underpinning service delivery

There is evidence to suggest that increased rates of the ecosystem processes underlying ecosystem services are associated with increased numbers of species or genes. There are also a number of examples where simplification of ecosystems has potentially led to a net loss of services. Understanding ecological processes that underpin ecosystem services plays a key part in our understanding of the link between diversity and the value of ecosystem services.

The infrastructure, insurance and resilience values of biodiversity

The infrastructure, or primary value of biodiversity is related to the fact that some combinations of ecosystem structure and composition are necessary to ensure the 'healthy' functioning of the system.

The insurance hypothesis states that biodiversity insures ecosystems against declines in their functioning because the more species the greater the guarantee that some will maintain functioning even if others fail.

The resilience hypothesis may be characterised as an ecosystem's flexibility to reconfigure itself in the face of external shocks. It suggests biodiversity per se may also have economic benefits if species richness enables an ecosystem, currently in a desirable state, to resist or recover from perturbations.

There is evidence for both terrestrial and marine ecosystems that lends support to the insurance and resilience hypotheses but little evidence from the UK to demonstrate the magnitude of these values or the habitats and services for which they are most applicable. Empirical research is limited by gaps in our understanding of the underpinning science and a consequent lack of relevant data.

The role of biodiversity in the direct delivery of ecosystem services

Bioprospecting. If biodiversity harbours potentially valuable species or compounds, as yet undiscovered, bioprospecting may be an economically rewarding activity. The focus of bioprospecting is on the world's biodiversity hotspots. The marginal pharmaceutical value of a species is estimated to be moderate or small in biodiversity hotspots. Such values are likely to be small in the UK too.

Maintaining genetic diversity Maintaining crop wild relatives, rare breeds and landraces offers potential benefits to domesticated crops as well as insurance type values. While a range of potential benefits to conserving such genetic diversity there is no evidence from the UK to demonstrate the marginal values associated with their conservation.

Non Use Values of Biodiversity

Components of biodiversity are valued directly by people for a variety of reasons. These include the appreciation of wildlife and of scenic places and the spiritual, inspirational, educational or religious benefits associated with the natural world. When attempting to value complex goods or services like the diversity value of biodiversity, human cognitive limitations limit the validity of monetary valuation techniques. Very few studies have actually attempted to isolate and value biodiversity per se but there is ample evidence to demonstrate existence and bequest type values of nature held by the UJK population. Membership subscriptions and legacies left to wildlife organisations are examples. Some reasons we value nature, notably ethical or spiritual motivations, cannot be captured using economic techniques.

Issues to consider in valuing biodiversity

A prime motivation for marginal valuations of ecosystem services is the belief that doing so can help us make substantially better decisions regarding land, marine and natural resource use. There are limits to economic valuation and some ecosystem service benefits lend themselves more successfully to monetary valuation than others. Some considerations which influence the scope and applicability of monetary valuation include the irreversibility of decisions and ecological threshold effects.

Estimating the costs of managing biodiversity in the UK

In lieu of economics values for the biodiversity of the UK, the cost of managing biodiversity is taken as an indication of the value society holds for biodiversity. This is based on the assumption that the political biodiversity targets and legal mechanisms that have been brought in to support biodiversity are a reflection of the desire of the public through the political process. The total costs of biodiversity management, based on achieving the UK's biodiversity action plan targets and managing protected areas in the UK are just over £1 billion a year.

Introduction

Chapter 5 of the NEA (Norris et al 2010) has identified the many potential relationships between biodiversity and the delivery of ecosystem services (ES) but emphasises the very poor quantitative evidence available. Overall conclusions are clear that microorganisms, fungi and plants play a role in underpinning support/regulatory services while vertebrates are more important for cultural services and have a smaller role to play in supporting provisioning/regulatory services. That specific ecosystem services are predominantly provided by specific biodiversity groups or trophic levels may be very important for land or resource management.

While ecosystem service thinking focuses on the link between ecological processes and human wellbeing, it is also important to consider the prior link between biodiversity and ecosystem functioning. There has been increasing scientific focus on this aspect of the chain but less economic attention because there is little data illustrating how service delivery actually changes with alterations to an ecosystem's species richness/composition. An issue of interest for resource management is whether prioritising land and marine resources to maximise the delivery of ecosystem services will result in very different outcomes compared with managing them on the basis of existing conservation based priorities. Global and UK scale studies reveal little correlation between areas rich in biodiversity (according to nature conservation designations) and high ecosystem service delivery (Naidoo et al 2007, Anderson et al 2009).

Chapter 5 of the NEA (Norris et al 2010) has established the role of different biodiversity groups in underpinning ecosystem services and other chapters of the NEA analyse the nature and value of these services. A further consideration is whether biodiversity *per se*, provides additional wellbeing benefits beyond these ecosystem services it supports. So, are species rich grasslands more nutritious or higher yielding than species poor ones? Or, do species rich systems reduce yield variability or enhance resilience to external shocks?

The value of Biodiversity per se

The NEA adopts the CBD definition of biodiversity. It incorporates

a) The attribute of diversity – a measure of variation between genes, species and ecosystems
b) The composition and relative abundance of living things

Taking a) and b) together means that all ecosystem services are either biodiversity (trees, crops, fish) or their delivery is underpinned by it in some way. For the purposes of this chapter, we focus on the potential additional values associated with diversity as other ecosystem services are being assessed separately in their respective chapters.

The role of diversity in underpinning service delivery

If diversity is positively correlated with service delivery, then greater biodiversity will increase the value of ecosystem services (see diagram 2a, Chapter 5 NEA). Much of the evidence demonstrating a positive link between biodiversity and ecosystem services has the limitation of being at small scale or laboratory based. However, in the greater number of experiments to date, increased rates of the ecosystem processes underlying ecosystem services are associated with increased numbers of species (Hector & Bagchi 2007; Hopper et al 2005). In a recent meta-analysis of 446 studies of the impact of biodiversity on primary production, 319 of which involved primary producer manipulations or measurements, Balvanera et al. (2006) find that there is 'clear evidence that biodiversity has positive effects on most ecosystem services', and specifically that there is a clear effect of biodiversity on productivity.

There are a number of examples where simplification of ecosystems has potentially led to a net loss of services. The loss of fishery resources may be an example where fishing down the food chain leads to loss of value. Another example is the loss of peat soils in the fens and Somerset levels which means that thousands of tonnes of carbon are being lost each year – rough calculations show that the value of the carbon loss exceeds the value of agricultural production. (awaiting permission to use Natural England report for REF). However, there are also examples of environments where increased biodiversity may not necessarily lead to additional services (estuaries), or instances where increasing biodiversity can potentially result in a decrease of services (non native invasive species).

The role of biodiversity in underpinning ecosystem services is highlighted throughout the NEA. Chapter 13, for example, (Smith et al 2010) estimates that twenty percent of the UK cropped area comprises pollinator dependent crops and note that a high proportion of wild flowering plants depend on insect pollination for reproduction. The authors offer a conservative estimate of the value of pollinators to UK agriculture of £430 million per annum. In the UK, over 10,000 colonies are imported each year to pollinate crops growing in glasshouses and polytunnels (Smith et al 2010).

The infrastructure, insurance and resilience values of diversity

There have been a number of theories linking biodiversity to the stability of ecosystem functioning. This section describes three interrelated explanations as to why diversity per se may generate values beyond the ecosystem services it supports.

The biological composition of ecosystems, measured as biodiversity, has a key role to play in ecosystem service delivery and some authors have observed that a value of a continued functioning ecosystem arises from the fact that it underpins ecosystem functioning and processes which is sometimes referred to as 'primary' (Turner 2003) or 'infrastructure' (Costanza et al 1997). Total system value exceeds Total economic value with the difference lying in that the operating system yields or possesses primary, 'glue' or infrastructure value, i.e. value related to the fact that some combinations of ecosystem structure and composition is necessary to ensure the 'healthy' functioning of the system, or system status (Gren et al). For this reason, Turner et al (2003) argue that the aggregate total economic value of a given ecosystem's functions, or combinations of such systems at the landscape level, will not be equivalent to the total system value. The continued functioning of a healthy ecosystem is more than the sum of its individual functions (components). While this concept has strong intuitive appeal, it is difficult to see how such value can be estimated in monetary terms.

According to the insurance hypothesis, biodiversity insures ecosystems against declines in their functioning because the more species the greater the guarantee that some will maintain functioning even if others fail. Yachi and Loreau (1999) develop a theoretical model which shows an insurance effect of species richness on ecosystem productivity with a reduction in the temporal variance of productivity. Additionally, Baumgartner (2007) has developed a theoretical model to consider how greater diversity in agro-systems can provide insurance against the uncertain provision of ecosystem services (in terms of yield variability) used by risk-averse economic agents. This suggests there may be an ''insurance value'' of biodiversity, which could be significant but difficult to value. Various studies have attempted to value the contribution of crop diversity to the mean and variance of agricultural yields and farm income. Birol et al (2006), for example, use a choice experiment to estimate farmer's valuation of agricultural biodiversity on Hungarian farms.

Option value may also be considered a form of insurance in that, increasing diversity offers more options for the future discoveries. This may help ensuring continued possibilities for ecosystem adaptation, in addition to the possibilities for discoveries of economically valuable compounds for use by people in uncertain future. This value in biodiversity is likely to be associated with the variety of different genes that can be expressed by organisms as potentially useful phenotypic traits or characters (different chemicals or functional behaviour). Not knowing which genes or compounds will be of value in the future, they must all be treated as having equal value. The greatest value for biodiversity conservation will come from ensuring the persistence of as many different genes or compounds as possible. Weitzman (1998) developed a cost-effectiveness approach to determining actual conservation priorities. The underlying model, which he termed the 'Noah's Ark Problem, 'addresses the problem of best preserving diversity under a limited budget constraint. Here, the optimal policy is always to spend the entire budget on a subset of the species. The implication being that it is best to concentrate conservation efforts rather than spending money more thinly on a broader range of species.

The resilience hypothesis may be characterised as an ecosystem's flexibility to reconfigure itself in the face of external shocks. It suggests biodiversity per se may also have economic benefits if species richness enables an ecosystem, currently in a desirable state, to resist or recover from perturbations. As argued by Perrings et al (1995) the importance of biodiversity lies in its role in preserving ecosystem resilience, by underwriting the provision of key ecosystem functions over a range of environmental conditions. Early studies lend support to the resilience concept by demonstrating, that over small scales (e.g. the crop-field level) an increase in on-farm species richness and the diversity of overlapping functional groups of species enhances the level of functional diversity, which, in turn, increases ecological stability (Tilman et al 1996) and resilience (Holling 1988 and Holling 1996).

In the marine environment, small scale experimental studies of biodiversity and ecosystem function also suggest that high species richness leads to greater resilience. In some cases this is exhibited as increased resistance, for example in seagrass (Hughes and Stachowicz 2004), or as enhanced recovery after a perturbation (Reusch *et al.* 2005).

Such small scale studies cannot necessarily be extrapolated to the entire marine environment but recent studies suggest the same phenomenon is observed at larger scales. High diversity kelp forest are considered to be more resilient that their lower diversity equivalents (Steneck *et al.* 2004).The simplification of food chains has also been found to have detrimental effects on the resilience of systems, for example the removal of predators, such as pelagic fish, can lead to an increased abundance of their prey, in this case plankton, resulting in plankton blooms (Hughes *et al.* 2005). Under similar environmental conditions increased species richness generally decreases susceptibility to invasion by exotic species. However several other factors may be more influential and mask the effects of species richness, such as disturbance regime and resource availability (Hooper *et al.* 2005).

Worm et al (2006) analyzed local experiments, long-term regional time series, and global fisheries data to test how biodiversity loss affects marine ecosystem services across temporal and spatial scales. They observed that collapses of fisheries occurred at a higher rate in species-poor ecosystems, as compared with species-rich ones. Species richness of fished taxa was negatively related to the variation in catch from year to year and positively correlated with the total production of catch per year. They attribute increased stability and productivity to a portfolio type effect whereby a more diverse array of species provides a larger number of ecological functions and economic opportunities, leading to a more stable trajectory and better performance over time.

Holling (1973) looks at resilience specifically in terms of the magnitude of a shock which can be absorbed by an ecosystem without losing functionality. An innovative approach to the problem of assessing resilience in this sense of sustainability is proposed by Mäler *et al* (2007, 2008) who considers the ability of an ecosystem to withstand stresses and shocks and so continue to provide services. Mäler et al., propose treating this ecological 'resilience' as a stock with a distinct asset value which can be degraded or enhanced over time. A lack of knowledge and data availability limits the potential for testing the resilience theory empirically. One attempt is Walker *et al*. (2010) who examine the value to agriculture in South-East Australia of maintaining a saline free water table. Here agricultural expansion depletes the stock of non-salinated soils leading to a loss in ecological resilience. The depleting process provides agricultural produce creating a trade-off between the benefits of depletion and the fact that losses of resilience may need to be reversed if stocks fall below some threshold level.

The role of biodiversity in the direct delivery of ecosystem services

Bioprospecting

There is potentially a value associated with genetic or species diversity which may directly contribute to some goods/benefits. If biodiversity harbours potentially valuable species or compounds, as yet undiscovered, bioprospecting may be an economically rewarding activity. One example of bioprospecting is pharmaceuticals. Many pharmaceuticals are based on, or derived from, plant compounds. The economic value of the pharmaceutical use of genetic material can be gauged by the world markets in pharmaceutical products derived from genetic resources, which is around US$500-800 billion (ten Kate and Laird, 2000). Biomedical research is dependent on animals and microbes. Chivian (2003) cites several cases where animal species native to forests offer potentially key insights to ongoing research (e.g. poison dart frogs in Central and South America relating to the study of the central nervous system).

Simpson, Sedjo, and Reid (1996), focus on the value of potential medicinal uses of the species. Prospects for discovery increase with the number of genes or species. However, they find the likely value of conserving individual species in current times for pharmaceutical research to be low. Reasons are that biodiversity is abundant and hence one extra species has low economic value; and there is extensive 'redundancy' in that, once a discovery is made, finding the compound again has no value. Costello and Ward (2006) revise the work of Simpson et al and provide empirical results which suggest comparatively greater values associated with bioprospecting than those found by Simpson et al. Nonetheless, they conclude that biodiversity incentives are unlikely to generate much private sector conservation.

While there are examples of biodiversity in the UK being used to develop pharmaceuticals, such as digitalis (foxglove) in heart complaints, the focus of bioprospecting is on the world's biodiversity hotspots. If the marginal pharmaceutical value of a species is small in biodiversity hotspots such values are likely to be small in the UK too.

There may be other bioprospecting economic applications for genes and species beyond pharmaceuticals. Biotechnology involves the use of living things, organisms such as bacteria, fungi and in all applications including engineering and technology (e.g. the fermentation of cheese and beer, breadmaking, biodynamics). Looking at the genomes of bacteria could help in finding and exploiting those genetic traits that may make more flavourful cheeses, wine, sausages, etc.

One potential area for further exploration may be in the UK's marine environment. Lloyd-Evans (2005) estimates the global market value for marine organisms, used in biotechnology, at $2.4 billion in 2002, and notes a predicted growth rate exceeding 10% per annum over the next three years. Marine organisms have a variety of possible applications ranging from health care to industrial cleaners. The report identifies 21 companies that appear to have some connection with the use or exploitation of marine resources in the UK. Some of the viable business opportunities identified that capitalise on the UK's marine environment include applications of biofilm knowledge in anti-fouling, use of marine viruses, development of new enzymes for biocatalysis and development of bioactives for infections (rather than cancers).

Maintaining genetic diversity

Crop wild relatives and landraces contain the progenitors of our present day crops. Villa and colleagues (2005) suggest a definition for a landrace as 'a dynamic population(s) of a cultivated plant that has historical origin, distinct identity and lacks formal crop improvement, as well as often being genetically diverse, locally adapted and associated with traditional farming systems'. Maintaining crop wild relatives and landraces offers potential benefits to domesticated crops as well as insurance type values. Improvements obtained through genetic transfer from wild relatives have included drought and salt tolerance, early ripening and increased nutritional values, such as protein and vitamin content. Poysa (1993) suggests that the economic returns from investment in crop wild relatives and landraces can be substantial; for example, genetic material from a tomato wild relative has allowed plant breeders to boost the level of solids in commercial varieties by 2.4 per cent, which was estimated to be worth $250 million annually to processors in California. In terms of insurance, modern agriculture involves growing a narrow range of cultivars, with a narrow genetic basis, over large areas. This may increase the vulnerability of production to changes in climate or land use, and exposure to biotic stresses, including new races of pathogens. The potential benefits of landraces have been described in a recent review (Newton et al 2010). They are a potential source of traits for improved nutrition of cereal crops. They also have the potential to improve mineral content, particularly iron and zinc, if these traits can be successfully transferred to improved varieties. Landraces have been shown to be valuable sources of resistance to pathogens. There is also potential, largely unrealised as yet, for disease tolerance and resistance to pest and various abiotic stresses, including to toxic environments. Increasingly however, landraces are being replaced by modern cultivars which are less resilient to pests, diseases and abiotic stresses. We may potentially be losing a valuable source of germplasm for meeting the future needs of sustainable agriculture in the context of climate change. Conserving genetic diversity has a range of potential economic benefits. Some of these may be obtainable through ex-situ, rather than in-situ conservation.

Perrings (2001) and Pascal and Perrings (2007) explain why the public goods nature of conserving genetic diversity may lead to a socially sub-optimal supply. Since the social insurance benefits of higher levels of crop genetic diversity are not rewarded in many current markets, farmers have little private incentive to conserve it . From a farmer's perspective, the most profitable decision is frequently to grow only a few crop varieties, and not to invest in conservation of the varieties that are less 'favoured' by the market. In the case of genetic diversity, those farmers who do maintain *in situ* crop genetic diversity are essentially conserving a global public good and thus they can be seen as net-subsidizers of modern agriculture and food consumers worldwide.

Conservation of rare breeds of animals may also entail an option value in preserving genetic information which may have future utility to human society. They may also have significant value in terms of their social heritage.

As discussed above, the role of pollinators, such as bees, in maintaining crop production is well documented and of high importance, in Europe as elsewhere in the world. Pollination is therefore an essential ecosystem service which maintains biodiversity and supports other vital ecosystem functions; including soil protection, flood control and carbon sequestration. There is strong evidence that loss of pollinators reduces crop yield and that the availability of a diverse pool of pollinators tends to lead to greater yields.

Non use values of biodiversity

Components of biodiversity are valued by people for a variety of reasons. These include the appreciation of wildlife and of scenic places and the spiritual, educational or religious benefits associated with the natural world. Biodiversity plays an important role in fostering a sense of place and historical meanings and cultural importance are benefits of ecosystems linked to folklore, intellectual and spiritual traditions, art and heritage. The NEA notes that biodiversity itself can therefore be regarded as an ecosystem service in that many components or attributes of the natural world, like charismatic species or landscape beauty, may be valued by people. Biodiversity has both use and non-use values and Krutilla (1967) articulated a range of non use values people may hold for nature including the existence value of species and bequest values associated with knowing future generations may benefit from the continued existence of species. Section 8 of this appendix (Cultural services, Mourato et al 2010) identifies proxies, like legacies left to wildlife charities, which lend strong support for the existence of such non use values. There are also other health and well being benefits from engagement with biodiversity where benefits may be positively correlated with species richness or other dimensions of biodiversity (Fuller et al 2007, Weinstein et al 2009). Survey approaches have been adopted to gauge the direct health or psychological benefits of nature but attempting to establish monetary values for the non use benefits poses major difficulties because of their inchoate nature.

Heal et al 2005) argues that stated preference techniques requires information to be available to describe the change in a biodiversity (or natural ecosystem) and that the change must be described in the survey design in a way people can understand. Applied valuation techniques appropriate for high experience goods and services may not be valid for low experience ones for which stable preference have yet to be formed (Bateman et al 2008). Human cognitive limitations can therefore undermine monetary valuation (particularly non-use estimation through stated preference techniques) because individual survey respondents are unable to properly reference frame the problem despite state of the art survey designs (Turner et al 2003, Morse Jones 2010).

A number of UK valuation studies do incorporate an element of non use values though a major limitation of stated preference valuation techniques is the inability to distinguish individual components of value. Despite these clear limitations, stated preference surveys will often include an element of non use values as one motivation yielding a positive preference for some component or aspect of biodiversity. Very few studies have actually looked purely at the 'diversity' value of biodiversity. We summarise a number of stated preference and other studies in Table 5 (1) below.

TABLE 1. Summary of UK valuation studies that have included some component of the 'diversity' value of biodiversity

Author	**Date**	**Geographical remit**	**Aspect valued**	**Method**	**Key conclusions – more details in the reports**
Beaumont et al for Defra[1]	2006	UK	Marine biodiversity	Market, CV, replace-ment cost	Range of values presented for 13 different services
Boatman et al for Defra[2]	2010	England	Landscape, carbon, wildlife benefits of Environmental Stewardship	CV and CE	£1,083 million p.a. (mid value)
Christie et al for Defra[3]	2010 – approx October	UK	Biodiversity Action Plan continuation and	CE	n/a

[1] *Marine Biodiversity: an economic valuation*

[2] *Estimating the Wildlife and Landscape Benefits of Environmental Stewardship*

http://www.defra.gov.uk/evidence/economics/foodfarm/reports/index.htm

			full implementation		
Christie et al[4]	2006	Survey conducted in Cambridgeshire and Northumberland	Diversity of Biodiversity	CE and CV	Public has positive valuation preferences for most, but not all, aspects of biodiversity. They appeared to be largely indifferent to how biodiversity protection was achieved.
Christie et al for Defra[5]	2004	England	Variety of aspects of biodiversity valued, and wtp for different policy options such as agri-environment tested	CE and CV	the total economic value of agri environmental scheme, habitat re-creation scheme and biodiversity loss as a result of development in Cambridgeshire were £16.55m, £12.25m and £10.10m per annum respectively, while in

[3] Christie et al, 2010, *Economic Valuation of the Benefits of Ecosystem Services delivered by the UK Biodiversity Action Plan*, Defra.
[4] Valuing the diversity of biodiversity, *Ecological Economics* 58 304– 317
[5] Christie et al, 2004, *Developing measures for valuing changes in biodiversity*, Defra.

					Northumberland, the values of the habitat re-creation scheme and protect against biodiversity loss from development schemes were £6.21m and £4.82m per annum respectively.
Foster et al	1998	UK	Bird-friendly bread	CE	UK value preventing the decline of nine species was estimated to be £246 million per year
McVittie, Moran[6]	2010	UK Marine	Marine biodiversity	CE	Revealed preferences for both halting the loss of or increasing marine biodiversity in UK waters
NERA for Defra	2007	England, Wales	Good ecological water quality (use and non-use values) of the Water Framework Directive	CV and CE	Very large range. The CV produced total value for households in England and Wales of £1 billion - £3.8 billion p.a to

[6] McVittie, A., Moran, D., Valuing the non-use benefits of marine conservation zones: An application to the UK Marine Bill, Ecol. Econ. (2010),

					achieve high water quality by 2015
SAC for Defra[7]	2008	UK	Non-use value resulting from implementation of the Marine Act – halting the loss of biodiversity	CV and CE	Aggregate mean values were £1,611 to £1,810 million per year, median aggregate £1,170.7 million per year.
White et al[8]	2001	North Yorkshire	Value of otters, water voles, red squirrels and brown hares	CV	Regional value for North Yorkshire ranged from £0.42 million for brown hares to £2 million for red squirrels
Willis et al[9]	2005	Somerset Levels and South Downs	Landscape value of environmentally sensitive area policy(agri-environment scheme) compared to intensive farming	CV	Range of values from residents and visitors. Somerset Levels visitor's aggregate wtp was £15 million, South Downs £77

[7] Scottish Agricultural College, 2008, Determining monetary values for use and non-use goods and services: Marine Biodiversity – primary valuation, Defra.
[8] White, P.C.L., Bennett, A.C. and Hayes, E.L.V , 2001, "The use of willingness to pay approaches in mammal conservation", *Mammal Review*, 31, 2, 151 - 167

[9]Willis K. G., Garrod G. D. and Saunders C. M., 1995, Benefits of environmentally sensitive area policy in England: a contingent valuation assessment, *Journal of Environmental Management*, 44, 2, 105 – 125.

					million.
Willis et al – Forestry Commission[10]	2003	England, Scotland and Wales	Social and environmental benefits of forests, including biodiversity	For biodiversity, used focus groups and benefit transfer	Biodiversity in British forests was approx £368 million p.a.

Issues to consider in valuing biodiversity

A prime motivation for marginal valuations of ecosystem services is the belief that doing so can help us make substantially better decisions regarding natural resource use. There are limits to economic valuation and some ecosystem service benefits lend themselves more successfully to monetary valuation than others. This section highlights some considerations which influence the scope and applicability of monetary valuation.

Irreversibility: Krutilla (1967) and Krutilla and Fisher (1975) clarified the economic theory underpinning many of the values associated with biodiversity. A useful taxonomy of environmental value used by economists is the Total Economic Value concept. The concept includes option and quasi option values as components. The first, introduced by Weisbrod (1964), is commonly defined as the price that individuals are willing to pay for conversion of a natural asset in view of its possible use in the future. It is not related to current use and is typically used to gauge the value attached to future use opportunities. Some authors have interpreted option value as comparable to a risk premium arising from uncertainty as to the future value of a natural asset if it were to be preserved. Other interpretations stress the inter temporal aspects of the problem and the irreversibility of any decision to convert a natural asset, such as a National Park, to alternative uses (Hanneman 1984). Quasi-option value, as discussed by Arrow and Fisher (1974), relates to the welfare gain associated with delaying a decision in the face of uncertainty regarding the payoffs of alternative choices, and where at least one of the choices involves an irreversible change in land or resource use. This value stems from the benefits obtained from information gained by delaying an irreversible decision to convert a natural environment. Some economic activities, such as mining, damming rivers or mineral extraction, can involve changes to the natural environment which are essentially irreversible. In addition, ecological interactions may transform seemingly reversible economic actions into irreversible ecological alterations. For

[10] Willis, K, Garrod, G, Scarpa, R, Powe, N, Lovett, A, Bateman, I, Hanley, N and Macmillan, D., 2003, *The social and environmental benefit of forests in Great Britain*, Forestry Commission.

example, competitive interactions may prevent a valuable wildlife population from ever recovering, even after the damaging economic activity has ceased.

In consideration of these values, Krutilla (1967) argued that, 'natural environments will represent irreplaceable assets of appreciating value with the passage of time'. This may occur as society becomes larger and wealthier and as cheaper alternatives are found for the economic activities which led to the alteration of the ecosystem (energy generation for example). Irreversibility is therefore a critical consideration for decision-making (Weisbrod 1964; Arrow and Fisher 1974, Dixit and Pindyk 1994, Maler 2008). Krutilla and Fisher (1975) propose a means of modifying Cost Benefit Analysis to account for the values associated with conservation. In a similar vein, the TEEB Foundation Report, Chapter 5 (2010 forthcoming) identify a critical factor in discounting as the importance of environmental draw-down (destruction of natural capital) to estimates of the future growth rate of per capita consumption.

While decision making techniques have been proposed to account for these values, attempts to actually measure them in monetary terms have proved inconclusive. Some literature has looked at this point by trying to assess people's willingness-to-pay for future environmental assets, but no clear answer has emerged from this literature (Fisher and Krutilla, 1974, Hanley and Splash, 1993; Desaigues and Point, 1993).

Threshold effects. The work of Krutilla and Fisher (1974) focused on irreversibility mainly in the sense of the loss of a unique landscape. The conversion of boreal forest and muskeg wetlands to recover oil from Canada's tar sands for example. Others have considered the issue from a more ecological perspective in relation to potential threshold effects on ecosystem functioning. These occur where the functioning of the ecosystem is shifted to an alternative stable state which may undermine human interests. Once a threshold is crossed, further ecosystem, or resource depletion either physically cannot be reversed and the costs of reversibility may substantially outweigh benefits. Bateman et al (2010) term this situation 'economic irreversibility'.

We frequently lack sufficient understanding of ecosystem dynamics to actually locate thresholds. This complicates decision making given that a key reason for understanding complex systems is to inform decision-makers about when, or under what circumstances, an undesirable, irreversible change is likely to occur. This uncertainty has led to a safe minimum standard (SMS) approach where the SMS is the minimum quantity of ecosystem structure and process (including diversity, populations, interactions, etc.), that is required to maintain a well-functioning ecosystem capable of supplying services (Fisher et al 2009). It can be thought of as a precautionary approach to the management of natural assets (Ciriacy-Wantrup, 1952). Under such an approach conventional economic decision making prevails until a threshold threat is identified. At this point the onus of proof shifts away from assuming that development is justified unless the costs to the environment do not justify proceeding, to a presumption that conservation is the right option unless the sacrifice (i.e. the opportunity costs) that it entails is intolerably high. The SMS approach provides a safety-first approach to ensuring the future sustainability of human society through maintenance of the ecosystem stocks and services upon which it is reliant (Bateman et al 2010). For practical management purposes, Haines-Young and Potchin (2007) distinguish ecological *thresholds* from *limits* which they define as 'the 'point or range of conditions beyond which the benefits derived from a natural resource system are judged unacceptable or insufficient'

Rockström et al (2009), identify planetary scale environmental limits which have clear implications for the use of conventional marginal valuation. The authors argue that to avoid catastrophic environmental change global society must stay within defined 'planetary boundaries' for a range of essential Earth-system processes despite not knowing which unit of change might actually cause ecosystem collapse. An additional complication for decision making arises if the transgression of one boundary poses serious risks to the safe levels for other processes.

Genes, Species and Ecosystems. Chapter 5 of the NEA (Norris 2010) identifies specific ecosystem services predominantly provided by specific biodiversity groups. With such services, like pollination, wildlife watching or wild plants maintained for crop breeding, specific species may provide the service directly. These services may be relatively straightforward to identify. For example Maxted et al (2007) identify 1,955 plant species in the UK that are genetically related to a plants of economic importance with 303 being related to a major food species. For many other ecosystem services (e.g. water purification, nutrient cycling, and climate regulation) the link between aspects of biodiversity and service delivery is less tractable. This is because it may be the genetic or ecosystem level or interactions between levels, that primarily determines service delivery. Understanding ecological processes that underpin ecosystem services therefore plays a key part in our understanding of the link between diversity and the value of ecosystem services.

Ecosystem functionality and complexity. The delivery of some ecosystem services depend on interactions beyond single ecosystems and many can vary in time and space. Landscapes consist of several functionally integrated ecosystems such that disturbances to one ecosystem can have complex and indirect effects on other ecosystems within the landscape. Implications for management are that it is not always possible to isolate any one ecosystem when evaluating ecosystem services. Water purification is a good example.

Perrings (2006) observes that the economy and its environment co-evolve through time, and that the coupled system is complex and adaptive, exhibiting path dependence, non-linearity, and sensitivity to initial conditions. This generates fundamental uncertainty about the future consequences of current actions and suggests that for any given set of technologies there is a sustainable scale of the global economy. Tallis et al (2008) recognise that for effective ecosystem management "interlocking production models of the full suite of ecosystem services are needed". A clear steer for future research directions

Marginality. The appropriate context for economic valuation is conditioned, among other things, by the scale of the environmental changes. Fisher et al (2009) note the difficulty of establishing what constitutes a marginal change in regards to ecosystem processes. Measurement of incremental values works best when the increments are small, so that a change in one service will have minimal feedbacks through the rest of the system.

Non economic notions of value What do we mean by the *value* of nature? Clearly notions of value are complex and multidimensional. One possible interpretation by ecologists could be the contribution of something to a condition of state of an ecosystem system. Structures and functions of natural systems, by this definition, have value (i.e. the value of a tree in perpetuating a forest ecosystem). Such notions, together with non anthropocentric notions of intrinsic value, which relate to nature's inherent worth, are beyond the scope of economics.

Estimating the costs of managing biodiversity in the UK

Metrick and Weitzman(1998) identify the defining limitation for using economics in biodiversity preservation as the lack of a common denominator or natural anchor articulating what biodiversity is and therefore the lack of an objective function for assessing cost effective delivery or how to determine basic priorities for maintaining or increasing diversity. They attempt to develop a cost-effectiveness criterion that can be used to rank priorities among biodiversity-preserving projects under a limited budget constraint. Other economists have also attempted to introduce economic considerations into conservation planning (Polasky et al 2005). A lack of agreement about what biodiversity is or what the objective should be have limited practical applications of cost effective approaches.

In lieu of economics values for the biodiversity of the UK, the cost of managing biodiversity is taken as an indication of the value society holds for biodiversity. This is based on the assumption that the political biodiversity targets and legal mechanisms that have been brought in to support biodiversity are a reflection of the desire of the public through the political process. Specifically expenditure should reflect the value of biodiversity as an ecosystem service itself (the cultural or scientific values) rather than the value of biodiversity in providing other ecosystem services. In particular, we discuss the costs of running the UK Biodiversity Action Plan (BAP) and protected areas in the UK. A weakness with this approach is that the level of funding might be inefficient or ineffective in achieving the desired biodiversity outcomes. Even if the level of funding is correct, and distributed efficiently, society's values could still be even higher. However, in lieu of better data this approach can be used as indicative of the minimum value society holds.

What and where is the BAP and protected areas?

The UK Biodiversity Action Plan (BAP) was established in 1994 to respond to the Convention on Biological Diversity's 1992 challenge to slow the rate of biodiversity loss. It consists of national strategies and action plans to identify, conserve and protect existing biological diversity and enhance it where possible.[11] There are 65 priority BAP habitats and 1150 species. Achieving the UK BAP would mean that these species are secure and that people will be able to benefit from their continued existence across the UK.

The 2010 target has not been achieved. However, this missed target has been replaced by the 2020 European target, which was adopted by EU Heads of States in March 2010 and aims at "*halting the loss of biodiversity and the degradation of ecosystem services in the EU by 2020, and restoring them in so far as feasible, while stepping up the EU contribution to averting global biodiversity loss*".[12] In the absence of any announced changes to biodiversity policy, it is assumed UK BAP will represent one important way of achieving this new goal.

Even if it is changed, or abandoned altogether, the BAP still represents the most comprehensively costed biodiversity plan in the UK. The costs of managing BAP in each of the UK countries were estimated for Defra in 2006, and recently updated (GHK 2010). The annual amount estimated as the cost to deliver the UK BAP for 2010-2015 is £837 million.

The UK's protected area network is another important part of the commitment society has made to biodiversity. Sites of Special Scientific Interest (SSSIs) in England, Scotland and Wales and Areas of Special Scientific Interest (ASSIs) in Northern Ireland represent a network of the best examples of natural features throughout the UK. They have been identified as special for habitats, animals, plants or geology. The Natura 2000 is made up of Sites of Special Protection Areas (SPAs) for Birds and Special Areas of Conservation (SACs). These are areas of European importance. These sites are legally protected under the Birds and Habitats Directives.Table 2 describes the area of each network in the four UK countries.

TABLE 2: Protected areas within the UK (hectares)

	England	**Scotland**	**Wales**	**NI**
Natura2000	851 558	930 427	219 179	78 515
SSSI/ASSI	1 015 269	1 019 683	253 716	90 508

In England and Wales all Natura 2000 sites are within the SSSI network. Nearly all Natura 2000 sites are included in the Scottish and Northern Irish SSSI/ASSI network, so it is going to be assumed in the following analysis that all are included.

[11] http://www.ukbap.org.uk/

[12] Although it is not clear what the new Coalition government will do with BAP, it will remain the most recent costed biodiversity action for the near term.

The BAP is not a spatially explicit plan, and good data on spatial extent is difficult to get. A recent estimate is shown in Table 3. These represent some of the most important of the 65 priority habitats. Additionally, there are marine habitats.

TABLE 3: Area ('000 Ha) of BAP habitats by region, country and UK .[13]

Area of habitat ('000 Ha)	**England**	**Scotland**	**Wales**	**Northern Ireland**	**UK**
All habitats in region	1981	2874.5	495.8	366.3	5648.8

Overlap between BAP and SSSIs/N2K

The overlap between the BAP priority habitats and SSSIs is difficult to assess. The data set for BAP habitats is very large and it has not been possible to obtain an overlap-free layer that can be used with other data layers (such as SSSIs). However, Natural England has produced one estimate of the proportion of priority BAP areas that are also SSSIs for the majority of the priority BAP habitats in England.[14] This analysis produced an estimate that meant 42% of SSSIs were not in the BAP network.

For our analysis, in the absence of better information we are going to take this proportion and apply it across the UK. This means that we will use the most recent GHK estimates of BAP costs, but assume there is 42% additional area of protected areas outside of BAP habitats. The next section discusses how we have estimated the costs for this non-BAP area.

Methodology for this cost assessment

Where there is an overlap between BAP and SSSIs/N2K, it is assumed that the management costs will be similar between designations, and the BAP cost is used on its own. As there is not a perfect spatial overlap between BAP and these protected areas, BAP does not represent all of the costs of managing biodiversity in the UK. Accordingly, additional costs for the parts of protected area management that do not overlap with BAP, and some marine management, are included.

[13] Aberystwyth BAP benefits 2010

[14] Pers Comm., 2010, George Hinton Natural England.

Costs used were sourced from a report commissioned by Defra, *UK Biodiversity Action Plan: Preparing Costings for Species and Habitat Action Plans* (2006). The summary of these costs is shown in Table 4.

TABLE 4:Summary costs of BAP 210-2015

Component of BAP	Revised Estimates, 2010-2015, UK, 2009 Prices
Habitat Action Plans	**516**
Individual Species Action Plans	***47***
Action for Widespread Species	***274***
Species Action Plans	**321**
UK BAP Costs	**837**

Using the BAP costings has the virtue of achieving a narrow estimate of financial requirements related to conservation targets (rather than site spend that might include social spending such as trails and hides). The BAP estimates also limit inclusion of central agency costs and overheads (administration in general). For this reason they can be considered as conservative.

Other reasons why this estimate can be considered conservative is that we have assumed the non-BAP protected areas terrestrial network is complete, which may not be the case for some SPAs. Additionally, the BAP process does not include other biodiversity costs in society, such as:

- Regulatory costs for regulation that protects species and habitats
- Planning costs for regulation that protects species and habitats (e.g. EIA production, survey work, negotiation).
- Water catchment management to meet directives such as Bathing Water Quality and Water Framework Directive.
- Work on Invasive Species, e.g. grey squirrel strategic control, Rhododendron control; and Japanese knotweed bio-control programme and physical removal, etc.
- Species and habitat conservation programmes outside the BAP – e.g. the SNH Species Action Framework; RSPB sea eagle work etc.

Non-BAP costs – protected areas and marine

This section sets out an indicative estimate of the financial requirements for the effective management of biodiversity outside of BAP priority habitats in the UK. Only Habitat Action Plans (HAPs) are assessed and used to estimate the costs of these non-BAP sites. This is because the SSSIs and N2K are site based management and do not generally incorporate species action plans.

Although there is likely to be higher restoration work outside of the BAP and in SSSI areas (as of 1 Feb 2009, based on a report run from NE's database, 44% of English SSSIs were in favourable condition, meaning the rest were unfavourable in some way, mostly recovering), there are not expansion costs as the terrestrial SSSI network is largely complete. The BAP costs are still likely to approximate the SSSI costs, as roughly 40% of the area covered by key priority HAPs have restoration or expansion plans, with similar costs for both.

As the total annual costs of managing HAPs is £516 million/year, 42% of this to represent the cost of biodiversity management for the non-PA areas (as discussed earlier) is £217 million a year.

The BAP costs include some freshwater and marine costs that will be similar outside of BAP. However, the establishment of new marine SAPs and marine conservation zones isn't included in BAP, and should be added on. Although some new IAs for new marine areas have been conducted (e.g. the Dogger Bank SAC), it is more appropriate to deal with the estimate for the total network. The marine costs for new marine conservation zones were estimated at £42-82 million each year. The mid-point here is £62 million. A one-off capital cost of £20-24 million has been divided by twenty years here, to give an average of £1.1 million. In total, marine areas will cost around £63 million.

Total costs

The total costs of biodiversity management in the UK are shown in Table 4. They represent just over £1 billion a year.

TABLE 5: total costs of biodiversity management in the UK

Habitat	**Cost (£ million)**
BAP total	837
Additional for protected areas	217
Marine –	63
TOTAL	1,117

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