Developing measures for valuing changes in biodiversity:

Final Report

Report to

DEFRA
London

From

Dr Mike Christie[1], Dr John Warren[1], Prof. Nick Hanley[2], Dr. Kevin Murphy[2], Prof. Robert Wright[3], Mr Tony Hyde[4] and Mr Nick Lyons[4]

[1]University of Wales Aberystwyth
[2]University of Glasgow
[3]University of Stirling
[4] Socio-Economic Research Services

January 2004

Table of contents

List of Tables

List of Figures

Executive summary

This document reports the findings from the DEFRA funded research project 'Developing measures for valuing changes in biodiversity'. The aim of the research was to develop an appropriate framework that will enable cost-effective and robust valuations of the total economic value of changes to biodiversity in the UK countryside. The research involved a review of ecological and economic literature on the valuation of biodiversity changes. The information gathered from this review, along with the findings from a series of public focus groups and an expert review of valuation methodologies, were used to develop a suite of valuation instruments that were used to measure the economic value of different aspects of biodiversity. Contingent valuation and choice experiment studies were administered to households in Cambridgeshire and Northumberland, while valuation workshops were conducted in Northumberland only. The data from these studies were also used to test for benefits transfer.

Review of ecological and economic literature

The key issues identified in the review of ecological literature included:

- There is no one simple measure of biodiversity.
- Ecologists agree that species richness (the number of species per unit area) is a useful starting point for measuring biodiversity. However, there are issues regarding definitions of species and identification of a suitable area in which to measure biodiversity.
- Ecologists also recognise that some species are likely to be more important than other species with respect to enhancing and conserving biodiversity, e.g. keystone species and umbrella species.
- It was also recognised that humans may have anthropocentric preferences for certain species (e.g. cute and charismatic species), even though these species may not necessarily be important in ecological / biodiversity terms.
- Biodiversity may also be measured in terms of habitat diversity and ecosystem diversity.

The economic review highlighted the following issues:

- The total economic value of biodiversity comprises direct values (use, passive-use and options values) and indirect values.
- There are a range of methodologies available to value biodiversity change including revealed preference, stated preference, and cost-based approaches. However, no one method was considered to be capable of valuing all aspects of TEV associated with biodiversity change.
- Revealed preference methods (e.g. travel cost method and hedonic pricing) are largely restricted to the measurement of use values.
- Stated preference methods (e.g. contingent valuation and choice experiments methods) are in theory capable of estimating both use and passive-use values. However, in practice they are less suited to measuring indirect issues such as ecosystem services. The choice experiments approach has the added advantage that it is also capable of valuing the component elements of biodiversity.
- Cost-based approaches (e.g. replacement costs, restoration costs, preventative expenditures) infer a value for natural resources (including ecosystem functions and services) by how much it costs to replace or restore a resource after it has been damaged. In other words, these techniques do not measure the utility or economic value accrued to individuals from improvements in biodiversity.
- A review of existing valuation studies identified that although a significant amount of work has been undertaken to investigate the value of biological resources (and in

particular the value of individual species and habitats), few studies have attempted to value biological diversity *per se*. Furthermore, very little research has attempted to disentangle the value of the components of biodiversity.

Methodology and results

A series of developmental focus groups were undertaken to explore public understanding of the biodiversity concepts identified in the review to ecological and economic literature. The key findings from the focus group were that public understanding of the term biodiversity is generally low. However, the public do have the capacity to understand the concepts of biodiversity if described in layman's terms. Furthermore, it was clear that the way in which the public consider biodiversity is different to the way in which ecological experts consider biodiversity. Thus an important lesson from this is that studies that value complex goods such as biodiversity need to be careful in the way they present information on that good.

The actual valuation study utilised three survey instruments: a contingent valuation study, a choice experiment study and a series of valuation workshops. The contingent valuation and choice experiment studies were combined into a single survey instrument, which was administered to 400 household in both Cambridgeshire and Northumberland. During these interviews, information on biodiversity was presented using an innovative MS PowerPoint presentation. Six valuation workshops were administered in Northumberland only. The format of the workshops initially followed that of the household surveys, but also included further discussion of biodiversity and the choice task, as well as a further series of choice experiment choice tasks.

Contingent valuation study

The contingent valuation study addressed three biodiversity enhancing and protection policies:

- An agri-environmental scheme that aimed to enhance biodiversity on arable land through the creation of conservation headlands and the reduced application of pesticides and herbicides. Biodiversity benefits from this scheme would include an increased diversity of plants, insects, small mammals and birds; some of which may be rare.
- A habitat re-creation scheme that would enhance biodiversity by creating new wetland habitats on existing farmland. The new wetland would provide habitats for a wide range of plants, insects, small mammals and birds, including a number of rare species. In addition, the wetland area would provide ecosystem services such as flood protection and enhanced water quality.
- The third scheme would aim to avoid biodiversity loss as a result of housing development on farmland managed under existing agri-environmental schemes. The types of biodiversity protected under this policy would be similar to those described in the agri-environmental scheme above.

The key findings from the contingent valuation study included:

- In Cambridgeshire, the value of the agri-environmental, habitat re-creation and protect against biodiversity loss from development policies were £74.27, £54.97 and £45.30 respectively, where these values related to annual WTP amounts per household over a five year period.
- In Northumberland, the values of the habitat re-creation scheme and protect against biodiversity loss from development schemes were £47.49 annually per household and £36.84 annually per household respectively.
- In all cases, these values were found to be significantly different from zero.

- Furthermore, the estimated WTP values for the alternative policy scenarios were not found to be statistically different from one another.
- There was some evidence of consistency in mean WTP values for policies between the two case study areas, however, this was not the case for the transfer of the bid functions.
- The key policy implications of the above findings are that the public are willing to pay a positive sum of money for biodiversity enhancing and protecting policies. However, there were no significant differences between the values of the alternative policy prescriptions. Thus, we are unable to make clear recommendations with regard to which types of biodiversity policy should take priority.

Choice experiment method

The second method utilised was the choice experiment method. The CE study assessed the value of four attributes of biodiversity:

- *Familiar species of wildlife*. This attribute was described to include the concepts of charismatic, familiar (recognisable) and locally symbolic species. Three levels of this attribute were presented: protection of rare familiar species, protection of rare and common familiar species, and the status quo (continued decline).
- *Rare, unfamiliar species of wildlife*. This attribute focused on those species that are currently rare or in decline which are unlikely to be familiar to members of the public. The three levels of this attributes included: the slow down of decline of rare unfamiliar species, the recovery of populations of rare unfamiliar species and the status quo (continued decline).
- Species interactions within a habitat. This attribute was used to represent the importance of species interactions within a habitat, as well as a proxy for the preservation of ecologically significant species such as keystone and umbrella species. Levels of provision of this attribute included: habitat restoration, habitat re-creation and the status quo (continued decline).
- *Ecosystem processes*. Ecosystem processes focused on biodiversity's role in preserving the health of ecosystem processes. Levels of this attribute included: preservation of ecosystem processes that directly affect humans, preservation of all ecosystem processes, and the status quo (continued decline).

The key findings from the CE study were as follows:

- The attribute that targeted the 'recovery of rare unfamiliar species' attained the highest implicit price (£115 and £189 respectively for Cambridgeshire and Northumberland). Furthermore, this attribute was the only one that was valued significantly higher than any of the other attributes.
- In contrast to the above, the 'slow down the rate of decline of rare unfamiliar species' was found to be negative in the Cambridgeshire sample, while the attribute level was not significant in the Northumberland CE model.
- In Northumberland, both the protection of 'rare familiar species' (£90.59) and 'both rare and common familiar species' (£97.71) were found to achieve consistently high implicit prices, while in Cambridgeshire the protection of 'rare familiar species' (£35.65) was found to be significantly lower than the protection of 'rare and common familiar species' (£93.49).
- In Northumberland, the 'habitat restoration' attribute (£71.15) was found to be similar to the 'habitat re-creation' attribute (£74.00), while in Cambridgeshire the 'habitat re-creation' attribute (£61.36) achieve a higher implicit price than the 'habitat restoration' attribute (£34.40).
- Finally, the 'ecosystem processes' attribute with direct impacts for humans was highly valued in both Cambridgeshire and Northumberland (£53.62 and £105.22 respectively. However, the all 'ecosystem processes' attributes (which included the

human impact level) was not significant in Northumberland and was lower than the human impact level in Cambridgeshire. The reason for this findings appears to stem from the fact that generally there was a lower level of understanding of this attribute and therefore people valued it less.

The key policy implications of the CE data is that the public do value most, but not all, biodiversity attributes and that they appear to be able to distinguish between alternative attributes (but perhaps not always attribute levels). In particular, there is evidence to support the continued funding of policies that target species, habitats and ecosystem processes. Of particular interest is the finding that the public have high values for the protection of rare unfamiliar species; thus policies should not be restricted to target only familiar and charismatic species. Second, the comparison of the results between Cambridgeshire and Northumberland for the rare familiar species attribute level and the habitat restoration attribute level are interesting in that it would appear that people in Cambridgeshire have low values for these two attribute levels as a direct result of the perception that Cambridgeshire currently does not support such biodiversity.

Valuation workshop

Six valuation workshops (53 participants) were administered in Northumberland. The format of the workshops followed that of the household surveys, but also included further opportunities to discuss biodiversity and five further choice experiment tasks. The key findings from the workshops included:

- The information presented in the PowerPoint presentation allowed participants to attain a good understanding of all biodiversity attributes apart from the ecosystem processes attribute.
- Although the extra discussions in the workshop improved participants understanding of biodiversity concepts, this extra level of knowledge did not significantly influence their values for the biodiversity attributes.
- The discussion of the participant’s choice strategies provided evidence that participants were using consistent and consider valuation choices.

Conclusions

We argue that this research has been successful in attaining meaningful and robust values for complex goods. Evidence supporting this claim comes from a number of sources including the validity tests for the alternative valuation studies and the responses from the valuation workshop. We, however, stress that valuing complex goods is challenging, and in particular a lot of effort needs to be undertaken in developing the hypothetical descriptions of the goods in question. In our study, this effort included an ‘expert’ (ecologists) review of biodiversity and a series of focus groups to ‘translate’ the expert view into a language which was both understandable and meaningful to the public. Also we presented the information using an innovative MS PowerPoint presentation.

With regard to developing a cost-effective and robust framework for valuing biodiversity change the conclusions are less clear. First, tests for benefits transfer between the two case study areas generally failed. Thus, we cannot advocate the transfer of (robust) benefit values from our study areas to other areas of the UK. Although the reasons for the failure of benefits transfer are unclear, it may be that it is due to differences in the existing levels of biodiversity within the two case study areas. Further work would be required to clarify this. Second, the failure of benefits transfer (which is considerably less expensive that undertaking original studies) means that we cannot use the study results to provide a low cost framework for valuing biodiversity in the future. On a more positive note, we believe that our approach was largely successful in providing a robust framework in which to value biodiversity change. In particular, we argue that the public were capable of understanding our descriptions of biodiversity policies and attributes (with perhaps the 'ecosystem processes' being the exception). Thus, we recommend the use of the contingent valuation method for the valuation of biodiversity programmes and the choice experiment method for biodiversity attributes. Finally, an interesting result from this research was that the value estimates from the six Northumberland valuation workshops (which included additional discussions on biodiversity) were largely equivalent to the 400 responses from the Northumberland household survey. If such equivalence could be demonstrated to be consistent in other areas (say for example if we repeated the workshop in Cambridgeshire and find equivalence), then undertaking valuation workshops in other counties of England and linking the values from these counties to either the value of a low biodiversity area (ie. Cambridge) or a high biodiversity area (Northumberland), this may provide a relatively cheap framework to allow a robust aggregation of this studies results to the UK as a whole.

1. Introduction

This document reports the research undertaken for a DEFRA funded project 'Developing measures for valuing changes in biodiversity'. The aim of the research contract is '*to develop an appropriate framework that will enable cost-effective and robust valuations of the total economic value of changes to biodiversity in the UK countryside'*.

Before getting into the detail of this report, it is first useful to provide some background on why one might be interested in measuring the economic value of biodiversity and second to identify some of the potential problems and challenges that researchers may encountered while undertaking such a valuation exercise.

1.1. Why value biodiversity?

Biodiversity is important to humans for various reasons. First, biodiversity may increase an individual's welfare directly. This may be through actual use of a biological resource (e.g. recreational use of natural areas) or through passive-use benefits (e.g. derived from the knowledge that biodiversity is being protected for future generations to enjoy). Biodiversity may also increase an individual's welfare indirectly through its contribution towards the maintenance of ecosystem functions such as the regulation of the water and carbon cycles (Fromm, 2000; Pimm *et al.,* 1995). The conservation of the Earth's biological resources is thus essential to preserve the well-being of both current and future generations.

Human activities, however, have also contributed towards the unprecedented decline in the Earth's biodiversity. This, in turn, may be threatening the stability of the Earth's ecosystem functions, as well as the capacity of the Earth to provide ecosystem services to man. In order to protect and maintain the Earth's biodiversity, human society needs to make difficult decisions regarding its use of biological resources. For example, policies may need to be adopted to reduce livestock stocking densities on farmland to promote biodiversity. Environmental valuation techniques can provide useful evidence to support such policies by quantifying the economic value associated with the protection of the Earth's biological resource. Pearce (2001) argues that the measurement of the economic value of biodiversity is a fundamental step in conserving the biological resource since '*the pressures to reduce biodiversity are so large that the chances that we will introduce incentives* [for the protection of biodiversity] *without demonstrating the economic value of biodiversity are much less than if we do engage in valuation*'. Assigning monetary values to biodiversity is thus important since it allows the benefits associated with biodiversity to be directly compared with the economic value of alternative resource use options. Such evidence is likely to greatly to assist in the formulation of policies that protect biodiversity. OECD (2001) also recognises the importance of measuring the economic value of biodiversity and identifies a wide range of uses for such values, including:

- Demonstrating the value of biodiversity: awareness raising showing the importance of biodiversity,
- Determining damages for loss of biodiversity: liability regimes,
- Revising the national economic accounts,
- Setting charges, taxes, fines,
- Land use decisions, e.g. to make a case for sustainable agriculture / forestry or to protect an area,
- Limiting biological invasions,
- Limiting or banning trade in an endangered species,
- Assessing biodiversity impacts of non-biodiversity investments, e.g. road building,
- Setting priorities for biodiversity conservation within a limited biodiversity budget.

Furthermore, it is evident that the role of environmental valuation methodologies in policy formulation is increasingly being recognised by policy makers. For example, the Convention of Biological Diversity's Conference of the Parties decision IV/10 acknowledges that '*economic valuation of biodiversity and biological resources is an important tool for well-targeted and calibrated economic incentive measures'* and encourages Parties, Governments and relevant organisations to '*take into account economic, social, cultural and ethical valuation in the development of relevant incentive measures'.* The EC Environmental Integration Manual (2000) provides guidance on the theory and application of environmental economic valuation for measuring impacts to the environment for decision-making purposes. The manual suggests that environmental valuation should be undertaken alongside Environmental Assessment studies. Within the UK, the HM Treasury's 'Green Book' provides guidance for public sector bodies on how to incorporate non-market costs and benefits into policy evaluations.

1.2. Valuing biodiversity: the challenge!

Although environmental valuation techniques are increasingly being utilised to aid policy formulation, there is however still some latent resistance to placing monetary values on biodiversity. In particular, some environmental analysts argue that nature has non-anthropocentric "intrinsic values" and thus non-human species possess moral interests or rights (O'Neil, 1997; Ehrenfeld, 1988). Such positions lead to the advocacy of environmental sustainability standards, which to some extent preclude the need for valuation. However, general consensus accepts that placing monetary values on biological resources makes explicit the fact that biodiversity is used for instrumental purposes in terms of productive and consumptive opportunities (Fromm, 2000; Nunes and van den Bergh, 2001) and therefore will help policy makers make more informed decisions regarding the use of biological resources.

The validity of the various valuation methodologies have also been questioned (Bate, 1993) and as a consequence many of these methods have been subject to intensive academic debate and scrutiny. One of the key assessments of the validity of stated preference valuation methods (and the contingent valuation (CV) method in particular) occurred in 1993 following a natural resource damage assessment of the Exxon Valdez Oil disaster off the coast of Alaska. As part of the damage assessment, a contingent valuation study was conducted to assess the passive-use values associated with the prevention of future oil spills. The results from this study sparked an intensive debate on the validity of CV. To resolve this debate, a NOAA (National Oceanic and Atmospheric Administration) blue ribbon panel of experts was set up to review the validity of CV. The conclusions from this review was '*that CV studies can produce estimates reliable enough to be the starting point of a judicial process of damage assessment, including lost passive use values'* (Arrow *et al.,* 1993). However, the NOAA panel recognised that there was a variability in the quality of CV studies and therefore produced a set of guidelines for CV.

In addition to concerns regarding the validity of valuation methods, there are also a number of concerns relating to the valuation of biodiversity that need to be considered. These include incommensurate values, lexicographic preference issues (Spash and Hanley, 1995; Spash, 1993), the problem of dealing with protest votes (Spash, 1993), intergenerational rights issues (Bromley, 1995), people's understanding of a complex good (Christie, 2001; Limburg *et al*., 2002).

Another major concern for the valuation of biodiversity relates to different levels of understanding of the complexities of biodiversity by both the general public and the scientific community. In particular, stated preference valuation methods require survey respondents to make value judgements on the environmental good under investigation. This requires information on these goods to be presented to respondents in a meaningful and understandable format, which in turn will enable them to express their preferences. Here lies the problem. Studies have consistently found that members of the general public have a low awareness and poor understanding of the term biodiversity. For example, quantitative research undertaken in 1988 found that 63% of a UK sample did not know what the words ‘biological diversity’ meant (MORI, 1988b). More recent work for the Scottish Office confirms that public understanding of environmental terminology, including ‘biodiversity’, is very low. However, a study valuing biodiversity in British forests reported that although respondents generally had a poor understanding of the importance of wildlife in itself, ‘environmentalists’ and ‘outdoor enthusiasts’ were found to have a clearer understanding of ecological systems (ERM, 1996). This was also the case in a study valuing endangered species (Macmillan *et al*., 2001b) where responses demonstrated an understanding of the environment in general and more specifically wildlife conservation. Furthermore, other studies found that the UK public disliked the phrase ‘biological diversity’, preferring the terms ‘variety of life’, ‘living diversity’ and ‘biological variety’ (MORI, 1988a), or ‘variety of wildlife’ (ERM, 1996). Other research has shown that once the concept of biodiversity was explained in layman’s term a high proportion of the general public (78%) considered that ‘biological diversity’ was important (MORI, 1991). The findings from these studies will have significant implications for the valuation of biodiversity. In particular, the lack of public understanding of the term biodiversity will make the valuation exercise extremely difficult.

The issues highlighted above indicate that research that attempts to value changes in biodiversity will be challenging. Not only will research need to address and overcoming many methodological issues associated with environmental valuation techniques, but it will also need to identify appropriate language in which biodiversity concepts can be meaningfully conveyed to members of the public, thus enabling them to express their preferences. The research described in this report aims to address these challenges.

1.3. Structure of report

This report is structured into nine sections. Following this introduction to the report, Section 2 provides a review of ecological literature on biodiversity. Included in this review are sections on defining, measuring, and predicting biodiversity changes. Section 3 then examines the economic literature related to the valuation of biodiversity changes. Topics covered include the total economic value of biodiversity, economic valuation methodologies, and a review of studies that have attempted to value biodiversity. Section 4 involves the development of a conceptual framework in which to describe biodiversity concepts to the public. This framework was then scrutinised during a series of public focus groups to refine this framework. Also, in Section 4 alternative valuation protocols are systematically assessed by experts using a suitability matrix scoring system (SMSS). In Section 5 of the report, we clarify the research aims and objectives. The principal aim of the research was to develop an appropriate framework to enable cost effective and valid valuations of the total economic value of changes to biodiversity in the UK. To achieve this, we also addressed a number of objectives, including the valuation of the attributes of biological diversity, the valuation of biodiversity policies, and the examination of benefits transfer. Section 6 reports the three methodologies that were used in this research, including a contingent valuation study that measured public willingness to pay for three biodiversity policies (agri-environmental scheme, habitat re-creation scheme and protection against biodiversity loss as a result of development projects), a choice experiment that measured public willingness to pay for four biodiversity attributes (familiar species of wildlife, rare unfamiliar species of wildlife, species interactions within a habitat and ecosystem services), and valuation workshops that aimed to further explore the issues relating to the valuation of complex goods. The results of these studies are reported in Section 7 and are then discussion in Section 8. Finally, we draw conclusions in Section 9.

2. An Ecologist's Perspective of Biodiversity

2.1. Defining biodiversity

The concept of biological diversity, originally simply meaning "number of species present", appears to have been first developed in the sense in which it is used today during the 1970s – early 1980s (Peet, 1974; Lovejoy, 1980a,b; Norse & McManus, 1980), despite attempts to strangle the idea at birth (Hurlbert, 1971). A few years later, Norse *et al.* (1986) defined biological diversity at the genetic (within-species), species (species numbers), and ecological (community) level. The contracted term "biodiversity" came from a "National Forum on Biodiversity" held in the USA in 1986, the Proceedings of which (Wilson, 1988) brought the term, and concept, into more general use.

Although there are many possible definitions, perhaps the most widely-accepted is that provided in Article 2 of the "Convention on Biological Diversity" (signed by 157 national and supra-national organizations) at the 1992 UN Conference on the Environment and Development:

> *"Biological diversity" means the variability among living organisms from all sources, including, inter alia, terrestrial, marine and other aquatic systems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems.*

More recently, Harper & Hawksworth (1995), in the preface to The Royal Society's review of the biological diversity concept, suggested that biodiversity is best considered at three levels, 'genetic', 'organismal', and 'ecological' (or 'community') biodiversity. There is general agreement today that this approach is appropriate in the study of biodiversity - environment relationships. Current major ecological research effort, worldwide, is focused on understanding the implications of biodiversity for ecosystem productivity and functioning (e.g. Aarssen, 1997; Diemer *et al.,* 1997; Hodgson *et al.,* 1998; Yachi & Loreau, 1999; Hughes & Roughgarden, 2000; Cottingham *et al.,* 2001); and also in assessing human impacts on biodiversity, and the ecosystems which support biological communities (e.g. Willoughby, 1992; Chapin *et al.,* 1998; Naeem *et al.,* 1995; Sala *et al.,* 2000).

For example, in the context of environmental-related human impacts (in this case acidification, eutrophication, potential CO_2 increase, and leisure-use increase) upon the aquatic plant biodiversity of European lake ecosystems, Murphy (2003) suggested that diversity responses at these three scales were an appropriate basis for assessment:

- Genetic level diversity. Genetic variation may be partitioned within or between populations and may be the basis of locally adapted populations, races, and sub-species. It may be quantified at the molecular level, although historically such variation was described based on the measurement of physiological or morphological traits (e.g. Pieterse *et al.,* 1984; Nielsen and Sand-Jensen, 1997; Vöge, 1997a,b; Madeira *et al.,* 1999; Hollingsworth *et al.,* 1995, 1996).
- Species level diversity. This may be quantified simply as the loss or gain of species (S) between different locations (e.g. Szmeja and Clément, 1990), or over time (e.g. Macan, 1977; Van Dam and Kooyman - Van Blokland, 1978; Wallsten, 1981) or in both time and space (e.g. Roelofs, 1983). Changes in the relative abundance of species within a community (changes in community structure or as one community changes into another, e.g. via succession or agricultural 'improvement') are another facet of species level diversity (e.g. Arts *et al.,* 1990). As with species loss this may result from human induced environmental changes.

- Functional / ecological diversity. This is a more complex concept (Steneck & Dethier, 1994; Hills *et al.,* 1994; Herrera *et al.,* 1997). It relates to the complexity of ecosystems processes (number of interactions) occurring within a community, which arise from the number of functional groups of organisms present (Farmer and Spence, 1986; Murphy *et al.,* 1990).

In addition to assessing what is meant by the term "biodiversity", Harper & Hawksworth (1995) also identified seven major questions related to measuring biodiversity, some of which have been addressed reasonably well at the present time, while others still require considerable research to answer properly.

The questions are summarized below:

1. Is biodiversity just the number of species in an area?
2. If biodiversity is more than the number of species, how can it be measured?
3. Are all species of equal weight?
4. Should biodiversity measures include intraspecific genetic variability?
5. Do certain species contribute more than others to the biodiversity of an area?
6. Are there useful indicators of areas where biodiversity is high?
7. Can the extent of biodiversity in taxonomic groups be estimated by extrapolation?

To this set we may add two additional questions, relevant to the issue of predicting change in biodiversity, and using such change to assess the value and health of ecosystems:

8. Can biodiversity be used as a measure, or indicator, of the health ("biointegrity") of ecosystems?
9. Is biodiversity a useful measure for environmental valuation purposes?

The above nine questions have been identified by ecologists as a useful approach to the definition and measurement of biodiversity changes. We now explore each of these questions in turn, highlighting how ecological concepts may be translated to form the foundation of the valuation exercise.

2.2. Measuring biodiversity

2.2.1. Is biodiversity just the number of species in an area?

Biodiversity is frequently divided into a hierarchy of three levels: ecosystems/habitats, species and genes. Ecosystems are defined as communities of co-occurring species of plants and animals plus the physical environment; as such they are difficult to define and delimit. At the other end of the spectrum, genes are currently still difficult to identify and count. Thus, species counting is the obvious tool for measuring biodiversity. Therefore, although biodiversity may be measured at levels from genome to biome (Colwell and Coddington, 1994; Roy and Foote, 1997; Hawksworth, 1995; Lovejoy 1995; Magurran 1988), measurement of the number of species present (S) within a defined target area is generally accepted as one of the simplest measures of biodiversity. This of course raises the problem of defining the target area in which to record the inventory of species, which again requires the ability to define and delimit ecosystems. Whittaker (1977) identified four levels at which it is useful to measure species diversity. The smallest of these scales is *point* diversity, or micro-habitat diversity in which the inventory area is defined as being a homogeneous habitat. Above this level is *alpha* diversity or within-habitat diversity, which is probably the most widely used scale for recording species numbers. The third scale of diversity, *gamma* diversity, is that measured at the landscape scale, the area involved may contain a number of different ecosystems within a local area, and may include areas such as islands. The largest scale is Whittaker's fourth-level; *epsilon* or regional diversity, which applies to large biogeographic area, and comprises the total diversity of a group of areas of *gamma* diversity. To be able to compare areas in terms of their diversity Whittaker coined three additional levels of differentiation diversity (*pattern* diversity, *beta* diversity and *delta* diversity). *Pattern* diversity is defined as the measure of differentiation diversity between samples taken within a homogeneous habitat. *Beta* diversity (or between habitat diversity) is by far the most widely used measure of differentiation diversity. It is defined as the change in species composition and abundance between areas of *gamma* diversity. *Beta* diversity can be estimated as change in species diversity along a gradient (Wilson & Mohler, 1983) or by comparing the species composition of different communities. *Delta* diversity is defined as the change in species composition and abundance between areas of *gamma* diversity, which occur within areas of *epsilon* diversity. As such it is used to represent differences in diversity over wide biogeographic areas. In addition to these four levels of species diversity, and three levels of differentiation diversity, ecologists also measure diversity in terms of the structural complexity of habitats and how it relates to niche width.

Once the area to be considered has been defined, Harper & Hawksworth (1995) identified a further problem in basing biodiversity measures on a simple taxonomic concept. They postulated a site of defined area, containing just two organisms (i.e. S = 2), one being a plant species of the genus *Ranunculus* (e.g. *Ranunculus acris*: meadow buttercup), and the other from a list including:

a. another species of *Ranunculus* from the same section of the genus (e.g. *Ranunculus repens:* creeping buttercup),
b. another species of *Ranunculus* from a different section of the genus (e.g. *Ranunculus ficaria*: lesser celandine),
c. a species from a different genus in the family Ranunculaceae (e.g. *Anemone nemorosa*: wood anemone),
d. a species from a different plant family, in a different order (e.g. a grass such as *Anthoxanthum odoratum*: sweet vernal grass),
e. a fungus of the genus *Agaricus,*
f. a rabbit.

In taxonomic terms the diversity within the site is generally increasing as we go down this list because the species involved are further apart in evolutionary terms. As Harper & Hawksworth (1995) point out "…*any measure of biodiversity which described all these sites as equal would be particularly uninformative*": as is clearly shown by the fact that the measure remains at S = 2 throughout this series.

The answer to this problem lies in clearly identifying the basis for comparison of S (e.g. between sites, or over time, or both) on a taxonomic basis (e.g. for specified plant groups down to species level only: Wilson *et al.,* 2003; for birds: Parish *et al.,* 1994); or on a functional basis (e.g. McGrady-Steed & Morin, 2000; Symstad *et al.,* 2000). Where "like with like" comparisons of change in S can be made, in this way, then potentially useful trends and changes in biodiversity can be identified which are of practical use for policy, conservation, or management purposes.

In practical terms, probably the majority of recent studies which have aimed at examining and/or predicting biodiversity change, in relation to human activities, have incorporated S as (at least one) indicator of biodiversity status. Examples from a disparate range of habitat types would include Dony & Denholm (1985), Parish *et al.* (1994), Ali *et al.* (2000), Bini *et al.* (2001), Chamberlain & Fuller (2001), Downie *et al.* (1999), Wilson *et al.* (2003), Marshall *et al.* (1996), and McIntyre & Lavorel (1994).

For the practical reasons outlined above, ecologists most frequently describe biodiversity as some function of the number of species per unit area, even when they are interested in defining habitat, ecosystem or regional diversity. Part of the driver for this methodological approach is the scientists need to quantify variation. However, the general public may not be motivated by the same desire and may value higher levels of diversity (e.g. habitat biodiversity) without reference to species counting. Indeed, our understanding of the way in which members of the general public think about and value biodiversity is limited. We do not know whether the public understands, or is even aware of, the ecological concepts such as species, habitats and ecosystems. Thus, one of the key issues that this research will explore will be the extent of public understanding of ecological concepts of biodiversity. This was achieved through a series of public focus groups and is reported in Section 4.

2.2.2. If biodiversity is more than the number of species how can it be measured?

Three possible approaches were considered to take into account the issue of taxonomic divergence when assessing diversity (as identified in the *Ranunculus* example above):

a. Taxic measures. This approach utilises counts of the number of higher taxa present, rather than species, to indicate the biodiversity of a site. An example is Williams *et al.* (1994), who found a strong relationship between number (per 0.1 ha) of seed plant species and number of families represented by those species, and went on to produce maps of plant family richness on a world scale. On the other hand, Prance (1995) showed that only 6.4% of the known plant species present in the neotropics (tropical Central and South America) belong to the c. 40 exclusive or near-exclusive neotropical plant families, suggesting that assessment of plant biodiversity at family level would seriously underestimate actual plant diversity. A further problem is related to the poor taxonomic understanding of the real degree of specification in certain families: for example the supposed 242 species of *Hieracium* (hawkweeds) in the Norwegian flora (Lid, 1952) are almost certainly "*…better indicators of taxonomic traditions than of the scale of natural biological diversity*" in this group (Harper & Hawksworth, 1995).

b. Molecular measures. Improving knowledge of the DNA and RNA genomes of organisms could potentially provide the basis for measuring diversity: the biodiversity of a community could theoretically be measured as the sum of the variety of genetic information coded in the genotypes of all the organisms present, or any given subset of these (Embley *et al.,* 1995). However, this is a long way from being practically applicable at present.

c. Phylogenetic measures. Some suggestions have been made that the optimal approach to assessing biodiversity at a given site is to work downwards through the main phylogenies represented at a given site: i.e. to assess the number of clades represented within each kingdom, then phyla per kingdom, orders per phylum and so on, in order to place a relative value on biodiversity which reflects the "taxonomic distinctiveness" of the organisms present, based on the degree to which a sister-group of organisms has shown independent evolutionary history within the phylogeny (May, 1995). A practical problem with this approach is that although reasonably good phylogenies are available for some biota (e.g. flowering plants: Chase *et al.,* 1993) these remain poorly or not at all developed for several major groupings of organisms.

A second problem is deciding how much weight to allocate to groups showing lengthy independent histories (i.e. an early evolutionary branch-off point within the phylogeny), as opposed to subsequent species radiation (i.e. the number of extant species forming the "sister-group"). One suggestion is that the value placed on each sister-group within a given phylogeny should be equal. Thus, for example, within the reptiles the two species of tuatara still surviving today (on a handful of islands off the coast of New Zealand) form a separate sister-group within the reptile phylogeny, having branched off from the rest of the reptiles before the Triassic (Daugherty *et al.,* 1990; May, 1995). A biodiversity scheme for the reptiles on this basis would give the same weighting to the tuataras as to the sum of all the other 6000 reptile species alive today. While this is an extreme idea, a more sensible weighting might reflect the topology of the phylogeny tree, placing values on sister-groups which reflect their relative degree of independent evolutionary history. To date however relatively few studies have adopted this concept, despite its obvious merit.

In summary, although there is continuing debate about the value of using information at a level other than the species as the basis for assessing biodiversity, the general consensus remains that species-level assessment is still probably of the highest practical value.

2.2.3. Are all species of equal weight?

There are two quite different issues here. One reflects scientific uncertainties attached to exactly what constitutes a species, across different groups of organisms (e.g. Claridge & Boddy, 1994). The other is a reflection of the values which human beings place on the presence or absence of different organisms.

2.2.3.1. The species concept

To take the first issue: for certain species there is little dispute as to what constitutes a "biological species". Whether based on traditional taxonomic identification, or on molecular and phylogenetic evidence, for example, the two common species of British oak (*Quercus robur*: pedunculate oak, and *Quercus petraea*: sessile oak) are clearly identifiable as separate species. However they hybridize easily and the hybrid is fertile. Should we therefore count two, or three, species as present in areas where both parents and the hybrid occur (especially given that the hybrid is commoner than the individual parents in many parts of the British Isles: Stace, 1991)?

In other, apomictic, (asexual) plants (such as the genus *Hieracium*, already mentioned) treating each apomictic "species" as separate would greatly overestimate measures of plant species richness within an area (however this is countered by the fact that very few botanists can actually distinguish these plants down to "species" level, so in practice micro species tend to be lumped together in sections: only 12 sections being given by Stace (1991) for the 258 currently-recognised micro species of British *Hieracium*). In spite of this fact conservationists frequently accept apomictic species as being as 'worthy' of protection. For example, three out of the four species in the IUCN red data list of critically endangered terrestrial species, which occur in the UK, are in the genus *Sorbus*, as is one of two endangered species and six of 17 threatened species (*Sorbus* being another genus containing many apomictic species).

A similar but less extreme complication surrounds the breeding behaviour within sexually reproducing species. The hemi-parasitic plant genus *Rhinanthus* provides a good illustration of the issue. Within the UK there are two members of the genus; *R. angustifolius*, an out-breeding species, and *R. minor*, a very similar and inter-fertile in-breeding species. Because of its out-breeding habit, populations of *R. angustifolius* contain high levels of genetic diversity, but there is little variation between populations. In contrast, individual populations of *R. minor* contain low levels of genetic variation but they differ markedly from other populations (consequently it has been divided into many sub-species, some of which have been awarded high conservation status because of their rarity). Thus we can see that the breeding system of a species will affect how it partitions its genetic variation, which in turn may affect its conservation status.

Not only is genetic variation within species partitioned differently depending on breeding system, it also varies across evolutionary time and across space. While the species concept is considered robust for a particular species at a particular time in its evolution, it may be less clear, where one species ends and the next begins over evolutionary time (termed chronospecies). But does this matter when considering the measurement of biodiversity? Because of the relative young age of the British Isles, few of our native species have been isolated long enough to have evolved into distinct species. However, several have apparently started along the path, with both the red grouse and the Scottish crossbill, for example, being recognised as distinct sub-species and arguably species in their own right. The problem for quantifying biodiversity, is therefore just how far along the evolutionary path does a species need to travel before it should be counted in its own right.

In bacteria the problem is the opposite, with the species concept being highly conservative in molecular terms. As an example, the strains known to exist within a single bacterial species, *Legionella pneumophila*, have been shown to possess DNA homologies as different as those which occur between mammals and fish (Harper & Hawksworth, 1995). Nearly all bacterial "species" have at least 70% DNA – DNA relatedness. If that rule was applied to the mammals the number of supposed "species" would decline dramatically. All known hominid species, past and present - with their 98% homology - would, for example, be treated as a single species, putting an end to supposed human evolution between *Australopithecus* and *Homo sapiens* at a stroke. This simplistic statement of course ignores the fact of the huge difference in genome size between a bacterium and a mammal but for comparative biodiversity studies it remains a problem.

Lichens pose another problem for the species concept with regards to conservation prioritisation. Individual species names are ascribed to individual lichens, although each symbiotic relationship may be constructed of up to seven different species of fungi, algae and blue-green algae each with their own species names. Thus, although the UK red data list includes more than 170 species of lichens, the species of algae they contain may be widespread as free-living individuals while the fungal partners may also occur within other more common lichen species. However, information of this kind is not available for most lichens.

Clearly, the above issues raises particular difficulties when "total" biodiversity present at a site is to be assessed, and tends to point once again (given the current state of knowledge) towards the wisdom of applying measures of biodiversity on a group-by-group basis, accepting the fact that there are substantial differences between groups of biota in terms of what exactly constitutes a species.

2.2.3.2. "Cuteness", charisma and rarity

The second issue is one of the values assigned, consciously or unconsciously, by people to different organisms. The "cuteness" concept is an obvious issue: furry and feathery organisms, and attractive plants, are preferred by most (though not all) people to poisonous snakes, weeds and the smallpox virus (May, 1995). Closely related to the 'cuteness concept' is that of flagship species or charismatic species. These are high-profile, impressive species (such as top predators), or species linked to local identity such as national birds or plants (Noss, 1990). Species which possess characteristics which humans value (such as speed) tend to be regarded in higher esteem than species that do not. Indeed, active conservation measures are taken to promote the survival of some organisms (e.g. tropical rainforest birds) at exactly the same locations where active measures are taken to discourage the survival of others (e.g. tropical *Anopheles* mosquitoes). It is also worth noting, in this context, that the ecosystem biointegrity concept (of which biodiversity is an excellent measure: see below for more on this) includes "absence of disease" as one of its attributes (Costanza, 1992), despite the fact that presence of disease organisms must, by definition, increase the total number of species present, so long as the disease does not force other species into extinction in that ecosystem (which can of course happen: an example being the impact of sleeping sickness trypanosomes on mammals in parts of Africa). Although cute and charismatic species are clearly important for biodiversity in terms of human values, there appears to be no scientific indicator or measure of the cuteness or charisma of a species and thus it is difficult to incorporate such attributes into measures of biodiversity.

Rarity (on whatever scale) is a second attribute which contributes to the assigned "value" of species within the biodiversity of an ecosystem or habitat. This concept is inherent in the wide range of active management measures in place for conservation (e.g. Biodiversity Action Plans (BAPs), Environmentally Sensitive Areas etc.: see Potter, 1988; Robinson, 1994; Brotherton, 1996; Simpson *et al.,* 1996); and their driving policy measures, worldwide (e.g. Article 19 of the EC Structure Regulation 797/85 for ESAs; EC Species Directive etc.). The issue often reflects basically irreconcilable structures (such as political boundaries versus natural distributions of organisms), and is commonly allied to public pressure for conservation of preferred "rare" organisms. There are, however, well-recognised difficulties in using rarity as a measure of value in biodiversity assessment (McIntyre, 1992). Furthermore, species may be rare for a variety of different reasons, not all necessarily deserving of higher conservation status. For example newly evolved species are likely to be rare by definition, many such species are likely to fail to become established, but does this matter? Species with very exacting habitat requirements or those at high trophic levels are unlike ever to have been abundant, but should they be awarded the same conservation priority as formally common species that have recently become rare at the hand of man? To take three examples:

a. The rare (in Europe) aquatic plant *Najas flexilis* (slender naiad) is a Red Data Book species, listed in Annex 2 of the EC Species Directive, with its own BAP. Yet this plant is common in North American lakes, and is virtually unheard of by the general public in Europe (Wingfield 2002; Murphy 2002). A high value has been placed on this species by the EC, largely due to scientific pressure, because the overall plant diversity of Europe would be affected by its vulnerability to extinction caused by human impacts on the few lakes where it occurs in Europe.

b. At the other extreme there is enormous political pressure in Europe to protect a migratory bird species, the osprey (*Pandion halietus*), which is common (for a predator species) with an estimated world population of 25,000 – 30,000 pairs (Poole, 1989), and which only occurs in Britain due to an active and expensive protection programme, hugely popular with the public.

c. Somewhere in between is the European beaver (*Castor fiber*), wiped out by human activities from the British Isles in the 17th Century, not uncommon in the rest of Europe, and possibly about to be re-introduced to Scotland (MacDonald *et al.,* 2000). This was the result of a political decision to implement EC biodiversity policy (Nolet, 1997), but with virtually no groundswell of popular pressure in support of the decision (although this is likely to increase once the general public becomes aware of the programme, as the cuteness factor is undoubtedly high in this case!). Interestingly while it was deemed a requirement of EC policy to reintroduce the European beaver, it was not seen as acceptable to simultaneously reintroduce the rabies virus (which many European beavers carry) – so not all species are created equal.

The principal issue is whether it is desirable to assign weightings to individual species (or groups of species) which reflect their perceived value to people, and, if so, how to do this, preferably in a quantitative manner. There have been several attempts to assign weightings to rare species, usually in the context of assessing the "conservation value" of a site for practical management terms. Examples include Dony & Denholm (1985) for small woodland sites in southern England; Murphy *et al.* (1998) for agricultural land in Scotland; and Ali *et al.* (2000) for desert vegetation in the Eastern Sahara. Such schemes usually incorporate some estimate of weighting for the rarer species based on their frequency of occurrence across a defined part of the planet's surface (whether on a local or broader scale). The scheme utilized by Dony & Denholm (1985), for example, assigned rarity scores based on the occurrence of woodland plants within Bedfordshire (obviously very local), then used the sum of total species number (i.e. S) per unit area, proportion of selected "rarer" species within the flora of each site, and the sum of rarity scores for each species present to assess the value of each woodland.

Another useful and practical approach to account for rarity is to make an assessment of the likely threat that a species will become extinct. Such a hierarchy of threat of extinction is currently used in the IUCN red data list, which identify five levels of endangerment: extinct, extinct in the wild, critically endangered, endangered, and vulnerable).

2.2.4. Should biodiversity measures include intraspecific genetic variability?

Within-species genetic variation can be considerable in some species. The example of the bacteria has already been discussed. There are numerous methods for assessing such diversity (e.g. Templeton, 1995). The issue has become one of public interest in the context of the introduction of GM strains of food plants (e.g. Crawley *et al.,* 2001; Watkinson *et al.,* 2000). A related aspect of genetic diversity currently of concern is that of the loss of genetic integrity that may arise following the introduction of alien species or genotypes. The ruddy duck / white-headed duck is a good example of this phenomenon in which the Eurasian white-headed duck faces the threat of extinction following hybridisation with the North-American ruddy duck. While the species involved may technically become extinct, it is possible that at the molecular level, all the genes involved may continue to survive in the new hybrid population. It seems unlikely that the public have much concept of such genetic level variation at the molecular level, however, when such variation is manifest as the occurrence of sub-species such as the red grouse or Scottish crossbill, the story may be very different. The complexity of how the public perceive and value diversity below the species level is an area of research that requires further exploration.

2.2.5. Do certain species contribute more than others to the biodiversity of an area?

Thus far, we have argued that species richness appears to be the most useful practical measure of biodiversity, and that the general public's preferences for individual species may be influenced by charismatic / anthropocentric factors such as cuteness or rarity. Such factors, however, have little meaning in terms of an ecologist's perception of the importance of a species. Ecologists have identified certain species which make significant contribution to enhancing the biodiversity of an area.

2.2.5.1. The keystone species concept

The value of certain species in influencing (positively or negatively) habitat or resource provision for other species is a primary consideration here. This can occur directly, as in the case of British oaks, which provide resources assisting the survival of a great range of other organisms, including food for a range of herbivores; nest and feeding sites for Lepidoptera, birds and arboreal mammals; habitat for obligate oak-associates such as gall-wasps; mycorrhizal fungi; bark- and leaf-dwelling fungi; a range of pathogens; and epiphytic bryophytes and lichens, to name but a few (Morris & Perring, 1974). Alternatively keystone species may be less abundant species from higher in the food-chain. The classic example is the North-American sea otter. After being hunted to the edge of extinction in the nineteen century, there was a dramatic increase in the sea urchin populations (a major component of the otter's diet) which in turn resulted in the disappearance of kelp forests along the American west coast. Thus, such keystone species are thought to be pivotal species about which the diversity of a large part of the community depends. However, that is not to say the community itself will cease to function and become unrecognisable following the loss a keystone species. Indeed the National Vegetation Classification system (Rodwell, 1991) recognises oak woodland communities even in the absence of oak trees! Allied to the keystone species concept is that of the ecological indicator species (Noss, 1990). Such species are easy to monitor and variations in their numbers are used to indicate that an environmental change has occurred that is likely to have produced perturbations in the population of several other species with similar habitat requirements.

In freshwater streams in Britain, submerged plants such as *Callitriche* (water starworts) can substantially enhance the biodiversity of the stream habitat by increasing the bioarchitectural complexity of the habitat, thereby increasing the number of macroinvertebrate species which can be supported by the stream system (e.g. O'Hare & Murphy, 1999). Such keystone species (some of which have much less obvious roles than oaks or water starworts: e.g. the role played by the herbivorous fish *Pterodoras granulosus* in Brazilian rivers for seed dispersal of terrestrial plants: Souza-Stevaux *et al.,* 1994) can greatly increase the biodiversity of a site at which they are present (Hawksworth *et al.,* 1994).

Other species may play a more indirect role in altering the biodiversity-support functioning of an ecosystem by, for example, influencing the physico-chemical characteristics of a site. An example of the positive influence of organisms on habitat provision would be the role played by lichens in commencing soil development, and a vegetation succession, in newly-opened habitats at the snout of a retreating glacier. At the other extreme, toxin-producing cyanobacterial blooms (e.g. *Microcystis*) in eutrophic lakes may have a negative effect on biodiversity by killing off fish or zooplankton populations in the lake.

Recently a number of studies have claimed that community structure and hence ecosystem function is regulated by a small number of dominant species, which can be predicted by trait variation between species, for traits such as seed size (Crawley *et al.,* 1999; Turnbull *et al.,* 1999; Rees *et al.,* 2001). This has been termed the 'selection effect' and appears similar to the keystone species concept. In contrast Loreau & Hector (2001) argued that 'complementarity' of resource partitioning resulting from different species exploiting different resources by processing different traits is more important in regulating community processes. Thus 'complementarity' theory maintains that all the species present in a community have a role in regulating ecosystem functioning. There is evidence that both selection and complementarity mechanisms are likely to operate in combination in regulating community structure (Price, 1995; Loreau, 1998). Recent interest in complementarity as a determinant of community structure results from the fact it implies that the loss of *any* species has potential important consequences for ecosystem function, which is currently a central issue in ecology. In contrast, keystone or selectionist theories imply that the loss of many species may produce little or no effect on ecosystem function. However, they warn that such species loss does matter, because the species concerned may be keystone species in other ecosystems or are potential keystone species in communities yet to evolve. Since keystone species are by definition likely to be abundant/dominant species they are unlikely to be given high conservation status directly. However, the communities they dominate (and arguably help regulate) may well be targeted by Biodiversity Action Plans, Agri-environment prescriptions etc.

While the above classification of species as keystone species plus the arguments about dominant species versus complementarity are based entirely on the theory of ecosystem functioning, ecologists also classify species in terms of their potential importance in conservation. Such species are therefore identified in part by their perceived ability to attract human interest. For example, the terms umbrella species and flagship species are used to describe two related concepts, which describe a species' potential impact in promoting conservation. Umbrella species typically require large areas of habitat for their conservation. These are typically large mammals or birds, which need a variety of habitat types or alternatively require large blocks of a single habitat. Thus promoting the conservation of such species (which almost by definition tend to be charismatic-mega fauna) also automatically promotes the conservation of large tracts of habitat plus all the other species that share this resource.

2.2.5.2. Equitability.

Most natural biological communities show a characteristic structure in terms of both the species present and their relative abundances. The classic case is dominance by only a few common species, with a "tail" of additional members of the community, present in decreasing numbers per species. Extreme cases tend towards more-even numbers per species (rarely encountered, unless managed to that aim by human intervention, and difficult to sustain even then: ask any gardener) or towards extreme domination by a single species, with a tail of other species present in low numbers (common: any arable field under normal agricultural management, where the dominant species will be the crop plant, with other species being primarily pest, disease organisms and weeds, kept at low density by active management measures – pesticides and other agronomic procedures).

Numerous indices of biodiversity have been developed to take account of the concept of equitability or the evenness of species, so that communities with similar values of S (number of species per unit area of habitat), but differing relative abundances, can be quantitatively differentiated. The most commonly-used measures are the Shannon Index and Simpson's Index (Ghent, 1991). Quite frequently, such indices are significantly correlated with S (e.g. Wilson *et al.,* 2003) and may provide no advantage over the simpler measure for practical purposes (e.g. modelling), though they are clearly useful where equitability of species occurrence needs to be taken into account for conservation or other purposes (Hawksworth, 1995; Bini *et al.,* 2001). Ecologists typically use such indices to separate communities with similar species lists of the kind illustrated in Figure 1, or to track change within a community over time. However, the idea of equability of species abundance within communities may be alien to many members of the public.

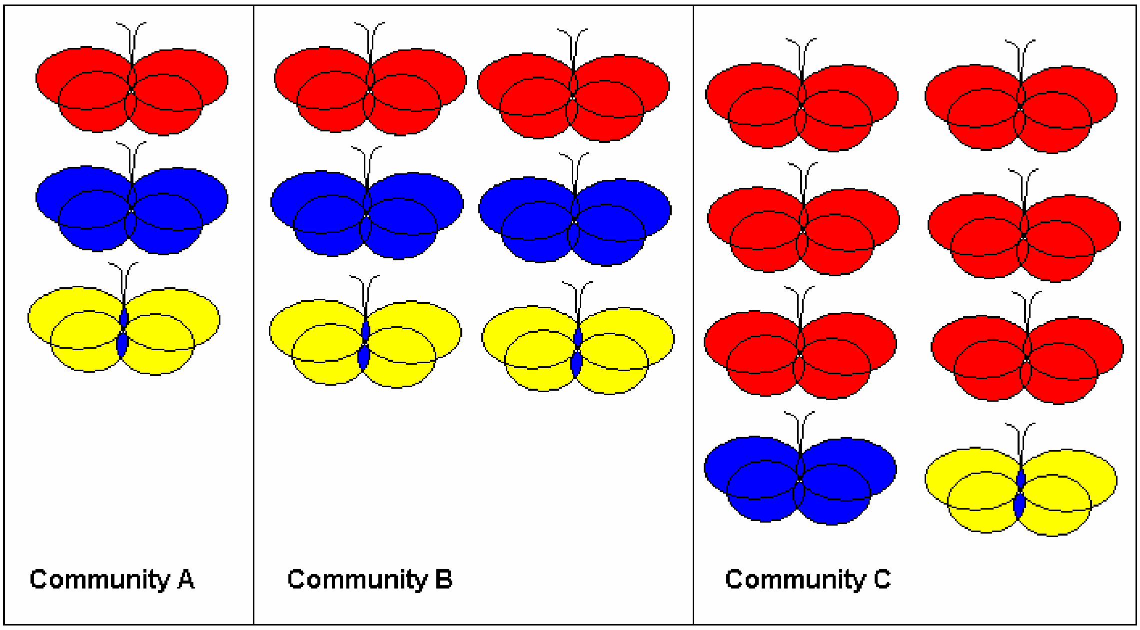

Figure 1: A theoretical example of three different communities comprised of the same three species, but which is the most diverse?

2.2.6. Are there useful indicators of areas where biodiversity is high?

Identification of biodiversity indicators which can show, for example, areas of the landscape where environmental and management factors combine to produce high overall diversity is an important issue in regard to policy formulation and implementation (Reid *et al.,* 1993), and we discuss the use of modelling approaches in this context later in this review. The concept of "biodiversity hot-spots" is well-established (e.g. Carey *et al.,* 1996). As an example, on a landscape level, Aspinall (1996) presented data to show the patterns of Shannon Index biodiversity (for either 34 or 20 taxonomic classes) at differing spatial resolutions for the whole of Scotland, illustrating major differences in diversity related to geographical factors.

The problem with this approach is that it is limited by the robustness of the species concept. While biologists generally agree that the species concept (see above) is sound across a limited range of organisms, it is becoming increasingly clear that the species concept differs between groups of species, such that species of bacteria differ from each other to a different degree than do species of birds or mammals. Thus taxonomic groups such as genera or families have little value when compared across taxonomies. This matters for example, when comparisons are made between the numbers of species found in a square metre of deep-sea mud, and a hectare of tropical rain forests. Such exercises are clearly meaningless, as the species being counted in each community do not share a common unit of discreteness.

2.2.7. Can the extent of biodiversity in taxonomic groups be estimated by extrapolation?

Estimation of diversity patterns on a large scale, e.g. on a continental scale: (Prance 1995; May 1995) provides both conceptual and practical difficulties. Species accumulation curves and rarefaction analysis offer possible solutions (e.g. Colwell and Coddington 1995; Downie *et al.,* 1999).

2.2.8. *Can biodiversity be used as a measure, or indicator, of the health of ecosystems?*

2.2.8.1. *Ecosystem biointegrity*

The definition of ecosystem health (or "biointegrity") is commonly based on a set of conceptual attributes (Costanza, 1992), including:

(1) homeostasis,
(2) absence of disease,
(3) diversity or complexity,
(4) stability or resilience,
(5) vigour or scope for growth,
(6) balance between system components.

For these attributes to be meaningful it is assumed that ecosystems have functions that can be related to some measure of diversity. Assessment of ecosystem biointegrity is often based on snapshot studies, sometimes on a comparative basis, but increasingly the approach taken is to use methods based on the changed-state concept, both on land and in aquatic ecosystems (e.g. Scheffer, 1998; Scheffer *et al.,* 2001).

The basis of the changed state concept is simple. The observed state of system quality (O) is compared with the state expected (E) at some designated historical point, usually prior to major human impact upon the system (e.g. 1850 in the case of the Scottish Standing Waters Classification Scheme: Fozzard *et al.,* 1999). A "hindcasting" approach may be used to model the E state (e.g. Moss *et al.,* 1994; Bodini 2000; Ferrier *et al.,* 1996; Dixit *et al.,* 1993; Allott & Monteith 1999). Alternatively, the state of sites is compared with baseline sites of high quality (reference sites) little-impacted or preferably unimpacted by human activities, and "typical" of the type of system under consideration. The approach may be used to classify the ecological quality of the system, and to monitor changes in that quality.

It has long been known that diversity as measured by species richness, peaks during vegetation succession, before climax vegetation communities have developed (typically woodland). Thus in the UK where the vast majority of habitats have been modified by man, the highest levels of species diversity are not associated with pristine, ancient woodlands, but with semi-natural agricultural habitats such as chalk grasslands. Therefore any simple mechanism for valuing biodiversity based on habitat naturalness is likely to give different results from one based on species counting.

Biodiversity provides a good measure (though not the only one) of the biointegrity of an ecosystem, and the biotic communities which it supports (Perlman and Adelson, 1997; Dickinson and Murphy, 1998). A major task in applied ecology is to predict the impacts of different scenarios of human impact (e.g. land management) on the biodiversity of plant and animal communities of ecosystems (Scheffer and Beets, 1994; Murphy and Hootsmans, 2002). Minimal linear models, which use biodiversity, or other ecosystem functional response variables, as indicators of changes occurring at ecosystem level, within a defined envelope of environmental conditions (Scheffer and Beets, 1994) have proved to be useful tools to understand the functioning of ecological communities within a range of different ecosystem types (e.g. Hilton *et al.,* 1992; Scheffer, 1992; Wilson *et al.,* 1996; Willby *et al.,* 1998; Ali *et al.,* 1999; Ali *et al.,* 2000; Murphy *et al.,* 2003).

One limitation to the study of ecosystem integrity has been the difficulty of defining ecosystem properties. This issue was responsible for the confusion surrounding the diversity – stability debate. The inter-relationship between diversity and ecosystem stability was much worked on by ecologists in the 1950s and 60s. Conventional wisdom at the time was that ecosystem stability increased with ecosystem complexity or diversity. More recently it has been argued that ancient diverse ecosystems containing many trophic interactions are likely to be more sensitive to species loss than are simpler ecosystems constructed of a few generalist species. The higher the proportion of specialist (monophagous) species within an ecosystem the more species are likely to be lost following the extinction of a single species. Grime (1998) argued that ecosystem stability was not a property of diversity *per se* but was related to the presence of dominant species which regulate ecosystem function. Lehman & Tilman (2000) clarified the stability – diversity debate by pointing out that community level stability (as defined by the ecosystems' ability to recover following perturbation e.g. fire) increases with diversity, however, the stability of individual populations of species within the ecosystem decreases with diversity.

The concept of ecosystem health therefore has a number of limitations with regard to its possible use in the valuation exercise. First, as noted above, biodiversity (as defined as the number of species) may not be directly correlated with ecosystem health. Second, there are major problems with respect to attaining a common indicator of ecosystem health across ecosystems. Finally, the term ecosystem health has now become common use language and therefore the term is largely meaningless for use in the valuation exercise.

2.2.8.2. Modelling biodiversity status of ecosystems

Attempts to model biodiversity (recently reviewed by Murphy & Hootsmans, 2002) have focused on predicting either change in the richness (S) of a target biota (e.g. McIntyre & Lavorel, 1994; Díaz & Tellería, 1994), or some measure of diversity incorporating equitability (such as the Shannon or Simpson's Index: e.g. Peet, 1974; Ali *et al.,* 2000; Aspinall, 1996), or have attempted to predict change in assemblage (e.g. Nilsson *et al.,* 1988), or have examined the impacts of change in biodiversity on other ecosystem parameters (e.g. Yachi & Loreau, 1999). It is envisaged that such models could, at least, identify the bounds to which biodiversity change is reported in a valuation exercise.

Diversity-environment relationships in plant communities, the "humpback model"

High biodiversity at intermediate intensities of environmental disturbance (*sensu* Grime, 1979), associated with a degree of temporal variability in habitat conditions (e.g. seasonal drying) is in line with the predictions of the "hump-back" model of biodiversity (Grime, 1979; Dickinson and Murphy, 1998; Willoughby, 1992). Hump-back relationships between plant diversity and environmental stress have been observed, for example, in relation to trophic status (an indicator of intensity of environmental stress) in Swiss lakes ranging from ultraoligotrophic to hypertrophic, where maximum diversity occurs in the mid-range of trophic class (Lachavanne, 1985). In a review of aquatic macrophyte diversity in Central and North America, Crow (1993) provided evidence for a correlation between higher diversity, and adaptation to seasonally variable wet/dry climates. He cited as an example the high diversity of *Utricularia* in Nicaragua, where 15 species of this genus occur, particularly in seasonally-wet sites.

From a comprehensive analysis of the literature on aquatic vegetation in 622 Scandinavian lakes, across a wide trophic spectrum, Rørslett (1991) found evidence that the principal predictors of macrophyte species richness were lake area, altitude, trophic state and several water quality variables. In Rørslett's Scandinavian study latitude did not influence species richness markedly, but compared with the American study of Crow (1993) the latitudinal range is relatively small. Low diversity was particularly observed in lakes experiencing a high intensity of environmental disturbance or stress, associated with lake regulation, acidification, or hypertrophication, the latter agreeing with Lachavanne's findings from Switzerland. In this

Scandinavian dataset, as in Swiss lakes, elevated macrophyte biodiversity was consistently associated with intermediate intensities of stress and disturbance. In general, meso-eutrophic lakes have the richest macrophyte flora.

Similar evidence has been found to support the hump-back hypothesis from many other habitats (e.g. agricultural land in Scotland: Wilson *et al.,* 2003; Canadian rangelands: Willoughby, 1992; desert vegetation in Egypt: Ali *et al.,* 2000; marshes: Day *et al.,* 1988). However, not all habitats cover a sufficiently broad range of conditions to show this type of relationship. Under such restricted circumstances of variation in stress or disturbance more linear relationships may become apparent between diversity status and environmental factors. Minimal linear models are appropriate to determine such relationships.

The 'humpback model' relationship between plant diversity and nutrient status can also be related to the age of the communities concerned. Low nutrient communities to the left of maximum diversity are typically, semi-natural ancient grassland types that have co-evolved over many generations, whereas the high nutrient, declining diversity communities to the right of the maximum are typically recent agricultural productive grasslands. This may have important consequences for understanding the selectionist versus complementarity debate (see above). Complementarity may be expected to have evolved in the low-nutrient semi-natural grasslands, whereas selectionist mechanisms are more likely to apply in recently created intensive grassland. However, Warren *et al.,* (2002) found no evidence for complementarity even in the fertile grassland communities to the right of the 'humpback' relationship. Even so it remains a reasonable assumption that diversity matters more to ecosystem functioning in ancient rather than recent communities.

Here again we have raised the issue of naturalness and the related concept of a pristine habitat. Ecologists routinely describe communities as natural, semi-natural, or man made. Restoration ecologists have recently added another term to this list – the facsimile community (a newly created habitat designed to mimic a desired target community of high conservation value). Few (if any) ecologists would not argue that natural habitats should be ascribed greater conservation value than semi-natural habitats, which in turn are of more value than man-made habitats and facsimile habitats. It becomes more complicated still when you try and compare good and bad examples of ancient woodlands with good and bad examples of semi-natural wild-flower meadows.

2.2.8.3. Modelling biodiversity change on agricultural land

In response to increasing reports of loss of biodiversity on farmed landscapes in the UK and elsewhere in the world (e.g. Paoletti *et al.,* 1992; Donald, 1998; Robinson & Sutherland, 2002; Andreasen *et al.,* 1996; Brickle *et al.,* 2000; Holland & Luff, 2000; Kleijn *et al.,* 2001), research has been undertaken in recent years to establish methodologies for predicting (at levels from qualitative to fully quantitative) change in biodiversity in response to changing agricultural practice (e.g. Díaz & Tellería, 1994; Parish *et al.,* 1994; Simpson *et al.,* 1996; Paoletti, 1999).

One such study at the University of Glasgow has developed a suite of models using land-use and agricultural management variables, plus functional attributes of the vegetation, as predictors of biodiversity response (for plants, selected invertebrate groups and birds) to changing agricultural land-use in Scotland (Abernethy *et al.,* 1996; Foster *et al.,* 1997; Downie *et al.,* 1998; Downie *et al.,* 1999; Downie *et al.,* 2000; Wilson *et al.,* 2003). The study took place within a land-use envelope ranging from upland sheep-grazed grasslands to lowland intensive arable systems. A multiple regression approach was used to model change in S resulting from land-use and agricultural management changes, which had shown substantial change in Scotland, in the years during which this study was undertaken. The model outcomes are valid, for predictive purposes, within the envelope of overall applicability of the models. However, the range of agro-geo-climatic conditions prevailing at the input data sites makes the models applicable over most of Scottish agricultural land. There is a reasonable probability that they would also be applicable to other areas of the British Isles experiencing comparable conditions.

Such knowledge based modelling approaches are generally limited to predicting the outcome of existing land-use practices. To address this limitation Warren & Topping (1999) developed a mechanistic vegetation model, which predicts changes to community composition based on simulation of the interaction of species in a three-dimensional arena in which a vast range of management prescriptions can be applied. Such modelling approaches can be used to predict rates of community change. This enables questions to be asked about how various degrees of community change over different time-scales are perceived and valued by conservationists and the public.

Models such as those of Wilson *et al.,* (2003), or Marshall *et al.,* (1996) – examining the contribution of field margins to plant diversity in agricultural landscapes - can be used to assess the likely impacts of such changes, in order to optimise land management strategies which would encourage the maintenance or enhancement of existing plant diversity. Once baseline values of S are established for the type of ecosystem under consideration, change in S can provide a measure of ecosystem response to altered land-use, for example resulting from policy decisions affecting management of the agricultural landscape (Fry, 1991). Where "like with like" comparisons of change in S can be made, in this way, then potentially useful trends and changes in biodiversity can be identified which are of practical use for policy, conservation, or management purposes.

Changes in biodiversity have been dramatic in Britain post-1945 (e.g. Robinson and Sutherland, 2002; Chamberlain & Fuller, 2001; Chancellor, 1985), and may be even more so in the near future, given the current climate of sea-change policy shifts in management of the UK agricultural landscape (Bignal *et al.,* 2001). In particular, loss of diversity, as discussed above, often signals ecosystem degradation (Prance, 1991; Murphy, 2003; Robinson & Sutherland, 2002).

2.3. The UK biodiversity resource

Finally, it is useful to outline the UK biological resource and also summarise the conservation policies that have been adopted to protect and enhance this resource.

Thanks to the Ordnance Survey and a long tradition of amateur naturalists, biodiversity within the UK is probably better recorded and audited than anywhere else on the planet; even so, we do not have distribution maps for many groups of organisms. What this information illustrates is that the diversity of flora and fauna in the British Isles is low. The reason for this limited diversity is threefold. First, although the rocks that make up the British Isles may be ancient, the species that inhabit the UK must have colonised it during the last 10,000 to 15,000 years following the last ice age. Many species simply failed to re-colonise the UK before the land bridge to continental Europe was broken. It is this fact (and not St Patrick) that explains the absence of snakes and other reptiles from Ireland. Second (and confounded with the first point), the proximity of the UK to continental Europe and its effectively young age have not allowed sufficient separation (in space or time) for the evolution of endemic British species to have occurred. This effect can be seen dramatically when comparing the UK's diversity with that of New Zealand (Table 1) which is a similar sized but more ancient and isolated island. The third factor responsible for limiting the diversity occurring in Britain is known as Rapoport's rule, which states that diversity increases towards the equator. The reason for this being that equatorial habitats receive more solar energy and hence, other things being equal, are capable of greater plant biomass production, which in turn is able to support more species.

In addition, temperate species are more ecologically adaptable to cope with the seasonal variability and as a consequence tend to have greater geographic ranges.

Table 1: IUCN red-data listed terrestrial species

List	United Kingdom	New Zealand
Extinct	0	20
Critically Endangered	4	10
Endangered	2	23
Vulnerable	17	67

Biodiversity conservation in the UK emerged from its tradition of amateur naturalists, non-government organisations, and interested landowners. Thus, in the first half of the 20th century conservation mechanisms were developed to protect specific species and important sites, with legislation deigned to prohibit damage. Following the 1992 Rio Convention, the UK government launched its UK Biodiversity Action Plan in 1994. This report identified and recommended activities for conservation over the next twenty years, recommended that a steering group be set up, and established principles for future biodiversity conservation in the UK. These principles emphasised the need for partnerships to be set up at all levels, targets to create measurable outcomes that addressed the needs of species and habitats, policy integration and public awareness.

The current approach has therefore developed into one in which biodiversity is no longer thought of as being restricted to particular isolated sites and rare species. Following Rio, an integrated view has emerged in which agri-environment schemes, earlier designations such as SSSIs, NNRs, SACs, are married together and focused through the BAP process to meet identified auditable biodiversity targets across the UK. Thus, urban area, recreational areas, and the wider farmed landscape are now all recognised as being important in supporting the UK's biodiversity resource.

2.4. Is biodiversity a useful measure for environmental valuation purposes?

The above review has highlighted a number of important issues with respect to the way that ecologists define measure and predict biodiversity. We now discuss these measures in the context of this research project and in particular discuss (1) the extent to which members of the general public understand these ecological concepts and (2) how useful the measures are for the quantification of the economic value of biodiversity.

It is first useful to re-emphasise the fact that there appears to be no one definitive measure of biological diversity. Thus, there is no simple way to present change in biodiversity to members of the public. Ecologists are, however, in general agreement that the number of species per unit of area provides a useful starting point. Although such a measure appears to be relatively straightforward, issues such as what constitutes a species and what size of area to use complicate this measure. Even if these questions were resolved, ecologists also recognise that some species, such as keystone species, may be more important and/or make a greater contribution to biodiversity than others. It is clear that ecologists find it difficult to agree amongst themselves the best way in which to measure biodiversity. The knock-on effect of this is that the task of describing biodiversity change to members of public will be challenging. A further complicating factor relates to the extent to which the public are capable of understanding ecologists' concepts. Existing research suggests that the public only have limited (if any) understanding of the term biodiversity (Hanley *et al.,* 2002). Clearly, ways in which biodiversity can be meaningfully described to members of the public will be a key issue to the success of this research project. It is also expected that members of the public will be greatly influenced by anthropocentric value such as 'cute and cuddly -ness' and charismatic species, which have no little or no ecological status. In order to address these issues, detailed qualitative research is undertaken to assess the extent of public understanding of the ecological measures of biodiversity, to identify which biodiversity concepts are considered to be important by members of the public, and then to establish appropriate language in which these measures can be meaningfully relayed to the public (See Section 4).

The use of biodiversity models (see Section 2.2.8.2 and 2.2.8.3) allows "what-if" scenarios of likely change in biodiversity to be established for possible future changes in land management, whether driven by policy change or other factors. Outputs from such scenarios can then be utilised in environmental valuation procedures: a principle aim of this study is to determine the utility of biodiversity as a measure for environmental valuation. In particular, such models may be used to identify the bounds of biodiversity change to be presented in the scenarios.

3. An Economists Perception of Biodiversity

In the introduction to this report, we presented arguments as to why economic valuation of biodiversity was important. We now further explore the various methodologies that economists have developed to quantify the economic value of the environmental and natural resources. In particular, we examine the types of value associated with biodiversity, explore the various valuation methodologies available for the valuation of biodiversity, and review current studies that have attempted to measure the economic value of biodiversity. A discussion of the potential of benefits transfer, an approach that adjusts values from existing studies to imply values in a new context, concludes this section.

3.1. Concepts of economic value of biodiversity

In welfare economics (and therefore in cost-benefit analysis), economic values are defined over some change in any factors that impact on utility, either directly, or indirectly through effects on production. When we speak of the "economic value" of a particular environmental resource, such as a forest, we implicitly refer to some change in its condition, for example whether the forest is felled or not. Likewise, any economic value for biodiversity should be defined with regard to a specific change in provision, such as a rise or fall in S, or some qualitative change in a given S such as a change in equitability. Also, as noted above, since economic values relate to effects on utility, the economic value of two equivalent S values may differ if people place higher subjective values on some species within S^1 compared to S^2 (for example, if S^1 contains more "cute" species than S^2). This will be so even if an ecologist regarded S^1 and S^2 as equally diverse.

3.1.1. Direct use values of biodiversity

What types of economic values could biodiversity have? The first class of impacts are *direct* impacts. In economic terms, the value of direct impacts may be considered as follows:

$$U = U(\mathbf{S}, \mathbf{Z}) \qquad (1)$$

where utility U depends directly on a vector of biodiversity measures **S**, and a vector **Z** of all other goods and services that people derive utility from[1]. The effect of any element S on U can occur in two ways:

- as a "use value": this means that people make direct use of biodiversity, for example by going bird-watching. The greater the number of species they see, the happier they are, and the higher is their utility. Utility may also increase if an element of **S** contains more species that people are particularly fond of or culturally attached to (ospreys might be a good example in Scotland, golden eagles might be another). Use may be non-consumptive as in bird watching, or consumptive as in fishing.
- as a "passive-use" value. People may simply care about species richness for a number of reasons; they feel species richness is important to ecosystem health; they have a care for "naturalness". Passive-use values can be split into three basic components, although these may overlap depending upon exact definitions. "Bequest values" relate to values gained from the knowledge that a species will be preserved for future generations. "Altruistic values" refer to the utility gained from preserving a species for the enjoyment of others today. Finally, "existence values" refer to an individual's willingness to pay to preserve the existence of a resource even though that individual has no actual or planned use of the resource for him/herself or anyone else. Extensive research in environmental valuation has found such passive-use values to be

[1] From hereon, any character in bold denotes a vector

statistically significant, and to be especially important where unique/rare environmental resources are concerned (Carson, *et al.*, 1999).

Where future demand for or supply of biodiversity is uncertain, then these use and passive-use values can incorporate a risk-premium (Ready, 1995); under certain conditions this will translate into an additional aspect of value, which has sometimes been referred to as an "option value".

3.1.2. Indirect use values of biodiversity

The second type of utility impacts we should consider are *indirect effects*. These occur when biodiversity is important in the functioning of those natural systems which people exploit in order to produce goods and services. Indirect use values are therefore derived from services provided by ecosystem functions. The most obvious examples are in terms of agriculture and fishing. In the literature, other terms including "contributory values", "primary values" and "infrastructure values" have been used to provide more precise definition of the components of indirect value (Norton, 1986). Suppose we partition the **Z** vector of equation (1) into two components; Z_b, those products whose production is influenced by the level of biodiversity and Z_n, those products not so influenced:

$$U = U\,(\mathbf{S};\ \mathbf{Z_b}\ (\mathbf{S}),\ \mathbf{Z_n}\,) \qquad (2)$$

Where we show the production of Z_b as implicitly dependent on the level of elements of **S**. This might be, for example, if the long-term resilience of agricultural systems is related to diversity, or if commercial fish stocks are also partly dependent for their long-term health on diversity. Then as long as $\delta Z_b/\delta S > 0$, and so long as $\delta U/\delta Z_b > 0$, then biodiversity has an indirect economic value. These indirect values may apply to whole natural systems, if diversity is crucial to their long-term survival. They also apply to pharmaceutical uses of diversity, since here Z would be medicines.

The "total economic value" (TEV) of a given level of S is then the sum of direct (use, passive-use, plus option) values, plus indirect values:

Total Economic Value = Use + Passive-use + Option + Indirect values

In this research, we are interested in establishing the TEV associated with a change in biodiversity. The various economic valuation methods used to determine TEV are reported in Section 3.2 below. It is, however, noted that different valuation approaches will have different abilities to value the different elements of TEV and that it is unlikely that one single approach will be capable of valuating all elements of TEV.

3.1.3. Other biodiversity value considerations

When considering the economic value, that is, the direct and indirect value of biodiversity, we also need to consider various other issues, including:

3.1.3.1. Biodiversity vs. biological resources.

Biodiversity (or biological diversity) refers to the variety of life, whereas biological resources refer to the manifestation of that variety (Nunes and van den Bergh, 2001). The distinction between the two is often confused and Pearce (1999) argues that although there are many studies that claim to value biodiversity, most are actually determining the value of a biological resource.

3.1.3.2. Holistic vs. reductionist approaches.

The holistic approach suggests that since biodiversity is an abstract notion it will be difficult to separate and measure due to the complexity and nature of the concept (Faber *et al.,* 1996). The reductionist perspective maintains that the total value of biodiversity can be divided into different value categories, particularly into direct and passive-use values (Pearce and Moran, 1994).

3.1.3.3. Local vs. global.

Biodiversity valuation studies are used at a local, regional, or national level for policy formulation, and biodiversity loss is generally defined in a global or worldwide context. Hammond *et al.* (1995) argue that biodiversity and biodiversity loss are relevant at multiple spatial levels, from local to global. The benefit could be at a local, regional, or global scale, depending on the aspect of biodiversity being valued, i.e. a single species, multiple species, an ecosystem, or ecosystem functions.

3.1.3.4. Expert vs. general public assessment.

There are various views as to who are the best individuals to participate in the valuation of biodiversity changes. Although the majority of valuation methods sample members of the public from all levels of education and life experiences, others rely on expert judgement, particularly from biologists and ecologists, as they are more appropriate to deal with the complexity of the subject (Nunes and van den Bergh, 2001). The inclusion of the general public in valuation exercises is important because consumer sovereignty is fundamental to cost benefit analysis. A compromising solution would be to let experts inform laypersons before carrying out the valuation exercise (Arrow *et al.*, 1993). The way in which information is presented to members of the public is critical for the valuation exercise since the information presented can (potentially) bias the resultant valuations. In valuation studies, information bias may be reduced through careful survey design and testing for information effects in developmental focus groups. Biodiversity is a complex concept and therefore it is essential that the impacts of information are thoroughly investigated both in the development of the survey instrument and to validate the valuation results.

3.2. Methods of estimating the economic value of biodiversity

Again, assuming for simplicity that biodiversity can be measured by S, how can we actually estimate the economic value of changes in S? This depends on whether we are concerned with direct or indirect impacts on utility. Utility itself is not measurable/quantifiable, thus we work with *monetary equivalents* of underlying utility changes. In standard economic theory, these monetary equivalents are either the maximum Willingness to Pay (WTP) of people for a welfare-increasing change, or their WTP to prevent a welfare-decreasing change; or the minimum compensation they would accept to forego a welfare-increasing change or tolerate a welfare-decreasing change, their minimum Willingness to Accept Compensation (WTAC). These two amounts may be different for a given change in S. Taking WTP for the moment, this is defined as:

$$V\,(\mathbf{S^1};\,\mathbf{Z_b}\,(\mathbf{S^1}),\,\mathbf{Z_n}\,;\,Y\text{-}WTP) = V\,(\mathbf{S^0}\,;\,\mathbf{Z_b}\,(\mathbf{S^0}),\,\mathbf{Z_n};\,Y) \qquad (3)$$

where $\mathbf{S^1 > S^0}$, Y is income, and V(.) is now indirect utility.

3.2.1. *Direct effects on utility*

In the case of S having a direct effect on utility, we wish to know by how much the monetary equivalent of U will change when **S** changes. There are two basic approaches to answering this question: through revealed preference, and through stated preferences.

3.2.1.1. *Revealed preference valuation techniques*

Revealed preference approaches rely on actual behaviour, and on finding some market-valued behaviour which is related to the level of S. The most obvious way in which this works is in the context of travel cost models for recreation. One way in which higher biodiversity increases utility, as mentioned above, is through use values. Bird watchers, fishermen, hunters and hill-walkers may all enjoy their recreational experiences more in ecosystems which are more diverse: more birds to see, more beautiful flower meadows to walk through, more types of fish to try and catch. For such users, travel to the recreational site is an essential input to the recreational activity, and this travel is costly both in terms of out-of-pocket travel costs and time costs. Travel costs models work by relating demand, that is the number of trips, to these travel costs. Under certain specifications, site characteristics could also be included, and one of these characteristics could be the level of diversity at a given site, S_j. For a group of recreational sites where a particular activity takes place, one could thus estimate, using data based on actual behaviour:

$$T_{ij} = f(C_{ij}, N_i, S_j, K_j) \qquad (4)$$

where T_{ij} are recreational trips (e.g. fishing trips) by individual i to sites j (j=1…J), C_{ij} are travel costs to individual i for visiting site j, N_i are socio-economic characteristics for individual i, Sj is biodiversity at site j, and K_j are other characteristics of the J sites. Using this type of equation, it is possible to work out the per-trip value of a site under current conditions, and how this would change when S changes. This seems attractive: however, since our measure of value relates now to actual use of the site, we can say nothing about passive-use values for S, or indeed about any option values. This is a big drawback in the current context. For a full discussion of the use of travel cost models, see Herriges and Kling (1999).

Another revealed preference method that has been used to infer a value of biodiversity, and ecosystem functions in particular, is averting behaviour. The principle underlying averting behaviour is that the costs incurred by individuals or firms to reduce or avoid the consequences of environmental damage infer the value of that environmental resource. For example, Ribaudo (1989) estimated the value of improvements to water quality based on the costs associated with reducing the discharge of pollutants in US waterways. Often, however, the costs of the action only represent a part of the environmental costs and therefore estimates based on averting behaviour should be regarded as a lower bound estimate of the value of that resource.

3.2.1.2. *Stated preference valuation techniques*

The second approach to estimating direct utility values is to use *stated preferences*. These are based on surveys of the population of people thought, *a priori*, to care about the change in S being evaluated. For example, suppose a project is being considered which will increase the number of bird species on farmland in Sussex. We might assume three groups of people would care about this: those living in Sussex, those living elsewhere who visit Sussex farmland on walks etc, and those living elsewhere who have a passive-use value for farmland birds. The most common stated preference technique is contingent valuation (CV). CV works through surveys which ask people, in a carefully structured way, how much at most they would be WTP to have the change in environmental quality go ahead, assuming it to be beneficial to them; or their WTAC to forgo it. Concentrating on the former option for present purposes, this would thus involve surveying a random sample of local residents, visitors and those living in other parts of England, informing them about the project in terms of its impacts on biodiversity, and why such projects are costly. Each respondent would then be required to state their maximum WTP to have the project go ahead (i.e., to benefit from an increase in diversity). For some people, this amount would be zero, if they do not care about increasing farmland bird diversity. A sample mean WTP would be calculated, from which a population mean is inferred.

Contingent valuation methods have become increasingly sophisticated, and a battery of validity tests developed. For a full discussion of the method, see Bateman and Willis (1999). The great advantages of CV are that (i) it is capable of estimating use, passive-use, and option values; and (ii) that it can be applied to a wide range of issues. The main worry about CV continues to be the divergence between stated values (how much you say you would pay) and actual values (how much would you pay if you really had to). Research suggests that the difference between these two magnitudes can be significant, but depends on both the nature of the good and the design of the CV (see the summary table in Hanley, *et al*., 2001a). Some researchers have put forward "calibration factors" which try and adjust stated WTP amounts to take account of hypothetical error, but these have been found to be context specific (Fox *et al.,* 1998). What is certain, however, is that CV has become very widely used in public decision-making, in the UK and elsewhere (see Bateman *et al.,* 2002; and Hanley, 2001). CV has also been widely applied to wildlife conservation, landscape and habitats.

An alternative to CV within the stated preference paradigm is Choice Experiments (CE). Choice experiments takes a somewhat different approach. Environmental resources are described in terms of their attributes. For instance, rivers in Hampshire could be described in terms of fish species present, low flow risks, condition of banksides, and visible pollution. A cost attribute is also included; in this case, perhaps the cost to households in terms of higher water rates of cleaning up rivers in Hampshire. Experimental design theory is then used to construct alternative scenarios of combinations of attributes and costs. People in populations expected to care about the policy change are then sampled (just as in CV), and asked to make choices amongst these alternative scenarios. These choices can be in terms of rating, ranking or strict preference. The analyst can then infer the maximum WTP of the sample for a change in any of the attributes, so long as they turn out to have been statistically-significant determinants of choice. So, for example, the analyst in this example could calculate the WTP of the sample for an increase in the number of fish species. Like CV, choice experiments approaches can measure use, passive-use and option values. They are also very versatile in what they can be applied to. Some use of CE has been made in valuing biodiversity in terms of landscape features and species diversity, in the UK as well as in other countries. For a full description of CE techniques, see Louviere *et al.* (2000) and Hanley *et al*. (2001a).

Most stated preference applications are administered using either postal questionnaires or in-person interviews. However, in recent years a number of applications have utilised more sophisticated methods of data collection including valuation workshops, market stalls and citizen juries. These approaches generally incorporate a period of time in which participants are able to gather and assimilate information from a wide range of sources and then discuss this information within a group context. Such features make these approaches particularly useful for the valuation of complex goods; such as biodiversity. A detailed review of these approaches can be found in MacMillan and Hanley (2002).

3.2.2. *Indirect impacts on utility*

Where biodiversity has indirect impacts on utility through its relationship with production, then matters are in one sense more straightforward. This is because produced goods and services have market prices and market demands which can be used to value changes in their supply. However, a key problem which remains is to uncover the quantitative link between the level of diversity, **S**, and production. For instance, suppose we suspect that higher species diversity has a positive relationship with commercial fisheries output, F. In other words, we suspect that the production function for commercial fisheries is:

$$F = F(E, B, \mathbf{S}) \qquad (5)$$

where E is an index of "effort" applied in the fishery (a combination of labour and capital), B is the biomass of the fish stock being exploited, and **S** as usual is biodiversity in the environment, in this case a marine ecosystem. What we require to estimate the indirect value of an increase in **S** is to know the quantitative relationship between **S** and F, namely $\delta F/\delta \mathbf{S}$. However, in practice this may be very hard to estimate, and may well involve stochastic terms and be time-dependent. The value of biodiversity through "bio-prospecting" is a special case of equation (5), where F(.) now measures the output of useful drugs. The other inputs here would be scientific effort in discovering, testing and developing a useful natural substance, and capital and variable costs in producing the drug once approved. We note some examples of this approach in the literature review below.

There are also a number of cost-based methods available including replacement costs, restoration costs, relocation costs, and preventative expenditures approaches. Essentially, these approaches infer a value of a natural resource by how much it costs to replace or restore it after it has been damaged. For example, Andreasson-Gren (1991) estimated the value associated with a wetland's nitrogen purification capacity by comparing it to the costs of using conventional nitrogen abatement technologies. Although these methods have been used to infer values of (predominantly) ecosystem functions, it should be stressed that these methods are based on costs and therefore do not strictly measure utility.

3.3. Review of studies that aim to value of biodiversity.

Above, we have discussed how economic valuation techniques may be used to measure biodiversity. In this next section we review a number of key studies that have attempted to measure the economic value of different elements of biodiversity. In particular, we distinguish between studies that have valued of biological resources (e.g. a particular species, habitat area, or ecosystem function) and those which have valued the biological diversity of those resources (e.g. components of biodiversity such as rarity or charismatic species). In the appendix to this report, we also summarise some of the key UK studies that have attempted to value the economic benefits of biodiversity.

3.3.1. *Review of studies that value the biological resource.*

'Biological resource' is a term used to describe the manifestation of the variety of life. In other words, a biological resource is a given example of a gene, species, habitat, or ecosystem. Biological resources are often easier to identify than biological diversity. As such, the majority of valuation studies that claim to value biodiversity have, in reality, valued a biological resource. In the following section, we review existing valuation studies according to three distinct categories of biological resources: genetic and species diversity, ecosystem and natural habitat biodiversity and ecosystem functions. We refer the reader to Nunes and van den Bergh (2001) for a more detailed review of these studies. A summary of the range of value estimates for each of these three categories of biological resources can be found in Table 2 below.

3.3.1.1. The value of genetic and species diversity.

This category of biological resources include both genetic and species diversity. Studies that have quantified genetic diversity have predominantly measured direct use benefits of biological resources in terms of inputs to the production of market goods such as new pharmaceutical and agricultural products. The majority of studies have based valuations on market contracts and agreements for bioprospecting by pharmaceutical industries. For example, Merck and Co. (the world's largest pharmaceutical firm) paid $1 million to the Instituto National de Biodiversidad to exploit Costa Rica's biological resources. Ten Kate and Laird (1999) provide an extensive review of such bioprospecting agreements. There is, however, only limited information available about the UK genetic resource. Franks (1999) provides a useful contribution on the value of plant genetic resources for food and agriculture in the UK and also the contribution of the UK's agri-environmental schemes to the conservation of these genetic resources.

There have been a large number of studies that have valued species diversity. Most of these studies have been undertaken in the US and utilise stated preference techniques (either contingent valuation or choice experiments), thus enabling both use and passive-use values to be assessed. Included in the passive-use values are existence values; that is, the value of the knowledge that a species exists. Nunes and van den Bergh (2001) provide an extensive review of valuation studies that have addressed both single and multiple species. Valuations for single species range from $5 to $126, and for multiple species range form $18 to $194 (Table 2). Nunes and van den Bergh note that care needs to be undertaken when interpreting the results from the valuation of single species since they often do not take account of substitute species. In the UK, there have been a limited number of studies that have valued both single and multiple species. For example, Macmillan *et al.* (2001b) estimated the value of wild geese conservation in Scotland, while White *et al.* (1997 and 2001) examine the value associated with the conservation of UK mammals including otters, water voles, red squirrels, and brown hare. Macmillan *et al.* (2001a) also takes a slightly different perspective by valuing the reintroduction of two species (the beaver and wolf) into native forests in Scotland. All these UK studies used stated preference methods and summaries are provided in the Appendix to this report.

3.3.1.2. The value of natural habitats

Biological resources may also be described in terms of the diversity within natural habitats. Studies have addressed the valuation of habitats from two perspectives. One approach is to link the value of biodiversity to the value of protecting natural areas that have high levels of outdoor recreation or tourist demand. Such studies have adopted various revealed preference methodologies including the travel cost method and tourism revenues. Since these studies are based on actual use of the natural resource, benefit values estimated are restricted to the measurement of use values only. Nunes and van den Bergh (2001) again provide an extensive review of these studies. UK examples include Klien and Bateman (1998) who used travel cost method to value Cley Marshes nature reserve and Willis (1990) who estimated the benefits associated with three SSSIs.

A second approach to the valuation of natural areas involves the use of stated preference methods. These techniques have the advantage over revealed preference techniques since they enable both use and passive-use values associated with habitat conservation to be elicited. Again, Nunes and van den Bergh (2001) provide an extensive review of these studies. Table 2 summarises the range of passive-use values elicited for terrestrial, coastal and wetland habitats. UK examples of CV studies that have valued habitats include: Garrod and Willis,

(1994) who examined the willingness to pay of members of the Northumberland Wildlife Trust for a range of UK habitat types; Bateman *et al.* (1993) who value wetland areas in the South Downs; Garrod and Willis, (1995) who valued chalk grasslands in the Norfolk Boards; Hanley and Craig (1991) who valued upland heaths in Scotland's flow country; and Macmillan and Duff (1998) who examine the publics WTP to restore native pinewood forests in Scotland. Again, summaries of the UK studies are provided in the Appendix to this report.

3.3.1.3. The value of ecosystem functions and services.

Ecosystem functions and services describe a wide range of life support systems including waste assimilation, flood control, soil and wind erosion, and water quality. Many of these functions and services are complex and it is likely that members of the public will possess a poor understanding of these issues. The consequence of this is that attempts to value ecosystem functions and services will be difficult, particular in methods (such as the stated preference methods) where respondents are required to make a value judgement based on the description of the good in question. Analysts often use other techniques including averting behaviour, replacement costs, and production functions to measure the indirect values of ecosystem functions. Again, Nunes and van den Bergh (2001) provide an extensive review of such studies. At the time of writing, we were no aware of any UK study that has attempted to value ecosystem function and services.

Table 2: Value ranges for biological resources

Life diversity level	Biodiversity value type	Value ranges	Method(s) selected
Genetic and species diversity	Bioprospecting	From \$175 000 to \$3.2 million	Market contracts
	Single species	From \$5 to 126	Contingent valuation
	Multiple species	From \$18 to 194	Contingent valuation
Ecosystems and natural habitat diversity	Terrestrial habitat (passive-use)	From \$27 to 101	Contingent valuation
	Coastal habitat (passive-use)	From \$9 to 51	Contingent valuation
	Wetland habitat (passive-use)	From \$8 to 96	Contingent valuation
	Natural areas habitat (recreation)	From \$23 per trip to 23 million per year)	Travel cost, tourism revenues
Ecosystems and functional diversity	Wetland life-support	From \$0.4 to 1.2 million	Replacement costs
	Soil and wind erosion protection	Up to \$454 million per year	Replacement costs, hedonic price, production function
	Water quality	From \$35 to 661 million per year	Replacement costs, averting expenditure

Source: Nunes and van den Bergh (2001)

3.3.2. Review of studies that value biological diversity

Pearce (2001) contends that despite the growing volume of environmental economic valuation literature, we still have little idea of the value of biodiversity in terms of biological diversity. Perhaps one of the main reasons why biological diversity has not been studied relates to the fact that biological diversity is a difficult concept to convey to the general public and therefore it is difficult to design a valuation study that addresses biological diversity. In this review, we explore two approaches to the valuation of biological diversity. The first relates to studies that have assessed willingness to pay for policies that aim to promote a general increase in biodiversity, while a second group of studies examine the valuation of specific components that describe biodiversity.

3.3.2.1. Valuation of general increases in biodiversity

A number of valuation studies have attempted to value biodiversity by explicitly stating to respondents that the implementation of a conservation policy will result in an increase in the biodiversity of an area. For example, Garrod and Willis (1997) estimated passive-use values for biodiversity improvements in remote upland coniferous forests in the UK. The improvements in forest biodiversity were described in relationship to a series of forest management standards that increased the proportion of broad-leaved trees planted and the area of open spaces in the forest. The marginal value of increasing biodiversity in these forests was estimated to range between £0.30 to £0.35 per household per year for a 1% increase and between £10 to £11 per household per year for a 30% increase in biodiversity-friendly forest area. Hanley *et al*. (2002) extend this work to examine public values for biodiversity across a range of woodland types. They conclude that '*non-use biodiversity values are particularly difficult to capture*'. In particular, they highlight that problems arise from (1) the fact that people have widely different preferences for wildlife thus the variance of the mean WTP is large, (2) people's WTP for biodiversity in British woodlands is a very small fraction of income, (3) biodiversity is a difficult concept for people to grasp.

Other studies have assessed public WTP to prevent a decline in biodiversity. For example, Macmillan (1996) measures public WTP to prevent biodiversity loss associated with acid rain.

3.3.2.2. Valuation of components of biodiversity

A number of studies have also attempted to elicit WTP for specific components of biodiversity. Macmillan *et al*. (2001b) used both contingent valuation and choice experiments to value the conservation of wild geese in Scotland. Of particular interest in this study was the finding that respondents were WTP between £2.83 to £16.50 extra for conservation policies that specifically targeted endangered species of geese. Samples *et al*. (1986) also found that information on endangered species increased WTP.

In a contingent valuation study of public WTP for four UK mammals, White *et al*. (1997 and 2001) examine the influence of species characteristics on WTP. They conclude that charismatic and flagship species such as the otter attracted significantly higher WTP values than less charismatic species such as the brown hare. They further suggest that species with a high charisma status are likely to command higher WTP values than less charismatic species that may be under a relatively greater threat or of more biological significance in the ecosystem. They conclude by stating that '*attaching too much emphasis to willingness-to-pay studies in nature conservation policy would therefore be at the expense of the less charismatic species and would probably lead to the inappropriate allocation of resources*'. In a meta-analysis of WTP for a range of species, Loomis and White (1996) also find that more charismatic species, such as marine mammals and birds, attract higher WTP values than other species.

The above review has demonstrated that from those studies that have claimed to value biodiversity, only a handful have actually examined biological diversity; most studies have alternatively tended to value biological resources. Furthermore, studies that have valued biological diversity currently only provided limited information on the value of the components of biological diversity. Research effort has yet to provide a comprehensive assessment of the value attached to the component of biological diversity such as anthropocentric measures (e.g. cuteness, charisma, and rarity) and ecological measures (e.g. keystone species and flagship species).

3.4. Benefits transfer and biodiversity.

Valuation studies are expensive and time-consuming. For this reason, the policy community has become increasingly interested in *benefits transfer* techniques. Benefits transfer (BT) is a method for taking value estimates from original studies, and adjusting them for use in some new context. The two main approaches to BT are:

- the transfer of adjusted mean values. Mean WTP estimates taken from the original study or studies are adjusted to account for differences in the environmental characteristics of the new site/context, and/or for differences in the socio-economic characteristics of the affected population at the new site.
- the transfer of benefit functions. Benefit functions are regression equations which explain variations in WTP across individuals according to variations in socio-economic factors and, in some cases, environmental characteristics. A benefits function can be used to produce estimates of WTP.

In both cases, meta-analysis (that is, the quantitative analysis of a collection of past studies) can be used to inform the BT process.

Much academic work has taken place in the past 10 years testing alternative BT methods and assessing their accuracy. The academic jury is still "out" on the validity of BT, even though BT is crucial for the wider use of environmental valuation in policy analysis. Studies by Bergland *et al.,* (1995) and Barton (2000) largely reject the validity of benefits transfer, both in terms of the transfer of adjusted mean values and the transfer of benefit functions. Brouwer (2000) surveys seven recent benefits transfer studies and finds that the average transfer error is around 20-40% for means and as high as 225% for benefit function transfers. Ready *et al.,* (2001) also found a transfer error of around 40% for a multi-country study on the health benefits of reduced air pollution. Shrestha and Loomis (2001) find an average transfer error of 28% in a meta-analysis model of 131 US recreation studies. As Barton points out, though, even fairly small transfer errors (11-26% in his case) can be rejected using the statistical tests favoured by economists. However, this has not stopped the development of large BT software packages, such as the North American-based EVRI package, developed by Environment Canada.

In the UK, the recently-developed Environmental Landscape Features (ELF) model finds some evidence in favour of BT in the context of landscape features on farmland (Oglethorpe *et al.,* 2000). ELF is a computerized transfer system based on the transfer of benefit functions from UK landscape valuation studies. It predicts per-hectare WTP values for a range of landscape features. The benefit functions in the model predict WTP per feature based on variations in the socio-economic characteristics of the population, such as household incomes, and for regional scarcity.

One debate on-going at present is whether more complex BT approaches necessarily do better than simple ones. Barton (op cit) finds a simple adjusted means transfer gets closer to original site values than the transfer of benefit functions. The opposite finding, however, is reported in Desvouges *et al.* (1998).

Finding acceptable benefits transfer methods is essential to the wider use of environmental valuation in policy. However, the standards of accuracy required in academic work may exceed those viewed as tolerable by policy-makers. Santos (1999) found that a meta-analysis based benefits transfer model for contingent valuation estimates of landscape features could get greater than 30% accuracy in 26% of cases; and greater than 50% accuracy in 44% of cases. This led Santos to question the added value of additional original studies. The key question is: how close is close enough for policy purposes? Finally, we are not aware of any BT system that has been developed for biodiversity *per se*, as distinct from use values for recreational resources, or landscape values for habitat types.

4. Identification of suitable approaches to value changes in biodiversity.

In the review of literature (Sections 2 and 3) we examined various ecological and economic definitions of biodiversity and reviewed alternative methodologies available to capture the total economic value of biodiversity change. It is clear from this review that the complex nature of biodiversity will make this valuation exercise extremely difficult. Furthermore, the review also highlighted that a number of studies have found that the general public have a very low knowledge and understanding of biodiversity (MORI, 1988b; ERM, 1996). This poor level of public understanding of biodiversity is likely to pose a major challenge to this study, particularly if the research is aiming to value the various ecological components of biodiversity. In order to identify suitable approaches for the valuation of biodiversity, we need to address three fundamental questions:

1. What aspects of biodiversity change are of most interest to this research?
2. How do we measure changes in biodiversity in a way that is meaningful to the public?
3. Which methodology is likely to be most suited to the valuation of this change?

We now address each of these questions in turn.

4.1. What aspects of biodiversity change are of most interest to this research

The first question that needs to be clarified in this research relates to what aspects of biodiversity change should this research attempt to value. A useful starting point to address this question can be drawn from gaps in current knowledge on the value of biodiversity identified in the review of literature. In particular, the review of existing valuation studies (section 3.3.2) highlighted that currently there is a lack of information about the value of ‘biological diversity’ (as opposed to the value of biological resources) and secondly that there is very little information on the values attached to the various ecological and anthropocentric measures of biological diversity. It is therefore proposed that the valuation of these components of biological diversity be the principal aim of this research. Members of the research steering group also identified this as an important area of research.

4.2. How do we measure biodiversity change in a way that is meaningful to the public?

First, we re-emphasise the fact that there was no *one* agreed approach to the measurement of biological change. Although ecologists are in general agreement that the number of species per unit of area (species richness) provides a useful starting point for the measurement of biological diversity, there are a number of complicating factors relating to the use of this measure, including what constitutes a species and what size of area to use. The precise definition of what constitutes a species is currently an issue of debate among ecologists and to some extent can be regarded as being pedantic in terms of the requirements of this research. There has also been much debate within ecological circles regarding the optimal size of area to measure biodiversity. It is, however, unlikely that members of the public will be aware of, or concerned about, the exact definition of these measures of biodiversity change.

Ecologists, however, also agree that species richness alone is insufficient to comprehensively measure biodiversity change. Other factors including ecological concepts (such as keystone species, and equitability etc.) and anthropocentric stimuli (such as cuteness, charismatic and rare species) also need to be considered and therefore potentially be incorporated into the valuation exercise. The identification of which of these components of biodiversity to measured in the study is likely to be one to the greatest challenges for this research; particular when you consider that ecologists themselves find it difficult to agree on the best way in which to measure biodiversity. A number of relevant issues are also of concern including (i) whether the general public are capable of understanding the ecological concepts and (ii) that the public's anthropocentric preferences (which largely have little or no ecological significance) may undermine the usefulness of a public valuation exercise (White *et al.* 1997, 2001). Intelligence gathered from exploratory discussions with members of the public indeed indicated that participants had a very low understanding of the term 'biodiversity' or of the ecological concepts that may be used to measure biodiversity. However, it was also found that these individuals were capable of understanding many of the ecological concepts if they were described in layman's terms. Thus, a further issue requiring investigation relates to what is the most appropriate language to meaningfully describe biodiversity to the public. It was also noted in the review of literature that it would be useful to distinguish between ecological concepts that were simply measures of biodiversity e.g. numbers of species and those which have some ecological and anthropocentric value, e.g. flagship species or rarity. Clearly, the public valuation of ecological concepts of biodiversity is likely to be challenging. The key to the success of this exercise will depend on whether the biodiversity concepts can be described in a way that is meaningful to members of the public. To address these issues, a conceptual framework of biodiversity was developed, and then tested on members of the public to identify which components of biodiversity the public understood and which components they thought were most important. We now report this exercise.

4.2.1. Development of a conceptual framework in which to present biodiversity.

In the review of ecological literature (Section 2), we identified over 21 different concepts that ecologists use to describe and measure biodiversity. Clearly, it would be extremely difficult to attempt to value all of these concepts. Furthermore, ecologists themselves often use different sub-sets of these measures for particular functions. In an attempt to simplify this, a conceptual framework was drawn up by members of the research team that aimed to provide a simplified and structured framework in which biodiversity could meaningfully be presented to members of the public. This framework (illustrated in Figure 1 and Figure 2 below) is split into two sections:

- Measures of biodiversity, i.e. the units that ecologists use to measure biodiversity.
- Biodiversity concepts, i.e. ecological and anthropocentric concepts of biodiversity.

Within each section of the framework, four levels (represented as rows in the two figures) are presented. The first three levels represent a hierarchal, structured framework of biodiversity concepts based on our interpretation of the ecological literature. These three levels are as follows:

- Level 1 (top row) relates to the general split between 'Measures of biodiversity' (Figure 1) and 'Biodiversity concepts' (Figure 2).
- Level 2 (row 2) relates to broad groupings of individual biodiversity concepts.
- Level 3 (row 3) includes all 21 alternative biodiversity measures and concepts identified in the ecological review of biodiversity (Section 2), which are now assigned to the Level 2 groups.

The final level of the framework (Level 4) represents our initial ideas on how the full range of biodiversity measures and concepts can best be meaningfully presented to be public. The structure of the Level 4 biodiversity groupings is based on the previous three levels, but modified to take account of comments made by participants of a series of focus groups. The discussions held in the focus groups aimed to identify the level of understanding that the public had for each of the elements of the framework and also to identify their views on the importance of each element. A variety of approaches were utilised to discuss these issues, including small and large group discussions, and written questionnaires. The proceedings of the focus group were also digitally recorded, thus enabling more detailed analysis of the discussions to be undertaken. A full report of the focus group discussions can be found in the 'Phase 1' and 'Phase 2' reports to DEFRA (Christie *et al.*, 2003a; Christie *et al.*, 2003b), however, here we simply report the key findings and recommendations from these discussions. The Level 4 framework was therefore developed to reflect the way in which the public identify with the complexities of biodiversity and thus will form the basis from which to develop the valuation survey instrument. Thus,

- Level 4 (last row) represents our initial thoughts on how biodiversity may be best presented to members of the public in a valuation exercise.

MEASURES OF BIODIVERSITY									
UNITS OF BIODIVERSITY				SCALE FACTORS					
Species richness	Individual groups of biota	Equitability	System naturalness	Genetic level	Point	Alpha	Community level	Gamma	Epsilon
It was concluded that it would not be appropriate to value the 'Measures of biodiversity' directly, but rather that these measures be used to describe the levels of provision of the biodiversity concepts (see Figure 2 below)									

Figure 1: Conceptual framework – Measures of biodiversity

BIODIVERSITY CONCEPTS										
ECOLOGICAL CONCEPTS					ANTHROPOCENTRIC CONCEPTS					
Keystone species	Umbrella species	Flagship species	Ecosystem function	Ecosystem Health	Endangered species	Rare species	Charismatic species	Cuteness	Familiar species	Locally important species
Species interactions within a habitat			*Ecosystem processes*		*Rare, unfamiliar species of wildlife*		*Familiar species of wildlife*			

Figure 2: Conceptual framework – Biodiversity concepts

We now describe our conceptual framework in more detail, and in particular outline our reasoning for the Level 4 proposal. However, before doing this we outline some general comments made by focus group participants on the general issue of their understanding of biodiversity.

One of the first issues discussed in the focus groups related to an assessment of the level of public understanding of the scientific terms and concepts associated with biodiversity. Discussions indicated that over half of the participants had never knowingly come across the term 'biodiversity' before. Furthermore, some of those who had indicated a familiarity with the term 'biodiversity' were unable to provide a clear or accurate definition of the concept. Alternative ways of describing biodiversity were discussed and the phrase '*the variety of different living organisms within a particular area or habitat*' was considered to be both useful and meaningful. Participants, however, indicated that they were familiar with some related terms including 'species', 'habitat' and 'ecosystem', but were not familiar with the majority of scientific concepts of biodiversity such as keystone species, flagship species etc. Clearly, these findings will have significant implications for this research in terms of the way in which the concepts of biodiversity are presented to members of the public. On a more positive note however, it was also found that most participants of the focus groups appeared to be capable of quickly picking up a basic understanding of most biodiversity concepts if these were explained in layman's terms. However, some participants indicated that they were often confused with regards to the precise definitions of the more closely related concepts. The conclusion from this is that research into the value of biodiversity would need to employ alternative, non-scientific terminology to meaningfully describe biodiversity.

Following the general discussions outlined above, the conceptual framework was then introduced to focus group participants. Much of the discussions of the framework centred on which aspects of biodiversity participants understood and considered to be the most important.

4.2.1.1. Measures of biodiversity

The first part of our conceptual framework (Figure 1) focuses on the various ways in which ecologists measure biodiversity. In Level 2 of our framework we identify two aspects to this.

First, we identify the range of units that ecologists often use to measure biodiversity, including species richness, individual groups of biota, equitability and system naturalness. Comments from focus group members highlighted that they experienced some difficulty in understanding the idea of equitability in particular, and to a lesser extent system naturalness. For this reason, it was considered that it would be futile to attempt to describe these units in a valuation exercise. The concepts of species richness and individual biota groups were generally understood. However, focus group participants indicated that they considered measures such as species richness as being too simplistic in itself since it did not take account of the fact that participants considered '*some species to be more important than others*'. Thus, it was concluded that rather than attempt to value the 'Units of biodiversity' *per se*, it might be more appropriate to use these units to describe the levels of the biodiversity concepts (Figure 2).

The second sub-category of biodiversity measures focused on what we term 'scale factors'. Within Level 3 of this sub-category, six alternative scale measures are identified by ecologists ranging from the genetic level to epsilon diversity. Focus group participants indicated that they were (i) unfamiliar with the ecological terminology used to describe the various scale levels and (ii) that they often found it difficult to distinguish between some of the more closely linked scale factors, e.g. point and alpha diversity scales. Furthermore, participants indicated that they would be more comfortable with an alternative scale based on the concepts of species, habitats and ecosystems than the scientific definitions of scale. Thus, it was concluded that it would be futile to attempt to directly value the alternative levels of biodiversity scale.

Thus, a key conclusion from the focus group discussions was that it would not be appropriate to base the valuation exercise directly on the various measures of biodiversity outline in Figure 1.

4.2.1.2. Biodiversity concepts

The second part of our conceptual framework (Figure 2) focuses on biodiversity concepts which were split into two broad grouping: ecological concepts and anthropocentric concepts.

The ecological concept grouping includes the concepts of keystone species, umbrella species, flagship species, ecosystem functions, and ecosystem health. Our initial proposal for the framework was to split the first three concepts into a group which we termed 'ecologically significant species' (which we later change to 'Species interactions within a habitat'), while the latter two concepts were grouped under the heading of 'ecosystem processes'.

The actual scientific terms keystone, umbrella and flagship species were poorly understood by focus group participants. However, the role which these species played in preserving biodiversity was considered to be important. In particular, the biodiversity outcomes associated with these species (namely, the protection and enhancement of species interactions within a habitat) was considered to be of more concern to focus group participants than the actual ecological concepts. Furthermore, participants found it difficult to clearly differentiate between the three ecological concepts. It was therefore proposed that rather than attempting to describe the ecological functions of these species, these concepts should be presented in terms of their output (i.e. preserving and enhancing 'species interactions within a habitat'). Further support for this decision came from the fact that focus group participants claimed to readily understand habitats as a concept. It was envisaged that by reducing the level of precision of definitions the concepts of umbrella and keystone species would be embedded into this category; thus this category would act as a proxy encompassing these ecologically significant species. The concept of 'flagship' species, however, was thought to overlap too much with the concept of charismatic species, and therefore it was considered that the example of the role that flagship species has on enhancing species interactions within a habitat should not be used as this was considered to lead to potential confusion within the overall framework.

Focus group participants also considered ecosystem processes to be important, however, they were not able to clearly recognise the differentiation between the terms 'ecological functions' and 'ecosystem health'. It was therefore concluded that these definitions be made less precise to allow these two concepts to be combined into a single category 'ecosystem processes' was appropriate. A further issue raised in the discussions related to the level of impact that ecosystem processes had on humans, and this was considered to be an important attribute of ecosystem processes worthy of further investigation.

The second group of biodiversity concepts identified related to anthropocentric (or human-related) concepts. Within this group, the concepts of rare and endangered species were both considered to be very important. There was, however, confusion regarding the precise definitions of these terms. For this reason, it was argued that these two concepts should be combined into a single category. Participants were also aware of the alternative levels of threat that a species may be under and they considered this to be very important.

The concepts of 'charismatic', 'cute' and 'familiar' species were all considered to have significant overlap and therefore it was considered that there would be no benefit from attempting to differentiate between the concepts. The concept of 'cute' species, however, was not considered to be helpful or important and therefore the concept could be dropped from the framework. 'Locally important' species were considered to be important both because people valued the fact that they would be able to see, first hand, the benefits from protecting these local species and because they valued the symbolic nature of local species. In all cases, a common theme was that these species were in some way or another likely to be familiar to the public. Thus, it was concluded that it would be useful to identify a group based on familiar species of wildlife.

Thus two distinct themes emerge from the anthropocentric concepts: familiarity and rarity. Based on evidence from the focus groups and comments from the research steering committee, it was proposed that a distinction should be made between familiar species and unfamiliar species, and that a level of rarity be considered within each grouping.

Based on this evidence, it is proposed that the Level 4 framework includes four concepts: familiar species of wildlife, rare unfamiliar species, species interactions within a habitat and ecosystem processes A fundamental new aspect of this framework is that it moves away from the ecological concepts (which focus group participants had some difficulties with) to an alternative framework based on simple concepts that the public appeared to be both familiar with and consider as being important. Finally, it is important to highlight the fact that we envisage that the four simple attributes outlined below will be considered as proxies for the more complex ecological concepts.

- *Familiar species of wildlife*. It is proposed that this attribute be described to include the concepts charismatic, familiar (recognisable) and locally symbolic species. It was also proposed that familiar species should be investigated in terms of common familiar species and rare familiar species.
- *Rare, unfamiliar species of wildlife*. It is proposed that this attribute would focus on those species that are currently rare or in decline which are unlikely to be familiar to members of the public. It was considered that this was an important policy question. Also, it was considered important to incorporate an assessment of the effect that the degree of protection from rarity has on values.
- Species interactions within a habitat. Species interactions within a habitat (which for the remainder of this report we shall refer to simply as the 'habitat' attribute) would be used as a proxy for the preservation of ecologically significant species such as keystone and umbrella species. A key feature of the habitat attribute will be to examine the totality of the habitat in terms of supporting a mix of species; rather than to focus on individual species.
- *Ecosystem processes*. Ecosystem processes will focus on preserving the health of ecosystem functions and services. It was also considered useful to distinguish between ecosystem processes which had a direct impact on humans (i.e. ecosystem services) and those which do not.

4.3. Which methodology is likely to be most suited to the valuation of this change?

The third fundamental question that needs to be addressed relates to which methodology is likely to be the most suited to the valuation of biodiversity change. The review of literature identified a range of value types associated with biodiversity change including direct values (use, passive-use and option values) and indirect values. In this study, we aim, if possible, to capture all components of total economic value associated with biodiversity change. Stated preference methods (including contingent valuation and choice experiments) appear to be the most flexible valuation approach since they are capable of capturing both use and passive-use values. Although both these stated preference methods have been used in the past to value some aspect of biodiversity, the choice experiments approach is more suited to the valuation of the components of biodiversity. Stated preference methods do, however, have limitations. For example, it is unclear whether stated preference methods are capable of eliciting indirect benefits associated with biodiversity (e.g. ecosystem functions and services). Also, there was an issue relating to whether traditional interview based stated preference approaches are capable of meaningfully describing the complexities of biodiversity change to survey respondents. Alternative approaches, such as the more deliberative valuation workshop approach, may therefore be more suited to this exercise. It was clear that a number of fundamental issues still need to be addressed with regard to the identification of which valuation method is the most suited to the valuation of biodiversity change. Therefore in what follows, we report a rigorous systematic assessment of the suitability of alternative methods for the valuation of biodiversity change. To achieve this, we adopted an approach employed by Murphy *et al.* (2002), namely, their Suitability Matrix Scoring System (SMSS).

4.3.1. Methodology: Suitability Matrix Scoring System (SMSS) for the valuation of biodiversity.

The aim of the Suitability Matrix Scoring System (SMSS) was to undertake a rigorous and robust assessment of the suitability of alternative valuation methodologies for the valuation of biodiversity change. The actual SMSS used in this research was based on a scoring system utilised by Murphy *et al.* (2002), who used such a system to assess the suitability of bioassessment schemes with potential to be used in freshwater systems for implementation of the EU Water Framework Directive in Britain. In our application, we test the suitability of fifteen alternative valuation methods (see Table 4) using five broad scoring criteria (see Table 3). A 'total' score for each of the valuation methods was estimated by summing the scores from the five scoring criteria. To ensure that a robust assessment was made, the SMSS was completed by ten of the UK's leading environmental economists (drawn to include a mix of academics, practitioners and policy makers). These economists were therefore asked to complete the SMSS (i.e. input their scores on an MS Excel template – see Table 5) based on their own use and knowledge of the techniques. A mean score was then estimated for each valuation method based on the responses from the ten economists (see Table 5). This mean score was then used as the basis of our assessment of the suitability of the alternative valuation methods. However, before discussing the results from this exercise, we now expand on the detail of the actual SMSS.

As stated above, the actual scoring system comprised five criteria. Within each criterion there were a series of sub-criteria (Table 3). The criteria used were developed based on the economic review (Section 3) and refined following discussions with the research steering group. The alternative valuation methods were assessed on how well the methods met each of the scoring criteria. The actual scores used in the SMSS generally comprised three levels:

0 = does not meet criterion
1 = partially meets criterion
2 = fully meets criterion.

Table 3 provides more detail of how these score were applied to each criterion. Full details on how the SMSS was to be completed can be found in the Appendix to the Phase 2 report (Christie *et al.* 2003b), where the 'Instructions for completing the SMSS' is reproduced.

Fifteen alternative valuation methods were assessed in this exercise (Table 4). The list of valuation methods was based on the methods identified in the review of valuation literature (Section 3). It should be noted that three alternative data collection methods were tested for the contingent valuation method and the choice experiments method; the formats tested included the postal questionnaire, in-person interviews and workshop.

Table 3: Scoring criteria used in the Valuation SMSS

Valuation SMSS scoring criteria	
1. Capacity to capture different elements of economic value associated with biodiversity	
Direct use values (a)	
Passive-use values, e.g. bequest, altruistic and existence values (a)	
Indirect values, e.g. ecosystem functions (a)	
Option / quasi option values, e.g. as yet undiscovered values for science etc. (a)	
Total system value (a)	
2. Suitability for the valuation of different components of a biodiversity programme.	
Capable of valuing the whole biodiversity programme. (a)	
Capable of valuing individual characteristics of a biodiversity programme? (a)	
Can estimate value of marginal changes to a biodiversity programme? (a)	
3. Tests for validity and relevance of results.	
Evidence from existing studies on reliability/validity. (a)	
Can account for lexicographic preferences / protests. (b)	
Can deal with scoping issues. (c)	
Will enable appropriate aggregation of results. (a)	
Capable of feeding into a workable benefits transfer system. (a)	
4. Handles "information problem"?	
Method allows adequate information to enable respondent to make a value judgement (a)	
Respondent can discuss information (a)	
Respondent can reflect on information (a)	
Allows qualitative data to also be collected. (a)	
5. Administration issues	
Ease of design / implementation (d)	
Cost/time to undertake. (e)	
Notes on scoring system	
(a)	0 = does not meet criterion; 1 = partially meets criterion; 2 = fully meets criterion.
(b)	0 = LP mean that valuations are meaningless, 1 = LP is a concern, but may be overcome / dealt with through good survey design, 2 = LP do not affect valuation affection valuation.
(c)	0 = scope effect are a problem and lead to meaningless valuations, 1 = scoping effects are a concern, but may be overcome / dealt with through good survey design, 2 = scoping effects are not an issue of concern.
(d)	0 = v. complex to design and implement; 1 = Relatively difficult to design and implement; 2 = Easy to design and implement.
(e)	0 = v. costly and time consuming; 1 = May be either costly or time consuming; 2 = Relatively cheap and quick to administer.

Table 4: Valuation methods assessed in the Valuation SMSS

Valuation method	Description of method
Replacement costs:	Costs of replacing a resource after it has been damaged.
Restoration costs:	Costs of restoring a resource after it has been damaged.
Relocation costs:	Costs of moving a resource.
Preventative expenditures:	Costs associated with preventing environmental damage.
Averting behaviour:	Costs involved to reduce or avoid consequences of environmental damage.
TCM:	Travel cost method.
HP:	Hedonic pricing.
CVM (P):	Contingent valuation method using postal questionnaire.
CVM (I):	Contingent valuation method using in person interview.
CVM (W):	Contingent valuation using valuation workshops (where participants are able to discuss details of the good in question).
CE (P):	Choice experiments using postal questionnaires.
CE (I):	Choice experiments using in person interview.
CE (W):	Choice experiments using valuation workshops (where participants are able to discuss details of the good in question).
Citizen juries.	Evidence is presented on the good being valued, and participants are required to come to an agreement on the value of the good.
Market stall:	Similar to valuation workshop, but respondents attend two sessions one week apart. This provides respondents with time to reflect on their preferences.

4.3.2. Results from the valuation SMSS

Table 5 reports means scores (along with it's standard deviation) for each SMSS criteria and sub-criteria for each of the 15 valuation methods. The methods that attain the highest total scores include market stall (31.05), choice experiments using workshops (31.08) and contingent valuation workshop (29.28). The postal and in-person interview formats of contingent valuation and choice experiments, along with citizen's juries, achieve scores in the mid twenties. The travel cost method attained a score of 14.37, while hedonic pricing scores 11.76. Finally, the cost based approaches receive scores around ten. Based on this assessment, it would appear that a form of valuation workshop or the market stall approach is likely to be the most suitable method for the valuation of biodiversity. The examination of the standard deviations from the mean of the total scores also supports this claim in that smaller standard deviations were found for the methods that achieved the highest scores. This illustrate that the economists were more consistent in allocating scores to those methods that attained the highest scores.

Examination of the five scoring criteria demonstrates that the stated preference valuation methods generally attained higher scores than the other methods in all criteria, except the 'administration issues' where the stated preference methods were considered to be more complex to design and more costly to administer. Perhaps the key area where the valuation workshop approaches gained extra points related to how these approaches handled the information problem. In particular, extra points were scored in relation to the fact that respondents could discuss and reflect on information provided, and also that they allow qualitative data to also be collected.

Table 5: Results from the SMSS

Valuation SMSS criteria	Replacement costs	Restoration costs	Relocation costs	Preventative expenditures	Averting behaviour	TCM	HP	CVM (P)	CVM (I)	CVM (W)	CE (P)	CE (I)	CE (W)
1. Capacity to capture different elements of economic value associated with biodiversity													
Direct use values (a)	0.80	0.80	0.67	0.90	0.90	1.50	1.00	1.60	1.80	1.80	1.80	1.90	1.90
	0.92	*0.92*	*1.00*	*0.88*	*0.88*	*0.53*	*0.67*	*0.52*	*0.42*	*0.42*	*0.42*	*0.32*	*0.32*
Passive-use values, e.g. bequest, altruistic and existence values (a)	0.50	0.50	0.33	0.30	0.20	0.00	0.10	1.70	1.90	1.90	1.80	2.00	2.00
	0.53	*0.53*	*0.50*	*0.48*	*0.42*	*0.00*	*0.32*	*0.48*	*0.32*	*0.32*	*0.42*	*0.00*	*0.00*
Indirect values, e.g. ecosystem functions (a)	0.70	0.70	0.33	0.60	0.50	0.00	0.20	1.20	1.40	1.70	1.30	1.50	1.80
	0.82	*0.82*	*0.50*	*0.52*	*0.53*	*0.00*	*0.42*	*0.42*	*0.52*	*0.48*	*0.48*	*0.53*	*0.42*
Option / quasi option values, e.g. as yet undiscovered values for science etc. (a)	0.40	0.40	0.44	0.30	0.20	0.00	0.00	1.20	1.40	1.70	1.30	1.50	1.80
	0.52	*0.52*	*0.73*	*0.48*	*0.42*	*0.00*	*0.00*	*0.42*	*0.52*	*0.48*	*0.48*	*0.53*	*0.42*
Total system value (a)	0.60	0.50	0.33	0.60	0.50	0.10	0.10	1.10	1.30	1.30	1.20	1.30	1.30
	0.70	*0.53*	*0.50*	*0.52*	*0.53*	*0.32*	*0.32*	*0.32*	*0.48*	*0.48*	*0.42*	*0.48*	*0.48*
Sub total	**3.00**	**2.90**	**2.11**	**2.70**	**2.30**	**1.60**	**1.40**	**6.80**	**7.80**	**8.40**	**7.40**	**8.20**	**8.80**
2. Suitability for the valuation of different components of a biodiversity programme.													
Capable of valuing the whole biodiversity programme. (a)	0.60	0.50	0.44	0.50	0.40	0.30	0.30	1.70	1.90	1.90	1.50	1.70	1.70
	0.84	*0.85*	*0.88*	*0.85*	*0.84*	*0.48*	*0.48*	*0.48*	*0.32*	*0.32*	*0.53*	*0.48*	*0.48*
Capable of valuing individual characteristics of a biodiversity programme? (a)	0.50	0.40	0.38	0.70	0.50	0.90	0.70	1.20	1.30	1.40	2.00	2.00	2.00
	0.71	*0.70*	*0.74*	*0.82*	*0.71*	*0.88*	*0.82*	*0.42*	*0.48*	*0.52*	*0.00*	*0.00*	*0.00*
Can estimate value of marginal changes to a biodiversity programme? (a)	0.60	0.60	0.38	0.60	0.50	0.70	0.90	1.40	1.50	1.60	2.00	2.00	2.00
	0.70	*0.70*	*0.74*	*0.84*	*0.71*	*0.95*	*0.74*	*0.70*	*0.71*	*0.70*	*0.00*	*0.00*	*0.00*
Sub total	**1.70**	**1.50**	**1.19**	**1.80**	**1.40**	**1.90**	**1.90**	**4.30**	**4.70**	**4.90**	**5.50**	**5.70**	**5.70**
3. Tests for validity and relevance of results.													
Evidence from existing studies on reliability/validity. (a)	0.63	0.63	0.43	0.43	0.29	1.50	1.33	1.20	1.30	1.22	1.22	1.22	1.11
	0.74	*0.74*	*0.53*	*0.53*	*0.49*	*0.71*	*0.71*	*0.42*	*0.48*	*0.44*	*0.44*	*0.44*	*0.33*
Can account for lexicographic preferences / protests. (b)	0.43	0.43	0.50	1.00	1.00	1.22	1.22	0.80	1.00	1.10	1.20	1.20	1.20
	0.79	*0.79*	*0.84*	*1.00*	*1.00*	*0.97*	*0.97*	*0.42*	*0.47*	*0.57*	*0.42*	*0.42*	*0.42*
Can deal with scoping issues. (c)	1.22	1.22	1.38	1.33	1.22	1.33	1.22	1.11	1.11	1.11	1.33	1.33	1.33
	0.97	*0.97*	*0.92*	*0.87*	*0.97*	*0.71*	*0.83*	*0.33*	*0.33*	*0.33*	*0.50*	*0.50*	*0.50*
Will enable appropriate aggregation of results. (a)	0.50	0.40	0.44	0.50	0.44	1.30	0.90	1.90	1.80	1.50	1.90	1.80	1.50
	0.71	*0.70*	*0.73*	*0.71*	*0.73*	*0.82*	*0.57*	*0.32*	*0.42*	*0.71*	*0.32*	*0.42*	*0.71*
Capable of feeding into a workable benefits transfer system. (a)	0.70	0.70	0.44	0.60	0.44	1.40	1.00	1.30	1.40	1.00	1.70	1.70	1.30
	0.67	*0.67*	*0.53*	*0.70*	*0.73*	*0.70*	*0.47*	*0.67*	*0.70*	*0.47*	*0.48*	*0.48*	*0.67*
Sub total	**3.48**	**3.38**	**3.19**	**3.86**	**3.40**	**6.76**	**5.68**	**6.31**	**6.61**	**5.93**	**7.36**	**7.26**	**6.44**
4. Handles "information problem"?													
Method allows adequate information to be presented (a)	0.44	0.44	0.38	0.44	0.38	0.67	1.00	1.10	1.30	1.90	1.30	1.40	2.00
	0.73	*0.73*	*0.74*	*0.73*	*0.74*	*0.50*	*0.93*	*0.32*	*0.67*	*0.32*	*0.48*	*0.70*	*0.00*
Respondent can discuss information (a)	0.33	0.33	0.25	0.22	0.13	0.22	0.22	0.60	1.00	2.00	0.60	1.00	2.00
	0.71	*0.71*	*0.46*	*0.44*	*0.35*	*0.44*	*0.44*	*0.70*	*0.82*	*0.00*	*0.70*	*0.82*	*0.00*
Respondent can reflect on information (a)	0.33	0.33	0.25	0.22	0.13	0.44	0.11	1.40	0.80	1.70	1.40	0.80	1.70
	0.71	*0.71*	*0.46*	*0.44*	*0.35*	*0.53*	*0.33*	*0.52*	*0.63*	*0.48*	*0.52*	*0.63*	*0.48*
Allows qualitative data to also be collected. (a)	0.57	0.57	0.33	0.43	0.33	0.22	0.11	1.10	1.10	1.90	1.20	1.20	2.00
	0.98	*0.98*	*0.82*	*0.79*	*0.82*	*0.44*	*0.33*	*0.74*	*0.74*	*0.32*	*0.79*	*0.79*	*0.00*
Sub total	**1.68**	**1.68**	**1.21**	**1.32**	**0.96**	**1.56**	**1.44**	**4.20**	**4.20**	**7.50**	**4.50**	**4.40**	**7.70**
5. Administration issues													
Ease of design / implementation (d)	1.56	1.56	1.13	1.22	1.00	1.44	0.56	1.20	1.10	1.10	1.20	1.10	1.10
	0.73	*0.73*	*0.64*	*0.67*	*0.89*	*0.88*	*0.53*	*0.42*	*0.32*	*0.57*	*0.42*	*0.32*	*0.57*
Cost/time to undertake. (e)	1.43	1.43	1.17	1.29	1.17	1.11	0.78	1.33	1.00	1.44	1.33	1.00	1.33
	0.53	*0.53*	*0.75*	*0.49*	*0.75*	*0.78*	*0.67*	*0.50*	*0.00*	*0.53*	*0.50*	*0.00*	*0.50*
Sub total	**2.98**	**2.98**	**2.29**	**2.51**	**2.17**	**2.56**	**1.33**	**2.53**	**2.10**	**2.54**	**2.53**	**2.10**	**2.43**
Total Score	**12.84**	**12.44**	**10.00**	**12.19**	**10.22**	**14.37**	**11.76**	**24.14**	**25.41**	**29.28**	**27.29**	**27.66**	**31.08**
Standard deviation of Total Score	*8.76*	*8.23*	*8.41*	*8.46*	*7.78*	*6.43*	*5.48*	*2.62*	*4.62*	*3.90*	*2.26*	*3.92*	*2.50*

In this next section, we discuss in more detail the suitability of the alternative methods to meet the five key scoring criteria. To aid this discussion, we will group the first five valuation methodologies into a single category, which we shall refer to as cost-based approaches.

4.3.2.1. Capacity to capture different elements of economic value

The elements of economic value examined in the SMSS included direct use values, passive-use values, indirect values, option values, and total system values. Generally, the cost-based approaches, the travel cost method, hedonic pricing and citizen's juries were considered to be restricted to the valuation of direct use values[2]. Although the stated preference methods attained high scores in all sub-criteria of value types, the CE and market stall methods scored consistently higher scores than the CVM method. It was also interesting to note that in the stated preference methods, the workshop format tended to have a higher score than the in-person interviews, which in turn was higher than the postal format. Thus, in terms of capturing a wide variety of value types, the workshop format of stated preference methods was considered to be most suitable.

---

[2] It should be noted that economic theory suggests that some of the responses received from the economists were technically incorrect with respect to this criteria. For example, cost based approaches only estimate the costs associated with an activity to protect or maintain an environmental resource and therefore do not technically measure the economic concept of value. In addition, the citizen jury method does not elicit economic values; it merely identifies the jury's decision on the best environmental outcome. A debrief discussion in a sample of the economists identified that they were not fully aware of the details of all of the valuation methods (and the cost based methods in particular). The fact that the standard deviations in the scores for these methods are high verifies this. Clearly, responses based on a poor level of knowledge of the various methods are of concern. However, it should also be noted that those methods that had the highest deviations were generally also those that achieved lower scores and therefore will be less likely to be chosen for the main valuation study to be undertaken in this research.

4.3.2.2. Suitability for the valuation of different components of a biodiversity programme.

These criteria examined the suitability of alternative methods for the valuation of the whole biodiversity programme, individual components of a biodiversity programme and marginal changes to a biodiversity programme. Again, stated preference methods consistently achieved higher scores than the other methods investigated. The market stall and contingent valuation methods gained the highest scores for the valuation of the whole biodiversity programme, while all formats of the choice experiment method gained the highest possible score for the valuation of programme components and marginal changes. Overall, the choice experiments approach achieved the highest score for this criterion and therefore considered to be most suitable.

4.3.2.3. Tests for validity and relevance of results.

The criteria used to assess the validity of the valuation methods included evidence from existing studies, tests for lexicographic preferences and scoping issues, an ability to aggregate results and a capacity to feed into benefits transfer. The stated preference methods, the TCM method and the HP method all achieved high scores in terms of evidence of reliability / validity from existing studies. There was little real variation in the scores for the lexicographic preferences or scoping issues. Both the TCM and the stated preference methods achieved high scores for aggregation issues and benefits transfer. It is interesting to highlight the fact that in both of these methods, the scores for aggregation issues and benefits transfer were higher for the postal and in-person interview formats than for the workshop format. Furthermore, choice experiments attained consistently higher scores than contingent valuation in terms of its suitability to feed into benefits transfer. Overall, choice experiment achieved the highest scores from this validity criterion.

4.3.2.4. Handles the "information problem"?

One of the key challenges which this research is likely to face relates to the difficulties of explaining the complexities of biodiversity to members of the public in a way that the public will understand. The handles the information problem criterion included four sub-criterion. Citizen's jury achieve the highest possible score in all four sub-criteria. This was closely followed by the market stall approach and the choice experiment valuation workshop which both attained the highest possible score in three of the four sub-criteria. The postal and in-person interview formats of the stated preference methods achieved significantly lower scores than the workshop formats. Finally, the cost-based approaches achieved the lowest scores. Based on this, it was concluded that the citizen's jury, market stall and valuation workshop are considered to be the best approaches in which to convey difficult concepts, such as biodiversity, to members of the public in valuation studies.

4.3.2.5. Administration issues

The final scoring criteria examined related to administration issues. Generally, there was not much variation in the scores relating to the ease of design and implementation of a study. In this criterion, the market stall and citizen's jury achieved the highest scores (i.e. most complex), while hedonic pricing received the lowest score. In terms of costs and time involved to undertake the studies, the market stall approach achieved the lowest scores (i.e. highest costs), followed by in-person interview formats of the contingent valuation and choice experiments approaches.

4.4. Conclusions from the SMSS exercise

The Suitability Matrix Scoring System exercise provided a wealth of information on the suitability of the fifteen alternative valuation methods for the valuation of biodiversity change. What is clear from this exercise was that the different approaches vary in terms of their perceived strengths and weaknesses, and importantly that there was no consensus about one method that stood out as being clearly better than the rest. Although the 'total score' attained from the exercise does provide a good indication of the approaches that are likely to be most suitable, more consideration is needed of the relative merits of each alternative approach.

To narrow down the decision, we first rejected those methods that were considered to be unsuitable. A review of the 'total scores' reveals that all of the cost-based approaches, the travel cost method, the hedonic price method and citizen's jury gained scores that were less than half the value of the scores for the stated preference methods (CV, CE and market stall). It could be argued that this fact alone probably provides enough evidence for rejecting these methods. However, it is important to be transparent about the reasons why these methods were considered to be unsuitable:

1. None of these methods are capable of valuing 'total economic value'. In other words, these methods are not capable of elicit benefit estimates for passive-use values, indirect use values, option use values or total system values.
2. All of these methods received low scores for the valuation of the different components of a biodiversity programme.
3. There was little evidence from existing studies that the cost-based approaches could value biodiversity change.
4. The cost-based approaches were considered to be unsuitable for aggregation of results.
5. The cost-based approaches would not be able to feed into a benefits transfer system.
6. All of the methods were likely to have difficulties handling information.

Based on these criteria, we concluded that the cost-based approaches, the travel cost method and hedonic price method are not thought to be suitable for the valuation of biodiversity change, and therefore rejected from this research.

The findings from the SMSS identified stated preference methods as more appropriate methods for valuing biodiversity. In particular, the market stall, choice experiment workshop, and contingent valuation workshop methods were found to achieve the highest overall scores. Other noteworthy approaches included the postal and in-person interview formats of the contingent valuation and choice experiments methods. What was clear from the analysis of the SMSS was that each approach had different strengths and limitations. Thus, the final decision with regards to which method is the most appropriate for the valuation of biodiversity required further consideration of the merits of each, as well as the overall score. In Table 6 below, we highlight the key issues that differentiate the main valuation method.
The final decision regarding which method is the most appropriate was considered in terms of the specific aims of this research project and the publics' ability to understand biodiversity change. In particular, the following issues were considered:

1. Do we wish to value all aspects of total economic value?
2. Do we wish to value all aspects of a biodiversity programme?
3. Do we wish to investigate benefits transfer?
4. Do we wish to aggregate the findings from the valuation exercise to the UK population?
5. How quickly can members of the public understand the complexities of biodiversity?

First, we address Questions 1 and 2 above. Should this research aim to value the total economic value of biodiversity and all aspects of a biodiversity programme? The answer to this is yes. Addressing these aspects of biodiversity was considered to be a fundamental aim of this research. Thus, based on the findings of the SMSS, it is recommended that either the choice experiments method or market stall method is adopted since these methods were found to be the most suited to the estimation of total economic value and the components elements of a biodiversity programme.

With regards to the third and fourth Questions, should this research aim to aggregate the value estimates to the UK and / or investigate benefits transfer? If these aims are to be achieved, it will be essential to use an approach that will allow large sample sizes to be attained. Such approaches include the postal and in-person interview formats of CVM and CE. Unfortunately, these methods were considered to be less suited to the handling complex information. It was clear from the focus group work that members of the public were unfamiliar with the concept of biodiversity. However, this work also found that the majority of individuals were capable of understanding biodiversity if described in layman's terms. Experience from the focus group work indicated that members of the public should be able to understand biodiversity during in-person interviews, however, it is unclear whether this would be the case in a postal survey.

The final Question relates to the publics' ability to understand the complexities of biodiversity. The valuation workshops and the market stall approaches were considered to be the most suited for dealing with complex issues. As mentioned above, the true extent to which the public can understand biodiversity needs to be tested in pilot studies. However, if these studies find that they can not adequately cope with this task, then it is recommended that valuation workshops or market stalls are used. Alternatively, these methods could be utilised to validate the data gathered in in-person interviews.

Table 6: Summary of strengths and limitations of alternative valuation methods

Method	SMSS Score	Key strengths	Key limitations
CE (workshop)	31.08	Valuation of biodiversity components Handles complex information	
Market stall	31.05	Valuation of biodiversity components Handles complex information	Costly to administer
CVM (workshop)	29.28	Handles complex information	
CE (interview)	27.66	Valuation of biodiversity components Suited to aggregation of results Suitable for benefits transfer	Costly to administer
CE (postal)	27.29	Valuation of biodiversity components Suited to aggregation of results Suitable for benefits transfer	
CVM (interview)	25.41	Suited to aggregation of results Suitable for benefits transfer	Costly to administer
CVM (postal)	24.14	Suited to aggregation of results Suitable for benefits transfer	

4.5. What can we conclude about the suitability of alternative methods to valuing biodiversity change?

The main conclusions from this assessment of the appropriateness of alternative methods to value biodiversity are as follows:

- There is not widespread understanding of the term “biodiversity”, but members of the public were found to be capable of understanding the term if it is explained to them.
- Species, habitats and ecosystem were all understood and thought to be important.
- The inclusion of all aspects of biodiversity identified in the literature would not provide a good “frame” for biodiversity for this study. This was because some of the aspects were not thought important by people, many were not understood, some could be more clearly presented if the precision of the definitions are relaxed to allow concepts to be merged, and all need re-describing in layman’s terms. The framing of biodiversity was considered to be very different at the level of public perception than at the level of the scientific community.
- Using focus groups, we identified the following simplified set of attributes as suitable for taking forward:
    - Familiar species of wildlife
    - Rare, unfamiliar species of wildlife
    - Species interactions within a habitat
    - Ecosystem processes
- Using a *Suitability Matrix Scoring System*, we assessed the abilities of 15 different environmental valuation methods to assess the economic value of biodiversity, described along these lines. This matrix made use of a range of criteria, including which elements of total economic value could be estimated using a given method, the availability of tests of validity/relevance, how the method handles the "information problem", and the ease of implementation and cost-effectiveness of the method.
- Ten environmental economists were asked to score each of the 15 methods on all of these criteria, using the matrix.
- The main conclusions that emerged were that stated preference approaches were scored most highly, and within these, those making use of market stall / valuation workshop techniques scored particularly well, due to their particular ability to handle the problem of unfamiliar goods. Cost-based methods scored worst, with revealed preference methods attaining intermediate scores.
- However, different strengths and weaknesses exist even amongst those stated preference approaches which scored highly.

5. Research aims and objectives

The remit for this research project was to:

- *Assess whether it is possible to attain meaningful and robust values for complex goods such as biodiversity;*
- *Develop an appropriate framework which will enable a cost-effective and robust valuation of the total economic value of biodiversity changes in the UK countryside.*

To address this, a review of 'biodiversity' from an ecologists and economists perspective was undertaken , along with the evidence gather from a series of public focus groups and an expert review of the suitability of alternative valuation methods. From this, four broad research objectives were identified and agreed with the research steering group. Below we list these objectives, outline the justification for setting these objectives, and briefly state how we set out to address these objectives. Full details of the research methodology are presented in the Section 6 of this report.

5.1. Valuation of the attributes of biological diversity.

The review of literature and in particular the review of valuation studies (Section 3.3) identified that there were gaps in current knowledge relating to the value of (i) biological diversity *per se* (as opposed to biological resources) and (ii) attributes of biological diversity. In other words, the majority of studies that claim to value biodiversity have tended to value a biological resource (e.g. a particular species or habitat) and have tended to value the totality of that resource rather than value the component attributes that contribute towards the diversity of that resource. The valuation of the attributes of biological diversity was therefore identified as an issue worthy of further research. In order to identify which attributes of biodiversity should be examined, a review of how ecologists define and measure biodiversity in a scientific context was undertaken (Section 2) and the findings from this was then presented to members of the public to identify which biodiversity elements they considered to be important (Sections 4.1 and 4.2). From this, four key attributes of biodiversity were identified: familiar species of wildlife; rare unfamiliar species of wildlife; species interactions within a habitat; and ecosystem processes. The review of valuation literature (Section 3.3) and the valuation SMSS (Section 4.3) identified that the choice experiments method was likely to be the most appropriate methodology for the valuation of the attributes of a resource.

The first objective of this research therefore aims to measure the economic value of the component attributes of biological diversity. It was proposed that the value of these attributes be measured using the choice experiment method.

5.2. Valuation of policy-relevant biodiversity changes on farmland.

The research steering group were also keen to attain an estimate of the economic value of changes in biodiversity that were policy-relevant. Three types of biodiversity changes were considered to be of particular relevance to policy makers, namely: biodiversity enhancement associated with agri-environmental schemes, biodiversity enhancements associated with the re-creation of wildlife habitats, and biodiversity loss from farmland associated with development activities (e.g. house building).

Information gathered in the review of economic literature (Section 3) and the Valuation SMSS indicated that the contingent valuation method would be the most appropriate method for such an exercise. The CV scenarios could be designed to directly elicit the values of the three proposed policy programmes. Contingent valuation thus seems a neater, more direct approach with regard to this second research objective.

The second objective of this research therefore is to measure public willingness to pay for policy programmes that (i) enhance biodiversity through adoption of agri-environmental schemes (ii) enhance biodiversity through the re-creation of new wildlife habitats, and (iii) protect against the loss of biodiversity on farmland as a result of development activity. It was proposed that the economic value of these biodiversity programmes be assessed directly using the contingent valuation method.

5.3. Examination of benefits transfer of biodiversity values

Benefits transfer (taking value estimates from original studies and adjusting them for use in another context) was highlighted in the literature review as being an issue of increasing interest to the policy-making community (Section 3.4), and was also identified as an area of interest to the research steering group. A common approach to testing benefits transfer is to compare value estimates at two discrete locations. Thus, it was proposed that the valuation studies be undertaken at two contrasting case study locations. It was also highlighted in Section 3.4 that there are two approaches to benefits transfer: the transfer of adjusted mean values and the transfer of benefit functions.

The third objective of the research aims to examine the success of transferring both adjusted mean values and benefit functions in the context of biodiversity changes at two discrete case study locations. For reasons that will be explained later, the case study locations chosen to test benefits transfer were Cambridgeshire and Northumberland.

5.4. Dealing with the information problem

One of the key challenges facing this research relates to the fact that members of the public generally have a low level of understanding of biodiversity. This was highlighted both in the review of literature (Sections 2 and 3) and in the public focus groups (Section 4). Work in the public focus groups, however, demonstrated that the public were able to understand biodiversity concepts if explained in layman's terms. Thus, the key to the success of the valuation exercises will be the success to which the complexities of biodiversity can be meaningfully explained to the public. In order to address this challenge, two approaches were proposed. First, an innovative multimedia presentation was proposed to present the complexities of biodiversity to the valuation study respondents. The utilisation of such an approach would enable sufficient and clear information to be presented to respondents in the relatively short timeframe available during household interviews. To further test the effects of information provision, it was proposed that the research also be undertaken in a valuation workshop setting. Such workshops would provide more time for participants to consider, discuss, and reflect on biodiversity and the valuation scenarios than would be possible in the standard household interviews. Thus, the adoption of valuation workshops would provide a sample of data where participants have attain a more comprehensive briefing on biodiversity issues and 'time for reflection and discussion' before they are required to make their valuation judgements. Such data will then enable tests to be undertaken to determine whether the value judgements made during the household interviews correspond to those made by the 'fully informed' workshop participants.

The fourth objective of this research will be to address the challenge of presenting enough *information provision on biodiversity to allow the household survey respondents to make precise and informed value judgements. Valuation workshops will also be used to further explore the effect of information provision.*

6. Research methodology

In specifying the research aims and objectives (Section 5 above), it was proposed that the research utilise both the choice experiment and the contingent valuation methods to value the components of biodiversity and biodiversity policies respectively. It was also proposed that the research utilise both household interviews and valuation workshops since the former allows relatively large sample sizes to be surveyed, while the latter enables further exploration of issues relating to information provision, and provides a methodological cross-check on the main survey results. Finally, two case study areas were also proposed to allow benefits transfer to be investigated. Clearly, the integration of all of the above components into a coherent study requires a systematic approach to the survey design. This was achieved as follows.

First, a common survey instrument was used in the household interviews and in the first sections of the valuation workshops. In other words, the valuation workshop required participants to complete the exact same questionnaire as used in the household surveys, before they were then asked to undertake further workshop activities.

Second, to maximise the amount of valuation information attained per interview, each household interview respondent was asked to complete a series of five choice experiment tasks plus one contingent valuation task. In the workshop, where a greater amount of time was available, each participant was asked to complete five choice experiment tasks (as in the household interviews), plus two (as opposed to one) contingent valuation questions, plus a further set of five choice experiment choice tasks. This number of tasks was decided following experimentation in the pilot studies.

Third, a single PowerPoint presentation was used in both the household interviews and valuation workshops to present the concepts of biodiversity and the various biodiversity improvement policies. The use of PowerPoint allowed the complexities of biodiversity to be presented in a stimulating and easy to understand manner. Furthermore, the use of a single presentation ensured consistency in the material presented.

Fourth, a target of 400 household interviews were undertaken at each of two case study locations (Cambridgeshire and Northumberland). This provided a large enough sample for data analysis and enabled tests for benefits transfer to be undertaken. The valuation workshops, on the other hand, were undertaken principally to test whether value statements were affected by the extra time for reflection / group discussion. Total sample size was less important here and therefore the number of workshops was restricted to six, with each comprising, on average, nine participants. In addition, the workshops were restricted to Northumberland only, since it was not thought that the effect of information was likely to vary between study areas.

The actual structure of the household interviews and valuation workshops is summarised in Figure 3 below. Further details of the component sections of these instruments can be found in Sections 6.1 to 6.9 below, while copies of the actual interviewer's script for the household interviews, the PowerPoint presentation on biodiversity, and moderator's script for the valuation workshop can be found in Appendices 2, 3 and 4 respectively.

The main household study comprised the following five sections.

Section A: Introduction to study
Section B: PowerPoint presentation of biodiversity
Section C: Choice experiment
Section D: Contingent valuation
Section E: Socio-economic data.

The valuation workshops followed the same format as the general household surveys (sections A to E above), but also included four additional sections:

Section F: Questionnaire de-brief on biodiversity
Section G: Questionnaire de-brief on choice task thought process
Section H: Repeat of choice experiments choice tasks
Section I: Review of consistency of choice tasks between Section C and H

Figure 3: Summary of the inter-related design of the household interviews and the valuation workshop.

We now explain each of these elements of the research approach in detail.

6.1. Section A: Introduction to study.

In both the household interviews and valuation workshops, survey respondents were welcomed and informed that the survey aimed '*to examine people's attitudes to the Cambridgeshire / Northumberland countryside*' and that the findings from the survey '*will be used to help the government design policies to improve the wildlife in the countryside*'.

Participants were then asked to complete two simple multiple choice questions on the biodiversity in Cambridgeshire / Northumberland. The aim of these questions was to ascertain the base level of knowledge of respondents on the region's countryside and on biodiversity. Also, these questions provided an easy introduction to the interview and helped respondents to relax.

6.2. Section B: PowerPoint presentation of biodiversity

The second section of the survey required respondents to watch a 20 minute MS PowerPoint presentation on biodiversity. The aim of this presentation was to provide adequate information on biodiversity and potential biodiversity enhancement policies to allow survey respondents to make valid value judgements. The presentation used in both the household interviews and the valuation workshops were identical, however, some of the detail was modified between the two case study areas to reflect the situation within those areas. For example, the species and habitats used to illustrate the various biodiversity concepts were modified according to those that exist in the actual case study area.

Below, we explain the reasons why we choose to use a MS PowerPoint for the presentation of biodiversity and then outline the actual content of the presentation.

6.2.1. Why MS PowerPoint was used to present biodiversity.

A key factor affecting the validity of stated preference valuation studies relates to the success to which the good under investigation can be meaningfully, accurately and consistently presented to survey respondents to enable them to make valid value judgements. Although this can be a challenge to most valuation studies, the very fact that only a small proportion of the public have knowingly heard of the term biodiversity before (see Section 4) presents a significant challenge to this research. In this study, the survey instrument was required to present a lot of information on biodiversity which is likely to be complex and new to respondents.

The majority of valuation studies tend to describe the environmental good under investigation using verbal descriptions, perhaps supported by some written script and / or pictorial images. Although such an approach to presenting the good can be successful with goods that are familiar to survey respondents, evidence gathered in the focus groups indicated that such a standard approach was unlikely to be suitable for presenting biodiversity which was found to be unfamiliar and considered complex. Feedback from focus groups also indicated that the large volume of new information required to be presented on biodiversity was found to lead to both confusion and respondent fatigue. The adoption of a more visual and interactive approach was therefore considered to be more suitable. The approach preferred by focus group participants was to use MS PowerPoint to present biodiversity. The advantages of this approach included:

- the ability of PowerPoint to present information in a range of formats including verbal narration, written bullet points, and visual images of actual species, habitats etc. The use of these various formats helps to stimulate respondents and also reduce respondent fatigue.
- the ability of PowerPoint to present this range of formats in a seamless manner; as compared to say using show cards etc.
- the use of PowerPoint also made it easy for respondents to assimilate the information by listening and watching; as opposed to having to physically read large volumes of text.
- the PowerPoint presentation, to some extent, was considered to reflect a television programme, which is a very common and easy way in which people today attain information.
- The use of a standard presentation to all survey respondents also allows consistency in the information presented to respondents.

There are, however, some disadvantages to using PowerPoint. The first is that computers are required to administer the presentation. Other than the expense of the computer hardware, the use of computers also means that interviews need to take place inside the respondent's house (as opposed to on the door step or street). Although the pilot studies indicated that such access was likely to be readily attained, a problem emerged in Cambridgeshire due to regional survey overload and 'Conmen' working in the areas. Notwithstanding these problems, it was considered that the use of PowerPoint constituted the best approach in which to present biodiversity to survey respondents in a stimulating manner.

6.2.2. Content of PowerPoint Presentation

The PowerPoint presentation was used in both the main household interviews and in the valuation workshops. The presentation was arranged into four main sections. A copy of the actual presentation, along with the narrative script, is reproduced in Appendix 4. Below, we summarise the main sections in the presentation.

Slide 1 : The first slide simply introduced the presentation. Survey respondents were informed of the content of the presentation and told that, following the presentation, they would be asked to answer questions on their views of the information presented and on the future of biodiversity in their local area.

Slides 2 – 8 : Slide 2 introduced survey respondents to a simple definition of biodiversity; '*biodiversity … is the scientific term used to describe the variety of wildlife in the countryside*'. This definition of biodiversity was the one that focus group participants preferred (see Section 4). The narrative that accompanied this slide provided further elaboration of this definition and provided examples to illustrate various aspects of biodiversity. Slides 3 to 8 then introduced the four component attributes of biodiversity that had been identified in the developmental focus groups (Section 4): familiar species of wildlife, rare unfamiliar species of wildlife, species interactions within a habitat, and ecosystem processes. Each attribute was defined, and the alternative levels of biodiversity enhancements associated with these attributes were introduced. Within these descriptions, named examples of relevant species, habitats and ecosystem processes were provided and images presented. These were included to help respondents attain a clearer understanding of the various aspects of biodiversity being discussed. Respondents were also made aware of alternative motivations that people may have for protecting the various aspects of biodiversity. For example, respondents were reminded that they '*might recognise an individual mammal, reptile, bird or even plant because it possesses impressive features such as being large or colourful, or alternatively that it has a particular significant in local culture*'. Following the presentation of this information, respondents were provided with an opportunity to discuss and clarify with the interviewer any issues of outstanding confusion. This discussion helped to ensure that all respondents had attained a reasonably level of understanding of the various biodiversity concepts.

Slides 9 – 12 : The case study area (Cambridge or Northumberland) was then introduced in Slides 9 to 12. Details presented included a description of the predominant land uses found within the case study areas, and the current levels of biodiversity that exist in those areas. Respondents were then informed that human activities, such as farming and development, are currently threatening overall levels of biodiversity in the area and the consequences of this on the four biodiversity attributes were outlined.

Slides 13 – 18 : Slide 13 informed respondents that the government could introduce policies to help protect and enhance biodiversity in the respective case study areas. Policies described included agri-environmental schemes and habitat re-creation schemes. Slides 14 – 17 then outlined how such policies could be introduced to specifically enhance the four aspects of biodiversity identified earlier. In each case, the potential improvements were described in terms of the attribute levels used in the choice experiment (see Section 6.3.2 for details). Respondents were then asked to think about which aspects of biodiversity they would like to see being protected and enhanced. Finally, at the end of the presentation respondents were given a further opportunity to clarify any issues of confusion / uncertainty regarding any aspect of the presentation.

Great effort was undertaken in the development of the above presentation (and in the survey as a whole) to use language which would be easily understood by members of the public. This effort included detailed discussions with members of the public in focus groups to identify which aspects of biodiversity they did and did not understand and to explore the most suitable language to use (details of these focus groups can be found in the Phase 2 report; Christie *et al.* 2003b). When actual scientific concepts and terms were used in the presentation, these were fully defined in layman's terms. Examples of species, habitats and ecosystem processes, along with relevant images, were also used in the presentation to aid understanding. The pilot studies also provided an opportunity to test the level of understanding that survey respondents had on the content of the presentation and a number of refinements made. The feedback from respondents of the pilot survey indicated that the majority of respondents understood the majority of concepts presented. Survey respondents, however, also indicated that the presentation of more information (to try to increase understanding) would likely be detrimental to the study as a whole since this would lead to respondent fatigue. Thus, the inclusion of opportunities for respondents to discuss issues of confusion with the interviewer was seen as a better option to ensure that respondents fully understood the information presented.

6.3. Section D: The choice experiment study

Following the PowerPoint presentation, respondents of both the household survey and valuation workshops were asked a complete a choice experiment exercise. The aim of the choice experiment was to evaluate people's willingness to pay for the four attributes of biodiversity. Below (Section 6.3.1) we outline how the choice experiment tasks were introduced to survey respondents and then in Section 6.3.2 we describe in detail the biodiversity attributes used in the choice experiment. Readers should note that the example we present here relates to the Cambridgeshire case study. The information presented in the Northumberland study was very similar to this, however, we highlight below any case study specific differences.

6.3.1. Implementation of the choice experiment

The choice experiment was introduced as follows:

> *In the presentation you were provided with information on different aspects of biodiversity. You were also informed that biodiversity within Cambridgeshire is under threat. We as a society have some options over how we respond to the threats to biodiversity. We are therefore interested in your opinions on what action you would most like to see taken.*
>
> *We are now going to show you five alternative sets of policy designs that could be used to enhance Cambridgeshire's biodiversity. In each set, you will be asked to choose the design which you prefer.*

An example of a choice task was then presented to respondents and the choice task was explained as follows:

> *On the card you will see three columns. Each column represents the biodiversity outcomes associated with different potential policy options for Cambridgeshire's biodiversity. Policy options A and B relate to policies that could be implemented to improve Cambridgeshire's biodiversity. The third column relates to a 'Do nothing' option; that is Cambridgeshire's biodiversity would continue to decline at current rates as discussed in the computer presentation. Each policy option on the card is described in terms of the four aspects of biodiversity outlined in the computer presentation, namely:*
>
> - *Familiar species of wildlife,*
> - *Rare, unfamiliar species of wildlife,*
> - *Species interactions within a habitat,*
> - *Ecosystem processes.*

> *In addition, there is one further aspect that you need to consider. Implementation of the biodiversity enhancing policies (i.e. policy options A and B) would be costly to you since the costs of biodiversity improvements would be passed onto you as a tax payer. You need to consider whether the biodiversity improvements associated with a particular policy is worth the extra tax burden placed on you. The 'Do nothing' option, however, would not cost you anything extra.*
>
> *We want you to indicate, for each of the five choice set that you are about to be presented with, whether you prefer policy option A, policy option B or the 'Do nothing' option. If you choose the 'Do nothing' option, you should assume that the current decline in biodiversity in Cambridgeshire would continue at current rates and that your tax bill will not change from its current level.*

Once the respondents had undertaken all five choice tasks, they were asked to indicate the main reason that they had for making the choice that they did. The reasons given included:

1. *I chose either policy option A or B because I thought that they were good value for my money.*
2. *I did not consider that the biodiversity improvements from either policy options A or B to be good use of my money.*
3. *I do not think that increases in taxation should be used to fund the biodiversity improvements shown in policy options A or B*
4. *I already contribute to environmental causes as much as I can afford*
5. *The costs of biodiversity improvement should be paid for by those who degrade biodiversity.*
6. *Other (please specify)*

Response 1 above was included to indicate a genuine positive valuation bid for biodiversity enhancements. Options 2 and 4 were used to indicate genuine zero bids, i.e. these respondents were not willing to contributes towards biodiversity enhancement programmes because they considered they would not benefits from it. Finally, options 3 and 5 were recorded as protest bids, i.e. respondents did not wish to contribute towards biodiversity improvements because they did not agree with the payment vehicle. The above summary describes how the choice experiment tasks were delivered to survey respondents. In the next section, we provide detail of the actual attributes used to describe biodiversity in the choice experiment.

6.3.2. Biodiversity attributes used in the choice experiment

The choice experiment was utilised to enable the value of the attributes of biological diversity to be measured. Attributes for the choice experiment were developed based on those aspects of biodiversity which the public considered to be important. The process of identifying key attributes was based on a review of ecological measures and definitions of biodiversity (Section 2), which were then scrutinised by members of the public (Section 4.1 and 4.2). The outcome of this exercise was four biodiversity attributes: familiar species of wildlife; rare unfamiliar species of wildlife; species interactions within a habitat; and ecosystem processes. Each of these attributes was then defined according to three levels of provision, including the status quo and two levels of biodiversity attribute enhancement. Details of the attributes and attribute levels were developed through a combination of discussions with members of the public in focus groups, through discussions with individuals who are actively managing biodiversity in the case study areas, and through computer predictions made using the

Glasgow biodiversity model (see Section 2.2.8.3). Table 7 below provides a summary of the four biodiversity attributes used in the choice experiment, along with the three levels of provision of each attribute. Further details of the biodiversity attributes and levels are then presented.

Table 7: Summary of biodiversity attributes and levels used in the choice experiment

	POLICY LEVEL 1			**POLICY LEVEL 2**		**DO NOTHING (Biodiversity degradation will continue)**
Familiar species of wildlife	Protect *rare* familiar species from further decline			Protect *both rare and common* familiar species from further decline..		Continued decline in the populations of familiar species
Rare, unfamiliar species of wildlife	Slow down the rate of decline of rare, unfamiliar species.			Stop the decline and ensure the recovery of rare, unfamiliar species		Continued decline in the populations of rare, unfamiliar species
Species interactions within a habitat	Habitat restoration, e.g. by better management of existing habitats			Habitat re-creation, e.g. by creating new habitat areas		Wildlife habitats will continue to be degraded and lost
Ecosystem processes	Only ecosystem services that have a direct impact on humans, e.g. flood defence are restored.			*All* ecosystem processes are restored		Continued decline in the functioning of ecosystem processes
Annual tax increase	10	25	100	260	520	No increase in your tax bill

First, it is useful to discuss the status quo or ‘Do nothing’ option. In choice experiment it is common practice to include a standard option within all choice tasks. In this study, we choose a ‘Do nothing’ policy option. The ‘Do nothing’ option was designed to reflect the situation where no new policies would be implemented to protect and enhance biodiversity on farmland in the case study areas. In other words, respondents were told that farmers would continue to farm as they currently are. The consequence for this option in terms of the four attributes of biodiversity was then reported. The actual wording used in the valuation study to describe the status quo is as follows:

> *So what will happen to Cambridgeshire’s biodiversity if we continue as we are and do nothing to help protect and enhance the county’s biodiversity?*
>
> *If additional effort is not made to protect Cambridgeshire’s biodiversity it is likely that:*

- *The populations of some familiar species will continue to decline, thus you will be less likely to see these in the countryside.*
- *The populations of some rare species will also continue to decline. This decline potentially may lead to the local extinction of some species from the Cambridgeshire countryside.*
- *The area of wetland and woodland habitats is likely to be further reduced and become more fragmented. Also the quality of the remaining habitat is also likely to decline. The consequence of this will be a general reduction in species diversity.*
- *Ecosystem processes will continue to be threatened, potentially leading to increased risk of future flooding, reduced water quality, and global warming.*

In addition to the 'Do nothing' option outlined above, each of the four biodiversity attributes was developed to represent two levels of biodiversity enhancement. We now describe these levels for each attribute in turn. We also refer the reader to the interviewer script on the PowerPoint notes page reproduced in Appendix 4.

6.3.2.1. Familiar species of wildlife

The attribute 'familiar species of wildlife' was defined as '*any bird, mammal, reptile or plant that is likely to be recognised by members of the public*'. Respondents were told that such species may be familiar to them *'... because it possess impressive features such as being large or colourful, or alternatively that it has a particular significance in local culture*'. Thus, in this definition, we aim to capture charismatic species and locally significant species. Survey respondents were also told that '*often these species will be familiar to you because they are commonly seen in the Cambridgeshire countryside'* while *'other species may be familiar to you because they are rare or endangered and therefore attract a lot of attention*'. Examples of both rare and common familiar species within the respective case study areas were then presented.

In the choice experiments design, the two levels of provision of familiar species of wildlife were presented as follows:

> ***Level 1: 'Protect rare familiar species from further decline.'***
>
> *The first possible outcome would be that the populations of familiar species that are currently rare would be protected from further decline. Policies would therefore only target those familiar species that are currently designated rare. These species would become more abundant and the threat of local extinction would be removed. Such a policy, however, would not target those familiar species that are not designated as being rare. Therefore the populations of these species would continue to decline at current rates, with some potentially becoming rare.*

Level 2: 'Protect both rare and common familiar species from further decline.'

The second possible outcome would be that the populations of both the rare and more common familiar species would be protected from further decline. Thus, policies would aim to remove the threat of local extinction of currently rare familiar species and would also stabilise the populations of other familiar species so that no more familiar species become rare in the future.

Thus, the 'familiar species of wildlife' attribute was designed to (i) assess the level of concern that the public have for species that they are familiar with and (ii) assess whether this concern is only for rare familiar species, or for both rare and common familiar species.

6.3.2.2. Rare unfamiliar species of wildlife

The second choice experiment attribute used was 'Rare unfamiliar species of wildlife'. This attribute was defined as '*... any species of wildlife that has officially been designated as being rare or endangered, but which members of the general public are unlikely to know about'*. Respondents were informed that such species might '… *include rare plants and insects, as well as some of the less well known birds and mammals'.* Examples of such species within the respective case study areas were given. Respondents were also informed that '*The very fact that so few of these rare unfamiliar species actual exist in the countryside means that you are unlikely to ever have seen one in the wild and also you are unlikely to ever see one in the future'.* Thus, the values estimated for 'Rare unfamiliar species' should be considered to reflect existence use values, rather than actual use values.

The two levels of enhancement of 'Rare unfamiliar species of wildlife' used in the choice experiment were as follows:

Level 1: Slow down or halt the decline in the populations of rare, unfamiliar species.

The first policy outcome would aim to slow down or halt the rate of the decline in the populations of rare unfamiliar species. The result from such a policy would be that the risk of locally and nationally extinction of rare unfamiliar species would be reduced. However, with this policy option it is likely that some rare unfamiliar species may still become locally and nationally extinct.

Level 2: Stop decline and ensure recovery of rare species

The second policy option would again target unfamiliar species that are locally or nationally rare in Cambridgeshire. However, policy would be such that it would aim to enhance the populations of these rare species to a level where the populations would recover and therefore the threat of local extinction would be removed.

Thus the rare unfamiliar species attribute was included to (i) assess whether the public have existence and / or moral value preferences for biodiversity enhancements and (ii) assess whether the public were able to distinguish between different levels of biodiversity enhancement (slow down decline versus recovery). The 'slow down decline' option was also considered to be very policy relevant since it reflects what is perhaps a more realistic situation, i.e. that it is often very difficult to design policies that would ensure complete recovery of rare species.

6.3.2.3. Species interactions within a habitat

The third attribute of biodiversity considered in the choice experiment related to 'Species interactions within a habitat' (which we shall simply refer to as the 'Habitat' attribute in this report).

'Habitats' were defined as '*places where groups of species live together*'. In Cambridgeshire, common habitats were stated to include wetland areas such as the fens and woodland areas. In Northumberland, common habitats included wet grassland, moorland and woodland. Respondents were informed that they would expect to find '*a wide variety of different types of species including plants, insects, birds and mammals'* within a habitat, however respondents would be '*unlikely to be familiar with the majority of species that live in that habitat.*'. This last sentence was included to ensure that respondents distinguished between the unfamiliar species found within habitats and the 'familiar species' attributes. Furthermore, respondents were informed that '*A key feature of a healthy habitat is the interaction between the different species that live in that habitat*'. Examples of such interactions were provided. Thus, the habitat attribute aimed to capture respondent's values for protection of a habitat area that supports a wide range of both unfamiliar and well-known species, rather than focusing on individual species as was the case in the previous two attributes. It should, however, be noted that although not directly presented to survey respondents, the habitat attribute was designed to include the outcomes (i.e. increased species interactions) associated with ecologically significant species such as keystone species and umbrella species.

The two policy levels aimed at protecting and enhancing habitats were described as follows:

> ***Level 1: Habitat restoration.***
>
> *Habitat restoration involves improved management on existing, but degraded wildlife habitats. Restoration policies tend to lead to moderate increases in species diversity within a particular habitat. Normally, it would take around 10 years before a restored habitat supports many species.*
>
> ***Level 2: Habitat re-creation.***
>
> *Habitat re-creation involves the re-establishment of wildlife habitats on land that is currently farmed. The re-creation of wildlife habitats will require large changes in the types of species found in an area. Normally, it would take around 50 years before a recreated habitat supports a wide diversity of species.*

The habitat attribute therefore moves away from the idea of the importance of particular species, to one in which it is the interaction of species is important. The levels presented relate to whether the public favour improvements on existing habitat features (which would involve a moderate change in species composition but would result in a high quality habitat) or favour the re-creation of new habitat areas on farmland (which would involve large changes in species composition, but result in only a moderate quality of habitat in the medium term). Also, the habitat attribute should be considered as a proxy for ecologically significant species.

6.3.2.4. Ecosystem processes

The final aspect of biodiversity that was considered in the choice experiment related to the role that biodiversity has in terms of providing ecosystem processes.

Ecosystem processes were described as the term '*used by scientists to describe a wide range of natural processes that help to keep nature in balance*'. Respondents were informed that some ecosystem processes may provide direct services to humans (for example, flood protection and maintaining water quality), while others would have less obvious impacts (carbon storage, nutrient cycling in soils). Respondents were also informed that although scientists do not yet fully understand all ecosystem processes, current knowledge suggests that ecosystem processes tend to be more stable with higher levels of biodiversity.

The two policy options to maintain and enhance the functioning of ecosystem processes were as follows:

> ***Level 1: Maintain only those ecosystem services that have a direct impact on humans.***
>
> *We take action to restore ecosystems to only maintain the supply of services that have a direct impact on humans. Thus, there will be a reduced risk of ecosystem services such as flood defences failing.*
>
> ***Level 2: All ecosystem processes are restored.***
>
> *We take action to restore ecosystems to maintain the supply of all processes reducing the risk of failure of those processes that affect humans such as flood defences and those processes which do not directly affect humans such as carbon storage or nutrient recycling.*

Thus, this attribute moves the focus of biodiversity away from wildlife examples to a focus on the role that biodiversity plays in supporting ecosystem processes. The levels used in the choice experiment were designed to examine whether the public differentiated between those ecosystem processes that directly affect man, and those ecosystem processes which do not.

6.3.2.5. Tax attribute (payment vehicle)

The payment vehicle used in the choice experiment was an increase in general taxation. The reasons for using this payment vehicle include the fact that biodiversity enhancement programmes are generally paid for through taxation and second that participants of the focus groups indicated that taxation was their preferred payment option. Six payment levels of taxation were used in the choice experiment, including the £0 level in the status quo option. The actual levels used were identified following a small open-ended pilot contingent valuation study which identified the likely range of bid levels for biodiversity enhancements. These levels were then tested in a pilot choice experiment. The bid levels were then refined following the analysis of the pilot study. Thus, the final tax levels used in the choice experiment were: £10, £25, £100, 260, 520, plus the no tax increase in the 'Do nothing' option. Tax rises were annual increases per household for the next five years

6.3.3. *Design of choice tasks*

Details of the choice experiment attributes and levels are presented above. In order to ensure that the attributes vary independently of one another, such that their individual effect on respondent's preferences can be isolated, an orthogonal experimental design is required. The 'Orthoplan' package of SPSS was used to generate a 3^4 X 5^1 fractional factorial experimental design, which created 25 choice options. A blocking procedure was then used to assign the options to 10 bundles of five choice sets. Thus each choice experiment respondent was presented with a bundle of five choice tasks, where each choice task comprises two policy options and a status quo. Both the household interviews and valuation workshop used this experimental design.

6.4. Section D: The contingent valuation study

The contingent valuation (CV) study was then undertaken immediately after the choice experiment section. The CV study was introduced as follows '*I would now like to discuss with you a particular biodiversity policy option within [Cambridge / Northumberland].*

Within Cambridgeshire, three policy programmes were examined:

- Agri-environmental scheme
- Habitat re-creation scheme
- Biodiversity loss due to development.

While only two programmes were examined in Northumberland:

- Habitat re-creation scheme
- Biodiversity loss due to development.

In the CV section of the household study, each respondent was provided with details of only one policy programme. The main reason for limiting the CV to only one scenario per respondent was based on the fact that attempts to introduce a second CV scenario resulted in respondent fatigue due to the total length of the survey. It was thus decided that it would be preferable to collect more robust data by restricting the household study to only one CV scenario per respondent. CV scenarios were therefore randomly assigned to respondents. In the valuation workshops, however, more time was available and respondents were asked to complete both of the Northumberland CV scenarios.

We now describe each of the CV policy scenarios in turn. The actual descriptions of these scenarios can be found in Appendix 3.

6.4.1. *CV scenario 1: Agri-environmental scheme*

Although there are a number of agri-environmental schemes currently being offered to UK farmers, each with slightly different objectives, the 'arable stewardship' scheme was considered to be of particular interest to this research. The key attraction of the arable stewardship scheme is that it aims to promote biodiversity on intensively managed arable land, which has little residual biodiversity. This contrasts with many of the other schemes which, for example, promote enhancements of biodiversity within existing, albeit degraded, habitats. Thus, through the examination of the arable stewardship scheme, the research will attain an understanding of how people value the promotion of biodiversity on intensively managed farmland. In the CV scenario, two aspects of the arable stewardship scheme were highlighted, namely the creation of conservation headlands and the reduction of fertiliser, herbicide and pesticide use on arable land. Details of these two management prescriptions and their likely biodiversity benefits were presented. The level of biodiversity change associated with these management prescriptions were established based on evidence from the arable stewardship scheme itself and on output from the Glasgow biodiversity model. Where possible the biodiversity benefits from the arable stewardship scheme were described according to the biodiversity attributes used in the choice experiment. The actual wording used to describe the benefits was:

- *It is likely that the diversity of plants within an arable field would more than double. In particular, agri-environmental schemes are likely to encourage the establishment of some of the more attractive and familiar plant species such as the poppy, as well as some rare less familiar species of plants such as corn cleavers and small-flower catchfly.*
- *These wildflowers would provide a source of food for rare birds such as grey partridge, skylark and corn bunting.*
- *The agri-environmental schemes would also provide habitats for a wide range of insects, butterflies and mammals.*

It should be noted that the agri-environment scheme scenario was only used in the Cambridgeshire case study, and not in Northumberland case study. The reason for not including this scenario in Northumberland was that only a relatively small proportion of agricultural land in Northumberland is under arable production and therefore an agri-environmental scheme similar to the arable stewardship scheme was not that relevant to Northumberland.

6.4.2. CV scenario 2: Habitat re-creation

The second scenario used in the CV study was habitat re-creation. In Cambridgeshire, the proposal was to re-create wetland habitats on existing farmland, while the Northumberland scenario proposed to re-create wet grassland on existing farmland. The various management activities required to re-create these habitats were explained. These included activities such as:

- The seasonal flooding of flood plains where possible.
- Maintenance and restoration of water quality.
- Cessation of usage of fertilisers and pesticides
- Reduction of grazing pressure
- Reintroduction of some wildlife species.

The biodiversity benefits associated with the re-creation of these habitats were again presented in terms of the choice experiment biodiversity attributes. In Cambridgeshire, these outputs were presented as:

- *The new wetland habitat will support a wide diversity of wildlife including plants, insects, birds and mammals.*
- *In particular, the new wetland would provide habitats for some familiar and unfamiliar rare species such as the water vole, bittern, and Desmoulin's whorl snail.*

- *The wetland area would also provide some ecosystem services such as flood protection and the enhancement of water quality.*

In Northumberland, similar biodiversity benefits were presented, however, the examples of species that would benefit from the programme were modified to take account of the local situation. Thus, the aim was to present the same level of biodiversity enhancement in the two case study areas, but to fine-tune the detail to the local situation.

6.4.3. CV scenario 3: Loss of biodiversity due to development

The third CV scenario aimed to examine public willingness to pay to prevent the loss of biodiversity on farmland as a result of urban development. Housing development projects, as opposed to industrial development, were the chosen form of development since housing developments were considered to be less likely to invoke protest votes. Respondents were thus presented with the following scenario:

> *The government is planning to build 70,000 new houses in Cambridgeshire. Many of these houses would be built on existing farmland. Although most of the farmland on which houses are likely to be built in Cambridgeshire is intensively managed and has low biodiversity value, it is likely that some of the houses would be built on farmland that is managed under an agri-environmental scheme.*

Examples of the type of biodiversity found on the threatened farmland were then described. Respondents were then told that a trust fund was being set up which would be used to ensure that at least 50% of farms that are currently managed under an agri-environmental policy are protected from urban development. Respondents were then asked to state how much they would be willing to pay into this fund the help ensure that the county's best examples of agri-environmental farmland is protected against urban development and therefore its biodiversity protected. The scenario used in Northumberland land was similar to this, other than the specific examples of species.

It is important to note that the three different CV scenarios for Cambridgeshire use two different payment vehicles. For agri-environment schemes and habitat creation a tax increase is used; for preventing development losses, a trust fund is employed. These choices reflect assumptions about which payment vehicles respondents would find most credible in each case. However, because they differ, the willingness to pay amounts from the trust fund scenario cannot strictly be compared to those from the tax scenarios, since the incentive properties differ across the payment vehicles. This is *not* a problem with the Northumberland CV scenarios, since all use the same payment vehicle.

6.4.4. The CV elicitation question

A critical component affecting the success of a CV study relates to the way the willingness to pay elicitation question is posed. In this study, we made every effort to follow the NOAA guidelines (Arrow *et al.*, 1993). Below, we outline the procedure that was followed.

Following the descriptions of the CV scenarios, respondents were first asked to state whether or not their household would be prepared in principle to contribute towards the biodiversity enhancement programmes. If the response to this question was 'no', respondents were asked to state the reason why they would not be prepared to contribute towards the policy. The list of reasons for no responses included:

1. *Biodiversity policies are not a good use of my money.*
2. *I do not think that there is a need to improve Cambridgeshire's biodiversity*
3. *I can not afford to pay towards biodiversity policies*
4. *I already contribute towards improving biodiversity in other ways*
5. *I would be prepared to contribute towards improving Cambridgeshire's biodiversity, but not by paying more tax*
6. *The costs of improving biodiversity should be paid by those that contribute to biodiversity loss*

Options 1, 2, 3 and 4 were included to represent genuine zero bids, while options 5 and 6 reflected protest bids.

Respondents who stated that they were prepared to contribute towards the biodiversity programmes were asked to consider the following before answering the CV WTP elicitation question.

> *I am now about to ask you how much your household would be prepared to pay, as an annual increase in your tax bill over a period of 5 years to secure the biodiversity benefits outlined above. The amount that you state would only go towards improving biodiversity in Cambridgeshire. Before you answer this question, please consider the following:*
>
> - *The amount that you state should reflect the benefit that you would receive from the biodiversity improvements in Cambridgeshire.*
> - *In order to make this payment, you may need to reduce the amount that you spend on other things.*
> - *If the total amount people are willing to pay is not enough, the policy may not be introduced.*

For CV scenarios 1 and 2 (agri-environmental scheme and habitat re-creation scheme) an annual increase in tax was the preferred payment vehicle. Respondents were therefore asked:

> *Please indicate which amount shown on the card your household would be willing to pay as an annual increase in your tax bill over a period of 5 years to achieve the biodiversity outcomes just described.*

In the third CV scenario, development loss, the payment vehicle used was altered to one of a trust fund, since this represented a more realistic option for this scenario. In all three scenarios, the WTP elicitation question was presented as a payment card. Two payment cards were used in the survey in an attempt to avoid starting point bias. Each payment included 11 value amounts ranging from £0.75 to £768 in the first payment card and £1.25 to £1280 in the second. Payment cards were randomly assigned to respondents.

Following the selection of the payment amount, respondents were asked to state, from the following list, the statement that most closely reflected how they came to their chosen amount:

1. *The amount I indicated reflects the benefits that I would gain from seeing increased biodiversity in Northumberland*
2. *The amount I indicated reflects the benefits that I would gain from knowing that the level of biodiversity in Northumberland had increased, even although I am unlikely to see it first hand.*
3. *The value reflects how much I think it would cost to improve biodiversity in Northumberland*
4. *The amount I indicated seems a reasonable amount to pay towards this type of policy.*
5. *I just picked a value at random.*

Options 1, 2 and 4 above were included to indicate a considered response to the WTP question, where option 1 reflects a likely use value and option 2 reflects a passive-use value. Options 3 and 5 were included to identify situations where respondents were not basing their value judgement on the likely utility gains, but on an alternative, less reliable basis.

6.5. Section E: Socio-economic data

The last section in the main household survey involved collecting data on respondent's socio-economic and attitudinal characteristics. Such data is required for the analysis of both the contingent valuation and the choice experiment WTP responses usually involves modelling the bid responses against these socio-economic and attitudinal characteristics. Survey respondents were therefore asked to provide information on their gender, age, martial status, household size, number of children, qualifications, work status, income and membership of environmental organisations.

6.6. Section F: Questionnaire debrief on level of understanding of biodiversity concepts

Sections A to E above were included in both the household interviews and the valuation workshops. The workshops, however, provided an opportunity to further explore participant's values of biodiversity. This additional work is described in Sections F to I.

The first additional exercise in Section F of the valuation workshops was to further explore the level of understanding of biodiversity concepts that participants had acquired before making their valuation judgements. This exploration was undertaken in three stages. First, participants were asked to indicate, on a likert scale, '*how well you think you understand the various biodiversity concepts, the levels of biodiversity protection associated with each concept and the two biodiversity policies*'. We shall refer to this question as the 'initial level of understanding'. Next, participants were asked to discuss, and then reflect on, each of the four biodiversity concepts and their associated levels. In these discussions, the moderator aimed to identify those concepts that respondents (i) understood and (ii) did not understand or were confused about. The moderator then provided further explanations of the concepts that were not understood. It should be noted that the above was the only way in which workshop participants got more information on biodiversity than the main household survey respondents. In the third stage, participants were again asked to indicate, on a likert scale, the level of understanding of biodiversity that the *now* (post discussion) think they had. The aim of these questions was to explore how the additional discussion and reflect on biodiversity affected their level of understanding of the biodiversity concepts. Analysis of this data on how changes in the level of understanding of biodiversity concepts affect valuations was used to validate the main household survey responses. This information was also used to examine 'social engineering'; that is whether the provision of 'too much' information results in advertising and socially engineers preferences upwards over and above their real value.

6.7. Section G: Reflection on the choice task

The workshop also provided an opportunity to explore how participants made their choices between the policy options in the choice experiment and also how they considered the cost implications in their choices. To address these issues, respondents were first asked to write down their thought processes which they underwent when deciding which policy option they preferred. This was followed by three closed questions:

Question G-2: Please indicate which of the follow statements most closely reflects how you choose your preferred policy option.

- *I considered the level of provision of all biodiversity attributes equally* ❑
- *I only considered the levels of provision of a couple of attributes which I considered to be important and largely ignored the others.* ❑
- *I only considered the level of provision of one attribute and ignored the rest.* ❑
- *I chose the cheapest option only* ❑
- *I chose options at random.* ❑

Question G-3: Please indicate which of the following statements most closely reflects how you considered the tax attribute in the choice sets.

- *I didn't really consider the tax attribute* ❑
- *I choose the options with the lowest tax level.* ❑
- *The tax amount was an important consideration when I made my choices and I rejected options where the tax amount was too high,* ❑
- *The tax amount was an important consideration when I made my choices and I tried to weigh up the level of tax with the environmental benefits.* ❑

Question G-4: To what extent did you consider whether you could actually afford the tax levels associated with your chosen option.

- *I fully considered what I would have to give up in order to pay the stated tax amounts* ❑

- *Although I considered the tax amounts, I didn't really think about whether I could afford it.* ❑

- *I didn't really think about whether or not I would be able to afford the tax amount.* ❑

The aims of the above three questions were to examine whether or not respondents undertook appropriate consideration of the biodiversity benefits from the policy options, and whether they made appropriate consideration of the cost implications to them. Finally, participant's responses to these questions were discussed in the group and participants were asked to further reflect on the way they made their choice tasks, and in particular the tradeoffs between biodiversity benefits and costs to respondents.

6.8. Section H: Repeat of choice experiments choice tasks

Following the group discussions on biodiversity and the choice task, workshop participants were asked to complete a second series of choice experiment choice tasks. This second set of choice tasks comprised the same choice sets as the original choice experiment, but the order of the choice tasks was varied to prevent participants from realising that they were completing the same tasks. The aim of this second round of choice tasks was therefore to test to see whether the additional information / insight that participants gained from the discussion affected their policy choices and therefore their valuations.

6.9. Section I: Review of consistency of choice tasks between Section C and H

The final section of the valuation workshop required participants to directly compare their initial choice experiment choices (from Section C) with the choices that they made following the discussions (Section H). Participants were then asked to consider the reasons why they may have changed their choice options.

6.10. Administration of survey

Details of the household interviews and valuation workshops are presented above. In this section we report detail of how these surveys were administered.

6.10.1. Administration of household survey

The household survey was conducted at two case study locations: Cambridgeshire and Northumberland. A random sample of 400 addresses was drawn from each location. The actual interviews took place either during weekday evenings or during weekends. Due to the need to use a computer for the PowerPoint presentation, all interviews were administered in respondent's homes. Interviewers made two attempts to contact the selected respondents at their home address. If no response was received during the second visit, the interviewer then approached neighbouring addresses until a willing respondent was found. In an attempt to avoid biasing the sample towards unemployed / elderly, interviews were not conducted during weekday working hours (9:00 am to 5:00 pm). This method of conducting the interviews was found to be very successful in Northumberland. However, interviewers found it difficult to gain access into people's houses in Cambridgeshire. Two facts contributed to these difficulties. First, a number of other surveys had recently been conducted in Cambridgeshire and there was a feeling within the area of survey overload. Second, there were also reports of Con Men working in Cambridgeshire who were illegitimately attempting to gain access to people's homes. The implications of these issues meant that often a large number of addresses in Cambridgeshire would need to be approached before a willing participant was found.

6.10.2. Administration of the valuation workshops

Six valuation workshops were undertaken during this research. All workshops were administered in Northumberland. In an attempt to attain a cross section of participants, the location where workshops were conducted was stratified between rural villages, towns and city. A sampling frame based on gender and age was used, and ten individuals were select on the day before the actual workshop. A £20 incentive was provided to encourage participation in the workshops.

6.11. Tests for benefits transfer

One of the aims of this research was to test whether the benefit estimates for biodiversity could be transferred between locations. To address this, two case studies locations were surveyed: Cambridgeshire and Northumberland. Two methods of benefits transfer were undertaking in this research. The first approach is to compare adjusted mean WTP values across case study locations, while the second approaches focus on the transfer of bid functions.

6.12. Description of case studies

It was outlined above that two case study locations were used in this research. The use of two study areas was considered to be important since it enabled the various biodiversity enhancement value estimates to be compared and contrasted between two rather different countryside settings, as well as enabling tests for benefits transfer to be undertaken. The criteria used to select the case studies included:

- Both case studies should address biodiversity protection and enhancement on farmland. The reason for choosing agricultural land, as opposed to other land uses, was that (i) it represented an area where marginal changes in biodiversity were likely to be seen in the future, (ii) it represents a policy area where biodiversity policies are likely to become of increasing importance in the future, (iii) its management is at the heart of DEFRA's remit.
- The case study areas were also chosen to represent contrasting levels of current biodiversity associated with different agricultural systems within the UK.

The two chosen case study areas were Cambridgeshire and Northumberland. In what follows, we briefly outline the biodiversity found within these two regions.

6.12.1. Biodiversity in Cambridgeshire

Cambridge was chosen to represent an area of intensively managed cereal production. Levels of on-farm biodiversity in Cambridgeshire are typically low, while the area as a whole contains many small, scattered nature reserves and SSSIs, which act as oasis for wildlife. Particular species of note associated with these reserves have included calcareous plants such as oxslip, marsh orchids and the pasque flower, rare butterflies (such as black hair streaks, swallowtails and large tortoiseshells). Vast numbers of birds winter on the Ouse washes, and several rare species also nest in the area such as black-tailed godwit and ruff. Much of Cambridgeshire's low lying county is capped by boulder clays and gravels left behind after the last ice age. The fertile soils associated with these conditions in combination with its moderate climate produce highly productive farmland, which have become biodiversity deserts of agricultural intensification. Even so, along the rivers Great Ouse, Little Ouse and the Cam, which wind across Cambridgeshire, a number of old, flood meadows and pastures still survive within their flood plains. Away from the river valleys but still on the boulder clay several fragments of ancient broadleaved woodlands remain. In habitat terms Cambridgeshire is most famous for its fenlands. Most of the fen is found in the northern half of the county, around the old Isle of Ely. In this area the soils are predominantly silt or peat based, and in former times much of the area was under shallow water or at least flooded in the winter. The fens support a range of specialist plants and invertebrates as well as being important for wetland and migratory birds. Large-scale agricultural draining of the fens started in the seventeenth century, and only recently have agri-environment schemes and technical developments allowed and promoted the reversion of this process. In the south of the county is a band of low rolling chalk hills. This region was probably never thickly wooded, being originally grazed by wild cattle and deer, but more recently by sheep. Only fragments of these species rich grasslands now survive, but they support a diverse range of plants and important butterfly populations. An interesting aspect of Cambridgeshire was that it has recently been subject to an arable stewardship pilot project, which aims to promote good farm management practices on arable land that encourage biodiversity. Furthermore, there are plans being constructed in Cambridgeshire to create new areas of wetland in the county. In particular, the National Trust has visions to acquire 3700 Ha of land over the next 100 years to extend Wicken Fen. Finally, Cambridgeshire is also subject to considerable developmental pressure, with 70,000 new houses being currently planned to be built in the region. It is proposed that these three situations be used as a basis for the CV scenarios.

6.12.2. Biodiversity in Northumberland

Northumberland was chosen to represent a contrast to Cambridgeshire. Northumberland is an area that contains one of the widest ranges of wildlife habitats of any English county. Northumberland's coastal region is known for its long beaches backed by flower filled sand dunes and the colonies of nesting sea birds. The sand dune of the North East are dramatically coloured in purple, gold and white of the bloody geranium, Danish milk vetch, birds foot trefoil and the fragrant dwarf burnnet rose. The coast also provides the only regular wintering site in Britain for the pale bellied Brent goose, which flies here from Spitsbergen. It is also the most important site for Wigeon, with Eider, mallard and shelduck also present in high numbers. Away from the agricultural coastal plain and the Tyne Valley the county rises to the west becoming rough sheep grazing and grouse moor. The uplands are divided into black and white hills. In the north the dome of the Cheviot is white being grass dominated. Here, Cheviot sheep have long grazed the mosaic of blanket bog and *Molinia* grassland, while in a few places remnants of an alpine flora survive. These areas support several species of rare sphagnum moss. Further south the dark heather moorlands appear black. These are found over acidic sandstones and shales. Here and in the north Pennine fells, where blanket bogs cover limestone the hills provide important strongholds for many upland birds. In these Pennine valleys a history of metal mining activity has scared the landscape, but also produced valuable habitat for a range of rare species including orchids, the mountain pansy, spring sand worts and alpine penny cress that are associated with these toxic conditions.

Like most of lowland Britain the coastal plain of Northumberland has seen dramatic declines in biodiversity in recent decades resulting from agricultural intensification. Many ancient species rich meadows have been lost, but a few survive in urban fringes, on soils too thin to plough and along the Whin Sill (along which runs Hadrian's Wall). Unlike the rivers of Cambridgeshire, the many rivers of Northumberland are fast flowing and mostly free of agriculture-related eutrophication. They provide an important network of wildlife habitats for spawning salmon and trout and for many river birds and mammals.

Along the coastal strip and low lying areas of the Tyne valley, the conurbation of Newcastle has spread resulting in the loss of important wet grassland habitats.

7. Results

This research utilises two valuation methodologies to value biodiversity changes in Cambridgeshire and Northumberland: the contingent valuation method and the choice experiments method. The majority of data was collected during household interviews in the two counties, however, a series of valuation workshops were also undertaken in Northumberland. In this section, we report the findings from the household contingent valuation survey, the household choice experiment survey and the valuation workshops (sections 7.1, 7.2 and 7.3respectively). A discussion of these results is presented in Section 8.

7.1. Analysis of the main household Contingent Valuation study

Contingent valuation data were collected in both the household interview survey and the valuation workshop. The household interviews were conducted at two locations: Cambridgeshire and Northumberland, while the valuation workshops were restricted to Northumberland only.

In practical terms, CVM analysis requires the estimation of multiple regression equations where willingness-to-pay amounts (or "bids") are related statistically to a series of individual characteristics thought to influence the amount that they would pay. These equations not only allow one to comment on which factors are important but also to comment on their relative importance. For example, one would expect income to be of particular importance given it relates directly to an individual's "ability to pay". Likewise, one would expect that preferences about the importance of biodiversity change would vary by education, age and other socioeconomic characteristics.

More formally, the basic structure of the linear CV model used in our study is as follows:

$$WTPi = \alpha + \boldsymbol{\beta X}_j + \boldsymbol{\gamma Z}_i + \varepsilon_i$$

where: "*WTP*" is the amount individual "*i*" is willing to pay; "***X***" is a vector of variables describing a scenario of biodiversity characteristics; "***Z***" is a vector of socio-economic characteristics relating to the individual (and/or his/her household and/or his/her family); and "*ε*" is a random error term. The main parameters of the model, $\boldsymbol{\beta}$ and $\boldsymbol{\gamma}$, describe the marginal effects of the included variables on willingness-to-pay (i.e. $\delta WTP/\delta \boldsymbol{X}$ and $\delta WTP/\delta \boldsymbol{Z}$).

The Cambridgeshire household survey comprised 341 interviews. These interviews were split between three policy scenarios, with each respondent of the household interviews only receiving one of these three scenarios. The number of interviews addressing each scenario was:

- Agri-environmental scheme (124 interviews)
- Habitat re-creation scheme (107 interviews)
- Protect against development loss (110 interviews)

In Northumberland, 395 people were interviewed in the household survey. Interviews were split between two scenarios:

- Habitat re-creation (209 interviews)
- Protect against development loss (186 interviews)

Also in Northumberland, 53 people took part in the valuation workshops. Each workshop participant received both of the Northumberland CV scenarios:

- Habitat creation (53 interviews)
- Protect against development loss (53 interviews)

In what follows, we report the analysis of the contingent valuation data according to:

- Comparison of WTP results across case study locations.
- Comparison of WTP results across biodiversity policies.
- Comparison of WTP from the household study and valuation workshops

7.1.1. Comparison of CV mean WTP results across case study locations – household interviews

Table 8 reports the CV WTP bids for all three 'merged' policy scenarios for Cambridgeshire, Northumberland, and the total, pooled sample across both study areas. In other words, Table 8 shows what people in Cambridgeshire / Northumberland are WTP for *any* policy to increase biodiversity. As may be seen, about one-third of respondents had a WTP of zero[3], in other words, did not value these increases in biodiversity. Mean WTP is higher for Cambridgeshire respondents (£58.87) than for those from Northumberland (£42.47): this difference is statistically significant at the 95% level. Median WTP is considerably less than mean WTP in all cases, illustrating a common finding in CV studies.

Table 8: Summary WTP Measures for *any* policy improvement scenario

SITE	N	Mean	Standard Error	95% Confidence Interval	95% Trimmed Mean	Median	Percentage with WTP = 0
Cambridgeshire	*341*	*£58.87*	*£5.84*	*£47.38↔£70.36*	*£42.84*	*£20.00*	*32.3%*
Northumberland	*395*	*£42.47*	*£3.97*	*£34.67↔£50.27*	*£30.09*	*£10.00*	*35.9%*
Both	*736*	*£50.07*	*£3.45*	*£43.29↔£56.85*	*£35.81*	*£20.00*	*34.2%*
Notes: t-test for difference in means: *t=2.3* and *p=0.02*							

Table 9 and Table 10 refine that above analysis by comparing WTP between Cambridgeshire and Northumberland for the habitat re-creation and development loss policy scenarios separately. We see that the value of habitat re-creation is greater in Cambridgeshire than in Northumberland (Table 9), and that this relationship is maintained when one compares WTP for preventing biodiversity loss due to development (Table 10). However, in neither case are these differences significant.

[3] Note that protest responses were coded as zeros: these mean WTP figures are thus conservative estimates.

Table 9: Summary WTP Measures - Habitat Re-creation Only

SITE	N	Mean	Standard Error	95% Confidence Interval	95% Trimmed Mean	Median	Percentage with WTP = 0
Cambridgeshire	*107*	*£54.97*	*£6.56*	*£41.96↔£67.98*	*£48.42*	*£24.00*	*29.9%*
Northumberland	*209*	*£47.49*	*£5.98*	*£35.70↔£59.27*	*£34.35*	*£12.00*	*27.8%*
Both	*316*	*£50.02*	*£4.53*	*£41.10↔£58.94*	*£39.12*	*£20.00*	*28.5%*
Notes: t-test for difference in means: *t=0.80* and *p=0.40*							

Table 10: Summary WTP Measures: Development Loss Only

SITE	N	Mean	Standard Error	95% Confidence Interval	95% Trimmed Mean	Median	Percentage with WTP = 0
Cambridgeshire	*110*	*£45.30*	*£7.82*	*£29.80↔£60.80*	*£31.26*	*£16.00*	*37.3%*
Northumberland	*186*	*£36.84*	*£5.07*	*£26.82↔£46.85*	*£25.29*	*£3.00*	*45.2%*
Both	*296*	*£39.97*	*£4.31*	*£31.49↔£48.47*	*£27.27*	*£5.00*	*42.2%*
Notes: t-test for difference in means: *t=0.9* and *p=0.37*							

Table 11 shows the bid curves for all respondents of the main survey ('both') and separately for the Cambridgeshire and Northumberland sample. In the regression equation, the dependent variable is WTP, while the independent variables include:

- age
- education
- sex
- marital status
- household size
- number of children
- employment status
- whether income was reported or not
- household income, if reported
- whether the respondent is a member of an environmental group

These variables were selected for inclusion in the full model, based on variables which have been shown to matter in past studies; and on theoretical grounds (income, for example, has a key role in demand theory).

Taking the pooled sample first ("Both"), we note that higher levels of education and household income increase WTP. No other socio-economic variables are significant at 95%. The dummy variable for whether the response comes from the Cambridgeshire or Northumberland sample is also significant, and shows that, even when we control for socio-economic differences in sample characteristics, people in Cambridgeshire valued the biodiversity increases more than people in Northumberland. This backs up the conclusions reported above which compared simple mean WTP across the two samples. The regression is overall significant at the 95% level ($p < 0.1$).

Turning to the bid curves for the two regions separately, we see that:

For Cambridgeshire:

- higher education increases WTP
- higher income increases WTP

For Northumberland:

- A-level/college education also increases WTP, relative to having neither
- households of a larger overall size have a higher WTP
- part-time workers have a lower WTP
- but income is not significant

Note that in no sample is the parameter on "missing income" a significant determinant of WTP, implying that there is no systematic relationship between income non-reporting and valuation.

Table 11: WTP Equations - Cambridgeshire and Northumberland (Absolute values of t-statistics in parentheses)			
	(1)	**(2)**	**(3)**
SITE:	**Both**	**Cambridgeshire**	**Northumberland**
VARIABLE:			
Age:			
< 25	*-16.71* *[1.1]*	*-11.69* *[0.5]*	*-31.63* *[1.9]*
26-44	--	--	--
45-65	*6.94* *[0.9]*	*10.77* *[0.7]*	*3.08* *[0.3]*
> 65	*-10.18* *[0.8]*	*2.24* *[0.1]*	*-22.47* *[1.5]*
Education:			
School leaver	--	--	--
A-levels/college	*21.11* *[2.5]*	*12.22* *[0.8]*	*27.55* *[2.9]*
Higher	*31.98* *[3.2]*	*43.94* *[2.6]*	*22.70* *[1.9]*
Gender:			
Female	--	--	--
Male	*10.41* *[1.5]*	*18.62* *[1.5]*	*3.11* *[0.4]*
Marital Status:			
Other	--	--	--
Married/cohabitating	*-13.59* *[1.5]*	*-14.73* *[1.9]*	*-14.33* *[1.5]*
Household size:	*1.71* *[0.4]*	*-8.82* *[0.6]*	*14.11* *[2.6]*
Number of children:	*-1.14* *[0.2]*	*5.52* *[0.6]*	*-6.51* *[1.0]*
Employment:			
Not working	--	--	--
Part-time	*-6.74* *[0.7]*	*24.07* *[1.4]*	*-25.74* *[2.2]*
Full-time	*-5.86* *[0.7]*	*5.54* *[0.3]*	*-7.95* *[0.7]*
Missing income:			
Yes	--	--	--
No	*0.38*	*10.28*	*2.50*

	[0.1]	*[0.4]*	*[0.1]*
Income:	*$7.13x10^{-4}$* *[3.1]*	*$1.10x10^{-3}$* *[2.9]*	*$3.07x10^{-4}$* *[1.1]*
Member of Env Group:			
No	--	--	--
Yes	*9.90* *[1.2]*	*10.19* *[0.7]*	*7.48* *[0.8]*
Site:			
Northumberland	--	*NA*	*NA*
Cambridgeshire	*12.90* *[1.9]*	*NA*	*NA*
Constant	*10.45* *[0.5]*	*23.38* *[0.7]*	*2.83* *[0.1]*
R^2(%)	*7.6*	*11.2*	*10.0*
F= (p=)	*3.9 (<0.01)*	*2.9 (<0.01)*	*3.0(<0.01)*
Mean WTP	*£50.07*	*£58.87*	*£42.47*
N	*736*	*341*	*395*
Chow test: F= (p=)	--	*3.1 (<0.01)*	

The Chow test[4] is used to test for differences between the bid functions in the two case study areas. This test again demonstrates that for all policy scenarios the parameter values for Cambridgeshire were significantly different to those in Northumberland. However, as with the mean WTP analysis undertaken on Table 8, the analysis of Table 11 merged all policy scenarios together. Table 12 and Table 13 repeats the Chow test for the habitat re-creation and development loss scenarios separately. In Table 12, it can be seen that the Chow test shows the parameter values for Cambridgeshire are different to those for Northumberland for habitat recreation: the inverse demand curves for the same scheme differ significantly between the two regions. In Table 13, however, the Chow test cannot reject equivalence. Note here, though, that the overall explanatory power of the bid curves is low, due mainly to small sample sizes in these sub-samples.

[4] That is, we test H0: Beta (Cambridgeshire) – Beta (Northumberland)

Table 12: WTP Equations - Habitat Re-creation Cambridgeshire and Northumberland (Absolute values of t-statistics in parentheses)			
	(1)	**(2)**	**(3)**
SITE:	**Both**	**Cambridgeshire**	**Northumberland**
VARIABLE:			
Age:			
< 25	*-39.41* *[1.9]*	*-17.75* *[0.1]*	*-64.67* *[2.5]*
26-44	--	--	--
45-65	*-8.26* *[0.7]*	*1.77* *[0.1]*	*-15.94* *[1.1]*
> 65	*-32.52* *[1.9]*	*-0.13* *[0.1]*	*-45.02* *[2.0]*
Education:			
School leaver	--	--	--
A-levels/college	*20.17* *[1.8]*	*6.50* *[0.4]*	*36.27* *[2.4]*
Higher	*15.83* *[1.2]*	*9.04* *[0.4]*	*26.19* *[1.5]*
Gender:			
Female	--	--	--
Male	*-2.40* *[0.3]*	*-4.87* *[0.3]*	*-1.59* *[0.1]*
Marital Status:			
Other	--	--	--
Married/cohabitating	*3.93* *[0.3]*	*4.63* *[0.3]*	*-8.19* *[0.5]*
Household size:	*10.93* *[1.9]*	*-6.53* *[0.8]*	*19.76* *[2.6]*
Number of children:	*-6.83* *[1.0]*	*4.31* *[0.5]*	*-16.08* *[1.7]*
Employment:			
Not working	--	--	--
Part-time	*-30.43* *[2.3]*	*25.73* *[1.3]*	*-45.32* *[2.5]*
Full-time	*-5.85* *[0.5]*	*48.67* *[2.7]*	*-24.20* *[1.6]*
Missing income:			
Yes	--	--	--
No	*-15.24*	*12.57*	*-24.30*

	[0.8]	*[0.4]*	*[0.9]*
Income:	*5.15x10^{-4}* *[1.9]*	*6.73x10^{-4}* *[1.8]*	*3.27x10^{-4}* *[0.8]*
Member of Env Group:			
No	--	--	--
Yes	*15.32* *[1.5]*	*-12.49* *[0.8]*	*16.50* *[1.2]*
Site:			
Northumberland	--	*NA*	*NA*
Cambridgeshire	*3.46* *[0.4]*	*NA*	*NA*
Constant	*30.19* *[1.2]*	*5.58* *[0.1]*	*36.00* *[1.1]*
R^2(%)	*11.9*	*22.5*	*16.2*
F= (p=)	*2.7 (<0.01)*	*1.9 (0.04)*	*2.7(<0.01)*
Mean WTP	*£50.02*	*£54.97*	*£47.49*
N	*316*	*107*	*209*
Chow test: F= (p=)	--	*1.9 (0.03)*	

Table 13: WTP Equations - Development Loss Only

Cambridgeshire and Northumberland
(Absolute values of t-statistics in parentheses)

	(1)	(2)	(3)
SITE:	**Both**	**Cambridgeshire**	**Northumberland**
VARIABLE:			
Age:			
< 25	*8.68* *[0.5]*	*13.06* *[0.3]*	*7.22* *[0.3]*
26-44	--	--	--
45-65	*20.01* *[1.7]*	*6.63* *[0.3]*	*29.98* *[2.1]*
> 65	*21.40* *[1.3]*	*37.12* *[1.2]*	*13.26* *[0.7]*
Education:			
School leaver	--	--	--
A-levels/college	*25.99* *[2.4]*	*13.42* *[0.7]*	*24.41* *[2.0]*
Higher	*39.00* *[3.0]*	*29.91* *[1.3]*	*25.70* *[1.6]*
Gender:			
Female	--	--	--
Male	*1.88* *[0.2]*	*-3.59* *[0.2]*	*2.61* *[0.2]*
Marital Status:			
Other	--	--	--
Married/cohabitating	*-18.98* *[1.6]*	*-27.41* *[1.1]*	*-20.25* *[1.4]*
Household size:	*2.66* *[0.4]*	*4.17* *[0.3]*	*5.40* *[0.7]*
Number of children:	*0.11* *[0.1]*	*-9.12* *[0.6]*	*4.78* *[0.5]*
Employment:			
Not working	--	--	--
Part-time	*16.41* *[1.3]*	*37.10* *[1.6]*	*8.34* *[0.5]*
Full-time	*1.53* *[0.1]*	*-23.94* *[1.0]*	*19.98* *[1.3]*
Missing income:			
Yes	--	--	--
No	*22.95*	*17.55*	*20.26*

	[1.3]	*[0.6]*	*[0.9]*
Income:	*3.74x10^{-4}* *[1.2]*	*8.98x10^{-4}* *[1.3]*	*2.71x10^{-4}* *[0.8]*
Member of Env Group:			
No	--	--	--
Yes	*-3.56* *[0.4]*	*2.11* *[0.1]*	*-4.60* *[0.4]*
Site:			
Northumberland	--	*NA*	*NA*
Cambridgeshire	*7.17* *[0.8]*	*NA*	*NA*
Constant	*-20.90* *[0.8]*	*-0.58* *[0.1]*	*-28.66* *[0.9]*
R^2(%)	*8.8*	*17.3*	*10.8*
F= (p=)	*1.8 (0.04)*	*1.4 (0.16)*	*1.5 (0.12)*
Mean WTP	*£39.97*	*£45.30*	*£36.84*
N	*296*	*110*	*186*
Chow test: F= (p=)	--	*1.6 (0.11)*	

The conclusions drawn from the above analysis is that, overall, there were significant differences between both the mean WTP values and the bid curves (that is value functions) across case study locations. However, when the data was analysed according by policy scenario, no difference was found in the mean WTP values or between the bid curves for development loss. However, significant differences were found between the bid curves for the two case studies for the habitat re-creation scenario.

7.1.2. *Comparison of mean WTP results across policy scenarios*

Table 14 presents results for Cambridgeshire only, and studies how WTP varies by policy scenario. WTP is highest for agri-environmental schemes (£74.27), and lowest for preventing development loss (£45.30). Habitat re-creation is valued between these two. This is of general interest, since the theory of loss aversion suggests that losses are often valued more than gains. However, these changes are not symmetrical in our case. What is more, these mean values are not statistically different from each other at 95% (p= 0.11). Table 16 (explained below) may be compared with this result. Note that sample sizes in each of the three treatments are quite small (n = 124, 107, 110). In Table 15, this analysis is repeated for Northumberland, where WTP across the two scenarios used (habitat re-creation and development loss) is compared. WTP is higher for the former, but again this difference is not significant (p=0.18).

Table 14: Summary WTP Measures by Type of Policy scenario (Cambridgeshire Only)

SITE	N	Mean	Standard Error	95% Confidence Interval	95% Trimmed Mean	Median	Percentage with WTP = 0
Agri-environment schemes	*124*	*£74.27*	*£13.26*	*£48.03↔£100.51*	*£53.28*	*£24.00*	*29.8%*
Habitat creation scheme	*107*	*£54.97*	*£6.56*	*£41.96↔£67.98*	*£48.42*	*£24.00*	*29.9%*
Development loss	*110*	*£45.30*	*£7.82*	*£29.80↔£60.79*	*£31.26*	*£16.00*	*37.3%*
ALL	*341*	*£58.87*	*£5.84*	*£47.38↔£70.36*	*£42.84*	*£20.00*	*32.3%*
Notes: F-test for difference in means: *F=2.2* and *p=0.11*							

Table 15: Summary WTP Measures by Type of Policy scenario (Northumberland Only)

SITE	N	Mean	Standard Error	95% Confidence Interval	95% Trimmed Mean	Median	Percentage with WTP = 0
Habitat creation Scheme	*209*	*£47.49*	*£5.98*	*£35.70↔£59.27*	*£34.35*	*£12.00*	*27.8%*
Development loss	*186*	*£36.84*	*£5.07*	*£26.82↔£46.85*	*£25.29*	*£3.00*	*46.8%*
ALL	*395*	*£42.47*	*£3.97*	*£34.67↔£50.27*	*£30.09*	*£10.00*	*35.9%*
Notes: t-test for difference in means: *t=1.4* and *p=0.18*							

This can be further tested using bid curve analysis by incorporating dummy variables for which scenario bids were stated. This is done in Table 16. Looking first at the Cambridgeshire bid curve, we see that neither of the dummies is significant (here, development loss is the excluded category). This finding carries over to the Northumberland bid curve, where the parameter on habitat re-creation is also insignificant. This confirms the results of Table 14 and Table 15: preferences do not seem to vary significantly according to *how* biodiversity is preserved, only whether it is or not.

Table 16: WTP Equations - Programme Variables Included

Cambridgeshire and Northumberland
(Absolute values of t-statistics in parentheses)

SITE:	**Cambridgeshire**	**Northumberland**
VARIABLE:		
Age:		
< 25	*-15.97 [0.6]*	*-31.27 [1.9]*
26-44	--	--
45-65	*9.17 [0.6]*	*2.49 [0.2]*
> 65	*0.91 [0.1]*	*-22.86 [1.5]*
Education:		
School leaver	--	--
A-levels/college	*12.88 [1.9]*	*27.27 [2.9]*
Higher	*42.41 [2.5]*	*21.59 [1.8]*
Gender:		
Female	--	--
Male	*17.97 [1.5]*	*3.11 [0.4]*
Marital Status:		
Other	--	--
Married/cohabitating	*9.28 [0.6]*	*-14.89 [1.4]*
Household size:	*-13.94 [1.8]*	*13.66 [2.5]*
Number of children:	*4.31 [0.5]*	*-6.12 [0.9]*
Employment:		
Not working	--	--
Part-time	*23.76 [1.4]*	*-24.45 [2.2]*
Full-time	*5.28 [0.3]*	*-8.23 [0.8]*
Missing income		
Yes	--	--
No	*8.64 [0.4]*	*1.99 [0.1]*
Income:	$1.09x10^{-3}$ *[2.9]*	$3.15x10^{-4}$ *[1.2]*
Member of Env Group:		
No	--	--
Yes	*10.50 [0.8]*	*5.58 [0.8]*
Programme:		
Development loss	--	--
Agri-environment	*19.85 [1.4]*	*NA*

Habitat creation	*2.09 [0.1]*	*5.95 [0.8]*
Constant	*16.58 [0.5]*	*1.19 [0.1]*
R^2(%)	*11.8*	*10.2*
Mean WTP	*£58.87*	*£42.47*
N	*341*	*395*

The conclusions we can draw from the above is that neither mean WTP or bid curves differ across schemes to a significant degree. It thus appears that people care about increasing biodiversity, but not how this is achieved.

7.1.3. Comparison of CV data from the household study and valuation workshops

One recent methodological advance built into the study design was to use valuation workshops as a cross-check on the main household survey CV results. These workshops allow greater opportunity for discussion and reflection than surveys. It is important to note that in this instance we standardised information sets across the two treatments (both received the same PowerPoint presentation). Valuation workshops were run in Northumberland only. Table 17 compares the main results from the workshops with those from the household survey. As can be seen, WTP was higher in the valuation workshop than in the main survey (£50.33 versus £42.47, across both policy scenarios); but this difference is not statistically significant (p = 0.40). Interestingly, the variance in the workshop sample is higher than in the main survey: this is probably due to the smaller sample used in the workshop. Again, however, note the difference in sample size. The small sample size for the valuation workshops (n=106) reflects the high costs of collecting valuation data in this way.

Table 17: Summary WTP Measures for 'pooled' policy scenarios: Valuation Workshop versus Main Survey (Northumberland Only)

SITE	N	Mean	Standard Error	95% Confidence Interval	95% Trimmed Mean	Median	Percentage with WTP = 0
Valuation Workshop	*106*	*£50.33*	*£9.08*	*£32.32↔£68.34*	*£35.22*	*£10.00*	*33.0%*
Main survey	*395*	*£42.47*	*£3.97*	*£34.67↔£50.27*	*£30.09*	*£10.00*	*35.9%*
Both	*501*	*£44.14*	*£3.67*	*£36.92↔£51.34*	*£30.96*	*£10.00*	*35.3%*
Notes: t-test for difference in means: *t=0.8* and *p=0.40*							

Table 18 and Table 19 repeat the comparison of workshop WTP with main household survey WTP, but this time by policy scenario. The same qualitative relationship between the WTP samples is found for habitat re-creation as in Table 17, with valuation workshop WTP being greater than that in the main survey. The pattern is reversed with the development loss scenario. But neither of these differences are statistically significant.

Table 18: Summary WTP Measures for Habitat Re-creation scenario: Valuation Workshop versus Main Survey (Northumberland Only)

SITE	N	Mean	Standard Error	95% Confidence Interval	95% Trimmed Mean	Median	Percentage with WTP = 0
Valuation Workshop	*53*	*£68.72*	*£15.89*	*£36.83↔£100.60*	*£52.95*	*£20.00*	*28.3%*
Main survey	*209*	*£47.49*	*£5.98*	*£35.70↔£59.27*	*£34.35*	*£12.00*	*27.8%*
Both	*262*	*£51.78*	*£5.76*	*£40.44↔63.12*	*£36.79*	*£12.00*	*27.9%*
Notes: t-test for difference in means: *t=1.3* and *p=0.22*							

Table 19: Summary WTP Measures for Development Loss scenario: Valuation Workshop versus Main Survey (Northumberland Only)

SITE	N	Mean	Standard Error	95% Confidence Interval	95% Trimmed Mean	Median	Percentage with WTP = 0
Valuation Workshop	*53*	*£31.94*	*£8.23*	*£15.42↔£48.46*	*£22.57*	*£5.00*	*37.7%*
Main survey	*186*	*£36.84*	*£5.07*	*£26.82↔£46.85*	*£25.29*	*£3.00*	*46.8%*
Both	*239*	*£35.75*	*£4.34*	*£27.19↔44.31*	*£24.59*	*£3.00*	*43.5%*
Notes: t-test for difference in means: *t=0.5* and *p=0.61*							

Table 20 shows a comparison of values across the two policy scenarios for the valuation workshop. WTP is now statistically different between habitat creation and development loss, with that for the former being more than double the latter. However, the variance of bids for habitat re-creation is high, as may be seen in the very wide confidence interval. Median WTP is 4 times higher for the habitat creation scenario than the development loss case.

Table 20: Summary WTP Measures: Valuation Workshop Only (Northumberland Only)

SITE	N	Mean	Standard Error	95% Confidence Interval	95% Trimmed Mean	Median	Percentage with WTP = 0
Habitat creation	*53*	*£68.72*	*£15.89*	*£36.83↔£100.60*	*£52.95*	*£20.00*	*28.3%*
Development loss	*53*	*£31.94*	*£8.23*	*£15.42↔£48.46*	*£22.57*	*£5.00*	*37.7%*
Both	*106*	*£50.33*	*£9.08*	*£32.22↔68.341*	*£35.22*	*£10.00*	*33.0%*
Notes: t-test for difference in means: *t=2.1* and *p=0.04*							

Finally, Table 21 compares the bid curves for the main survey in Northumberland with those from the valuation workshop. It will be recalled that a simple comparison of mean values showed no significant differences between these two treatments, except in that respondents in the latter were more able to distinguish between schemes than those in the main survey. In Table 21, model (1) pools WTP values for all schemes and includes a variable for whether the response came from a valuation workshop or the main survey[5]. As may be seen, this dummy is insignificant (t=1.0), which again shows that there is no significant effect on mean WTP of obtaining data from the workshops relative to the main survey. Models (2) and (3) split responses by scenario. What emerges here is that the "habitat re-creation" model shows a valuation workshop effect which is significant at 90%, although not at 95%. The sign on this variable indicates that being in a valuation workshop increases WTP for habitat creation, with such respondents being WTP a predicted £25.66 extra compared to main survey respondents. No such effect is found for the development loss scenario. However, we stress that this valuation workshop effect on habitat creation values is not significant at the more typical hurdle of 95% significance.

[5] Small sample size for the valuation workshop means that estimating separate bid curves for this treatment is an inferior option.

Table 21: WTP Equations - Valuation Workshop and Main Survey

Northumberland Only
(Absolute values of t-statistics in parentheses)

	(1)	**(2)**	**(3)**
SITE:	**Both programmes**	**Habitat re-creation**	**Development loss**
VARIABLE:		**only**	**only**
Age:			
< 25	*-42.22* [3.0]	*-68.80* [0.3]	*-10.54* [0.6]
26-44	--	--	--
45-65	*1.3* [0.1]	*-15.99* [1.1]	*21.73* [1.8]
> 65	*-32.71* [2.2]	*-51.25* [2.2]	*-5.75* [0.3]
Education:			
School leaver	--	--	--
A-levels/college	*21.73* [2.4]	*33.12* [2.4]	*12.90* [1.2]
Higher	*13.03* [1.2]	*17.73* [1.1]	*11.34* [0.8]
Gender:			
Female	--	--	--
Male	*0.31* [0.4]	*-4.36* [0.4]	*0.99* [0.1]
Marital Status:			
Other	--	--	--
Married/cohabitating	*-16.03* [1.7]	*-9.22* [0.6]	*-20.23* [1.7]
Household size:	*12.90* [2.8]	*18.37* [2.7]	*4.65* [0.8]
Number of children:	*-5.00* [0.9]	*-11.92* [1.4]	*1.94* [0.3]
Employment:			
Not working	--	--	--
Part-time	*-32.43* [3.0]	*-49.23* [2.9]	*-4.90* [0.4]
Full-time	*-18.00* [1.8]	*-37.84* [2.6]	*10.25* [0.8]
Missing income:			
Yes	--	--	--
No	*6.97*	*-6.04*	*12.91*

	[0.5]	*[0.3]*	*[0.7]*
Income:	*$3.21x10^{-4}$* *[1.3]*	*$3.22x10^{-4}$* *[0.8]*	*$3.25x10^{-4}$* *[1.0]*
Member of Env Group:			
No	--	--	--
Yes	*15.97* *[1.9]*	*24.05* *[1.9]*	*6.45* *[0.6]*
Sample:			
Main survey	--	--	--
Valuation Workshop	*9.48* *[1.0]*	*25.66* *[1.7]*	*-5.84* *[0.5]*
Constant	*12.81* *[0.6]*	*30.94* *[1.0]*	*-0.39* *[0.1]*
R^2(%)	*9.6*	*16.5*	*7.6*
F= (p=)	*3.4 (<0.01)*	*3.2 (<0.01)*	*1.2(<0.26)*
Mean WTP	*£44.14*	*£51.78*	*£35.75*
N	*501*	*261*	*239*

Summarising these comparisons, it appears that the main survey results are confirmed by the valuation workshop results, in that no significant differences between the two treatments were found. However, one important difference is that workshop participants did differentiate significantly between the two policy scenarios (habitat re-creation and development loss), where this did not happen in the main survey. Although it is unclear as to what the reason for this may be, it is suggested that this may reflect a greater variance in WTP bids in the workshop sample than in the household survey (which reflects the smaller sample used in the workshop)..

7.2. Choice Experiment Results

The second methodology utilised was choice experiments. In the CE study, respondents were presented with a series of choice tasks in which they were asked to choose their preferred policy option from a list of three options; one of which included the status quo. Each policy option was described in terms of a bundle of four biodiversity attributes plus the price (tax) attribute, where each policy option was presented at various levels according to an orthogonal experimental design. The four biodiversity attributes (each with three levels of provision) used in the choice experiment were:

- Familiar species of wildlife: continued decline; protect rare familiar species from further decline; protect rare and common familiar species from continued decline
- Rare, unfamiliar species of wildlife: continued decline; slowing down the rate of decline; stopping decline and ensuring recovery
- Species interactions within a habitat: wildlife habitats continue to be degraded and lost; habitat restoration; habitat re-creation
- Ecosystem processes: continued decline in functioning; only services with direct impact on humans restored; all ecosystem services restored.
- In addition, a price term was included, being an annual tax increase to pay for these policies. Six levels were used: no increase, £10, £25, £100, £260, £520.

In the main household survey, 741 respondents (343 in Cambridgeshire and 398 in Northumberland) each undertook five choice tasks. In the valuation workshop, 53 respondents undertook five choice tasks before the discussion and five choice tasks after the discussion.

The analysis of respondent choices was based on random utility theory (RUT). According to RUT, the respondent's utility function is comprised a deterministic, observable component and a random, unobservable component (Hanemann *et al.*, 1994). It is important to point out that the respondent has full knowledge of their utility function. Utility is only random from the point of view of the researcher. Let the utility of alternative i from choice set C be represented by

$$U_i = V_i + \varepsilon_i \tag{1.1}$$

where U_i represents the utility of choosing alternative i, V_i represents the deterministic component, and ε_i represents the random error term. Note that the choice set C comprises three alternatives (Choice A, Choice B and the status quo). The selection of alternative i implies that the utility of alternative i is greater than the utility of any other alternative. Thus, the probability of an individual choosing alternative i can be expressed as

$$\begin{aligned} \Pr\left[i \mid C\right] &= \Pr\left[U_i > U_j\right], \forall j \in C \\ &= \Pr\left[\left(V_i + \varepsilon_i\right) > \left(V_j + \varepsilon_j\right)\right] \\ &= \Pr\left[\left(V_i - V_j\right) > \xi\right], \end{aligned} \tag{1.2}$$

where $\xi = \varepsilon_j - \varepsilon_i$. By assuming that the error term, ξ, is distributed according to a double log (Gumbel) distribution, the probability of choosing alternative i can be expressed as

$$\Pr\left[i \mid C\right] = \frac{\exp\left(\mu V_i\right)}{\sum_{j \in C} \exp\left(\mu V_j\right)}, \tag{1.3}$$

where μ represents a scale factor; which we assume to equal one.

In the CE study, where there were three choice alternatives, the choice probabilities have a convenient closed-form solution known as the conditional logit model. The conditional logit model is structured such that the probability of choosing alternative i depends on the utility of that alternative relative to the utility of all other alternatives. The CE utility function (Equation 1.5) represents the utility of the different options in the conditional logit model and in its basic format comprises the attributes of the policy option, as well as the bid and the intercept. Thus, the CE utility function can be expressed as

$$V_i = \alpha_i + \beta\left(Y - Tax_i\right) + \gamma\left(Z_i\right) \tag{1.5}$$

where $i = 1,...,N$ indexes options available, α_i is an alternative specific constant that captures the effect of systematic but unobservable factors on the respondent's choice, Y is income, and β represents a parameter, Z_i is composed of variables measuring attributes of choice site and γ represents its parameter.

Welfare estimates in the form of compensating surplus can be derived from both the conditional logit model using the following formula

$$CS = -\frac{1}{\beta_M}\left[In\left(\sum_i \exp(V_0)\right) - In\left(\sum_i \exp(V_1)\right)\right], \tag{1.6}$$

where β_M is the marginal utility of income (assumed to be equal to the negative of the coefficient of the monetary variable); V_0 and V_1 represents the indirect utility functions before and after the change under consideration. Equation (1.6) can be used to estimate the compensating surplus associated with changes in quality of environmental goods where there are multiple sites. However, the choice set usually only includes a single change in a policy option. In such situations, equation (1.6) may be reduced to

$$CS = -\frac{1}{\beta_M}\left(V_0 - V_1\right). \tag{1.7}$$

A further reduction is possible if the marginal value of a change with a single attribute is estimated. This implicit price (which is sometimes referred to as the part-worth) can be estimated as a ratio of coefficients

$$IP = -\frac{\beta_{Attribute}}{\beta_M}. \tag{1.8}$$

7.2.1. Choice experiment results

Table 22 shows results from the choice experiment data for both Cambridgeshire and Northumberland, based on a conditional logit model. The pseudo-R2 value is higher for the latter sample, and is very close to the 20% level suggested by Louviere *et al.* (2000) as indicating a very good fit in this kind of data.

The Cambridgeshire model shows significant estimates for all the attribute parameters. In almost all cases, parameter signs are in accord with *a priori* expectations. As may be seen, improving the familiar species attribute from continued decline to either protecting rare species only '*familiar species (rare)*'or protecting all species '*familiar species (rare and common)*' increases utility; moving the habitat attribute from continued decline to habitat restoration '*habitat (restoration)*' or habitat recreation '*habitat (recreation)*' is positively valued; moving the ecosystem processes attribute from continued decline to a recovery of either directly-relevant services alone '*ecosystem (human)*' or all ecosystem processes '*ecosystem (all)*' creates higher utility. The only exception is for the rare, unfamiliar species attribute. Here, although a move from continued decline to stopping decline and ensuring recovery '*rare unfamiliar species (recovery)*' increases well-being, a move to slowing decline '*rare unfamiliar species (slow down)*' is negatively valued. All tax increases reduce utility, as expected.

For Northumberland, the same pattern is repeated, except that the '*Ecosystem (all)*' and '*Rare unfamiliar species (slow down)*' attribute levels are not significant. This means that any biodiversity improvements in the habitat or familiar species attributes are positively and significantly valued, as is an improvement in directly-relevant *ecosystem processes 'ecosystem (human)'* - although not an improvement in all ecosystem processes '*ecosystem (all)*'services. This implies the Northumberland group only cared about ecosystem processes that directly impact on their well-being. The Northumberland group also had a negative value for '*rare unfamiliar (slow down*)', but since this estimate is insignificant, this is unimportant.

Table 22: Logit models for Cambridge and Northumberland CE samples

Cambridgeshire

Attribute	*Parameter estimate*	*t-value*
FAMILIAR SPECIES (RARE)	0.126	2.1
FAMILIAR SPECIES (RARE + COMMON)	0.331	5.2
RARE UNFAMILIAR SPECIES (SLOW DOWN)	-0.165	-3
RARE UNFAMILIAR SPECIES (RECOVER)	0.408	5.7
HABITAT (RESTORATION)	0.122	2.3
HABITAT (CREATION)	0.217	3.5
ECOSYSTEM (HUMAN)	0.19	3.2
ECOSYSTEM (ALL)	0.15	2.2
PRICE	-0.004	-15.2
Pseudo R^2	14%	
N (Individuals)	343	

Northumberland

Attribute	*Parameter estimate*	*t-value*
FAMILIAR SPECIES (RARE)	0.309	5.1
FAMILIAR SPECIES (RARE + COMMON)	0.334	5.2
RARE UNFAMILIAR SPECIES (SLOW DOWN)	-0.08	-1.5
RARE UNFAMILIAR SPECIES (RECOVER)	0.645	8.1
HABITAT (RESTORATION)	0.243	4.7
HABITAT (CREATION)	0.253	4.3
ECOSYSTEM (HUMAN)	0.359	5.9
ECOSYSTEM (ALL)	0.064	1
PRICE	-0.003	-15.3
Pseudo R^2	19%	
N (Individuals)	398	

The statistical equivalence of the parameter estimates of the two models can be compared using a Likelihood Ratio test. The probability value for this test is < 0.01, indicating that the models are different. In other words, the valuation of biodiversity attributes varies significantly between the two samples.

7.2.2. Implicit prices for biodiversity attributes

Table 23 shows the implicit prices estimated from the logit model results in Table 22. These implicit prices show the marginal WTP on average of moving from one level - the excluded level, which in our case is always the worst-case, do nothing level - to a higher level. For example, the value of £35.65 for '*Familiar species (rare)*' for Cambridgeshire means that people were on average willing to pay £35.65 extra per year in higher taxes to move from continued decline in familiar species to a situation where rare, familiar species are protected from further decline. These are "ceteris paribus" values, so should be treated with care in a cost-benefit context. We can see from Table 23 that a scale effect is present in almost all cases for Cambridgeshire, meaning that higher levels of protection are valued more highly for each attribute, with the exception of the odd result on '*Rare unfamiliar species (slow down)*', and in the case of '*Ecosystem (all)*', where the value of protecting only directly-relevant ecosystem processes is higher than that of protecting all. The highest benefits in per-person terms come from ensuring the recovery of rare, unfamiliar species.

For Northumberland, the implicit prices for '*Rare unfamiliar species (slow down)*' and '*Ecosystem (all)*' are omitted, since the parameter estimates were not significantly different from zero. Furthermore, there was little evidence that the Northumberland sample considered the scale effects between the levels of the familiar species and for habitat attribute. Highest WTP is associated with ensuring the recovery of rare, unfamiliar species - the same result as for Cambridgeshire.

Table 23: Implicit prices for Cambridge and Northumberland CE samples

Cambridgeshire

Attribute	*Implicit Price*	*SE*	*95%lower*	*95%upper*
FAMILIAR SPECIES (RARE)	35.65	17.19	1.95	69.34
FAMILIAR SPECIES (RARE + COMMON)	93.49	18.03	58.15	128.82
RARE UNFAMILIAR SPECIES (SLOW DOWN)	-46.68	15.88	-77.80	-15.55
RARE UNFAMILIAR SPECIES (RECOVER)	115.13	21.22	73.53	156.72
HABITAT (RESTORATION)	34.4	15.32	4.37	64.42
HABITAT (CREATION)	61.36	17.52	27.02	95.69
ECOSYSTEM (HUMAN)	53.62	16.97	20.35	86.88
ECOSYSTEM (ALL)	42.21	19.23	4.51	79.90

Northumberland

	Implicit Price	*SE*	*95%lower*	*95%upper*
FAMILIAR SPECIES (RARE)	90.59	19.24	52.87	128.30
FAMILIAR SPECIES (RARE + COMMON)	97.71	18.47	61.50	133.91
RARE UNFAMILIAR SPECIES (SLOW DOWN)	n/a			
RARE UNFAMILIAR SPECIES (RECOVER)	189.05	25.28	139.50	238.59
HABITAT (RESTORATION)	71.15	16.29	39.22	103.07
HABITAT (CREATION)	74	17.51	39.68	108.31
ECOSYSTEM (HUMAN)	105.22	17.7	70.52	139.91
ECOSYSTEM (ALL)	n/a			

7.3. Valuation workshop results

The third methodology adopted was the valuation workshops. The structure of the workshops was based around the main survey; i.e. workshop participants were presented with exactly the same information as respondents of the main survey and were also asked to complete the same five choice experiment choice tasks and two CV valuations. However, workshop participants were also asked to further discuss and reflect on the descriptions of biodiversity and the choice tasks before they were asked to repeat the choice experiment exercise.

In total, 53 people participated in the workshops. The workshops were undertaken at three locations in Northumberland: two were held in Morpeth, two in Alnwick and two in Hexham. These locations were chosen to reflect both rural and urban areas within Northumberland. Comparison of workshop participants with respondent of the main Northumberland survey demonstrated consistency in the socio-economic characteristics between both samples.

7.3.1. Analysis of participants understanding of biodiversity concepts

Workshop participants were asked to indicate, on a likert scale, their level of understanding of biodiversity concepts both before and after the period of discussion and reflection. In particularly, they were asked to consider their level of understanding of (i) the choice experiment biodiversity attributes (ii) the levels of provision of the choice experiment biodiversity attributes and (iii) the contingent valuation policy scenarios. Table 24 summaries the main findings from this self assessment exercise. Within this Table, higher values reflect a greater level of understanding. General observations from these results indicate that the 'ecosystem processes' attribute was less well understood (both before and after the discussions) than the other biodiversity attributes presented in the choice experiment. Also, participants indicated that they generally understood the descriptions of the biodiversity attributes more than the descriptions of the levels of the attributes. Finally, levels of understanding of the CV policies were generally lower than that of the choice experiment attributes. Assuming that the midway point in the likert scale (i.e. '3') represents a reasonable level of understanding of the biodiversity concepts, we can conclude that all aspects of the descriptions of biodiversity (other than the 'ecosystem processes' attribute and attribute level before the discussion period) were, on average, reasonably well understood. On this basis, it should also be noted that the 'ecosystem processes' attributes was also reasonably well understood after the discussion period. One further important finding from this exercise is that the opportunity to discuss and reflect on the biodiversity concepts increased participants understanding of these concepts in *all* cases. Thus, workshop participant's second set of choice experiment choices would have been based on a better level of understanding of biodiversity than what their first set of choice task. Finally, it should be noted that the figures reported in Table 24 reflect mean scores. Examination of the actual responses on the likert scale provides a further insight into changes in levels of understanding of the biodiversity concepts following the discussion period. For example, 15% of participants indicated that their level of understanding of the familiar species attribute was either scored '1' or '2' on the likert scale before the discussions, and that following the discussion all participant indicated a level of understanding above '3'. In contrast, 34% of participants indicated a score of '1' or '2' for the ecosystem processes attribute before the discussion, while 10% remained at '2' or below after the discussion.

Table 24: Analysis of level of understanding of biodiversity (Before and After discussion)

Understanding of CE attribute	*(Before)*	*(After)*
Familiar species	3.42	4.02
Rare unfamiliar species	3.38	4.00
Habitat	3.51	3.96
Ecosystem services	2.64	3.32
Understanding of CE attribute levels		
Familiar species	3.13	3.83
Rare unfamiliar species	3.11	3.68
Habitat	3.23	3.81
Ecosystem services	2.51	3.25
Understanding of CV policies		
Habitat re-creation	3.08	3.75
Development loss	3.30	3.79

NB: The figures in the above table relate to mean scores, where the scores ranged from 1 (poor understanding) to 5 (good understanding)

Workshop participants were also provided with the opportunity to openly discuss the biodiversity concepts. It was apparent from these discussions that participants were familiar with the concepts of ‘familiar species’, ‘rare species’ and ‘habitats’. Several participants commented that the PowerPoint presentation simply gave more precise definitions to these concepts. A small number of participants, however, stated that they had difficulty with the concept of ‘ecosystem processes’. One participant commented that she had *"...never thought…well, I knew about or at least I've heard about things like the greenhouse effect but you don't know much do you? But when you start to think about the local services like you mentioned…well its all part of it. I feel that I need to know more...It's all linked and too big to be able to decide properly”.* Another participant admitted her lack of understanding of the ‘ecosystem processes’ attribute had led her to ignoring that attribute in her policy choices. Her choices were instead based on the familiar and, to her, more understandable species and habitat attributes. This finding might explain why the ‘ecosystem processes’ attribute was found to be insignificant in the Northumberland choice model (see Table 23 in Section 7.2.2). These discussions support the perception that the level of understanding of ecosystem processes was generally lower than for the other attributes. It was also clear from these discussions that the actual level of understanding of biodiversity concepts closely reflected the stated levels of understanding reported in Table 24 above.

In the workshop discussions, participants were asked whether the information presented influenced anyone's choice. One response to this question was "*you've told us nothing we didn't already know*". He went on to comment that the presentation simply "*told us where things fit…what belongs in which group; rabbits in 'familiar' and such like*". This theme was often repeated; the information presented was a clarification of definitions and language for concepts already known to the participants.

One further issue which arose from the discussion was a general feeling that much of Northumberland had escaped the industrialisation of neighbouring areas (e.g. shipbuilding along the Tyne, coal mining in the east of the county). As a result of this, the prevailing attitude of participants was that Northumberland’s ecosystems were generally healthy. This fact might also contribute to the reason why the ‘ecosystem services (all)’ attribute in the choice experiment was insignificant.

7.3.2. Analysis of how participants made their choices in the choice experiment.

Workshop participants were also asked to consider and then discuss the choice making strategy they used to decide their preferred policy option in the choice experiment. Three issues were addressed here: how respondents considered the various attributes in the choice task (Question G-2), how respondents considered the ‘price’ attribute (Question G-3) and how respondents considered the actual level of the ‘price’ attribute (Question G-4).

In response to the Question G-2, just under half of the respondents (43.4%) indicated that they considered all attributes in the choice task, while 47.2% indicated that they considered only some of the attributes (Table 25). Five percent of respondents stated that they chose the cheapest option. Noteworthy, are the findings that none of the responses stated that they only considered one attribute, and none stated that they chose options at random. These findings indicate that the majority of participants made some form of considered compensatory choice.

Table 25: Choice making strategy: level of consideration of choice experiment attributes

Choice strategy	*% of responses*
1 Consider all attributes equally	43.4
2 Consider only some attributes	47.2
3 Consider only one attribute	0.0
4 Chose cheapest option	5.7
5 Chose at random	0.0
0 Not answered	3.8

Participants were then asked (Question G-3) to reflect on how they consider the ‘Price’ attribute when they made their choices (Table 26). Encouragingly, from the perspective of methodological validity, over 75% of participants claim to have examined and evaluated the price attribute; 41.5% rejected policies where the price was considered too high and 34% thought the price amount important. A further 7.5% chose the lowest price level which, although it may indicate consideration of the price amount in a modification of the compensatory choice, points to consideration of policy cost. A substantial minority (13.2%) claim not to have considered price in their choice of policy.

Table 26: Choice making strategy: level of consideration of the ‘price’ attribute.

Choice strategy	*% of responses*
1 Didn't consider the price attribute	13.2
2 Chose lowest price level	7.5
3 Reject policies where the price was too high	41.5
4 Price amount was important consideration	34.0
5 Not answered	3.8

Finally, participants were asked to indicate how they considered the actual size of price attribute when making their choices (Table 27). Almost half of respondents (49.1%) claim to have made full consideration of the price level. Again, it is encouraging that few gave little consideration to the price level (Response code 3). However, 30.2% of respondents considered the price amount but not whether they could afford it.

Table 27: Choice making strategy: level of consideration of level of the ‘price’ attribute.

Choice strategy	*% of responses*
1 Fully considered what I would have to forego in order to pay	49.1
2 Considered the price amount but didn't really consider whether I could afford it	30.2
3 Didn't consider if I would be able to afford it	13.2
4 Not answered	7.5

Workshop participants were then provided with an opportunity to discuss their choice making strategies. It was apparent from these discussions that a number of choice strategies existed.

First, it was clear that many of the participants imposed a ceiling on the price which they would be willing to pay for biodiversity enhancements. For example, several participants commented that they did not even inspect the attributes of a policy option which they considered beyond their price limit. Others, however, claimed greater flexibility to their limit in that they would inspect policy options where the price attribute was high, but would seek additional biodiversity benefits for the extra cost.

The second element of choice strategy was concerned with the way in which the range of attributes was considered. Two strategies were identified: fully compensatory choice, and limited choice strategy. A fully compensatory choice making strategy was claimed by over 40% of respondents (i.e. they considered all attributes when deciding their preferred option). Two approaches to the fully compensatory choice strategy were identified. In the first, participant would order the attributes according to their perceived importance, and then evaluate the policies with reference to this lexicon. Thus, these participants appeared to have a clear set of preferences for the alternative biodiversity attributes. The second fully compensatory approach involved participants considering all attributes to be of approximately equal worth. Thus, when faced with the choice between the policy options, participants would *"add up the benefits; more benefits equals best policy”*. In this case, participants appeared not to have preferences for individual biodiversity attributes, but rather they were aiming to maximise the overall biodiversity benefits.

In the limited choice strategy, participants would base their choice on a selected subset of the biodiversity attributes. Participants indicated that the subset would be selected to exclude those attributes which they did not consider to be important or (in two cases) those attributes they did not fully understand (i.e. ecosystem processes). Just under half of the participants indicated to use a limited choice strategy. This finding may explain the reason why the '*ecosystem processes (all)*' attribute was insignificant. It is useful to note that workshop participants indicated that they applied a consistent choice method over all five policy choices.

7.3.3. Choice experiment: comparison of main study and valuation workshop

In order to assess the impact that the opportunity to further discuss and reflect on the information presented and the discussion on the choice task had on respondents willingness to pay for biodiversity attributes, workshop participants were asked to complete two sets of choice experiment exercise; before and after the discussion. Table 28 reports these two workshop choice experiment models (we label the one near the outset of the workshop, after receiving the same information as the main survey participants as 'Before', and one near the end, having had a chance to discuss and reflect on the issues further 'After'), along with the choice model from the main household survey. The first issue to note regarding the workshop choice models is that neither model fits very well due to the small sample size. However, we can note that the number of significant variables in the models increases from 3 before the discussion to 7 after the discussion. Also, the overall fit of the models also improves following the discussion. In other words, a learning effect seems to be present. Looking at the second choice model ('After'), we see that it compares quite well with the main household survey CE results for Northumberland, with only '*Rare unfamiliar species (slow down)*' having a negative sign, and with '*Ecosystem (all)*' still being insignificant. The workshop choices also show the '*habitat (restoration)*' attribute to have an insignificant effect on utility. Implicit prices are also very similar, with a complete recovery of rare, unfamiliar species having the highest welfare gain. Finally, we note that a formal LR test shows that the parameters of the main survey CE model for Northumberland are not significantly different than either the 'Before' or 'After' models from the valuation workshops. In this sense, the valuation workshops provide a similar support for the main household survey choice experiment results as was the case for contingent valuation.
`

Table 28: Choice experiment results: workshop versus main survey, Northumberland

	Main Survey		Workshop: 'Before'		Workshop: 'After'	
ATTRIBUTE	***Parameter***	***t-statistic***	***Parameter***	***t-statistic***	***Parameter***	***t-statistic***
FAMILIAR SPECIES (RARE)	*0.309*	*5.1*	*0.172*	*1.1*	*0.327*	*2.0*
FAMILIAR SPECIES (RARE + COMMON)	*0.334*	*5.2*	*0.257*	*1.6*	*0.343*	*2.0*
RARE UNFAMILIAR SPECIES (SLOW DOWN)	*-0.080*	*-1.5*	*-0.028*	*-0.2*	*-0.316*	*-2.1*
RARE UNFAMILIAR SPECIES (RECOVER)	*0.645*	*8.1*	*0.166*	*0.8*	*0.654*	*3.0*
HABITAT (RESTORATION)	*0.243*	*4.7*	*0.093*	*0.7*	*0.149*	*1.1*
HABITAT (CREATION)	*0.253*	*4.3*	*0.323*	*2.0*	*0.332*	*2.0*
ECOSYSTEM (HUMAN)	*0.359*	*5.9*	*0.386*	*2.4*	*0.319*	*2.0*
ECOSYSTEM (ALL)	*0.064*	*1.0*	*0.116*	*0.6*	*0.211*	*1.2*
TAX	*-0.003*	*-15.3*	*-0.004*	*-6.2*	*-0.004*	*-5.8*
A_OPTA	*-0.012*	*-0.1*	*0.823*	*2.3*	*-0.295*	*-0.8*
A_OPTB	*-0.205*	*-1.5*	*0.894*	*2.4*	*-0.081*	*-0.2*
-2*lnL	*3172.6*		*417.4*		*440.7*	
p-value	*<0.01*		*<0.01*		*<0.01*	
Pseudo R^2	*19.2%*		*16.7%*		*18.7%*	
N (Individuals)	*398*		*53*		*53*	

8. Discussion

In Section 7 above, we presented the analysis of the contingent valuation and choice experiment data. In the following section we discuss the implications of this analysis. In particular we address the following questions:

- Do members of the public value the protection and enhancement of biodiversity?
- If so, what aspects of biodiversity do the public value most?
- How robust are our value estimates?
- Can our benefit estimates be transferred to value other situations?

8.1. Do members of the public value protection and enhancement of biodiversity?

Perhaps one of the key questions of interest to policy makers is 'Do members of the public have positive value preferences for biodiversity protection and enhancement'? In other words, does the evidence from the empirical work undertaken in this research support the thesis that the public value biodiversity (as opposed to having zero or negative value preferences). We address this issue by examining evidence from the contingent valuation study and then from the choice experiment study.

8.1.1. Evidence from the CV study to support the thesis that the public do value biodiversity.

The first source of evidence relating to whether the public value biodiversity comes from data on the proportion of respondents that indicated they would be willing to pay *something* towards biodiversity enhancements. This can be addressed by reference to Table 29 below, which reports the responses to Question D-1 of the household study which asked respondents to state '*Would your household be prepared to pay extra tax to contribute towards this policy to improve ... biodiversity?'* Two thirds of the CV respondents stated that they would be willing to pay some amount towards *any* of the biodiversity enhancements scenarios; this finding was consistent between the two case study areas. There were, however, slight differences between the proportions of respondents stating a willingness to pay towards the different policy scenarios. Approximately 73% of respondents stated that they would be willing to pay towards the agri-environmental scheme and habitat re-creation scenarios, while 58% stated a willingness to pay to avoid biodiversity losses due to development. The second piece of evidence comes from the mean WTP values reported in Table 8. Here, we see that mean WTP for any biodiversity policy scenario was £58.87 and £42.47 for Cambridgeshire and Northumberland respectively. Furthermore, these values were significantly different from zero. Thus, overall, the evidence from the CV household survey indicates that the majority of respondents were willing to contribute some positive amount towards biodiversity enhancement and protection policies.

Table 29: Proportion of household CV respondents stating that they would be willing to pay towards biodiversity.

Area		**CV scenario**			
		Agri-environmental scheme	**Habitat re-creation**	**Development loss**	**All scenarios**
Cambridge	% WTP	70.16	70.09	62.73	*67.74*
	% Not WTP	29.8	29.9	37.3	*32.3*
	n	*124*	*107*	*110*	*341*
Northumberland	% WTP		73.68	56.45	*65.57*
	% Not WTP		26.32	43.55	*34.43*
	n		*209*	*186*	*395*
Both areas	***% WTP***	**70.16**	**72.47**	**58.78**	***66.58***
	% Not WTP	**29.84**	**27.53**	**41.22**	***33.42***
	n	***124***	***316***	***296***	***736***

It is also useful to examine the reasons why respondents stated a willingness to contribute towards biodiversity improvement policies. Such data are reported in Table 30. Within this Table, reasons 1, 2 and 4 were included to indicate a considered response to the WTP question. Overall, 84% of respondents stated a considered response to the CV WTP question. Within this, 7.8% chose reason 1 (reflecting a likely use value), 17.2% chose reason 2 (reflecting a passive-use value), and 59.5% chose reason 4 (reflecting the amount that respondents consider to be reasonable for this type of policy). Reasons 3 and 5 were included to identify situations where respondents were not basing their value judgement on the likely utility gains. Seven percent (6.9%) stated reason 4 (which reflected the costs of the improvements) and 3.9% chose reason 6 (picked a value at random). Overall, these findings suggest that the majority of CV respondents made considered responses to the CV valuation question.

Table 30: Stated reasons why CV respondents were WTP towards biodiversity scenarios

Reason for stating WTP	**CV scenario**			
	Agri-environmental scheme	**Habitat re-creation**	**Development loss**	**All scenarios**
1: The amount I indicated reflects the benefits that I would gain from seeing increased biodiversity	6.9 %	7.1 %	9.2 %	**7.8 %**
2: The amount I indicated reflects the benefits that I would gain from knowing that the level of biodiversity had increased, even although I am unlikely to see it first hand.	21.8 %	18.1 %	13.8 %	**17.2 %**
3: The value reflects how much I think it would cost to improve biodiversity	5.7 %	5.7 %	9.2 %	**6.9 %**
4: The amount I indicated seems a reasonable amount to pay towards this type of policy	54.0 %	61.5 %	59.8 %	**59.5 %**
5: I just picked a value at random.	8.0 %	1.8 %	4.6 %	**3.9 %**
6: Other	3.4 %	5.7 %	3.4 %	**4.5 %**
Number of respondents	*87*	*226*	*174*	***487***

It is also interesting to note the reasons why household CV respondents stated that they were not willing to contribute towards a biodiversity policy. This information is presented in Table 31 below. Reasons 1, 2, 3 and 4 were included to represent genuine zero bids. Overall, 43.4% of respondents stated a zero response reason; the majority of which (29.8%) stated that they could not afford to pay towards biodiversity improvements. Reasons 5 and 6 reflected protest bids. Overall, 38.4% of responses were coded as protest bids. The majority of these (25.6%) stated that the costs of improving biodiversity should be paid by those that contribute to biodiversity loss, while 12.8% stated that the costs should not be paid through increased taxation.

Table 31: Stated reasons why CV respondents were NOT WTP towards biodiversity scenarios

Reason for not stating WTP	CV scenario			
	Agri-environmental scheme	Habitat re-creation	Development loss	All scenarios
1 Biodiversity policies are not a good use of my money.	16.2 %	2.3 %	2.5 %	4.5 %
2 I do not think that there is a need to improve the county's biodiversity	-	2.3 %	5.0 %	3.3 %
3 I can not afford to pay towards biodiversity policies	24.3 %	38.4 %	25.2 %	29.8 %
4 I already contribute towards improving biodiversity in other ways	8.1 %	5.8 %	5.0 %	5.8 %
5 I would be prepared to contribute towards improving biodiversity, but not by paying more tax	24.3 %	12.8 %	9.2 %	12.8 %
6 The costs of improving biodiversity should be paid by those that contribute to biodiversity loss	16.2 %	20.9 %	31.9 %	25.6 %
7 Other reason	10.8 %	17.4 %	21.0 %	18.2 %
Number of respondents	33	71	94	198

8.1.2. Are choice experiment respondents willing to pay anything towards biodiversity enhancement scenarios?

The above analysis was repeated for the choice experiment household data. Here, however, we were interested in whether respondents chose a biodiversity enhancement policy option (Options A or B) as opposed to the 'do nothing' option. Data presented in Table 32 shows that, overall, 15% of respondents chose the 'Do nothing' option. In other words, these respondents were not willing to pay additional taxes to achieve biodiversity enhancements. Eighty five percent of the choices made by CE respondents were for choice options A or B. This demonstrates that the majority of respondents were willingness to pay some amount of additional taxation to attain biodiversity enhancements. The choices were equally distributed between Options A and B; as you would expect in CE.

Table 32: Proportion of household CE respondents choosing the alternative biodiversity options

	Choice Option A	*Choice Option B*	*Do nothing Option*	*N (choice tasks)*
Cambridgeshire	41.4 %	41.0 %	17.7 %	1715
Northumberland	44.9 %	42.3 %	12.8 %	1990
Both	*43.3 %*	*41.7 %*	*15.1 %*	*3705*

The reasons given by CE survey respondents for making these choices are presented in Table 33. Over half of the respondents (52.6%) stated that they considered that the biodiversity improvements stated in policy options A or B were 'good value of my money'. Genuine zero bids were indicated by reasons 2 and 4 below. Three percent of respondents stating a zero bid stated that the biodiversity improvements were not good use of their money, while five percent stated that they already contribute to environmental causes. Protest votes included 'I do not think that increases in taxation should be used to fund biodiversity improvements' (6.5%) and 'The costs of biodiversity improvement should be paid for by those who degrade biodiversity' (14.2%). Eighteen percent of the respondents stated other reasons for their choices.

Table 33: Stated reasons for making CE choices.

Reasons for response	**AREA**		
	Cambridgeshire	**Northumberland**	***Both***
1. I chose either policy option A or B because I thought that they were good value for my money.	52.2%	53.0%	*52.6%*
2. I did not consider that the biodiversity improvements from either policy options A or B to be good use of my money.	2.6%	4.0%	*3.4%*
3. I do not think that increases in taxation should be used to fund the biodiversity improvements shown in policy options A or B.	8.5%	4.8%	*6.5%*
4. I already contribute to environmental causes as much as I can afford.	6.4%	3.8%	*5.0%*
5. The costs of biodiversity improvement should be paid for by those who degrade biodiversity.	14.6%	13.8%	*14.2%*
6. Other reason	15.5%	20.4%	*18.1%*
Number of respondents	*341*	*395*	*736*

In conclusion, the evidence gathered in the household study indicates that two-thirds of CV respondents were willing to pay some amount towards biodiversity enhancements, while 85% of choice experiment choices were in favour of paying additional taxation to gain biodiversity enhancements. Analysis of the motivation underlying these statements of support for biodiversity shows that the majority of respondents had made genuine consideration of the benefits they would gain from biodiversity enhancements.

Of the third of CV respondents that stated they were not willing to contribute towards biodiversity enhancements, about half (43%) stated a genuine zero response, while 38% were protest bids. In the CE study, only eight percent of respondent stated genuine zero bids.

8.2. *What aspects of biodiversity do the public value the most?*

In this section we examine the mean WTP amounts from the CV household surveys and then the CE household survey for the biodiversity policies and attributes respectively.

8.2.1. The value of biodiversity policies

The value of biodiversity policies were examined in the CV study. An analysis of this data was reported in Section 7.1 above. We now further discuss these findings.

The mean willingness to pay values found in the Cambridgeshire household CV survey were £74.27, £54.97 and £45.30 respectively for the agri-environmental scheme, habitat re-creation scheme and protect against biodiversity loss from development. In Northumberland, the values of the habitat re-creation scheme and protect against biodiversity loss from development schemes were £47.49 and £36.84 respectively. A number of key issues are important here. First, all of these mean values were found to be significantly different from zero. In other words, the survey respondents had positive willingness to pay values for these policy options. Second, these values lie within a range of between £35 to £75 per household per year. For policy making purposes it may be sufficient to simply note the general scale of these benefits (in the region of around £50), as compared to other possible policy priorities which might have values of around £1 or £1000.

Comparison of the actual WTP values for each policy option suggests that highest values were attained for the agri-environmental scenario, followed by habitat re-creation and then protection against biodiversity loss. However, these values were found not to be significantly different from each other at the 5% level. Thus, based on the evidence from the CV study, we conclude that the public do value the three biodiversity policies, but we cannot provide statistically significant recommendations as to which policy provides the highest public values.

We can also use these mean WTP values to derive the total value of the biodiversity policy scenarios for the two counties as a whole. This is achieved by multiplying the mean WTP per household with the total number of households in the two counties (222,873 households in Cambridgeshire and 130,780 households in Northumberland: Office of National Statistics, 2001). Thus, for Cambridgeshire the value of biodiversity enhancements associated with an expansion of agri-environmental programmes and habitat re-creation programmes is £16.55m and £12.25m annually over five years respectively, while the total willingness to pay to avoid loss of biodiversity due to housing development was £10.10m annually over a five year period. Similarly, in Northumberland, the value of biodiversity enhancements due to habitat re-creation programmes was £6.21m annually over five years, while willingness to pay to avoid loss of biodiversity due to development was £4.82m annually over five years.

It terms of the implications of the above for biodiversity policy, the highest mean WTP amount (Cambridgeshire only) was found for biodiversity enhancements associated with an increase in the area of farmland managed under an agri-environmental scheme. This policy aims to enhance biodiversity through the creation of conservation headlands and the reduced usage of fertilisers, pesticides and herbicides on arable land. The biodiversity benefits from this scheme were described to include a doubling of the diversity of plants within arable fields, the creation of habitats for insects, butterflies and small mammals, and the protection of rare birds.

The habitat re-creation scheme was also valued by survey respondents. In both counties, this policy aimed to create new wetland habitats on existing farmland. Survey respondents were informed that this would be achieved through the restoration of natural river courses and the creation of rapids and pools; seasonal flooding; planning reeds and other wetland plants; and the reintroduction of some wildlife species. The biodiversity benefits from this programme are likely to include the protection of familiar and unfamiliar rare species of wildlife such as the water vole and butterflies, and the enhancement of ecosystem services.

The final programme was to protect at least 50% of land currently being managed under an agri-environmental scheme from being lost to housing development. The biodiversity benefits from this programme would be largely equivalent to those for the agri-environmental programme outlined above. It is therefore interesting to note that the mean WTP for the 'development loss' programme is lower (although not significantly lower) than the mean WTP for the agri-environmental scheme. This finding may appear counter-intuitive since one tends to expect that people value the protection against the loss of a resource more than the creation of a new resource. Although the reason for this finding is unclear, there are two possible explanations. First, the biodiversity losses described in the 'development loss' scenario does not include 'irreversible losses'. In other words, we do not suggest that the development of new houses would lead to any species becoming extinct. Thus, people may be less concerned about biodiversity losses that are not considered to be irreversible. Second, it may be that some respondents were concerned about the shortage of housing within Cambridgeshire, and therefore the lower value in the 'development loss' scenario may include positive benefits from housing developments.

8.2.2. The value of biodiversity attributes.

The choice experiment examined public willingness to pay for four biodiversity attributes: familiar species, rare unfamiliar species, habitat and ecosystem processes. The implicit prices for enhancement of these biodiversity attributes from a base level of decline are report in Table 23 (see Section 7.2.2). Generally, the values of these attributes were both positive and significant in the regression model. The highest implicit price was attained for '*Rare unfamiliar species (recover)*' i.e. recovery of rare unfamiliar species to a stable level which was £115 and £189 in Cambridgeshire and Northumberland respectively. Furthermore, the 95% confidence interval for '*Rare unfamiliar species (recover)*' is found to be significantly higher than the 95% confidence intervals for the '*Familiar species (rare)*', '*Habitat (Restoration)*' and '*Habitat (Creation)*' attribute levels. This finding provides strong evidence that people are now appreciating the value of the 'non-charismatic species'; which in turn provides evidence supporting policies such as species Biodiversity Action Plans. The protection of both rare and common familiar species '*Familiar species (rare + common)*' and habitat re-creation '*Habitat (Creation)*' were also found to have high implicit prices in both case study locations. Two attributes were found not to be valued by respondents. These were '*Rare unfamiliar species (slow down)*' (which describes a slow down in the rate of decline of rare species) which was found to be negative in the Cambridgeshire model and not significant in the Northumberland model. '*Ecosystem (all)*' (which describes protection of ecosystem processes that have both direct and indirect impacts on man) was also found not to be significant in the Northumberland study. We now examine each of the four attributes in turn and discuss the policy implications for that attribute.

8.2.2.1. Familiar species of wildlife

The familiar species of wildlife attribute was defined as any species that the public are likely to recognised, including charismatic and locally important species. In the choice model, familiar species attained positive and significant implicit prices (Table 23). In

Cambridgeshire, scale effects were evident in that the implicit price for the protection of both rare and common familiar species (£93.49) was significantly higher than the protection of only the rare familiar species (£35.65). This was not, however, the case in the Northumberland sample, where the two levels of protection had similar implicit prices (£90.59 and £97.71 respectively for the protection of rare only and rare and common familiar species). The implication from the Cambridge result is that the respondents appear to attain additional utility from the knowledge that common familiar species would not become rare in the future. However, similar evidence was not found in the Northumberland sample where the utility from rare familiar species accounted for the majority of the value estimate. In conclusion, evidence from the choice experiment suggests that the public do support policies that target rare familiar species of wildlife, but the evidence is less clear for the value of common familiar species has. Two possible explanations for this may include the fact that Cambridgeshire respondents were better able to distinguish between the two policy levels, whereas the Northumberland sample simply considered the attributes as a whole and did not consider the levels. However, evidence from the Northumberland workshops (Table 24) indicates that participants were able to distinguish between levels. An alternatively explanation may be that the Cambridgeshire sample did not believe that rare familiar species existed in the county and therefore they did not support the policy to protect rare familiar species. Unfortunately, it was not possible to clarify this from the information gathered in the study.

8.2.2.2. Rare unfamiliar species of wildlife

The second attribute addressed in the choice experiment related to rare unfamiliar species of wildlife. Two levels of provision were addressed. '*Rare unfamiliar species (slow down)*' which aimed to '*slow down the rate of the decline in the populations of rare unfamiliar species. .... it is likely that some rare unfamiliar species may still become locally and nationally extinct*'. The second level '*Rare unfamiliar species (recover)*' aimed to '*stop decline and ensure recovery of rare unfamiliar species*'.

The findings for the '*Rare unfamiliar species (slow down)*' attribute level were interesting since it was found to be negative in the Cambridgeshire sample (indicating that negative utility would be gained from a slow down in the decline of the population of rare unfamiliar species – which was not predicted), while the attribute level was not significant in the Northumberland CE model. The implications of this finding was that it appears that the public are unwilling to support policies that simply delay the time it takes for a species to become (locally) extinct. This conclusion was further emphasised by the fact that highest implicit prices were attained from the '*Rare unfamiliar species (recover)*' attribute level which promised full recover of the populations of rare unfamiliar species. A number of policy implications can be drawn from these findings. First, it would appear that people do appreciate the value of unfamiliar species (that is species which are neither charismatic nor locally significant). This finding directly contradicts those found by White *et al.* (1997 and 2001) and Loomis and White (1996) who found that more charismatic species were likely to attract higher WTP values than less charismatic species. Second, the public appear to only support policies that aim to achieve recovery of the populations of rare species, rather than those that simply attempt to slow down decline in population numbers. A further implication of these findings was that the survey respondents were told that they were unlikely to ever see these rare, unfamiliar species. Thus, these values can be considered to represent passive-use values.

8.2.2.3. Species interactions within a habitat

The habitat attribute was included to assess whether the public valued the restoration of existing habitats '*Habitat (Restoration)*' or the re-creation of new habitats on farmland '*Habitat (Recreation)*'. Both attribute levels were found to be positive and significant in the two case study locations. In Cambridgeshire, the value for habitat restoration (£35.65) was half that for habitat re-creation (£61.36), while similar values were attained for both levels in Northumberland (£71.15 and £74.01 respectively). The reason for this difference may be similar to those stated above for the familiar species attribute. In other words, the Cambridgeshire respondents may have considered that there were very few existing habitats within Cambridgeshire which would benefit from restoration. Again, evidence was not collected to verify this. However, there was evidence that the public would support policies that aimed to protect and enhance species interactions within habitats and the mix of species that reside within them, although the value of the implicit prices were found to be slightly lower than those found for the two species attributes. One further implication of this result is that the public value ecologically significant species such as keystone species, umbrella species and flagship species, all of which have important roles for the protection of habitats and the species that reside within them.

8.2.2.4. Ecosystem processes

The ecosystem processes attribute was included to assess whether the public valued ecosystems that only had a direct impact on humans '*Ecosystem (Human)*' and all ecosystem processes including those which did not directly affect humans '*Ecosystem (all)*'. The ecosystems services that had direct impacts on humans were found to be both positive and significant. However, the '*Ecosystem (all)*' attribute level was not significant in the Northumberland model and was lower than the '*Ecosystem (Human)*' attribute level in the Cambridge sample. It would thus appear that survey respondents 'cared' about ecosystem functions that affect humans, but were less interested in the other ecosystem processes. The discussions held in the Northumberland valuation workshops may help to explain the '*Ecosystem (All)*' finding. In particular, it was found that the ecosystem services attribute was poorly understood by many workshop participants, and that this led to some participants ignoring this attribute in their choice decisions. Assuming that this was also the case in the main survey, this might explain the insignificant or low value estimates found for the ecosystem services attribute.

8.3. How robust are our value estimates?

Three methodologies (choice experiments, contingent valuation and valuation workshops) were utilised in this research to examine the value of biodiversity. In this section, we examine the robustness of the value estimates attained from these methods by first reference to various validity tests and then undertaking a critique of the methods used .

8.3.1. Validity tests

The robustness of the value estimates may be tested using a number of validity tests. First, theoretical validity involves the comparison of observed results with those expected in theory. Usually this involves modelling WTP bids against socio-economic and attitudinal variables. Validity would be supported if *a priori* expectations relating to the significance and direction of co-efficients of the explanatory variables are met. A number of regression models were undertaken for the CV data (reported in Section 7.1). Generally, the direction of the range of socio-economic variables included in these models behaved as expected. Furthermore, the R^2 value for these models ranged between 10% and 20%; which is considered reasonable for this type of analysis. Thus, we argue that the CV models were theoretically valid. Regression models (attributes only) were also undertaken for the CE study (Section 7.2.1). The direction of attribute co-efficients were again largely as predicted and the R^2 values for the Cambridgeshire and Northumberland models were 14% to 19% respectively. Thus, it is argued that the CE models were also theoretically valid.

A second test of validity (convergent validity) involves the comparison of the survey results with those from other studies. Two approaches to convergent validity are considered. First, it could be possible to compare the findings from the contingent valuation study with those from the choice experiment. In theory, this could be achieved by adding up the implicit prices of the relevant choice experiment attributes to match the level of attributes found in the CV policy programmes. Although it had been the initial intention of this research to design the two studies to allow such a comparison to be undertaken, the reality was that such an exercise was not possible since the CV policy programmes could not be directly mapped with the choice experiment attributes. Thus, a robust convergent validity test could not be undertaken between the two methods. However, in general terms, a summation of the value of the choice experiment values tend to be higher than the value of the individual CV policy scenarios. This result is expected since the choice experiment values related to biodiversity changes to those attributes across the counties as a whole, while the CV scenarios relate to more specific areas within the counties. A second approach to convergent validity testing is to compare the results from our study with those from other studies that value biodiversity. Table 2 in Section 3.3.1.3 provides a summary of the values attained in other studies. Mean willingness to pay values for single and multiple species range between \$5 to \$126 and \$18 to \$194 respectively, while values for habitats ranges from \$8 to \$101. Our values for the familiar species, rare unfamiliar species and habitats choice experiment attributes are comparable to these values. A number of other studies allow more direct comparisons. Garrod and Willis (1994) who found that members of the Northumberland Wildlife Trust were willing to pay an average of £10.05 to create a single new wildlife habitat reserve in Northumberland; the value for our habitat creation CV scenario was £54 for habitat improvements throughout the two counties. Hanley *et al.,* (2001) found that average WTP for increases in the area of field margins in Cambridgeshire was £11.53 and £14.70 respective for a 5% and 25% increase; the value of our CV agri-environmental scenario was £74 and included both conservation headland and the reduction in pesticides, herbicides and fertilisers. Macmillan *et al.* (2001b) found the average willingness to pay for conservation policies that targeted endangered wild geese was between £2.83 and £16.50; our value for the protection of *all* rare familiar species in Cambridgeshire and Northumberland was £35 and £90 respectively. Thus, base on the above, we argue that the is evidence that generally supports the values elicited in this study, although it would appear that these value do lie within the upper bound of estimates for similar studies.

A third test of validity undertaken examined content validity. Here, the aim was to test to determine whether the way survey respondents interpret the information presented corresponds to that intended by the researcher. The valuation workshops provided an opportunity to test content validity. In particular, the workshops provided participants with further opportunities to discuss and reflect on the biodiversity concepts and the choice tasks before undertaking the second series of CE choice tasks. Section 7.3.3 reports the regression equations for the CE both before and after these discussion. The Likelihood Ratio test found that the two models were not significantly different from each other. This evidence suggests that the extra discussions held within the valuation workshop did not affect respondent's choices, and therefore implies that the information presented in the household studies was adequate to enable survey respondents to make informed value judgements. Thus, we argue that the content (information set) was appropriate for the study.

Based on the above analysis, it is argued that the findings from this study represent valid results.

8.3.2. *Critique of methodologies used*

This research has aimed to provide a robust framework for the valuation of biodiversity. It was clear from the outset that such an exercise was likely to be challenging. In particular, the fact that the general population has a low level of understanding of biodiversity imposes a major obstacle to this work. In what follows, we assess the extent to which this study successfully addresses a number of key challenges including the valuation of complex goods and scoping issues.

8.3.2.1. *Valuing complex goods*

As stated above, one of the key challenges to this research was the fact that the general public have a low level of knowledge and understanding of biodiversity. Clearly, valid valuations can only be attained if survey respondents have adequate information to allow them to make informed decisions. This research addressed this issue in a number of ways. First, a lot of effort (in the form of focus groups) was undertaken during the initial stages of the research to find out which aspects of biodiversity the public were most concerned about. Not only did the findings from these focus groups help us to identify the scope of the research project (in terms of which aspects of biodiversity to address), but it also helped to identify the most appropriate language in which to describe these aspects of biodiversity. For example, the focus group discussions were key to the realisation that the public view biodiversity in a very different way from that of biodiversity 'experts' (i.e. ecologists). To provide an example, it was clear from the focus groups that the public were not overly concerned with the dynamics of how biodiversity enhancements are achieved, instead, they were more concerned with the biodiversity outcomes (e.g. that a range of species within a habitat were being protected). Ecologists, on the other hand, tend to focus on the mechanisms that achieve biodiversity enhancements (e.g. the role of keystone or umbrella species within a habitat). A study that simply attempted to value biodiversity in terms of an ecologists perspective would likely confuse respondents and therefore result in meaningless value estimates.

The second way in which we tackled the problem of the public's poor knowledge of biodiversity was the use of the MS PowerPoint presentation. Most valuation studies tend to present information on the good in question through verbal descriptions, perhaps supplemented with pictorial show cards. It was clear from the focus groups and our initial pilot study that such an approach would have limited success. The adoption of a PowerPoint presentation allowed for a more dynamic presentation of the complexities of biodiversity involving the seamless combination of verbal descriptions, which were re-emphasised with written bullet point notes and visual images. Furthermore, the use of a computer presentation helped to stimulate respondents and therefore helped to minimise respondent fatigue. This was particularly important due to the amount of new information that we were presenting to respondents. The adoption of a 'standard' verbal approach to presenting this level of information would simply have been inadequate. There are, however, issues with the use of PowerPoint presentations. First, it requires the use of laptop computers for all interviews. In addition to the obvious cost implications, there are also practical issues, namely the requirement to have access to a power source during interviews. In this study, this was overcome by undertaking interview inside respondent's homes. It was found that once inside a respondent's house, gaining access to a power source was easily attained. Being inside the respondent's house also had the added advantage that the respondent was more relaxed (than in say street interviews) and was also more willing to spend time completing the interview (which was important since an interview took between 30 and 40 minutes to complete). Thus, it would appear that the PowerPoint presentation had several advantages over standard methods of conveying information in valuation studies.

Another important question relates to how successful the PowerPoint presentation actually was in conveying the complexities of biodiversity. The findings from the valuation workshop provide an insight into this question. First, workshop respondents indicated that they had a reasonably good understanding of the biodiversity concepts following the PowerPoint presentation; i.e. other than the ecosystem services attribute, respondents scored their level of understanding of biodiversity concepts at 3 or more out of 5 in the likert scales (Table 24). However, it is also noted that respondent's level of understanding of biodiversity was further improved following the workshop discussions on biodiversity (Table 24). Thus, it is clear that the PowerPoint presentation alone did not fully inform respondents. Reference to the data presented in Table 28 however indicates that the parameters of the main survey choice model do not differ significantly from either the 'Before' workshop model (i.e. before the opportunity to discuss biodiversity further) or the 'After' workshop model (i.e. after the discussion). Thus, it would appear that the extra knowledge gained during the additional discussions of biodiversity in the workshop did not influence WTP amounts for the biodiversity attributes in the choice models. Given that researchers need to attain a balance between (i) providing sufficient information to enable respondents to make informed value judgements and providing too much information that may bias value estimates (social engineering) and (ii) the amount of time spent presenting information to respondents and the costs of doing so, it is argued that the amount and level information presented during the PowerPoint presentation was sufficient to enable respondents to provide robust assessments of their WTP for biodiversity attributes. It is therefore concluded that the use of the PowerPoint presentations provide a useful tool to enable researchers to present a lot of complex information to survey respondents.

8.3.2.2. Scoping issues

Scoping is another key methodological issue in valuation studies. Scoping is the term used to describe the size or extent of the good / policy being valued. Researchers therefore need to ensure that respondents are fully aware of the scope of the good being valued and that their value estimates are based on this scope. In this research, the scope of the biodiversity policies were based around the two counties: Cambridgeshire and Northumberland. Throughout the valuation interview (and in particular the PowerPoint presentation) respondents were repeatedly informed that the biodiversity policies which they were considering would be restricted to either Cambridgeshire or Northumberland (depending on the location the interviews took place). Within the actual valuation questions respondents were again repeatedly reminded that they should only be considering biodiversity improvements in Cambridgeshire / Northumberland. Given this level of reinforcement of the scope of the proposed policies, we fully expected that the majority of respondents would have taken this scope into account while making the value judgements. To test whether this was the case, we asked the participants of the workshop to '*indicate* the *locations you think the biodiversity policies that we have been talking about will be targeted'.* The results from this are reported in Table 34 below.

Table 34: Workshop participant's perceptions of scope of biodiversity policies

Location	*% responses*
An individual farm	2.2%
Locations around this town	4.4%
Northumberland	48.9%
England	6.7%
UK	33.3%
Europe	4.4%

These findings are of concern since only 48.9% of the workshop participants correctly identified the scope of the valuation study. Almost one-third of participants considered that the study was addressing biodiversity policy throughout the UK. Unfortunately similar data was not collected in the main valuation study; however, it is unlikely that the main study would differ significantly from those reported in Table 34 for the workshop. Clearly these findings are of concern for the reliability of the WTP estimates. Thus, we recommend that the results be interpreted with caution.

What lessons can be learned from the above? First, a shortfall of our study was that it appears to have largely failed to ensure that respondents were fully aware of the geographic scope of the study when making their value judgements in that almost a third of respondents considered that the biodiversity policies related to the UK as a whole rather than Cambridgeshire / Northumberland. This finding is of concern for two reasons. First, it implies that our mean WTP estimates include some estimates that have a larger geographic scope than was intended. Thus it is likely that our mean WTP values are over-estimates of the true WTP for the two counties. Second, and perhaps of more concern to stated preference studies as a whole, is the fact that given the number of times that respondents were repeatedly informed that the biodiversity policies were restricted to the county level[6], that many respondents still failed to appreciate the scope to the study. The lack of appreciation of scope was certainly not expected. How could we prevent this happening in the future? One possible solution may have been to explicitly state to respondents that the geographic scope of the programmes was the county level and '*not the UK as a whole*'. However, it is unclear as to whether the additional of such a statement would significantly alter the findings reported in Table 34. This leads us to our second lesson learnt. In the valuation workshops a question was included to identify respondent's perceptions of the scope of the study (reported in Table 34 above). Such a question clearly provides useful information with regards to the whether or not respondents were actually valuing the same scope of policy that was intended by the researcher. If such a question had been included in the main study, then it would have been possible to exclude those respondents that did not match the intended scope. In order to prevent these problems in the future, it is recommended that future valuation studies including a question to check that respondents are basing their estimates on the appropriate scope of good.

8.4. Can our benefit estimates be transferred to other situations?

Tests of benefits transfer may be undertaken either by comparing mean WTP values (CV data) or implicit prices (CE data) across case study areas; and by comparing the bid curves (CV data) or indirect utility functions (CE data) across the case study areas.

8.4.1. Tests for benefits transfer from the CV data.

If we first examine the comparison of mean WTP responses from the main CV study. Data presented in Table 8 demonstrates that the mean WTP for all policy programmes was significantly different between the two case study areas. Thus, this finding does not support simple benefits transfer. However, in this comparison the Cambridgeshire dataset included estimates for an agri-environmental policy, which was not included in the Northumberland study. A more refined analysis that examines the transfer of mean WTP for the individual policy scenarios (habitat re-creation and development loss) was presented in Table 9 and Table 10 respectively. This analysis found that there were no significant differences between the mean WTP values for individual policy programmes across case study areas. This provides some evidence supporting benefits transfer between programmes.

[6] We refer the reader to the questionnaires reproduced in the Appendix to appreciate the actual number of time that the geographical scope was referred to in the study.

The transferability of bid curves for the CV data were also analysed. In Table 11, a dummy variable for case study area was found to be significant in the 'both' (pooled) dataset. This suggests that even after taking account of socio-economic differences people in Cambridgeshire valued biodiversity differently than people in Northumberland. A Chow test[7] was also undertaken to analyse whether the bid curve underlying the Cambridgeshire data was different to that underlying the Northumberland data (Table 11). This test shows that the inverse demand curves (bid curves) are different: it would therefore be incorrect to estimate WTP in either region using the bid curve parameters from the other. In this sense, we reject benefits transfer for the Contingent Valuation data.

The bid curve analysis was also repeated separately for the habitat re-creation and development loss scenarios separately. The Chow test for habitat recreation shows significant differences between the two regions (Table 12), while the same analysis for the development loss scenario does not. Thus, in the bid curve analysis there was some evidence to support benefits transfer, but this certainly is not conclusive.

In the above analysis, benefits transfer was not supported when the entire CV dataset was used; however, there was some evidence of equivalence when the analysis is undertaken separately for the policy programmes. The fact that the 'merged' dataset was different between the two case study areas (i.e. the Northumberland dataset does not include an agri-environmental scheme scenario) may bring into question the validity of the tests using the entire dataset. One way in which to test the impact of this was to examine the equivalence of WTP for individual policy scenarios across case study areas (Table 14, Table 15 and Table 16). Evidence from these tests indicates that there were no significant differences between mean WTP between case study areas, and no difference in the bid curves for the development loss scenario. However, significant differences were found between the bid curves for the habitat re-creation scenario. Thus, there does appear to be some evidence supporting the possibility of transferring benefit estimates from the CV study. However, it should also be pointed out that acceptance of the benefits transfer hypothesis is inflated by the imprecision of WTP estimates (ie as confidence intervals widen, it is harder to reject benefits transfer in terms of overlapping mean WTP values).

8.4.2. Tests for benefits transfer from the choice experiment data.

Benefits transfer was also tested in the choice experiment data. Here a Likelihood Ratio test was used to compare the beta values (parameter estimates) between the Cambridgeshire and Northumberland models (Table 22). The probability value for this test is < 0.01, indicating that the two models were different. Based on this evidence we would reject the transfer of the indirect utility functions between the two areas.

Another test for benefits transfer undertaken on the choice experiment data was to test whether the implicit prices for each attribute were significantly different from each other between the Cambridgeshire and Northumberland samples. Evidence from Table 23 indicates that the 95% confidence intervals for implicit prices do overlap between the models in two out of six cases - for '*Familiar (rare + common)*' and '*Habitat (Creation)*'; however, this is largely due to the large standard errors on the implicit prices (the same problem was referred to above for CV data). So again, there is little evidence in support of benefits transfer in the choice experiment data.

[7] That is, we test Ho: Beta (Cambridgeshire) = Beta (Northumberland)

8.4.3. Benefits transfer implication

In the above analysis, it was concluded that there was some evidence supporting benefits transfer using the CV data on individual policy programmes, but there was little evidence to support benefits transfer using the choice experiment dataset. In both cases, the evidence for accepting benefits transfer was stronger when mean WTP transfers were made, as opposed to bid function transfers. Taking into account that fact that it is generally accepted that the transfer of bid functions is a more refined approach to benefits transfer than the transfer of mean WTP amounts, we argue that there is little evidence to support the benefits transfer from this study. Based on this finding, we also argued that it would be inappropriate to attempt to aggregate the values found from the two case studies to the value of England as a whole.

9. Conclusions

The overall remit for this research was to assess whether it is possible to develop a framework to enable meaningful and robust values for biological diversity in the UK. Important to this aim is the assessment of whether it is possible to gain meaningful and robust values for complex non-market goods (biodiversity being an example of such a complex good), and whether the framework developed would be suitable for other similar practical applications within the UK. In addition to this overall aim, a number of specific objectives were also addressed including: the measurement of the economic value of the component attributes of biological diversity; the measurement of the economic value of policy programmes to enhance and protect biodiversity; an examination of the feasibility of benefits transfer; and a discussion of methodological issues.

The research involved a number of elements. Preparatory work included a review of ecological and economic literature relevant to valuing biodiversity, an expert review of the suitability of alternative valuation methodologies for valuing biodiversity, a series of focus groups to assess the level of public understanding of biodiversity. Three alternative valuation approaches were used in the final study including a contingent valuation study to address the value of three biodiversity policies, a choice experiment to value the attributes of biological diversity and a series of valuation workshops to further address various methodological issues. In this concluding section, we first address the specific research objectives relating to the valuation of biodiversity policies and attributes, and the feasibility of benefits transfer. Finally, we conclude by commenting on the overall suitability of the adopted approaches for the valuation of complex goods.

9.1. Measurement of the economic value of policy programmes which enhance and protect biodiversity.

A contingent valuation survey was used to examine public WTP for three biodiversity policies:

- An agri-environmental scheme (based on the pilot Arable Stewardship scheme) that aims to enhance biodiversity on arable land through the creation of conservation headlands and the reduced application of pesticides and herbicides. Biodiversity benefits from this scheme would include an increased diversity of plants, insects, small mammals and birds; some of which may be rare.
- A habitat re-creation scheme that would enhance biodiversity by creating new wetland habitats on existing farmland. The new wetland would provide habitats for a wide range of plants, insects, small mammals and birds, including a number of rare species. In addition, the wetland area would provide ecosystem services such as flood protection and enhanced water quality.
- A scheme that would aim to avoid biodiversity loss as a result of housing development on farmland managed under existing agri-environmental schemes. The types of biodiversity protected under this policy would be similar to those described in the agri-environmental scheme above.

The key findings from the CV study were as follows:

- The value of the three policies in Cambridgeshire were £74.27, £54.97 and £45.30 respectively (figures relate to £ per household per year over a five year period), while in Northumberland, the values of the habitat re-creation scheme and protect against biodiversity loss from development schemes were £47.49 and £36.84 respectively. In all cases, these values were found to be significantly different from zero. However, these values should be interpreted with caution since a finding from the valuation workshop indicated that approximately one-third of workshop participants interpreted the scope of the study to relate to the UK as a whole as opposed to the individual counties. Assuming that the respondents to the main household survey also misinterpreted the scope of the biodiversity policies, it is likely that these estimates are within the upper bounds of the true value.

- The estimated WTP values for the alternative policy scenarios were not found to be statistically different from one another. The implications of this are that although people do value biodiversity enhancements and protection, they appear to be indifferent with regard to how such enhancements are achieved – perhaps suggesting that they are happy that biodiversity 'experts' decide on the best way to protect and enhance biodiversity.
- There was some evidence of consistency in mean WTP values for policies between the two case study areas, however, this was not the case for the transfer of the bid functions.
- Analysis was also undertaken to estimate the overall value of these biodiversity policies to the two counties. This found that 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 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. Again, caution should be undertaken in interpreting these values since they are likely to represent upper bounds of the true values.

The key policy implications of the above findings are that the public are willing to pay a positive sum of money for biodiversity enhancing and protecting policies. Thus, we provide evidence in support of further investment in biodiversity enhancing and protecting policies in the future. However, there were no significant differences between the values of the alternative policy prescriptions. Thus, we are unable to make clear recommendations with regard to which types of biodiversity policy should take priority. However, we note that the agri-environment policy scenario did attain the highest willingness to pay amount out of the three policy options. Although, this finding was restricted to the Cambridge case study, it does indicate that the public appreciate the role that agriculture has in preserving the UK's biodiversity.

9.2. Measurement of the economic value of the component attributes of biological diversity.

The second method utilised was the choice experiment method. The CE study assessed the value of four attributes of biodiversity: familiar species of wildlife, rare unfamiliar species of wildlife, species interactions within a habitat and ecosystem services. The key findings from the CE study were as follows:

- The public were willing to pay some positive amount for all but two of the biodiversity attributes. As with the contingent valuation study, these findings provide evidence that the public support biodiversity policies.
- The attribute that targeted the 'recovery of rare unfamiliar species' attained the highest implicit price (£115 and £189 respectively for Cambridgeshire and Northumberland). Furthermore, this attribute was the only one that was valued significantly higher than any of the other attributes. This finding is significant in that it demonstrates that the public do value the protection of rare non-charismatic species. In terms of a policy context, this finding provides support for species Biodiversity Action Plans which often target rare, unfamiliar species.
- In contrast to the above, the 'slow down the rate of decline of rare unfamiliar species' was found to be negative in the Cambridgeshire sample (indicating that negative utility would be gained from a slow down in the decline of the population of rare unfamiliar species – which was not predicted), while the attribute level was not

significant in the Northumberland CE model. The implications of this finding was that it appears that the public are unwilling to support policies that simply delay the time it takes for a species to become (locally) extinct. In other words, the public appear to only support biodiversity policies that ensures the continued survival of species as opposed to simply delaying the inevitable.

- In Northumberland, both the protection of 'rare familiar species' (£90.59) and 'both rare and common familiar species' (£97.71) were found to achieve consistently high implicit prices, while in Cambridgeshire the protection of 'rare familiar species' (£35.65) was found to be significantly lower than the protection of 'rare and common familiar species' (£93.49). Although the reason for these differences between areas is unclear, it is suggested that the policy implications of this finding is that the public appear to support policies that target rare familiar species of wildlife, but are less willing to support the protection of common species.
- Similar results to the above were also found for the 'species interactions within habitats' attribute. In Northumberland, the 'habitat restoration' attribute (£71.15) was found to be similar to the 'habitat re-creation' attribute (£74.00), while in Cambridgeshire the 'habitat re-creation' attribute (£61.36) achieve a higher implicit price than the 'habitat restoration' attribute (£34.40). Again data was not collected to ascertain the reason for these differences, but it was suggested that the low value in Cambridgeshire for the 'habitat restoration' attribute may be due to the perception that Cambridgeshire does not support habitats that could be restored, and therefore survey respondents did not support these policies.
- Finally, the 'ecosystem processes' attribute with direct impacts for humans was highly valued in both Cambridgeshire and Northumberland (£53.62 and £105.22 respectively. However, the all 'ecosystem processes' attributes (which included the human impact level) was not significant in Northumberland and was lower than the human impact level in Cambridgeshire. The reason for this findings appears to stem from the fact that generally there was a lower level of understanding of this attribute and therefore people valued it less.

The key policy implications of the CE data is that the public do value most, but not all, biodiversity attributes and that they appear to be able to distinguish between alternative attributes (but perhaps not always attribute levels). In particular, there is evidence to support the continued funding of policies that target species, habitats and ecosystem processes. Of particular interest is the finding that the public have high values for the protection of rare unfamiliar species; thus policies should not be restricted to target only familiar and charismatic species. Second, the comparison of the results between Cambridgeshire and Northumberland for the rare familiar species attribute level and the habitat restoration attribute level are interesting in that it would appear that people in Cambridgeshire have low values for these two attributes as a direct result of the perception that Cambridgeshire currently does not support such biodiversity.

9.2.1.1. Is benefits transfer feasible?

A series of tests for benefits transfer of the CV and CE data between the two case study areas were also undertaken. Although there was some evidence of equality of mean WTP values, there was little evidence of the transferability of the benefit functions between case study areas. Thus, based on these findings we reject the feasibility of benefits transfer. There are a number of issues that need to be considered in light of this conclusion. First, in this research we aimed to transfer values between two contrasting case study areas: Cambridgeshire with minimal existing biodiversity and Northumberland which has a relatively rich biological resource. It may be that it is these differences that led to the rejection of benefits transfer. In order to test this, it would be interesting to repeat the study in another location that has similar levels of biodiversity to the case study areas, e.g. Bedfordshire for Cambridgeshire. There are, however, wider implications of this finding. If we can not demonstrate that benefits transfer is feasible within a single country (i.e. England), then there would appear to be little hope of benefits transfer between countries. Clearly, the implication of this need to be considered if policy makers plan to use the EVRI database in the future. Benefits transfer, however, is a relatively new concept and further research needs to be undertaken to explore the feasibility of transferring values between situations and also research needs to be undertaken to explore more sophisticated ways of transferring the values.

9.2.1.2. Methodological issues raised in this research

The valuation of changes in biological diversity was clearly a challenging application of valuation methodologies. Although we believe that we were largely successful in our study, we feel that it is important to highlight some of the innovative approaches used in this research to address these challenges, as well as identify some of the shortcomings.

- First, this research provides one of the first valuation applications which incorporates a comprehensive examination of a wide range of biodiversity attributes. Previous studies have tended to focus on a single species or habitat. By attempting to capture a comprehensive range of biodiversity attributes, this study enables direct comparisons to be made across the relative value of different biodiversity attributes; for example familiar species against ecosystem services. Most other studies would require this comparison to be made across different studies, which is likely to be affected by some element of transfer error. Thus, this research provides a useful policy tool to enable direct comparisons to be made between the relative values of alternative biodiversity attributes.
- Secondly, biodiversity is a complex issue. This research has demonstrated that through careful survey design and the adoption of innovative information presentation methods, stated preference valuation methods are capable of valuing complex goods. We recommend that researchers need to consider new and innovative information presentation methods, particularly when considering complex goods.
- Furthermore, the use of valuation workshops provided an opportunity to further explore respondent's understanding of biodiversity concepts and the valuation task itself. Such an exercise can provide further evidence to support the findings of a valuation study. We recommend that valuation studies should utilise valuation workshops as a tool to help validate the valuation instrument and therefore the research results.
- One issue of concern with this research relates to scoping issues. A finding from the valuation workshop indicated that one-third of respondents mis-understood the geographic scope of the valuation question (i.e. they thought that the scope of the study related to the UK as a whole, as opposed to the county level). This finding is of concern for valuation studies as a whole since respondents were repeatedly informed throughout the interview that the study was based on the county. In other words, in this study standard approaches to conveying the scope of the study failed! A recommendation stemming from this finding is that all valuation studies should include a follow-up question to ask respondents to identify the scope of the good that they thought they were valuing. Responses from this question may be used to exclude responses that do not accurately address the scope of the study.

9.2.1.3. Is it possible to gain meaningful and robust values for complex goods such as biodiversity?

This final section of the report specifically focuses on the key questions that this research aimed to address, namely

- *Assess whether it is possible to attain meaningful and robust values for complex goods such as biodiversity;*

- *Develop an appropriate framework which will enable a cost-effective and robust valuation of the total economic value of biodiversity changes in the UK countryside.*

Turning to the first aim, we argue that we have been successful in attaining meaningful and robust values for complex goods. Evidence supporting this claim comes from a number of sources including the validity tests for the alternative valuation studies and the responses from the valuation workshop. We, however, stress that valuing complex goods is challenging, and in particular a lot of effort needs to be undertaken in developing the hypothetical descriptions of the goods in question. In our study, this effort included an 'expert' (ecologists) review of biodiversity and a series of focus groups to 'translate' the expert view into a language which was both understandable and meaningful to the public. An important issue here was the realisation that the way the public considered biodiversity was different to that from the experts. In other words, the public were more concerned with the biodiversity outcomes, where ecologists tended to focus on the processes affecting biodiversity change.

With regard to developing a cost-effective and robust framework for valuing biodiversity change the conclusions are less clear. First, tests for benefits transfer between the two case study areas generally failed. Thus, we cannot advocate the transfer of (robust) benefit values from our study areas to other areas of the UK. Although the reasons for the failure of benefits transfer are unclear, it may be that it is due to differences in the existing levels of biodiversity within the two case study areas. Further work would be required to clarify this. Second, the failure of benefits transfer (which is considerably less expensive that undertaking original studies) means that we cannot use the study results to provide a low cost framework for valuing biodiversity in the future. On a more positive note, we believe that our approach was largely successful in providing a robust framework in which to value biodiversity change. In particular, we argue that the public were capable of understanding our descriptions of biodiversity policies and attributes (with perhaps the 'ecosystem processes' being the exception). Thus, we recommend that contingent valuation method for the valuation of biodiversity programmes and the choice experiment method for biodiversity attributes. Finally, an interesting result from this research was that the value estimates from the six Northumberland valuation workshops (which included additional discussions on biodiversity) were largely equivalent to the 400 responses from the Northumberland household survey. If such equivalence could be demonstrated to be consistent in other areas (say for example if we repeated the workshop in Cambridgeshire and found equivalence), then undertaking valuation workshops in other counties of England and then linking the values from these counties to either the value of a low biodiversity area (ie. Cambridge) or a high biodiversity area (Northumberland), this may provide a relatively cheap framework to allow a robust aggregation of this studies results to the UK as a whole.

10. References

Aarssen, L.W. (1997). High productivity in grassland ecosystems: effected by species diversity or productive species? Oikos 80, 183-184.

Abernethy, V.J., McCracken, D. I., Adam, A., Downie, I., Foster, G. N., Furness, R.W., Murphy, K.J., Ribera, I., Waterhouse, A., Wilson, W.L. (1996). Functional analysis of plant-invertebrate-bird biodiversity on Scottish agricultural land. In: Simpson I. A. & Dennis P. (eds), The Spatial Dynamics of Biodiversity: Proceedings 5th Annual IALE Conference, Stirling 1996, International Association for Landscape Ecology, Stirling, pp. 51-59.

Ali M.M., Murphy K.J. & Abernethy V.J. (1999). Macrophyte functional variables v. species assemblages as predictors of trophic status in flowing waters. *Hydrobiologia* 415, 131-138.

Ali, M.M., Dickinson, G. and Murphy, K.J. (2000). Predictors of plant diversity in a hyperarid desert wadi ecosystem. J. Arid Environm. 45, 215-230.

Allott, T.E.H. and Monteith, D.T. (1999). Classification of lakes in Wales using integrated Biological data. Countryside Council for Wales. Science Report No 314.

Andreasen, C. Stryhn, H. and Striebig, J.C. (1996). Decline in the flora in Danish arable fields. J. Appl. Ecol. 33, 619-626.

Arrow, K. Solow, R Portney P. R., Learner E. E., Radner R., and Schuman, H. (1993). Report of the NOAA Panel on Contingent Valuation, *Fed. Regist.* **58**, 4601-4614.

Arts, G.H.P., Roelofs, J.G.M. and de Lyon, M.J.H. (1990). Differential tolerances among softwater macrophyte species to acidification. Can. J. Bot. 68, 2127-2134.

Aspinall, R.J. (1996). Some issues in measuring and modeling (bio)diversity. In: Simpson I.A. and Dennis P. (eds), The Spatial Dynamics of Biodiversity: Proceedings 5th Annual IALE Conference, Stirling 1996, International Association for Landscape Ecology, Stirling, pp. 43-49

Barton, D. (2001) The transferability of benefits transfer. PhD thesis, Agricultural University of Norway, Aas.

Bate, R. 1994. *Pick a Number: A Critique of Contingent Valuation Methodology and its Application in Public Policy*. Washington, D.C.: Competitive Enterprise Institute.

Bateman, I. and Willis, K. (1999). Valuing environmental preferences: theory and practice of the contingent valuation method. Oxford: Oxford UP.

Bateman, I., Willis, K. and Garrod, G. (1993). Consistency between contingent valuation estimates: a comparison of two studies of UK National Parks. Regional Studies, 28, 5, 457-474.

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

Bignal, E.M., Jones, G. and McCracken, D. (2001). Future directions in agriculture policy and nature conservation. Brit. Wildlife 13, 16-20.

Bini, L.M., Thomaz, S. M. and Souza, D. C. (2001). Species richness and beta-diversity of aquatic macrophytes in the Upper Paraná River floodplain. Arch. Hydrobiol. 151, 511-525.

Bodini, A. (2000) Reconstructing trophic interactions as a tool for understanding and managing ecosystems: application to a shallow eutrophic lake. Can. J. Fish. Aq. Sci. 57, 1999-2009

Brickle, N.W., Harper, D.G.C., Aebischer, N.J. and Cockayne, S.H. (2000). Effects of agricultural intensification on the breeding succes of corn buntings *Miliaria calandra.* J. Appl. Ecol. 37, 742-755.

Brotherton, I. (1996). Biodiversity, spatial extent and protectability. In: Simpson, I.A. and Dennis, P. (eds), The Spatial Dynamics of Biodiversity: Proceedings 5th Annual IALE

Conference, Stirling 1996, International Association for Landscape Ecology, Stirling, pp. 149-160.
Brouwer, R. (2000). Environmental value transfer: state of the art and future prospects. Ecological Economics, 32 (1), 137-152
Carey, P.D., Hill, M.O. and Fuller, R.J. (1996). Biodiversity Hotspots in Scotland. SNH Research, Survey, Monitoring Report No. 77. Scottish Natural Heritage, Edinburgh.
Carson, Flores and Mitchell (1999). The theory and measurement of passive-use values. In . Bateman IJ and Willis KG. (Eds) *Contingent valuation of environmental preferences: assessing theory and practice in the USA, Europe and Developing countries*. Oxford University Press.
Chamberlain, D.E. and Fuller, R.J. (2001). Local extinctions and changes in species richness of lowland farmland birds. Global Ecol. Biogeog. 10, 399-409.
Chancellor, R.J. (1985). Changes in the weed flora of an arable field cultivated for 20 years. J. Appl. Ecol. 22, 491-501.
Chapin, F.S. and 13 others (1998). Ecosystem consequences of changing biodiversity – experimental evidence and a research agenda for the future. Biosci. 48, 45-52.
Chase, M.W. and 41 others (1993). Phylogenies of seed plants – an analysis of nucleotide sequences from the plastid gene *rbc***L.** Ann. Missouri Bot. Garden 80, 528-580.
Christie M (2001) A comparision of alternative contingent valuation elicitation treatments for the evaluation of complex environmental policy. *Journal of Environmental Management.* 62, 255-269.
Christie M, Warren, J, Hanley, N, Murphy, K and Wright R (2003a). *Developing measures for valuing changes in biodiversity: Phase 1: a review of relevant ecological and economic literature*. Interim report to DEFRA: :London.
Christie M, Warren, J, Hanley, N, Murphy, K and Wright R (2003b). *Developing measures for valuing changes in biodiversity: Phase 2 report*. Interim report to DEFRA: :London.
Claridge, M.F. and Boddy, L. (1994). Species recognition systems in insects and fungi. In: Hawksworth, D.l. (ed.) The identification and characterisation of pest organisms. CAB International, Wallingford, pp. 261-274.
Colwell, R.K. and Coddington, J.A. (1995). Estimating terrestrial biodiversity through extrapolation. Phil. Trans. Roy. Soc. London B 345, 101-118.
Cooper, E. and MacKintosh, J. (1996). NVC review of Scottish grassland surveys. SNH Review No. 65. Scottish Natural Heritage Edinburgh.
Costanza, R. (1992) Toward an operational definition of ecosystem health. In: Costanza, R., Norton, B.G. & Haskell, B.D. (eds.) Ecosystem health: new goals for environmental management. Island Press, Washington DC, pp. 239-256.
Cottingham, K.L., Rusak, J.A. and Leavitt, P.R. (2001). Increased ecosystem variability and reduced predictability following fertilization: evidence from palaeolimnology. Ecol. Lett. 3, 340-348.
Crabtree, B., Macdonald, D. and Hanley, N. (2002). Non-market benefits associated with mountain regions. Report for Highlands and Islands Enterprise and Scottish Natural Heritage.
Crawley, M.J., Brown, S.L., Heard, M.S. and Edwards, G.R. (1999). Invasion-resistance in experimental grassland communities: species richness or species identity? Ecology Letters, 2 (3), 140-148.
Crawley, M.J., Brown, S.L., Hails, R.S., Kohn, D.D. and Rees, M. (2001). Transgenic crops in natural habitats. Nature 409, 682-683.
Crow, G.E. (1993). Species diversity in aquatic angiosperms, latitudinal patterns. Aquat. Bot. 44, 229-258.
Daugherty, C.H., Cree, A., Hay, J.M. and Thompson, M.B. (1990). Neglected taxonomy and continuing extinction of tuatara. Nature 47, 177-179.
Day, R.T., Keddy, P.A., McNeill, J. and Carleton, T. (1988). Fertility and disturbance gradients: a summary model for riverine marsh vegetation. Ecology 69, 1044-1054.
Desvouges, W.H., Reed Johnson, F. and Banzhaf, H.S (1998). Environmental Policy Analysis with Limited Information. Cheltenham: Edward Elgar.

Diamond, P.A. and Hausman, J.A. (1994). Contingent valuation – is some number better than no number. Journal of Economic Perspectives, 8 (4), 45-64.

Díaz, M. and Tellería, J.L. (1994). Predicting the effects of agricultural changes in Spanish croplands on seed-eating overwintering birds. Agric. Ecosyst. Environm. 49, 289-298.

Dickinson, G. and Murphy, K.J. (1998). Ecosystems, a Functional Approach. Routledge, London, UK. 190 pp.

Diemer, M., Joshi, J., Korner, C., Schmid, B. and Spehn, E. (1997). An experimental protocol to assess the effects of plant diversity on ecosystem functioning utilized in a European research network. Bull. Geobotan. Inst. ETH 63, 95-107.

Dixit, S.S., Cumming B.F., Birks H.J.B., Smol J.P., Kingston J.C., Uutala A.J., Charles D.F. and Camburn K.E.. (1993). Diatom assemblages from Adirondack lakes (New York, USA) and the development of inference models for retrospective environmental assessment. Journal of Palaeolimnology 8, 27-47.

Donald, P.F. (1998). Changes in the abundance of invertebrates and plants on British farmland. Brit. Wildlife 9 (5), 279 – 283.

Dony, J.G. and Denholm, I. (1985). Some quantitative methods of assessing the conservation value of ecologically similar sites. J. Appl. Ecol. 22, 229-238.

Downie, I.S., Abernethy, V.J., Foster, G.N., McCracken, D.I., Ribera, I. and Waterhouse, A. (1998). Spider biodiversity on Scottish agricultural land. Proceedings 17th European Colloquium on Arachnology, 1997, Edinburgh: pp. 311-317.

Downie, I.S., Ribera, I., McCracken, D.I., Wilson, W.L., Foster, G. N., Waterhouse, A., Abernethy, V.J. and Murphy, K.J. (2000). Modelling populations of *Erigone atra* and *E. dentipalpis* Araneae: Linyphiidae across an agricultural gradient in Scotland. Agr. Ecosyst. Environ. 80, 15-28.

Downie, I.S., Wilson, W.L., McCracken, D.I., Foster, G.N., Ribera, I., Waterhouse, A., Abernethy, V.J. and Murphy, K. J. (1999). The impact of different agricultural land-uses on epigeal spider biodiversity in Scotland. J. Insect Conserv. 3, 273-286.

Embley, T.M., Hirt, R.P. and Williams, D.M. (1995). Biodiversity at the molecular level: the domains, kingdoms and phyla of life. In: Hawksworth, D.L. (ed.) Biodiversity: measurement and estimation. Chapman and Hall, London, pp. 21-33

Ernfeld D (1988) Why put a value on biodiversity? In: Wilson (ed) Biodiversity. Washington DC: national Academy Press 212-216.

E.R.M. (1996). Valuing Management for Biodiversity in British Forests: Final Report for the Forestry Commission, April 1996.

Faber, M., Manstetten, R. and Proops, J. (1996). Ecological Economics. Concepts and Methods. Edward Elgar, Cheltenham, Brookfield.

Farmer, A.M. and Spence, D.H.N. (1986). The growth strategies and distribution of isoetids in Scottish freshwater lochs. Aquat. Bot. 26, 247-258.

Ferrier, R.C., Malcolm, M.A., McAlister, E., Edwards, A. and Morrice, J. (1996) Hindcasting of in-loch phosphorus concentrations based on land cover classification. Report to Scotland and Northern Ireland Forum for Environmental Research, Stirling, UK.

Foster, G. N., McCracken, D. I., Blake, S., Ribera, I. (1997). Species biodiversity and conservation value in agriculture: ground beetles Carabidae as a case study. Proceedings Biodiversity in Scotland: status, trends and initiatives. Edinburgh, 1996. The Stationery Office, London, Edinburgh.

Fox, J. A., J. F. Shogren, D. J. Hayes and J. B. Kliebenstein (1998), 'CV-X: Calibrating Contingent Values with Experimental Auction Markets', American Journal of Agricultural Economics 80, 455-465.

Fozzard, I., Doughty, R., Ferrier, R.C., Leatherland, T. and Owen, R. (1999). A quality classification for management of Scottish standing waters. Hydrobiologia 395/396, 433-453

Franks, J.R. (1999). In situ conservation of plant genetic resources for food and agriculture: a UK perspective. Land Use Policy, 16 (2), 81-91.

Fromm, O. (2000). Ecological structure and functions of biodiversity as elements of its total economic value. Environmental and Resource Economics 16(3), 303-328.

Fry, G.L.A. (1991). Conservation in agricultural ecosystems. In: Spellerberg, I.F., Goldsmith, F.B. and Morris, M.G. (eds.) The scientific management of temperate communities for conservation, Blackwell, London, pp. 415-443.

Garrod, G.D. and Willis, N.D. (1994). Valuing biodiversity and nature conservation at a local level. Biodiversity and Conservation 3, 555-565.

Garrod, G.D. and Willis, K.G. (1995). Valuing the benefits of the South Downs Environmentally Sensitive Area. *Journal of Agricultural Economics* 46(2) 160-173.

Garrod, G.D. and Willis, K.G. (1997). The non-use benefits of enhancing forest biodiversity: a contingent ranking study. *Ecological Economics* 21, 45-61.

Garrod, G.D. and Willis, K.G. (1999). Economic valuation of the environment: methods and case studies. Edward Elgar, UK.

Ghent, A.W. (1991). Insights into diversity and niche breadth analyses from exact small-sample tests of the equal abundance hypothesis. Amer. Midl. Nat. 126, 213-255.

Grime, J.P. (1979). Plant Strategies and Vegetation Processes. Wiley, Chichester, UK.

Grime, J.P. (1998). Benefits of plant diversity to ecosystems: immediate, filter and founder effects. Journal of Ecology, 86 (6), 902-910.

Hammond, D.L., Ricklefs, R.E., Cowling, R.M., Samways, M.J. (1995). Magnitude and distribution of biodiversity. In: Global Biodiversity Assessment, UNEP, pp. 21-106.

Hanley, N. and Craig, S. (1991). Wilderness development decisions and the Krutilla-Fisher model: the case of Scotland's 'low country'. Ecological Economics, 4, 145-164.

Hanley, N. (2001). "Cost-benefit analysis and environmental policy making" Environment and Planning C, 19, 103-118.

Hanley, N., Mourato, S. and Wright, R. (2001). Choice modelling approaches: a superior alternative for environmental valuation? Journal of Economic Surveys, 15, 3, 453-462.

Hanley, N., Oglethorpe, D., Wilson, M. and McVittie, A. (2001). Estimating the value of environmental features, Stage 2 – Draft Final Report to MAFF.

Hanley, N., Willis, K., Powe, N. and Anderson, M. (2002) Non-market benefits of forestry Phase 2: Valuing the benefits of biodiversity in forests. Report to Forestry Commission.

Hanemann, W.M. (1994). Valuing the environment through contingent valuation. Journal of Economic Perspectives, 8 (4), 19-43.

Harper, J.L. and Hawksworth, D.L. (1995) Preface. In: Hawksworth, D.L. (ed.) Biodiversity: measurement and estimation. Chapman and Hall, London, pp. 5-12.

Hawksworth, D.L. (ed.) (1995). Biodiversity: measurement and estimation. Chapman and Hall, London, 140 pp.

Hawksworth, D.L., Lodge, D.J. and Ritchie, B.J. (1994). Fungal and bacterial diversity in the functioning of tropical forests. In: Orians, G.S. and Dirzo, R. (eds.) Ecosystem function and biodiversity in tropical forests. J.Wiley, New York.

Herrera, R.A., Ulloa, D.R., Valdés-Lafont, O., Priego, A.G. and Valdés,A.R. (1997). Ecotechnologies for the sustainable management of tropical forest diversity. Nature and Resources 33, 2-17.

Herriges, J. and Kling, C. (1999). Valuing recreation and the environment Cheltenham: Edward Elgar.

Hills, J.M., Murphy K.J., Pulford, I.D. and Flowers, T.H. (1994). A method for classifying European riverine wetland ecosystems using functional vegetation groups. Functional Ecology 8, 242-252.

Hilton, J., Irish, A.E., Reynolds, C.S. (1992). Active reservoir management. A model solution. In: Sutcliffe, D.W. and Jones, J.G. (eds.) Eutrophication research and application to water supplies. Freshwater Biological Association, Ambleside, pp. 185-196.

Hodgson, J.G., Thompson, K., Wilson, P.J. and Bogaard, A. (1998). Does biodiversity determine ecosystem function? The Ecotron experiment reconsidered. Funct. Ecol. 12, 843-848.

Holland, J.M. and Luff, M.L. (2000). The effects of agricultural practices on Carabidae in temperate ecosystems. Integr. Pest Manage. Rev. 21, 1-21.

Hollingsworth, P.M., Gornall, R.J. and Preston, C.D. (1995). Genetic variability in British populations of *Potamogeton coloratus* (Potamogetonaceae). Plant. Syst. Evol. 197, 71-85.
Hollingsworth, P.M., Preston, C.D. and Gornall, R.J. (1996). Genetic variability in two hydrophilous species of *Potamogeton: P. pectinatus* and *P. filiformis*. (Potamogetonaceae). Plant. Syst. Evol. 202, 233-254.
Hughes, J.B. and Roughgarden, J. (2000). Species diversity and biomass stability. Amer. Nat. 155, 618-627.
Hurlbert, S.H. (1971). The nonconcept of species diversity: a critique and alternative parameters. Ecology 52, 577-586.
Kleijn, D., Berendse, F., Smit, R. and Gilissen, N. (2001). Agri-environment schemes do not effectively protect biodiversity in Dutch agricultural landscapes. Nature 413, 723-725.
Lachavanne, J.B., (1985). The influence of accelerated eutrophication on the macrophytes of Swiss lakes, abundance and distribution. Verh. Int. Verein. Limnol. 22, 2950-2955.
Lehman, C.L. and Tilman, D. (2000). Biodiversity, stability, and productivity in competitive communities. American Naturalist. 156 (5), 534-552. See page 16.
Lid, J. (1952). Norsk Flora. Norske Samlaget, Oslo.
Limburg KE, O'Neill RV, Costanza R, Farber S (2002) Complex systems and valuation. Ecological Economics 41 (3) 409-420.
Loomis J.B. and White D.S. (1996). Economic benefits of rare and endangered species: summary and meta-analysis. *Ecological Economics* 18, 197-206.
Loreau, M. (1998). Biodiversity and ecosystem functioning: A mechanistic model. Proceedings of the National Academy of Sciences, USA. 95 (10), 5632-5636.
Loreau, M. and Hector, A. (2001). Partitioning selection and complementarity in biodiversity experiments. Nature, 412 (6842), 72-76.
Louviere JJ, Hensher DA and Swait JD (2000). *State Choice Methods: Analysis and application*. Cambridge University Press: Cambridge
Lovejoy, T.E. (1980a). Foreword. In: Soulé, M.E. and Wicox, B.A. (eds.) Conservation Biology: an evolutionary-ecological perspective. Sinauer Assoc., Sunderland, Massachusetts, pp. v – ix.
Lovejoy, T.E. (1980b). Changes in biological diversity. In: Barney, G.O. (ed.). The Global 2000 report to the President, Vol. 2 (The Technical Report). Penguin Books, Harmondsworth, pp. 327-332.
Lovejoy, T.E. (1995). The quantification of biodiversity: an esoteric quest or a vital component of sustainable development? In: Hawksworth, D.L. (ed.) Biodiversity: measurement and estimation. Chapman and Hall, London, pp. 81-87.
Macan, T.T., 1977. Changes in the vegetation of a moorland fishpond in twenty-one years. J. Ecol. 65, 95-106.
Macdonald, D. W., Tattersall, F. H., Rushton, S., South, A. B., Rao, S., Maitland, P. and Strachan, R. (2000). Reintroducing the beaver (*Castor fiber*) to Scotland: a protocol for identifying and assessing suitable release sites. Animal Conservation 3, 125-133.
Macmillan, D.C, Hanley, N, and Buckland, S (1996). Contingent valuation of uncertain environmental gains. *Scottish Journal of Political Economy* 43(5), 519-533.
Macmillan, D.C., Duff, E.I (1998). The non-market benefits and costs of native woodland restoration. *Forestry* 71(3), 247-259.
Macmillan, D.C., Duff, E.I. and Elston, D. (2001a). Modelling non-market environmental costs and benefits of biodiversity projects using contingent valuation data. Environmental and Resource Economics, 18(4), 391-340.
Macmillan, D., Daw, M., Daw, D., Patterson, I., Hanley, N., Gustanski, J-A., and Wright, R. (2001b). The economic value of geese. Aberdeen: Department of Agriculture and Forestry, University of Aberdeen.
Macmillan, D.C and Hanley, N.D. (2002) New approaches to data collection in contingent valuation. University of Aberdeen working paper.

Madeira, P.T., Van, T.K. and Center, T.D. (1999). Integration of five Southeast Asian accessions into the world-wide phenetic relationships of *Hydrilla verticillata* as elucidated by random amplified polymorphic DNA analysis. Aquat. Bot. 63, 161-167.

Magurran, A.E. (1988). Ecological diversity and its measurement. Princeton University Press.

Marshall, E.J.P., Baudry, J., Moonen, C., Fevre, E. and Thomas, C.F.G. (1996). Factors affecting floral diversity in European field margin networks. In: Simpson, I. A. and Dennis, P. (eds), The Spatial Dynamics of Biodiversity: Proceedings 5th Annual IALE Conference, Stirling 1996, International Association for Landscape Ecology, Stirling, pp. 97-105.

May, R.M. (1995) Conceptual aspects of the quantification of the extent of biological diversity. In: Hawksworth, D.L. (ed.) Biodiversity: measurement and estimation. Chapman and Hall, London, pp. 13-20.

McGrady-Steed, J. and Morin, P.J. (2000). Biodiversity, density compensation, and the dynamics of populations and functional groups. Ecology 81, 361-373.

McIntyre, S. (1992). Risks associated with the setting of conservation priorities from rare plant species lists. Biol. Conserv. 60, 31-37.

McIntyre, S. and Lavorel, S. (1994). Predicting richness of native, rare, and exotic plants in response to habitat and disturbance variables across a variegated landscape. Conserv. Biol. 8, 521-531.

Morris, M.G. and Perring, F.H. (1974)The British oak: its history and natural history. E.W. Classey, Faringdon.

Moss, B., Johnes, P. and Phillips, G. (1994). August Thienemann and Loch Lomond - an approach to the design of a system for monitoring the state of north-temperate waters. Hydrobiologia, 290, 1-12.

Murphy, K.J. and Hootsmans, M.J. (2002). Predictive modelling of aquatic community attributes: biomass, biodiversity, biointegrity and biomonitoring. Acta Limnol. Bras. 14 (3), 43 - 60.

Murphy, K.J. (2003). Plant communities and plant diversity in softwater lakes of Northern Europe. Aquatic Botany 73, 287 - 324.

Murphy, K.J., Dickinson, G., Thomaz, S.M., Bini, L.M., Dick, K., Greaves, K., Kennedy, M.P., Livingstone, S., McFerran, H., Milne, J.M., Oldroyd, J. and Wingfield, R.A. (2003). Aquatic plant communities and predictors of diversity in a sub-tropical river floodplain: the upper Rio Paraná., Brazil. Aquatic Botany 77, 257 - 276

Murphy, K.J., McCracken, D.I., Foster, G.N., Furness, R.W., Waterhouse, A., Abernethy, V.J., Downie, I., Wilson, W.L., Adam, A. and Ribera, I. (1998). Functional analysis of plant-invertebrate-bird biodiversity on Scottish agricultural land. Vols. 1 - 3. Final Report to SOAEFD. University of Glasgow and Scottish Agricultural College, Glasgow. 266 pp.

Murphy, K.J., Kennedy, M.P., McCarthy, V., Ó'Hare, M.T., Irvine, K. & Adams, C. (2002). A review of ecology based classification systems for standing freshwaters. Report to SNIFFER/ EA. University of Glasgow and Trinity College Dublin.

Murphy, K.J., Rørslett, B. and Springuel, I. (1990). Strategy analysis of submerged lake macrophyte communities: an international example. Aquatic Botany 36, 303-323.

Naeem, S., Thompson, L.J., Lawler, S.P., Lawton, J.H. and Woodfin, R.M. (1995). Empirical evidence that declining species diversity may alter the performance of terrestrial ecosystems. Phil. Trans. R. Soc. Lond 347, 249-262.

Nielsen, S.L. and Sand-Jensen, K. (1997). Growth rates and morphological adaptations of aquatic and terrestrial forms of amphibious *Littorella uniflora* L. Aschers. Plant Ecol. 129, 135-140.

Nilsson, C.. Grelsson, M. and Sperens, U. (1988). Can rarity and diversity be predicted in vegetation along river banks? Biol. Conserv. 44, 201-212.

NOAA – National and Atmospheric Administration. (1993). Report of the NOAA Panel on Contingent Valuation, Federal Register US, 58(10), pp. 4601-4614.

Nolet, B.A. (1997) Management of the beaver (*Castor fiber*): towards restoration of its former distribution and ecological function in Europe. Report, Council of Europe, Strasbourg.

Norse, E.A. and McManus, R.E. (1980). Ecology and living resources biological diversity. In: Environmental Quality 1980: The 11th Annual Report of the Council on Environmental Quality. Council on Environmental Quality, Washington DC, pp. 31-80.
Norse, E.A., Rosenbaum, K.L., Wilcove, D.S., Wilcox, D.A., Romme, W.H., Johnston, D.W. and Stout, M.L. (1986). Conserving biological diversity in our National Forests. The Wilderness Soc., Washington DC.
Norton B (1986). *The preservation of species: the value of biological diversity*. Princeton University Press: Princeton.
Noss, R.F. (1990) Can we maintain biological and ecological integrity. Conservation Biology 4, 241-243.
Nunes, P.A.L.D. and van den Bergh, J.C.J.M. (2001). Economic valuation of biodiversity: sense or nonsense? Ecological Economics 39, 203-222.
O'Hare, M.T. and Murphy, K.J. (1999). Invertebrate hydraulic microhabitat and community structure in *Callitriche stagnalis* Scop. patches. Hydrobiologia 415, 169-176.
O'Neil, J. (1997). Managing without prices: the monetary valuation of biodiversity. *Ambio* 26, 546-555.
OECD (2001). Valuation of biodiversity benefits: Selected studies. OECD: Paris.
Oglethorpe, D., Hanley, N., Hussain, S. and Sanderson, R. A. (2000). Modelling the Transfer of the Socio-Economic Benefits of Environmental Management, Environmental Modelling and Software, 15, 343-356.
Paoletti, M.G. (1999). Invertebrate biodiversity as indicators of sustainable landscapes – practical use of invertebrates to assess sustainable land use. Elsevier, Amsterdam.
Paoletti, M.G., Pimentel, D., Stinner, B.R. and Stinner, D. (1992). Agroecosystem biodiversity: matching production and conservation biology. Agric. Ecosyst. Environm. 40, 3 – 23.
Parish, T., Lakhani, K. and Sparks, T.H. (1994). Modelling the relationship between bird population variables and hedgerow and other field margin attributes. 1. Species richness of winter, summer and breeding birds. J. Appl. Ecol. 31, 764-775.
Pearce, D. (2001). Valuing biological diversity: issues and overview. In (Eds) OECD. Valuation of biodiversity benefits: Selected studies. OECD: Paris.
Pearce, D. and Moran, D. 1994. The Economic Value of Biodiversity. Earthscan Publications Limited, London, UK.
Peet, R.K. (1974). The measurement of species diversity. Ann. Rev. Ecol. Syst. 5, 285-307.
Perlman, D.L. and Adelson, G. (1997). Biodiversity, Exploring Values and Priorities in Conservation. Blackwell, Oxford.
Pieterse, A.H., Ebbers, A.E.H. and Verkleij, J.A.C. (1984). A comparative study on isoenzyme patterns in *Hydrilla verticillata* l.f. Royle from Ireland and north-eastern Poland. Aquat. Bot. 18, 299-303.
Pimm, S.G.R, Gittleman, J, Brooks,T (1995). The future of biodiversity. Science 269, 247-350. 21 July.
Poole, A.F. (1989). Ospreys, a natural and unnatural history. Cambridge University Press, Cambridge.
Portney, P.R. (1994). The contingent valuation debate – why economists should care. Journal of Economic Perspectives. 8 (4), 3-17.
Potter C. (1988). Environmentally Sensitive Areas in England and Wales: an experiment in countryside management. Land Use Policy 5, 301-313.
Prance, G.T. (1991). Rates of loss of biological diversity: a global view. In: Spellerberg I.F., Goldsmith F.B. and Morris M.G. (eds) The scientific management of temperate communities for conservation. Blackwell, London, pp. 27-44.
Prance, G.T. (1995). Species biodiversity in the neotropics. In: Hawksworth, D.L. (ed.) Biodiversity: measurement and estimation. Chapman and Hall, London, pp 89-99
Price, G.R. (1998). The nature of selection. Journal of Theoretical Biology. 175 (3), 389-396.
Ready, R. (1995). "Environmental Valuation Under Uncertainty" in Handbook of Environmental Economics edited by D. Bromley. Blackwells.

Ready, R.C., Navrud, S. and Dubourg, W.R. (2001). How do respondents with uncertain willingness to pay answer contingent valuation questions? Land Economics, 77 (3), 315-326.
Reid, W.V., McNeely, J.A., Tunstall, D.B., Bryant, D.A. and Winograd, M. (1993). Biodiversity indicators for policy makers. World Resources Institute, Washington DC.
Rees, M., Condit, R., Crawley, M., Pacala, S. and Tilman, D. (2001). Long-term studies of vegetation dynamics. Science. 293 (5530), 650-655.
Robinson, G.M. (1994). The greening of agricultural policy: Scotland's Environmentally Sensitive Areas (ESAs). J. Environm. Planning and Manage. 37, 215-231.
Robinson, R.A. and Sutherland, W.J. (2002). Post-war changes in arable farming and biodiversity in Great Britain. J. Appl. Ecol. 39, 157-176.
Rodwell, J.S. (1991-1995). British Plant Communities: Volumes 1 - 4. Cambridge University Press, Cambridge.
Roelofs, J.G.M. (1983). Impact of acidification and eutrophication on macrophyte communities in soft waters in The Netherlands. I. Field observations. Aquat. Bot. 17, 139-155.
Rørslett, B. (1991). Principal determinants of aquatic macrophyte richness in northern European lakes. Aquatic Botany, 39:173-193.
Roy, K. and Foote, M. (1997). Morphological approaches to measuring biodiversity. Trends Ecol. Evol. 12, 277-281.
Sala, O.E. and 18 others (2000). Biodiversity – global biodiversity scenarios for the year 2100. Science 287, 1770-1774.
Samples K.C , Dixon, J.A. and Gowen M.M. (1986). Information disclosure and endangered species valuation. *Land Economics* 62, 306-312.
Santos, J.M.L. (1999) "Accounting for variation in contingent valuation estimates of landscape benefits: genuine value differences or method-led divergences?" Paper presented to annual EAERE conference, Oslo, June.
Scheffer, M. and Beets, J. (1994). Ecological models and the pitfalls of causality. Hydrobiologia, 275/276, 115-124.
Scheffer, M. (1992). Fish and nutrients determine algal biomass: a minimal model. Oikos 62, 271-282.
Scheffer, M. (1998). Ecology of shallow lakes. Chapman and Hall, London.
Scheffer, M., Carpenter, S., Foley, J.A., Folkes, C. and Walker, B. (2001). Catastrophic shifts in ecosystems. Nature 413, 591-596.
Shrestha, R.K. and Loomis, J.B. (2001). Testing a meta-analysis model for benefit transfer in international outdoor recreation. Ecological Economics, 39 (1), 67-83.
Simpson, I.A. and Dennis, P. (eds), The Spatial Dynamics of Biodiversity: Proceedings 5th Annual IALE Conference, Stirling 1996, International Association for Landscape Ecology, Stirling, pp?.
Simpson, I.A., Hanley, N., Parsisson, D. and Bullock, C.H. (1996). Indicative prediction of botanical diversity change in Environmentally Sensitive Areas. In: Simpson I.A. and Dennis P. (eds), The Spatial Dynamics of Biodiversity: Proceedings 5th Annual IALE Conference, Stirling 1996, International Association for Landscape Ecology, Stirling, pp. 71-78.
Spash C.L. (1993). Economics, ethics and long term environmental damages. Environmental Ethics 15, 117-132.
Spash C.L. and Hanley, N. (1995). Preferences, information and biodiversity preservation. Ecological Econoimcs, 12, 191-208.
Souza-Stevaux, M.C., Negrelle R.R.B. and Citadini-Zanette, V. (1994). Seed dispersal by the fish Pterodoras granulosus in the Paraná River Basin, Brazil. J. Trop. Ecol. 10, 621-626.
Srestha, R.K. and Loomis, J. (2001). Testing a meta-analysis model for benefit transfer in international outdoor recreation. Ecological Economics, 39 (1), 67-84.
Stace, C. (1991). New Flora of the British Isles. Cambridge University Press, Cambridge, UK. 1226 pp.

Steneck, R.S. and Dethier, M.N. (1994). A functional group approach to the structure of algal-dominated communities. Oikos 69, 476-498.
Symstad, A.J., Siemann, E. and Haarstad, J. (2000). An experimental test of the effect of plant functional group diversity on arthropod diversity. Oikos 89, 243-253.
Szmeja, J. and Clément, B. (1990). Comparaison de la structure et du déterminisme des *Littorelletea uniflorae* en Poméranie (Pologne) et en Bretagne (France). Phytocoenologia 19, 123-148.
Templeton A.R. (1995) Biodiversity at the molecular genetic level: experiences from disparate macroorganisms. Phil. Trans. Roy. Soc. London B 345, 59 – 64
Ten Kate K and Larid S (1999). *The commercial use of biodiversity*. Earthscan, London.
Turnbull, L.A., Rees, M. and Crawley, M.J. (1999). Seed mass and the competition/colonization trade-off: a sowing experiment. Journal of Ecology, 87 (5), 899-912.
Turner, R.K., Brouwer, R. and Georgiou, S. (2001). Ecosystem functions and the implications for economic evaluation. English Nature Research Report Number 441. English Nature, Northminster House, Peterborough, UK.
van Dam, H. and Kooyman-van Blokland, H. (1978). Man-made changes in some Dutch moorland pools as reflected by historical and recent data about diatoms and macrophytes. Int. Rev. ges. Hydrobiol. 63, 587-607.
Vöge, M. (1997a). Plant size and fertility of *Isoetes lacustris* L. in 20 lakes of Scandinavia: a field study. Arch. Hydrobiol. 139, 277-287.
Vöge, M. (1997b). Number of leaves per rosette and fertility characteristics of the quillwort *Isoetes lacustris* L. in 50 lakes of Europe: a field study. Arch. Hydrobiol. 139: 415 - 431.
Wallsten, M. (1981). Changes of lakes in Uppland, central Sweden, during 40 years. Symb.. Bot. Ups.. 23, 3, 1-84.
Warren, J. and Topping, C.J. (1999). A space occupancy model for the vegetation succession that occurs on set-aside. Agriculture Ecosystems and Environment, 72 (2), 119-129.
Warren, J., Wilson, F. and Diaz, A. (2002). Competitive relationships in a fertile grassland community - does size matter? Oecologia, 132 (1), 125-130.
Watkinson, A.R., Freckleton, R.P., Robinson, R.A. and Sutherland, W.J. (2000). Predicting biodiversity responses to GM-herbicide-tolerant crops. Science 289, 1554-1556.
White, P.C.L., Gregory, K.W., Lindley, P.J. and Richards, G. (1997). Economic values of threatened mammals in Britain: A case study of the otter Lutra lutra and the water vole Arvicola terrestris. Biological Conservation, 82 (3), 345-354.
White, P.C.L., Bennett, A.C. and Hayes, E.J.V. (2001). The use of willingness-to-pay approaches in mammal conservation. Mammal Review, 31, 2, 151-167.
Whittaker, R.H. (1977). Evolution of species diversity in land communities. Evolutionary Biology, 10, 1-67.
Willby, N.J., Murphy, K.J. and Pulford, I.D. (1998). A Scottish flood-plain mire: the Insh Marshes, Strathspey. Scottish Geog. Magazine, 114, 13-17.
Williams P.H., Humphries C.J. and Gaston K.J. (1994). Centres of seed-plant diversity: the family way. Proc. R. Soc. Lond. B 256, 67-70.
Willis, K.G. (1990). Valuing non-market wildlife commodities: an evaluation and comparison of benefits and costs. Applied Economics, 22, 13-30.
Willis, K.G., Garrod, G.D. and Saunders, C.M. (1995). Benefits of Environmentally Sensitive Area Policy in England: Contingent Valuation Assessment. Journal of Environmental Management, 44, 105-125.
Willoughby, M. (1992). Species diversity and how it is affected by livestock grazing in Alberta. Range Notes No. 13, Alberta Forestry, Lands and Wildlife Publication No. T/207, Lethbridge, Alberta.
Wilson, E.O. (1988). Biodiversity. National Acad. Press, Washington DC.
Wilson, M.V. and Mohler, C.L. (1983). Measuring compositional change along gradients. Vegetatio, 54 (3), 129-141.

Wilson, W.L., Abernethy, V.J., Murphy, K. J., Adam, A., McCracken, D.I.,Downie, I.S, Foster, G.N., Furness, R. W., Waterhouse, A. and Ribera, I. (2003). Prediction of plant diversity response to land use change on Scottish agricultural land. Agric. Ecosystems Environment 94, 249 - 263

Wilson, W.L., Day, K.R., Hart, E.A. (1996). Predicting the extent of damage to conifer seedlings by the pine weevil (*Hylobius abietis* L.): a preliminary risk model by multiple logistic regression. New Forests 12, 203-222.

Wingfield, R. and Murphy, K.J. (2002). Assessing and predicting the success of *Najas flexilis*, a rare aquatic macrophyte in relation to lake environmental conditions *Proc. 11th EWRS International Symposium on Aquatic Weeds, Moliets, France,* 59 - 62..

Yachi, S. and Loreau, M. (1999). Biodiversity and ecosystem productivity in a fluctuating environment: the insurance hypothesis. Proc. Natl. Acad. Sci. USA 96, 1463-1468.

11. Appendices

Appendix 1: Review of UK valuation studies of biodiversity.

Appendix 2: Valuation SMSS. Mean score (Standard deviations in parenthesis)

Appendix 3: Interviewer's scripted used in the valuation studies

Appendix 4: PowerPoint presentation used in the valuation study

Appendix 5: Interviewer scripted used in the valuation workshops, Northumberland