Forest Research

Links between biodiversity and rotation length

September 2017

Vadim Saraev
Nadia Barsoum
Robin Gill
Laura Henderson
Andrew Peace
Chris Quine
Gregory Valatin

The Research Agency of the
Forestry Commission

Forest Research is the Research Agency of the Forestry Commission and is the leading UK organisation engaged in forestry and tree related research. The Agency aims to support and enhance forestry and its role in sustainable development by providing innovative, high quality scientific research, technical support and consultancy services.

Contents

Introduction 8

Aims and Objectives 12

Methodology 13

Results 14

Ecological evidence on biodiversity 14
Studies on changes in biodiversity with stand age 14
Studies on impacts of forest management interventions on biodiversity 16
Defra Environmental metrics and biodiversity 18
Birds and Mammals 19
Evidence from the UK Biodiversity Assessment Project 20
Economic evidence on Biodiversity 22

Discussion and Recommendations 26

Quantifying biodiversity 26
Links between biodiversity and stand age 27
Links between biodiversity and optimal rotation length 28
Economic valuation of biodiversity 30

References 33

Appendices 40

Annex I: Biodiversity: Definitions and Policy Context 40

Definitions 40
Policy context 41
References for Annex I 42

Annex II: Evidence Review: Links between biodiversity and rotation length 44

Introduction 44
Types of biodiversity and stand age relationship 44
Literature Search 47
Results 49
Discussion 50
Reference for Annex II: 51

Annex III: Forest Management Interventions to increase Biodiversity 52

Introduction 52
Methods 52
Summary of literature 53
Species-level research and biodiversity 53

Biodiversity Indicators and silviculture 59
Limitations of indicators 60
Recommendations for management 60
Forestry and Metrics on Biodiversity 65
Conclusion 67
References for Annex III: 69

Annex IV: Literature search protocols 75

Data 78

Annex V: Valuing Biodiversity 79

Methods for valuing biodiversity 79
Biodiversity valuation estimates 83
References for Annex V 85

List of Figures

Figure 1. Number of studies indicating an increase, decrease or neutral response to stand age by taxonomic group/biodiversity indicator 16
Figure 2. Relationship between vertebrate species richness and relative abundance and stand age in British forests 20
Figure 3. Biodiversity response through the stages of forest growth from pre-thicket to mature, in Sitka spruce upland forests 22
Figure 4. Schematic of some possible relationships between biodiversity (y axis) and rotation length (x axis) 44
Figure 5. Non-market valuation methods 80

List of Tables

Table 1. Management systems and biodiversity 18
Table 2. Summary information collected for each of the reviewed papers. 47
Table 3. Number of studies showing temporal change in species diversity with stand age and model types that best describe these temporal changes 51
Table 4. Generalised characteristics of different management regimes as positive or negative for biodiversity. 56
Table 5. Feature criteria and condition assessment categories used in biodiversity offsetting metric (Natural England 2010). 66
Table 6. Tags used to describe papers (in alphabetic order) 76
Table 7. Examples of searches (through selected papers) with tags 77
Table 8. Woodland biodiversity value estimates 84

Executive Summary

Biodiversity has emerged as a major focus of international environmental policy and practice (see Annex I for definitions). Its crucial function in sustaining life on earth and underpinning the provision of all ecosystem services has become increasingly recognised, including in the 1992 Convention on Biological Diversity and in ecosystem assessments such as the Millennium Ecosystem Assessment and the UK National Ecosystem Assessment.

Currently the approach to achieving UK forest biodiversity objectives tends to be primarily in terms of regulation and pursuit of specific opportunities and less often through mainstream forest management practices (e.g. rotation length decisions). However, there may be potential for integrating biodiversity into such mainstream forest management decisions as part of a wider 'ecosystem services' approach, taking account of evidence linking biodiversity to stand age and forest structure.

The overarching aim of this study was to examine links between biodiversity and rotation length with a view to exploring how biodiversity could be incorporated into an optimal rotation length model. This is part of a wider agenda of accounting for the multiple benefits provided by woodlands in decision-making frameworks for sustainable forest management. Extending traditional models focusing upon timber production to the wider benefits of woodlands will facilitate more comprehensive comparisons between management alternatives characterised by different rotation lengths.

The methodology adopted for the study mainly relied on a review of international literature and analysis of UK biodiversity assessment project data. Four distinct reviews were carried out covering links between biodiversity and stand age (Annex II), optimum management interventions for biodiversity and a review of existing indicators and metrics (Annex III), a re-analysis of data from the Forestry Commission Biodiversity Assessment Project and a review of approaches to economic valuation of biodiversity with examples (Annex V).

On the issue of quantifying biodiversity and its links to a stand age the study reached the following conclusions:

● the concept of biodiversity is complex with no consensus at present in either the ecological literature or the economic literature on how best to quantify it. A consequence is the difficulty of encompassing all the aspects of interest in a single metric.

● although there is no universal response to stand age, with variations between taxa and sites (as might be expected), overall there is more ecological evidence in published studies of increasing species richness with stand age, than of a fall (or of no change).

● the literature on habitat requirements for birds and mammals in British forests suggests that after a brief initial increase in species richness and relative abundance with stand age, these both then fall in the thicket stage (with a minimum at around 20 years), thereafter increasing again (though some of the increase is only obtained at ages beyond typical rotation lengths). (Thicket stage: the stage of forest growth after canopy closure when lower branches of the trees meet and interlace to form a dense, often impenetrable growth. Especially applied to conifer forests of some 10 to 20 years of age.)

● evidence on the impact of alternative forest management approaches such as continuous cover forestry and how these differ from traditional even-aged stands with respect to biodiversity is currently sparse.

The review found that economic literature linking biodiversity and rotation length is also sparse. In particular economic aspects of biodiversity and rotation length are very rarely adequately treated in research papers or their relationship investigated in any depth. Nevertheless, some strands of research were identified of potential help in progressing investigation of links between biodiversity and rotation length:

● a direct approach to including biodiversity in optimal rotation modelling is possible where one can link biodiversity values and rotation length in an economic framework that includes robust existing monetary estimates of biodiversity (or its proxy) and a knowledge of how biodiversity changes over a rotation. However, such circumstances are rare at present.

● an indirect approach is possible when the biodiversity value is unknown but a set of management constraints imposed (e.g. on leaving standing deadwood) in order to conserve biodiversity affects the choice of the optimal rotation length. In this case incorporating these constraints and associated costs in the optimisation problem is required.

● optimisation with biodiversity benefits included may be best accomplished in a multi-stand model (i.e. a mosaic of different stand ages) rather than the more common single stand approach, as different species will respond differently to forest stand age

● as some forestry production processes, e.g. the relationship between profit and production of ecosystem services, e.g. timber production and biodiversity, are non-linear, it may be optimal to apply separate management objectives to different stands within a forest in some cases. This is a further reason to favour multi-stand modelling.

As a result of the above findings the following recommendations are proposed:

Recommendation 1: *Further work on multi-stand forest modelling and the nature of the relationship between profits and the production of ecosystem services by UK forestry is needed. This would permit the identification of conditions under which it is optimal to separate the management of stands by forestry objective.*

Recommendation 2: *Given the presence of climate change uncertainties and still largely unknown ecological interactions among species, empirical work is needed to investigate conditions under which ecosystem resilience can be maximised (i.e. fostering high levels of redundancy, particularly among species with known important functional roles).*

Recommendation 3: *For vertebrates in British forests, adopting the U-shaped generalised response to stand age estimated from their habitat requirements is recommended, instead of hypothesizing a simple linear relationship with stand age up to a maximum.*

Recommendation 4: *Work linking evidence on biodiversity and stand age to wider research on climate change impacts is needed if mistakes associated with assuming that future relationships will be the same as those that held in the past are to be minimised.*

Recommendation 5: *Fundamental work to develop methods which can provide robust estimates of woodland biodiversity values is still needed, and should take account of insights from behavioural economics on the influence of cognitive factors on stated preferences (e.g. related to framing, information provision and format, and learning).*

Recommendation 6: *Where the reliability of existing monetary biodiversity estimates is in doubt, consideration should be given to employing a range of estimates in order to reflect existing uncertainties, or to incorporating biodiversity in optimisation models using other approaches, such as multi-criteria analysis. Where mandatory forest management measures to preserve biodiversity are incorporated as costly constraints in forestry optimization, the impact of these costs on the NPV may serve as proxy for the value society places on woodland biodiversity conservation in the locality focused upon.*

Links between biodiversity and rotation length

Introduction

Biodiversity is the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems. It is a complex concept which is difficult to quantify and therefore also to value.

Nevertheless, biodiversity conservation and restoration has become a major focus of international environmental policy and practice. Its vital role in sustaining life on earth has become increasingly recognised, including in the Convention on Biological Diversity (CBD) signed by 150 governments at the 1992 Rio Earth Summit. The Millennium Ecosystem Assessment characterizes biodiversity as underpinning the provision of all ecosystem services (Millennium Ecosystem Assessment (MA), 2005), with the UK National Ecosystem Assessment (UK National Ecosystem Assessment, 2011, p. 64) similarly viewing it as sustaining and supporting the functioning of all ecosystems and thus the delivery of all ecosystem services.

Biodiversity may be thought of as an intergenerational, global public good (Helm and Hepburn, 2012). It has a role not only directly in the delivery of 'final goods' such as recreation and tourism, landscape and aesthetic amenity and ecological knowledge, but can affect other ones too (e.g. timber production) indirectly through its contribution to 'intermediate' ecosystem services such as soil fertility and pest control (eftec, 2011; UK National Ecosystem Assessment, 2011).

Estimating aggregate economic values for biodiversity, as opposed to valuing marginal changes, can be theoretically problematic (Helm and Hepburn, 2012, p. 6). However, it is notable that biodiversity values estimated in previous studies are relatively large compared to most other woodland ecosystem service values. For example, the non-use value of biodiversity for 0.5 million ha (less than 20% of the total area) of British forests has been estimated (reflated to 2013 prices) at £500 million p.a. (Willis et al., 2003), more than double the estimated current annual revenue from UK timber production.(For the definition of 'non-use value', see the glossary below). Valuing the 10 million green tonnes of softwood and 0.5 million green tonnes of hardwood produced in the UK in 2012 (FC, 2013) at an average price for Forestry Commission coniferous standing sales of £14 per cubic metre overbark (approximately £17 per green tonne) in the year to September 2012 (FC Timber statistics, http://www.forestry.gov.uk/forestry/infd-7aql5b), would suggest an annual revenue of UK timber production of the order of £170 million currently. Almost 90% of the hardwood came from private sector (non FC/FS) woodlands for which standing sales price data is sparse.

Glossary

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

One measure of biodiversity is species diversity which is most commonly associated with **species richness** i.e. the number of individual species present in an area but may also include measures of abundance. **Relative species abundance** in a particular ecosystem (e.g. lowland coniferous forests) is calculated by dividing the number of species found at one field site by the total number of species found at all field sites. For example, one may have found 1,000 fern species across 20 field sites; then if 500 species of fern is found at a particular site its relative species abundance is 50%.

Public good is a good that is both non-excludable and non-rival in that individuals cannot be effectively excluded from use and where use by one individual does not reduce availability to others.

Non-use value is the value that people assign to goods / services even if they never have and never will use them. Non-use values are associated with knowledge that environmental resources continue to exist (existence value), or are available for others to use now (altruistic value) or in the future (bequest value).

An **arthropod** is an invertebrate animal having an exoskeleton (external skeleton), a segmented body, and jointed appendages (paired appendages). Arthropods form the phylum Arthropoda, which includes the insects, arachnids, myriapods, and crustaceans. (from Wikipedia, accessed 2 June 2016)

Hartman optimal rotation. Richard Hartman in 1976 (Hartman, 1976) considered how the optimal rotation length changes if one also considers other ecosystem services provided by forests that depend on the age of the forest. These "amenity services" of forests are considered to include recreation, wildlife, wilderness, visual amenity, water quality, carbon sequestration and biodiversity. It is Hartman's assumption that these services depend only on forest age.

Ecological resilience can be defined as the amount of disturbance that an ecosystem could withstand without changing self-organized processes and structures (defined as alternative stable states), i.e. before flipping into a different stable state (Gunderson, 2000).

Currently the approach to achieving UK forest biodiversity objectives (Defra, 2011; FC 2011) tends to be primarily in terms of regulation (e.g. conservation of areas associated with protected species and habitats), and pursuit of specific opportunities (e.g. for habitat restoration, landscape connectivity, structural and tree species diversity), and less often through mainstream forest management practices (e.g. rotation length decisions). However, there may be potential for adopting a more nuanced approach to mainstream forest management decisions based upon a wider 'ecosystem services' (ES) approach, taking account of evidence linking biodiversity to stand age and forest structure. The ES approach was adopted as the primary framework for action under the CBD (http://www.cbd.int/ecosystem/) and described by the Conference of the Parties at its Fifth Meeting in 2000 as 'a strategy for the integrated management of land, water and living resources that promotes conservation and sustainable use in an equitable way' (http://www.cbd.int/decision/cop/default.shtml?id=7148).

There is also potential for accounting for changes in biodiversity in modelling other forestry management alternatives at stand level – including Continuous Cover Forestry (CCF), in forest management planning at landscape (and wider) scales, as well as in appraising forestry policy options more widely. For example, the Slowing the Flow study (Nisbet *et al.*, 2011) illustrates an approach where biodiversity is characterized as varying with stand age.

However, agreeing a definition (or definitions) of biodiversity is a necessary key initial step prior to any such development. As noted above, biodiversity is a complex concept. It can be considered at multiple levels (e.g. the level of genes, populations, species, communities and ecosystems) and can be measured in different ways. Definitions of biodiversity are still subject to active debate (Helm and Hepburn, 2012), but some of the most authoritative and widely used are presented in Annex I.

The Forestry Commission currently publishes statistics on species richness (i.e. number of different species) and vegetation condition for GB coniferous and broadleaved woodlands based upon Countryside Survey data (FC, 2013, Table 5.4). This confirms that broadleaved woodlands are generally associated with a greater diversity of plant species than coniferous ones, but does not show how species richness varies with stand age. Biodiversity can also vary across different bio-geographical zones and depend upon the species mix, forest management approach and stand structure. Although sometimes viewed as a particular feature of old-growth forests, the relationship between stand age and biodiversity may be complex, as some species may thrive when trees are younger, or prior to canopy closure. Furthermore, the relationship with stand age may also be influenced by geographical location (e.g. related to limits in the range of particular species, soil quality), land use history (e.g. continuity of woodland presence), tree species present, origin of the forest stand (natural regeneration vs. planted) and potentially affected by climate change too.

For biodiversity to have a noticeable impact on optimal rotation length it must change significantly with age and / or have a high value (relative to timber) – as can be seen by considering solutions to the Hartman model (Hartman, 1976).

Aims and Objectives

The overarching aim of the study was to examine links between biodiversity and rotation length with a view to exploring how biodiversity could be incorporated in an optimal rotation length model. This is part of a wider agenda of accounting for the multiple benefits provided by woodlands in decision-making frameworks for sustainable forest management. An optimal rotation length model currently under development by Forest Research covers several ecosystem services (e.g. carbon sequestration and timber production) as well as accounting for specific risks (e.g. windthrow) that may be affected by climate change. Extending traditional models focusing upon timber production via inclusion of the wider benefits of woodlands will facilitate more comprehensive comparisons between management alternatives to be made, including those which may involve leaving stands unmanaged, or with minimal harvesting intervention.

The primary objectives to be addressed by the study were:

1. To review approaches taken in previous work incorporating biodiversity in rotation length models.
2. To review evidence of how biodiversity varies with the age of woodlands, between woodland types, using woodland biodiversity indicators and considering different forest management approaches (mixed age/single age stands, mixed species/single species, coppice rotations) and climatic conditions.
3. To provide recommendations on the approach to adopt to incorporate biodiversity in developing optimal rotation length models, including the potential to extend these models to include impacts of climatic change. In addition, to highlight any associated research needs or gaps identified during the course of the study.

Following consultation with the Forestry Commission Corporate and Forestry Support, the study was broadened to address the following additional objectives:

4. To discuss whether and how biodiversity can be reliably monetised in an optimisation model – including the extent to which this is only feasible at a whole woodland scale, rather than at stand level.
5. In considering evidence on a) biodiversity supported by different stages of stand development and b) whether there are some stages that support higher overall levels of biodiversity than others, give particular attention to elements that have known cultural and ecosystem function value (e.g. butterflies, wood ants, earthworms).

6 To consider evidence on the influence of spatial and landscape scale issues, including open space within woodlands, on biodiversity.

7 To consider evidence on the impact of wider forest management approaches such as CCF and how these differ from traditional even-aged stands with respect to biodiversity.

8 To explore existing metrics of biodiversity and whether these can be reliably monetised for UK woodlands.

Methodology

The study was primarily based on a review of international literature in line with the Government Social Research Service (GSR) Rapid Evidence Assessment (REA) guidance (GSR, 2013) and extensive field data analysis from the 52 stand plots of the Forestry Commission biodiversity assessment project established in 1993 (Humphrey, Ferris and Quine, 2003). Ecological and economic evidence is summarised separately, before drawing together these strands in discussing broader underlying issues, overall study conclusions and recommendations.

The study adopts an ecosystem services approach whereby the contribution of biodiversity to human wellbeing is the focus. The Convention on Biological Diversity (CBD) defines the ecosystem approach as "a strategy for the integrated management of land, water and living resources that promotes conservation and sustainable use in an equitable way" (Secretariat of the Convention on Biological Diversity, 2005, Decision V/6 The ecosystem approach). Ecosystem services are the means by which ecosystems provide benefits to people and ultimately an ecosystem services approach relies on valuation of ecosystem services that benefit people (Ingram, Redford and Watson, 2012). This may lead to issues with species without utilitarian or economic value and ecological processes that do not directly benefit people (which may include critical primary and intermediary ecological functions).

To keep the main body of the report succinct, supporting material and analyses have been placed in Appendices, where specific methods adopted are detailed. Annex I provides definitions of biodiversity, annexes II and III provide ecological reviews of links between biodiversity and rotation length and management approaches, annex IV contains details of an initial literature search and annex V provides a review of approaches to economic valuation of biodiversity with examples.

Results

Results are presented in two parts. First, we present ecological evidence on biodiversity and stand age. This evidence would be required to underpin any economic analysis. The evidence comes from literature reviews and analysis of experimental data from field plots. Second, we present economic evidence on biodiversity and stand age modelling and valuation.

Ecological evidence on biodiversity

The ecological evidence comprises five components: a literature review of how biodiversity varies with stand age; a literature review of biodiversity and forest management; a review of existing biodiversity indicators and metrics; a review focusing on birds and mammals; a re-analysis of data from Forestry Commission Biodiversity Assessment Project. These are summarised in the following sections, with fuller versions provided in the supporting papers presented in annexes II and III.

Studies on changes in biodiversity with stand age

A literature search identified 124 studies of potential relevance to determining the existence and nature of links between biodiversity and stand age. Many of these compared species diversity in stands of different ages at a given location (i.e. biodiversity for a 'chronosequence' of stands). Changes in species diversity were most frequently reported in terms of species richness (i.e. number of species present), but proxy biodiversity indicators were also sometimes assessed (e.g. volume of coarse woody debris, stand structural diversity). Analysis of the papers aimed to establish the shape of the response curves between biodiversity (or taxonomic group/species) and rotation age. A suite of 13 potential types of response curves is identified in Annex II drawing upon prior expert judgement.

A high proportion of the studies focussed on arthropod diversity and approximately half of these investigated beetles, quantified through field work. Ground vegetation and lichen also featured frequently with fewer studies on birds, mammals and fungi.

For many of the taxa, stand age had mixed effects on species diversity, depending on the study and species group. For example, with regard to arthropod species diversity there were 18 cases of a significant positive effect (i.e. increasing species diversity) with stand age, 12 cases of a negative effect (i.e. declining species diversity with stand age) and 6 cases where stand age had no effect. A clearer trend was observed for lichens, where the majority of studies (n=20) revealed a positive effect of stand age on species diversity with many of these following a linear relationship. Coarse woody debris was an indicator also found most frequently to be increasing with stand age.

Annex II provides a summary table of the response curves identified by taxonomic group or indicator. The summary demonstrates that for some taxa and indicators there is considerable diversity in the functional response (e.g. whether positive, negative, linear, non-linear, etc.). This suggests that the total biodiversity response to stand age (i.e. comprised of all measured taxa) could be rather flat, given that different components move in different directions. For instance, arthropods had 11 different response curves to stand age, ground vegetation had 9, coarse woody debris had 5. For many of the taxonomic groups – especially those which displayed less diversity in their responses (bryophytes, fungi, mammals etc.) - the sample sizes were insufficient to indicate a clear trend in response.

It is clear from the range of responses revealed by the studies that an overarching relationship between biodiversity (or even diversity within taxonomic groups) and stand age is not easily identified. This is likely to be due to the many influences on biodiversity in woodlands: site history, tree species composition, topography, climate etc. Overall, however, there was most evidence of an increase in biodiversity with stand age: of the studies with statistically significant results, where p was at least <0.05, 66 indicated an increase in species richness with stand age, 33 a decrease and 10 were neutral. These results are summarised in Figure 1 below.

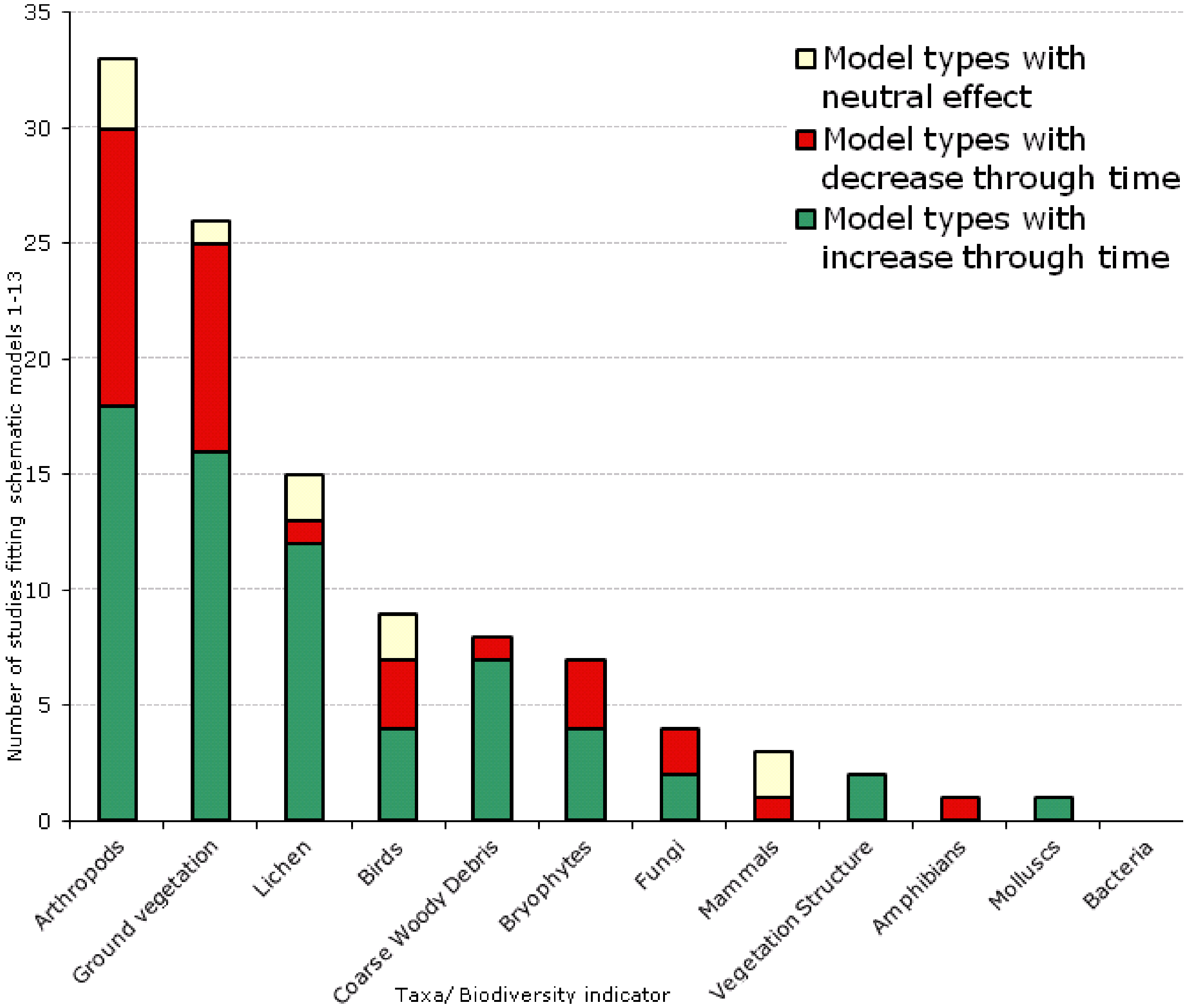


Figure 1. Number of studies indicating an increase, decrease or neutral response to stand age by taxonomic group/biodiversity indicator

Studies on impacts of forest management interventions on biodiversity

A literature search identified 167 studies of potential relevance to elucidating the impacts of different forest management interventions on biodiversity. As was found in the process of reviewing evidence on stand age and biodiversity, the review revealed many complications associated with aggregating results from individual studies in an attempt to assess general trends in total biodiversity. The most substantial problems identified were:

- The usefulness of measuring overall species richness or abundance is limited and can favour early successional species at the expense of the woodland specialists most under threat.
- Time scales of studies are often too short to determine an effect.
- The choice of taxon or taxa (for study) is biased or not sufficiently representative of biodiversity as a whole.
- Results are often contradictory and different taxa respond in different ways.
- Site conditions are often not truly comparable. (Conditions are also not always sufficiently well described to determine whether sites are 'equivalent' or not).
- The scales of studies are rarely equivalent

The difficulties associated with comparing studies meant that there was insufficient evidence in the literature to identify which interventions might be most beneficial for enhancing / optimising biodiversity. These would depend on individual site conditions and on a multi-scaled approach: incorporating multiple stands and the broader landscape.

The literature did reveal certain characteristics of specific management systems that are beneficial or detrimental to aspects of biodiversity and these are summarised in Table 1 below. (A more detailed table is available in the supporting paper in Annex III linking these characteristic to species groups of interest.)

Table 1. Management systems and biodiversity

Silviculture	Characteristics identified as positive for biodiversity	Characteristics identified as detrimental to biodiversity
Clear cut	Large open spaces Refuge for grassland species in intensively managed arable landscapes Provision of edge habitat Providing horizontal diversity on a landscape scale	Even aged Structure Lack of horizontal and vertical stand complexity Structure favours generalists and excludes woodland specialists Management technique precludes many species Lack of natural regeneration Lack of tree species diversity
Coppice	Permanent and temporary open space Standard trees Varied ground flora Structural diversity Deadwood in abandoned coppice	Lack of deadwood in active coppice Lack of tree species diversity Lack of structural diversity associated with abandoned or over mature coppice
Selection felling	Stand continuity Structural complexity Standing biomass Tree Age distribution Gap release and open areas Horizontal diversity	Few refuges for species susceptible to disturbance Open areas can be too small to benefit a full suite of open habitat species Absence of large veteran trees
Shelterwood	Structural Diversity in mid storey Canopy trees Seedling regeneration	Lack of open space, ground flora and microhabitats Lack of horizontal diversity Even aged structure and lack of mature forest

Consideration of the factors in Table 1 could help to inform management decisions as they also broadly equate to the indicators that are widely recognised as proxies for biodiversity (e.g. deadwood, structural diversity etc.). Such indicators have obvious limitations – they *indicate* rather than *determine* - and they do not take account of rare species or species with particular cultural value. However, they can be used to inform principles of management applicable to all silviculture systems that increase the potential for biodiversity i.e. high tree species diversity, diverse age distribution of trees, areas of open space etc. The literature identified that certain taxonomic groups may be more likely to benefit from structural characteristics of different management systems (see extended table in supporting document). However, as the response to management is not consistent within taxonomic groups, conservation of particular species of ecosystem service interest (e.g. butterflies, wood ants, earthworms etc.) would require a more individually targeted response taking specific site, species and landscape characteristics into consideration.

Defra Environmental metrics and biodiversity

The criteria used in environment metrics by Defra for valuing different habitats for environmental offsetting were examined but found to be insufficiently refined, in their current form, for determining the value of different forms of woodland biodiversity. The existing metric focuses upon the condition and 'distinctiveness' of a habitat, with its ultimate worth based on an accumulation of non-monetary units . As the metric is a multi-habitat approach designed to identify equivalence of one habitat with another, the criteria for establishing woodland condition are very broad (i.e. diverse tree age and structure; deadwood over 20 cm diameter is present etc.). The factors have no thresholds or weightings (e.g. what constitutes diverse is not quantified). Furthermore the criteria for condition are based on semi-natural woodland and do not take into consideration the structure of plantations or commercially managed forests.

Agreement on a full suite of woodland biodiversity indicators would be required to develop this metric for the purpose of valuing forest biodiversity. Threshold values or weightings which were applicable across all woodland types would also need to be established. For instance, how many tree species represent ‘good’ diversity and whether this was of equivalent worth to biodiversity as the presence of deadwood etc., would need to be established. Furthermore it would need to be established if the metric was to be applied to all woodlands or tailored to individual specific management types. Whilst it may, therefore, be possible to develop such a metric in the long term, this would require additional research or expert consultation. The indicators identified in this report as most associated with biodiversity and forest management could viably form the basis of such a metric. Any requirement to incorporate economic assessments into such a metric would also require investigation into the specific monetary values of biodiversity indicators in the UK for which there is currently little robust data.

Birds and Mammals

A separate analysis was undertaken for birds and mammals, as these often feature prominently in management objectives.

The response of birds and mammals to stand age, in terms of species richness and relative abundance was assessed from evidence of habitat requirements reported in the literature (Fuller, 2003; Harris and Yalden, 2008). Use of each age class was scored 0–3 representing the frequency of use for each species (0 none; 1 occasional; 2 moderate; 3 high). In total 73 bird species and 49 mammal species were considered.

The results for relative abundance of both birds and mammals indicated a small peak in the use of young stands with a small decline in the thicket stage, followed by an increase in the relative use of mature stands (Figure 2).

The results point to diversity generally increasing with rotation length, with a dip in the early part of mid-rotation (at around 20 years) and the greatest value occurring in mature stands (Figure 2). These changes reflect the broad changes in vegetation structure accompanying stand ageing, namely the initial establishment of herb and shrub layers followed by replacement by thicket and then a mature tree canopy. The development of a mature stand is typically associated with partial regrowth of ground and shrub vegetation as well as increasing amounts of dead wood and canopy depth.

The response of species richness to stand age was much weaker than that for relative abundance, suggesting that age-related changes affect habitat suitability for many species but the overall number of species remains relatively constant.

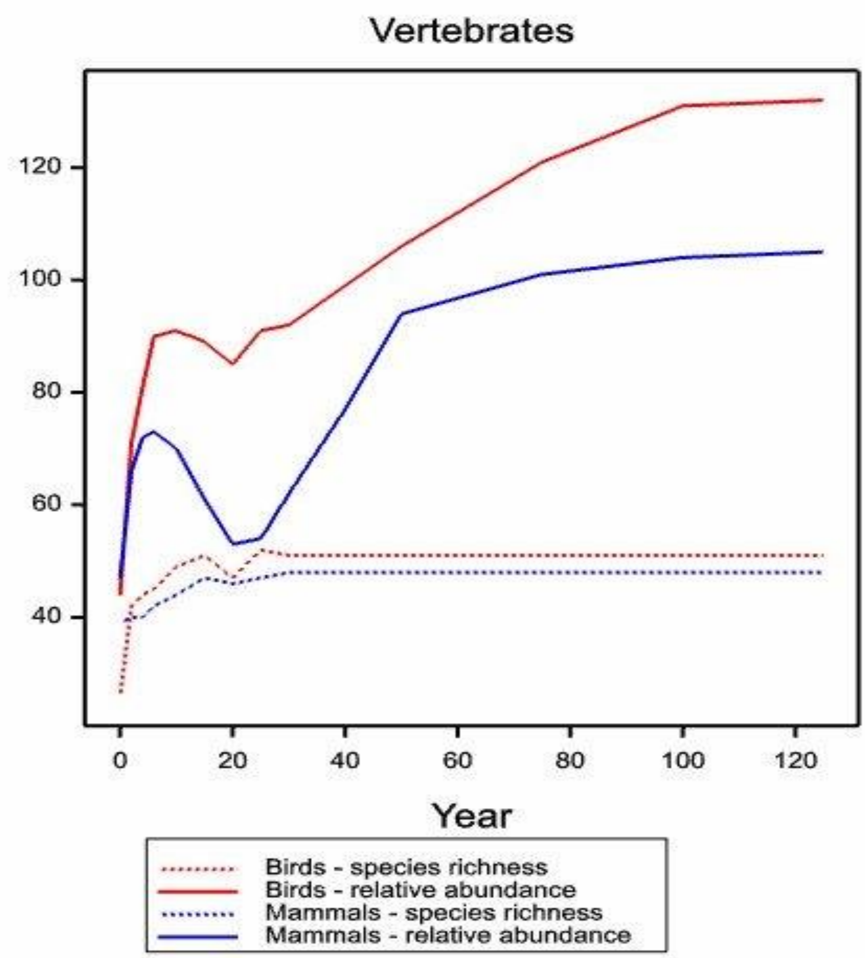


Figure 2. Relationship between vertebrate species richness and relative abundance and stand age in British forests

Evidence from the UK Biodiversity Assessment Project

To obtain evidence of the functional relationship between biodiversity and stand age, we chose to re-examine data that had been collected in a range of forests in the UK. The biodiversity assessment project was established in 1993 in response to a growing awareness of the value of forests for wildlife and the need to develop management practices suited to promoting it. The project involved an assessment of presence and abundance of species in 52 even-aged forest stands distributed across the UK and included sites encompassing a range of tree species, site types and plantation ages (Humphrey et al., 2003). The data obtained from the project represents the most extensive dataset on biodiversity that currently exists for UK forests. We explored relationships with stand age for all of the taxa and vegetation structure variables that had been assessed under the biodiversity assessment project.

The presence and abundance of a number of key taxonomic groups had been assessed in one hectare plots at each site, including fungi, bryophytes, pteridophytes, vascular plants and three insect orders (*Syrphidae* (Hoverflies); *Cicadomorpha* (Leafhoppers) and *Coleoptera* (Beetles)). In total, 679 species of fungi, 140 species of mosses and lichens, 169 species of vascular plants and 616 species of insect were identified in these plots. Assessments also included measures of stand structure, dead wood, canopy variables and site characteristics.

Initial analyses of the data focused on using generalised linear models with a view to obtaining evidence of simple relationships between species diversity and stand age. Although these analyses yielded some evidence of trends, there were some shortcomings with this approach. For some taxa, differences amongst stand age classes were apparent but not necessarily indicating a linear trend. Further, it was clear that environmental factors, including elevation, previous land use, tree species and climatic regime were all influencing diversity, and potentially masking the influence of stand age to some extent. An alternative approach based on non-linear modelling was implemented.

For each tree species and elevation class (lowland, foothills and upland) the dataset includes two sites considered similar ('replicates') with up to four different stand ages at each site, representing different stages of development from pre-thicket to over-mature. Thus, the dataset contains a maximum of eight observations for each tree species.

We re-examined the data with a view to obtaining evidence of the functional relationship between biodiversity and stand age. However, for some tree species and elevation classes it proved impossible to obtain a full chronosequence of stands (comprising different stand ages for similar sites), with the result that there was limited scope for determining a general relationship between biodiversity and stand age. The best statistical fit in more than half of all cases was obtained by assuming no change in biodiversity with stand age. This may be partly a consequence of the limited number of observations in each category.

However, it was feasible to obtain chronosequence of comparable stands of Sitka spruce in upland sites. These revealed a clear trend, with biodiversity lower in mid-rotation than in young and mature stands (Figure 3). Sitka spruce trees are particularly efficient at light interception, with the result that forest vegetation is forced through acute structural change as the under-storey is almost entirely eliminated during transition from young to mid rotation phases.

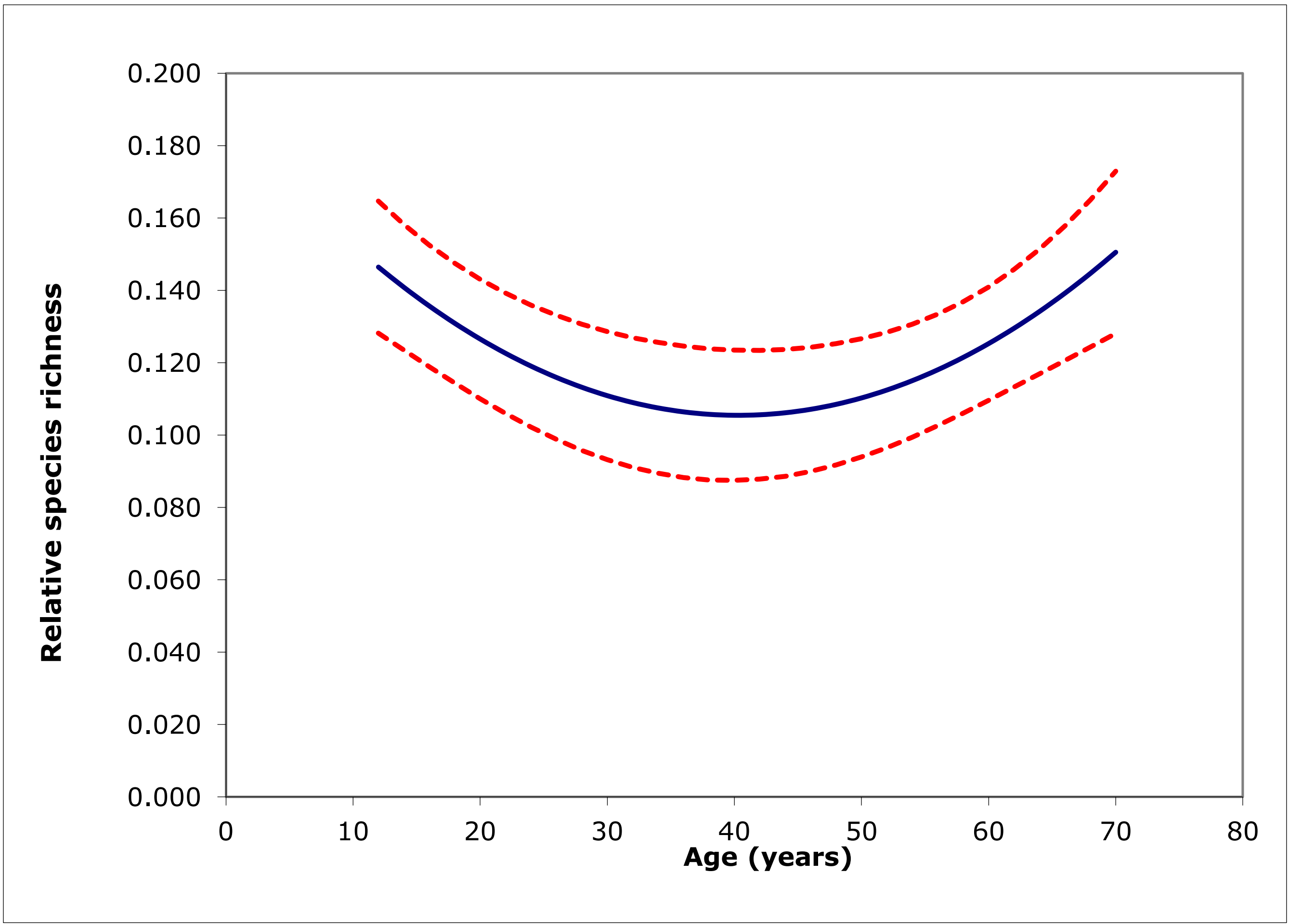


Figure 3. Biodiversity response through the stages of forest growth from pre-thicket to mature, in Sitka spruce upland forests

Economic evidence on Biodiversity

An extensive literature search covering the last 20 years uncovered surprisingly few references where the three key research interests: economics, biodiversity and rotation length appear together and their relationship was investigated. That is, although the studies contained terms that belong to each of our three key research interests, further investigation showed that in many instances not all three were interlinked and researched in depth but often mentioned rather briefly and/or tangentially.

28 of the 398 studies that were identified as 'possibly relevant' in the initial search (see Annex for details of search terms, etc.) were tagged as relating to economic evidence. These studies, together with the ones identified in later searches, were undertaken in a wide range of contexts, including:

- biodiversity and carbon sequestration (McCarney *et al.*, 2008);
- joint production of multiple benefits and sustainable forest management (Backeus *et al.*, 2006; Bergseng *et al.*, 2012; Berkes and Davidson-Hunt, 2006; Duncker *et al.*, 2012; McCarney *et al.*, 2008; Messier and Kneeshaw, 1999; Vierikko *et al.*, 2010; Vierikko *et al.*, 2008);
- effects of fuelwood harvesting and biomass production on biodiversity (Bouget *et al.*, 2012; Dauber *et al.*, 2010);
- cost-efficiency of various conservation measures and approaches with the majority of studies focusing on old-growth preservation and leaving behind retention trees or coarse woody debris (Carlen *et al.*, 1999; Gustafsson and Perhans, 2010; Jonsson *et al.*, 2010; Jonsson *et al.*, 2006; Juutinen *et al.*, 2004; Juutinen and Monkkonen, 2007; Naesset *et al.*, 1997; Perhans *et al.*, 2011; Polasky *et al.*, 2012; Ranius *et al.*, 2005; Wikberg *et al.*, 2009);
- design of instruments (subsidy and tax) for biodiversity maintenance (Hanley *et al.*, 2012; Koskela *et al.*, 2007b; Miteva *et al.*, 2012);
- forest biodiversity valuation (alone or as a constituent part of some scenario set) by the public though not linked directly to rotation length (Christie *et al.*, 2011; Czajkowski *et al.*, 2009; Garrod and Willis, 1997; Hanley *et al.*, 2002; Horne *et al.*, 2005; Nijkamp *et al.*, 2008; Thiene *et al.*, 2012; Tyrväinen *et al.*, 2013).

Reviewing the studies revealed two strands of primary interest: (i) 'best fit' papers, which addressed the three key research interests adequately, (ii) papers that considered economics and biodiversity but did not consider these factors in relation to rotation length. These two groups of papers are discussed below.

For the first group of studies linking together economics, biodiversity and rotation age (Koskela *et al.*, 2007a; McCarney *et al.*, 2008; Miettinen *et al.*, 2013; Nghiem, 2013), none were undertaken in the UK. Furthermore, all of the papers in this group used bespoke data sets and estimated functional relationships for their particular study areas, so that although the studies reveal how the problem could be tackled in principle their results are not easily transferable to a UK context.

In a study from Vietnam (Nghiem, 2013) management strategies balancing economic gains and biodiversity conservation are explored for a forest comprising a number of stands. Optimal rotation age is calculated by maximising the discounted revenues from timber and carbon sequestration subject to a biodiversity maintenance constraint, which is proxied by a bird density indicator being equal to or exceeding a minimum viable population threshold. The bird density depends positively on stand age and size. The results showed that the inclusion of biodiversity conservation led to a longer rotation age compared to the case where timber and carbon sequestration values alone are considered. Increasing rotation age by 36% in order to meet the biodiversity constraint led to a decrease in the net present value (NPV) of 12%.

Similarly, another study (Bergseng *et al.*, 2012) focussed on southeast Norway (main tree species – Norway spruce and Scots pine) showed that NPV can fall by up to 43% in the case of biodiversity protection measures. The biodiversity conservation measures included:

- not disturbing growth in key areas;
- selective or shelterwood felling;
- maintaining at least 30% of the woodland as old forest;
- maintaining a 20% share of the woodland as deciduous forest;
- 10 retention trees per ha;
- increasing the rotation cycle by 50%;
- no thinnings;
- using only natural regeneration.

Two studies from Finland (Koskela *et al.*, 2007a; Miettinen *et al.*, 2013) attempt to incorporate biodiversity conservation into an optimal rotation length framework as developed by Faustmann and Hartman. The first study (Koskela *et al.*, 2007a) applies the extended Hartman rotation framework and ran a simulation based on Finnish data for a southern pine stand. In the study biodiversity conservation is modelled through the volume of green tree retention (GTR) at final felling. GTR differs from conventional shelterwood or seed systems in that retention trees are left permanently uncut attempting to mimic and restore natural disturbance regimes. GTR's main biodiversity benefit is derived from maintaining the permanent cover of trees, a steady flow of dead wood and promotion of understory vegetation. The biodiversity index used in the study was a weighted sum of scaled values of various structural elements present in the stand: volumes of different tree species, timber volumes in 10 cm-diameter classes, and volumes of deadwood components (standing deadwood and downwood of different tree species). The index value is determined from a contingent valuation study (Pouta, 2005). The study suggests longer rotation periods by comparison with the standard Faustmann framework and a higher volume of GTR (volume depends positively on biodiversity valuation). The second study (Miettinen *et al.*, 2013) examines alternative whole-tree and stem-only harvesting regimes when water quality, biodiversity conservation and carbon sequestration ecosystem services are included in the analysis on drained peatland forests. However, due to data problems the study could not estimate optimal rotation age and biodiversity conservation impact. Both studies depend crucially on specific data from the study areas.

The other study (McCarney *et al.*, 2008) is from Canada's boreal mixed woodland region in north-central Alberta. Biodiversity is represented by a wildlife habitat quality index for five vertebrate species. The model maximises the net present value of timber harvest under constraints on carbon management and a minimum level of wildlife habitat quality (a proxy for biodiversity conservation). The model developed is quite complex, very specific and relies on a number of proprietary software tools. The results show a reduction in the NPV of timber production for a firm faced with incremental increases in minimum habitat area requirements. Wildlife species that prefer mature areas of aspen and mixed woodland cover are likely to benefit from carbon sequestration incentives, whereas habitat benefits associated with conifer forests are less clear.

In the second group, none of the papers which consider forest management issues together with economics and biodiversity (but not links to stand age or rotation length) focuses upon UK woodlands. A European study linking different forest management alternatives, economics and biodiversity (Duncker *et al.*, 2012) illustrates how high management intensity (productive forests for timber and biomass) can negatively impact on biodiversity indicators (woody debris, tree size distribution and a number of tree species). This supports the well-known trade-off between maximising the economic value of timber production and the preservation of biodiversity at stand level (Boscolo and Vincent, 2003; Carlsson, 1999; Hunter, 1999). The study's simulated forest management units represented Central European forest ecosystems in the submontane vegetation zone under a humid–temperate climate with acidic soils.

Although not considering forest management issues or changes with stand age, a number of biodiversity valuation studies of UK woodlands have been undertaken. A brief summary of recent papers valuing European woodland biodiversity and associated monetary value estimates is provided in an Annex V.

A number of papers lacking developed economic models or valuations were considered to be worth reviewing. Those focusing on links between forest biodiversity and structural stand stages (including age) were considered of potential use in future work aimed at adding an economic component to help close existing research gaps (Brockerhoff *et al.*, 2008; Humphrey, 2005; Smith *et al.*, 2008), and some papers in shedding further light on choice of biodiversity indicators (Duncker *et al.*, 2012; Juutinen and Monkkonen, 2004; McCarney *et al.*, 2008; Smith *et al.*, 2008; Vierikko *et al.*, 2010), including a few focusing on policy decision-making (Failing and Gregory, 2003; Lindenmayer *et al.*, 2000).

Of papers in this third group, an Irish study (Smith *et al.*, 2008) linking biodiversity to major structural stand stages based on experimental data from 44 study sites of commercial forestry plantations of Sitka spruce and ash, was considered one of the most useful due to similarities between forests in Ireland and those in the UK. Biodiversity for five taxonomic groups - bryophytes, vascular plants, spiders, hoverflies and birds was measured with compositional, structural and functional indicators. The best structural biodiversity indicator was stand stage. The study found that biodiversity trends over the forest cycle and between tree species differ among the taxonomic groups studied. Canopy cover was the main structural indicator and affected other structural variables such as cover of lower vegetation layers. Other structural indicators included deadwood and distances to forest edge and to broadleaved woodland. Functional indicators included stand age, site environmental characteristics and management practices. The study recommended early thinning and at regular intervals for spruce plantations to prevent complete canopy closure and improve biodiversity, although associated net costs were not considered.

Although a single stand perspective is the traditional approach in forest economics, some recent studies suggest adopting a broader scale approach that views the forest as comprised of a number of stands of differing characteristics (age, species, amount of open space, canopy structure, etc.). Given the habitat requirements for different species, optimisation with biodiversity benefits included may be best accomplished in a multi-stand model rather than with the current well-developed and widely applied single stand approaches of Faustmann and Hartman. Such models can be developed to cover a number of stands but are more complex and tend to have a fixed time horizon for current examples (unlike Faustmann type models, which consider an infinite series of rotations).

Support for a multi-stand approach also comes from the fact that some forestry production processes are non-linear. For example, the relationship between profit and production of ecosystem services such as timber and biodiversity. Such non-linear behaviour may arise, for example, from fixed forest management costs and administration constraints. In this situation it may be optimal to spatially separate management objectives for different stands in some cases (Noack *et al.*, 2010; Robert and Stenger, 2013). Thus one stand may be best managed for optimal timber production while another is optimised for biodiversity or the delivery of ecosystem services such as carbon sequestration and recreation.

Discussion and Recommendations

Quantifying biodiversity

The review highlighted a number of important issues affecting the quantification of biodiversity and the appropriate metrics for economic modelling. These include:

● the concept of biodiversity is complex with no consensus at present in either the ecological literature or the economic literature on how best to quantify it. One problem, noted above, is the difficulty of encompassing all of the aspects of interest in a single metric. Boundary issues complicate the choice (e.g. do species found in woodlands and other habitats, or part of the year in UK woodlands and part in other countries, count equally in quantifying 'UK woodland biodiversity'?) However, a more fundamental difficulty may lie at the root of the lack of consensus that relates to a potentially irreconcilable opposition between analysis and synthesis. On one hand, traditionally one is analysing a complex problem by isolating it from the rest of the environment and simplifying it by focusing on a few major interactions (often a single one). On the other hand, it seems that biodiversity must be analysed in its entirety, holistically.

● the emergent nature of scientific understanding of the relative importance of its constituent parts in ecosystem functioning and in underpinning ecosystem services, as well as concerning critical thresholds. Some argue (Helm and Hepburn, 2012), that more is currently known, for example, about Sun–Earth system interactions and the Earth's atmosphere dynamics – major constituents in climate change models, than about many ecological systems. No evidence was found in the studies reviewed linking biodiversity and stand age to evidence on climate change impacts.

● differing perceptions exist of what constitutes beneficial (e.g. 'native' to UK, or to the EU) and detrimental (e.g. 'non-native', or pests and invasive species) elements of biodiversity from an ecological perspective, as do perceptions of the contribution of different elements to human wellbeing. For example, to what extent do midges, mosquitoes, cockroaches, rats, poisonous fungi or snakes add to or detract from human wellbeing?

● measures of biodiversity such as total 'species richness' and 'total abundance', although perhaps intuitively easy to grasp, and metrics such as the Shannon Index of species diversity, are overly simplistic as measures of woodland biodiversity. They are also unfamiliar to many non-ecologists and would require some education work for non-specialists to express preferences and base estimates of values.

Links between biodiversity and stand age

The principal findings of the reviews of the ecological evidence on links between biodiversity and stand age or stand age class include that:

● although there is no universal response to stand age, with variations between taxa and sites (as might be expected), overall there is more ecological evidence in published studies of increasing species richness with stand age, than of a fall (or of no change). This is especially the case for lichens (and, unsurprisingly, also for coarse woody debris – often used as a biodiversity indicator). For a few species – notably mammals and amphibians, existing evidence appears to suggest declining species richness with stand age, but this conclusion draws on relatively few studies (see Fig 1 above). Factors associated with stand age that would have an impact (e.g. light levels) were not investigated.

• initial analysis of standardised species richness scores for fungi, lichens and bryophytes, vascular plants and invertebrates from the UK Biodiversity Assessment Project pointed to an overall fall in biodiversity with stand stage in coniferous woodlands, but increases for some species (e.g. Norway spruce), and a 'U-shaped' relationship for Sitka spruce (the highest biodiversity being in the pre-thicket stage). However, as the influence of other factors such as elevation and previous land use may mask the true effect of age class, regression analysis is needed to separate out the effects (currently ongoing).

• the literature on habitat requirements for birds and mammals in British forests suggests that after a brief initial increase in species richness and relative abundance with stand age, these both then fall in the thicket stage (with a minimum at around 20 years), thereafter increasing again (though some of the increase is only obtained at ages beyond typical rotation lengths).

• There is no clear evidence on the impact of alternative forest management approaches such as CCF and how these differ from traditional even-aged stands with respect to biodiversity. Each silvicultural management alternative has its advantages and disadvantages for biodiversity. Although general characteristics beneficial for biodiversity can be identified, including in relation to spatial and landscape scale issues such as open space within woodlands, no single method of managing forests for optimal biodiversity conservation exists, with the best approach depending on specific circumstances.

Links between biodiversity and optimal rotation length

The review found that economic literature linking biodiversity and rotation length is also sparse. Economic aspects of biodiversity and rotation length are very rarely adequately treated or their relationship investigated in depth (perhaps unsurprising given the complex and multi-faceted nature of biodiversity). While a few studies focus on high value areas such as biodiversity of tropical forests, or wetlands and coastal areas, very little research has been done on plantations and productive forestry linking biodiversity to rotation length. Nevertheless, four strands of research were identified of potential help in progressing investigation of links between biodiversity and rotation length:

• a direct approach where an economic framework includes robust existing monetary estimates of biodiversity (or its proxy) and a knowledge of how biodiversity changes over a rotation although such studies are rare at present. Where, for example, biodiversity values increase with forest age, incorporation of biodiversity leads to a longer optimal rotation length (Koskela *et al.*, 2007a).

● where robust values are not available, biodiversity may still affect optimal rotation length through a set of management constraints required for biodiversity conservation (Bergseng *et al.*, 2012; Duncker *et al.*, 2012). In this case an indirect approach is possible with conservation constraints and associated costs incorporated in the NPV optimisation problem for a forest unit. Such problems are often solved using a linear programming approach. This approach does not consider the value of the conserved or enhanced biodiversity per se, which is mostly unknown, but only the cost of associated management measures (the biodiversity conservation objective is considered to be an absolute requirement).

● the third strand focuses primarily on ecological research linking biodiversity and forest age or stage (Smith *et al.*, 2008), also proposing some useful indicators to assess and monitor biodiversity conservation (Failing and Gregory, 2003; Vierikko *et al.*, 2010). It shows that different taxa/species respond differently to forest age and structural characteristics (canopy closure) and that optimal management may be location specific and require disaggregating biodiversity objectives to focus upon specific elements.

● partially linked to the third strand above, some studies (Hyde, 2012; Noack *et al.*, 2010; Robert and Stenger, 2013) suggest adoption of a broader scale approach that views the forest as comprised of a number of stands of differing characteristics (age, species, amount of open space, canopy structure, etc.). Optimisation with non-timber benefits included may be best accomplished in a multi-stand model (Hyde, 2012, p. 98) rather than with currently well developed and widely applied single stand approaches of Faustmann and Hartman. Such models can be developed to cover a number of stands (McCarney *et al.*, 2008; Nghiem, 2013) but are more complex and are not exactly Faustmann-like, and tend to have a fixed time horizon. Some of these models are explicitly spatial in nature.

● forestry production processes can be non-linear – including relationships between rotation length and biodiversity, as well as production of timber and other ecosystem services ,. Non-linear relationships can also stem from, for example, from fixed forest management costs and administration constraints (Boscolo and Vincent, 2003; Bowes *et al.*, 1989; Swallow *et al.*, 1990). In some cases, non-linear relationships can imply that it is optimal to spatially separate management objectives for different stands (Noack *et al.*, 2010; Robert and Stenger, 2013). For example, one stand may be best managed for optimal timber production while another is optimised for biodiversity, or the delivery of ecosystem services such as carbon sequestration and recreation.

The current state of knowledge and available methods do not permit the inclusion of biodiversity in optimum rotation length models at present except in special cases.

◇ **Recommendation 1:** *Further work on multi-stand forest modelling and the nature of the relationship between profit and production of ecosystem services in UK forestry is needed. This would permit the identification of conditions under which it is optimal to separate management of stands by forestry objective, and where biodiversity objectives are best pursued in isolation rather than as part of a multi-objective optimisation approach (as implicit in sustainable forest management).*

The primary focus of the current study is on how biodiversity varies with stand age and how biodiversity could be incorporated into optimal rotation length modelling. However, the review found that incorporating biodiversity in this way is not uncontroversial.

Some argue (Perry, 2013) that in the presence of climate change uncertainties and largely unknown ecological interactions among species, the pursuit of objectives based on optimisation approaches may be misguided. Instead, they suggest the focus should be on increasing ecosystem resilience and functional diversity based on the precautionary principle. However, empirical evidence demonstrating the superiority of forestry management approaches based upon maximising ecosystem resilience appears to be lacking at present.

◇ **Recommendation 2:** *Empirical work is needed to investigate conditions under which it is optimal to maximise ecosystem resilience rather than biodiversity.*

The emergent nature of scientific understanding of many aspects of biodiversity, and, in particular, the paucity and immaturity of research on how woodland biodiversity changes with forest age and management, highlights existing uncertainties that preclude being overly-prescriptive in making recommendations. Nonetheless, given the current state of knowledge, it is possible to provide tentative recommendations on the approach to use in including biodiversity in optimisation models and ecosystem services assessments.

◇ **Recommendation 3:** *For vertebrates in British forests, adopting the U-shaped generalised response to stand age estimated from their habitat requirements - which includes a minimum around year 20, is recommended, instead of assuming no change or a simple linear increasing relationship with stand age up to a maximum level .*

◇ **Recommendation 4:** *Work linking evidence on biodiversity and stand age to wider research on climate change impacts is needed. This will be important if risks of mistakes associated with assuming that future relationships will be the same as those that held in the past are to be minimised.*

Economic valuation of biodiversity

Can biodiversity be reliably valued in monetary terms in order to include it within a standard economic objective function within an optimisation model, or for the purposes of wider ecosystem services assessments? Economic opinions remain divided and there are a number of issues to consider in answering this question.

Firstly, as discussed above, there is incomplete ecological evidence at present on the level of UK woodland biodiversity in different types and locations of forests and how it varies with stand age. The dearth of evidence appears especially true for elements with known cultural and ecosystem function value such as butterflies, wood ants, earthworms (which were not covered in the UK Biodiversity Assessment Project, for example).

Secondly, effects in survey–based economic valuation studies (such as hypothetical bias associated with respondents overstating values because they do not expect to pay the costs involved), and large differences between the values expressed for willingness to accept and willingness to pay for specific changes, has led to continuing debate in the literature about the robustness and validity of existing valuation methods (Haab *et al.*, 2013; Hausman, 2012; Lamb, 2013).

Furthermore, given the complexity of the concept of biodiversity, survey-based valuation studies are likely to be susceptible to learning effects. Unfamiliarity with biodiversity concepts and with different approaches to its measurement implies that detailed information has to be provided to participants on aspects being valued. This may induce learning effects that alter preferences and values expressed. (On evidence of learning effects and other cognitive influences that can influence survey-based valuation estimates, see Moseley and Valatin, 2013). Thus, akin to Heisenberg's Uncertainty Principle (the impossibility of simultaneously measuring exactly the velocity and position of an object), undertaking a survey-based valuation study may alter the values expressed. This can raise doubts about whether value estimates are credible (see also discussion in Diamond and Hausman, 1994).

None of the valuation studies of UK woodland biodiversity identified in the review use non-survey-based approaches, apart from a single study focusing upon legacies in wider contexts. (A summary of existing valuation approaches to non-market goods and services, including biodiversity, together with a review of recent papers valuing European woodland biodiversity and associated monetary value estimates, is provided in an Annex V).

Providing one accepts the unresolved nature of the continuing debate about the robustness of survey-based valuation methods and that values expressed can change as a consequence of learning effects, the evidence review illustrates how biodiversity can be valued in monetary terms. The reliability of these estimates remains a more open question, however.

◇ **Recommendation 5:** *Fundamental work on developing methods to provide robust estimates of woodland biodiversity values is still needed, and should take into account insights from behavioural economics on the influence of cognitive factors on stated preferences (e.g. related to framing, information provision and format, and learning).*

Even if we accept that at present methods such as contingent valuation are deeply flawed (Diamond and Hausman (1994)) and do not provide a good basis for informed policy making (Hausman, 2012) – issues on which consensus is lacking within the study team, this would not mean that biodiversity cannot be incorporated into optimal rotation length models. As discussed above, a strand of the economic literature exists that incorporates biodiversity through including conservation constraints and associated costs, which by-passes the issue of valuing biodiversity. Alternatively, a more direct approach is to incorporate non-monetary measures of biodiversity in the objective function being maximised. However, weighting these with monetary elements in an objective function may then have to rely upon expert judgement.

◇ **Recommendation 6:** *Where the reliability of biodiversity value estimates is in doubt, consideration should be given to ranging estimates to reflect existing uncertainties, or incorporating biodiversity in optimisation models using other approaches, such as multi-criteria analysis. Where mandatory forest management measures to preserve biodiversity are incorporated as costly constraints in forestry optimization, the impact of these costs on the NPV may serve as proxy for the value society places on woodland biodiversity conservation.*

Acknowledgements

We would like to thank many people who helped in compiling literature and especially Rachel Goldstein - an intern from University of Rochester, New York, who started the review process with an initial search. Thanks to Pat Snowdon, Richard Haw and Steven Hendry for comments on earlier drafts.

References

BACKEUS, S., WIKSTROM, P. AND LAMAS, T. (2006). Modeling carbon sequestration and timber production in a regional case study. *SILVA FENNICA* **40**, 615-629.
BARNOSKY, A. D., MATZKE, N., TOMIYA, S., WOGAN, G. O. U., SWARTZ, B., QUENTAL, T. B., MARSHALL, C., MCGUIRE, J. L., LINDSEY, E. L., MAGUIRE, K. C., MERSEY, B. AND FERRER, E. A. (2011). Has the Earth/'s sixth mass extinction already arrived? *Nature* **471**(7336), 51-57.

BERGSENG, E., ASK, J. A., FRAMSTAD, E., GOBAKKEN, T., SOLBERG, B. AND HOEN, H. F. (2012). Biodiversity protection and economics in long term boreal forest management — A detailed case for the valuation of protection measures. *Forest Policy and Economics* **15**(0), 12-21.

BERKES, F. AND DAVIDSON-HUNT, I. J. (2006). Biodiversity, traditional management systems, and cultural landscapes: examples from the boreal forest of Canada. *INTERNATIONAL SOCIAL SCIENCE JOURNAL* **58**, 35+.

BOSCOLO, M. AND VINCENT, J. R. (2003). Nonconvexities in the production of timber, biodiversity, and carbon sequestration. *Journal of Environmental Economics and Management* **46**(2), 251-268.

BOUGET, C., LASSAUCE, A. AND JONSELL, M. (2012). Effects of fuelwood harvesting on biodiversity — a review focused on the situation in Europe11 This article is one of a selection of papers from the International Symposium on Dynamics and Ecological Services of Deadwood in Forest Ecosystems. *Canadian Journal of Forest Research* **42**(8), 1421-1432.

BOWES, P. M. D., KRUTILLA, J. V., ECONOMICS, R. F. T. F. F. AND PROGRAM, P. (1989). *Multiple-Use Management: The Economics of Public Forestlands*. Taylor & Francis.

BROCKERHOFF, E. G., JACTEL, H., PARROTTA, J. A., QUINE, C. P. AND SAYER, J. (2008). Plantation forests and biodiversity: oxymoron or opportunity? *Biodiversity and Conservation* **17**(5), 925-951.

BURNS, F., EATON, M. A., GREGORY, R. D., AL FULAIJ, N., AUGUST, T., BIGGS, J., BLADWELL , S., BRERETON, T. M., BROOKS, D. R., CLUBBE, C., DAWSON, J., DUNN, E., EDWARDS, B., FALK, S. J., GENT, T., GIBBONS, D. W., GURNEY, M., HAYSOM, K. A., HENSHAW, S., HODGETTS, N. G., ISAAC, N., MCLAUGHLIN, M., MUSGROVE, A. J., NOBLE, D. G., O'MAHONY, E., PACHECO , M., ROY, D., SEARS , J., SHARDLOW, M., STRINGER, C., TAYLOR, A., THOMPSON, P., WALKER, K. J., WALTON, P., WILLING, M. J., WILSON, J. AND WYNDE, R. (2013). State of nature. Available at: http://nora.nerc.ac.uk/502092/.

CARLEN, O., MATTSSON, L., ATLEGRIM, O. AND SJOBERG, K. (1999). Cost efficiency in pursuing environmental objectives in forestry. *JOURNAL OF ENVIRONMENTAL MANAGEMENT* **55**, 111-125.

CARLSSON, M. (1999). A method for integrated planning of timber production and biodiversity: a case study. *Canadian Journal of Forest Research* **29**(8), 1183-1191.

CHRISTIE, M., HYDE, T., COOPER, R., FAZEY, I., DENNIS, P., WARREN, J., COLOMBO, S. AND HANLEY, N. (2011). Economic Valuation of the Benefits of Ecosystem Services delivered by the UK Biodiversity Action Plan. Final Report to the Department for Environment, Food and Rural Affairs (DEFRA). August 2011. Aberystwyth.

CZAJKOWSKI, M., BUSZKO-BRIGGS, M. AND HANLEY, N. (2009). Valuing changes in forest biodiversity. *Ecological Economics* **68**(12), 2910-2917.

DAUBER, J., JONES, M. B. AND STOUT, J. C. (2010). The impact of biomass crop cultivation on temperate biodiversity. *GLOBAL CHANGE BIOLOGY BIOENERGY* **2**, 289-309.

DECLERCK, F. AND SALINAS, A. (2011). Measuring biodiversity. Ch. 3. In: (Eds) B. RAPIDEL, F. DECLERCK, J. F. LE COQ AND J. BEER. Ecosystem services from agriculture and agroforestry: measurement and payment. Earthscan, London. p. 66.

DEFRA (2011). Biodiversity 2020: A strategy for England’s wildlife and ecosystem services. DEPARTMENT FOR ENVIRONMENT FOOD AND RURAL AFFAIRS (DEFRA). London. Available at: http://www.defra.gov.uk/publications/2011/08/19/pb13583-biodiversity-strategy-2020/.

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.

DUNCKER, P. S., RAULUND-RASMUSSEN, K., GUNDERSEN, P., KATZENSTEINER, K., DE JONG, J., RAVN, H. P., SMITH, M., ECKMÜLLNER, O. AND SPIECKER, H. (2012). How Forest Management affects Ecosystem Services, including Timber Production and Economic Return: Synergies and Trade-Offs. *Ecology and Society* **17**(4).

EFTEC (2011). Scoping Study on Valuing Ecosystem Services of Forests Across Great Britain. Final Report for the Forestry Commission. October 2011. London. Available at: http://www.forestry.gov.uk/pdf/VES_FinalReport_eftec_Oct2011.pdf/$file/VES_FinalReport_eftec_Oct2011.pdf

FAILING, L. AND GREGORY, R. (2003). Ten common mistakes in designing biodiversity indicators for forest policy. *Journal of Environmental Management* **68**(2), 121-132.

FC (2011) Forests and Biodiversity - UK Forest Standard Guidelines Forestry Commission, Edinburgh. 1 i-iv 68pp.
http://www.forestry.gov.uk/pdf/FCGL001.pdf/$FILE/FCGL001.pdf

FC (2013). Forestry Statistics 2013. Forestry Commission (FC). Edinburgh. Available at: http://www.forestry.gov.uk/website/forstats2013.nsf/LUContentsTop?openview&RestrictToCategory=1.

FOREST EUROPE (2010). Enriching Forest biodiversity, FOREST EUROPE is implementing Convention on Biological Diversity provisions. FOREST EUROPE (Ministerial Conference on the Protection of Forests in Europe): Aas, Norway. Available at: http://www.foresteurope.org/documentos/EnrichingForestBiodiversity_print.pdf.

FULLER, R. J. (2003). Bird Life of Woodland and Forest. Cambridge University Press.

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.

GSR (2013) *Rapid Evidence Assessment (REA) toolkit*. Available at: http://webarchive.nationalarchives.gov.uk/20140305122816/http://www.civilservice.gov.uk/networks/gsr/resources-and-guidance/rapid-evidence-assessment (Accessed: 23 September 2016).

GUSTAFSSON, L. AND PERHANS, K. (2010). Biodiversity Conservation in Swedish Forests: Ways Forward for a 30-Year-Old Multi-Scaled Approach. *AMBIO* **39**, 546-554.

HAAB, T. C., INTERIS, M. G., PETROLIA, D. R. AND WHITEHEAD, J. C. (2013). From Hopeless to Curious? Thoughts on Hausman's "Dubious to Hopeless" Critique of Contingent Valuation. *Applied Economic Perspectives and Policy*.

HANLEY, N., BANERJEE, S., LENNOX, G. D. AND ARMSWORTH, P. R. (2012). How should we incentivize private landowners to 'produce' more biodiversity? *Oxford Review of Economic Policy* **28**(1), 93-113.

HANLEY, N., WILLIS, K., POWE, N. AND ANDERSON, M. (2002). Social & Environmental Benefits of Forestry Phase 2: VALUING THE BENEFITS OF BIODIVERSITY IN FORESTS. Report to the Forestry Commission. April 2002. Centre for Research in Environmental Appraisal & Management, University of Newcastle.

HARRIS, S. AND YALDEN, D. W. (Eds) (2008). *Mammals of the British Isles: Handbook*. Southampton, UK: The Mammal Society (4th Edition).

HARTMAN, R. (1976) The Harvesting Decision When a Standing Forest Has Value. Economic Inquiry 14:52-58. doi: 10.1111/j.1465-7295.1976.tb00377.x

HAUSMAN, J. (2012). Contingent Valuation: From Dubious to Hopeless. *Journal of Economic Perspectives* **26**(4), 43-56.

HELM, D. AND HEPBURN, C. (2012). The economic analysis of biodiversity: an assessment. *Oxford Review of Economic Policy* **28**(1), 1-21.

HORNE, P., BOXALL, P. C. AND ADAMOWICZ, W. L. (2005). Multiple-use management of forest recreation sites: a spatially explicit choice experiment. *Forest Ecology and Management* **207**(1–2), 189-199.

HUMPHREY, J. W. (2005). Benefits to biodiversity from developing old-growth conditions in British upland spruce plantations: a review and recommendations. *Forestry* **78**(1), 33-53.

HUMPHREY, J. W., FERRIS, R. AND QUINE, C. P. (Eds) (2003). Biodiversity in Britain's Planted Forests: Results from the Forestry Commission's Biodiversity Assessment Project. Edinburgh: Forestry Commission.

HUNTER, M. L., ED. (1999). *Maintaining Biodiversity in Forest Ecosystems*. Cambridge University Press.

HYDE, W. F. F. (2012). *The Global Economics of Forestry*. Taylor & Francis.

DEFRA (2011) 'Biodiversity 2020: A strategy for England's wildlife and ecosystem services'. Edited by Department for Environment Food and Rural Affairs (DEFRA). London. Available at: http://www.defra.gov.uk/publications/2011/08/19/pb13583-biodiversity-strategy-2020/.

GSR (2013) *Rapid Evidence Assessment (REA) toolkit*. Available at: http://webarchive.nationalarchives.gov.uk/20140305122816/http://www.civilservice.gov.uk/networks/gsr/resources-and-guidance/rapid-evidence-assessment (Accessed: 23 September 2016).

HUMPHREY, J. W., FERRIS, R. AND QUINE, C. P. (2003) 'Biodiversity in Britain's Planted Forests: Results from the Forestry Commission's Biodiversity Assessment Project'. Edinburgh: Forestry Commission. Available at: http://www.forestry.gov.uk/PDF/fcrp004.pdf/$FILE/fcrp004.pdf.

INGRAM, J. C., REDFORD, K. H. AND WATSON, J. E. M. (2012) 'Applying ecosystem services approaches for biodiversity conservation: benefits and challenges', *SAPI ENS, Surveys and Perspectives Integrating Environment and Society [Online]*. Institut Veolia Environnement, (5.1). Available at: http://sapiens.revues.org/1459.

JONSSON, M., RANIUS, T., EKVALL, H. AND BOSTEDT, G. (2010). Cost-effectiveness of silvicultural measures to increase substrate availability for wood-dwelling species: A comparison among boreal tree species. *SCANDINAVIAN JOURNAL OF FOREST RESEARCH* **25**, 46-60.

JONSSON, M., RANIUS, T., EKVALL, H., BOSTEDT, G., DAHLBERG, A., EHNSTROM, B., NORDEN, B. AND STOKLAND, J. N. (2006). Cost-effectiveness of silvicultural measures to increase substrate availability for red-listed wood-living organisms in Norway spruce forests. *BIOLOGICAL CONSERVATION* **127**, 443-462.

JUUTINEN, A., MANTYMAA, E., MONKKONEN, M. AND SALMI, J. (2004). A cost-efficient approach to selecting forest stands for conserving species: A case study from northern Fennoscandia. *FOREST SCIENCE* **50**, 527-539.

JUUTINEN, A. AND MONKKONEN, M. (2004). Testing alternative indicators for biodiversity conservation in old-growth boreal forests: ecology and economics. *ECOLOGICAL ECONOMICS* **50**, 35-48.

JUUTINEN, A. AND MONKKONEN, M. (2007). Alternative targets and economic efficiency of selecting protected areas for biodiversity conservation in boreal forest. *ENVIRONMENTAL & RESOURCE ECONOMICS* **37**, 713-732.

KOSKELA, E., OLLIKAINEN, M. AND PUKKALA, T. (2007a). Biodiversity conservation in commercial boreal forestry: The optimal rotation age and retention tree volume. *FOREST SCIENCE* **53**, 443-452.

KOSKELA, E., OLLIKAINEN, M. AND PUKKALA, T. (2007b). Biodiversity policies in commercial boreal forests: Optimal design of subsidy and tax combinations. *FOREST POLICY AND ECONOMICS* **9**, 982-995.

LAMB, W. (2013). Commentary on economic valuations of biodiversity. *Ecological Economics* **89**(0), 170-173.

LINDENMAYER, D. B., MARGULES, C. R. AND BOTKIN, D. B. (2000). Indicators of Biodiversity for Ecologically Sustainable Forest Management
Indicadores de Biodiversidad para el Manejo Ecológicamente Sostenible de Bosques. *CONSERVATION BIOLOGY* **14**(4), 941-950.

MACLEAN, I. M. AND WILSON, R. J. (2011). Recent ecological responses to climate change support predictions of high extinction risk. *Proc Natl Acad Sci U S A* **108**(30), 12337-12342.

MCCARNEY, G. R., ARMSTRONG, G. W. AND ADAMOWICZ, W. L. (2008). Joint production of timber, carbon, and wildlife habitat in the Canadian boreal plains. *CANADIAN JOURNAL OF FOREST RESEARCH-REVUE CANADIENNE DE RECHERCHE FORESTIERE* **38**, 1478-1492.

MESSIER, C. AND KNEESHAW, D. D. (1999). Thinking and acting differently for sustainable management of the boreal forest. *FORESTRY CHRONICLE* **75**, 929-938.

MIETTINEN, J., OLLIKAINEN, M., NIEMINEN, T. M., UKONMAANAHO, L., LAURÉN, A., HYNYNEN, J., LEHTONEN, M. AND VALSTA, L. (2013). Whole-tree harvesting with stump removal versus stem-only harvesting in peatlands when water quality, biodiversity conservation and climate change mitigation matter. *Forest Policy and Economics*.

MILLENNIUM ECOSYSTEM ASSESSMENT (MA) (2005). *MA Releases Second Report: Biodiversity and Human Well-being*.

MITEVA, D. A., PATTANAYAK, S. K. AND FERRARO, P. J. (2012). Evaluation of biodiversity policy instruments: what works and what doesn't? *Oxford Review of Economic Policy* **28**(1), 69-92.

MOSELEY, D. AND VALATIN, G. (2013). Insights from behavioural economics for ecosystem services valuation and sustainability. Forestry Commission Research Report. Forest Research. Edinburgh. Available at: http://www.forestry.gov.uk/pdf/FCRP022.pdf/$FILE/FCRP022.pdf

NAESSET, E., GOBAKKEN, T. AND HOEN, H. F. (1997). Economic analysis of timber management practices promoting preservation of biological diversity. *SCANDINAVIAN JOURNAL OF FOREST RESEARCH* **12**, 264-272.

NGHIEM, N. (2013). Optimal rotation age for carbon sequestration and biodiversity conservation in Vietnam. *Forest Policy and Economics*.

NIJKAMP, P., VINDIGNI, G. AND NUNES, P. A. L. D. (2008). Economic valuation of biodiversity: A comparative study. *Ecological Economics* **67**(2), 217-231.

NISBET, T. R., MARRINGTON, S., THOMAS, H., BROADMEADOW, S. AND VALATIN, G. (2011). Slowing the Flow at Pickering. Final Report to Defra FCERM Multi-objective Flood Management Demonstration project RPM5455. April 2011. Forest Research. Available at: http://www.forestry.gov.uk/fr/INFD-7ZUCQY#final1

NOACK, F. A. W., MANTHEY, M., RUITENBEEK, J. H. AND MOHADJER, M. R. M. (2010). Separate or mixed production of timber, livestock and biodiversity in the Caspian Forest. *Ecological Economics* **70**(1), 67-76.

PERHANS, K., GLÖDE, D., GILBERTSSON, J., PERSSON, A. AND GUSTAFSSON, L. (2011). Fine-scale conservation planning outside of reserves: Cost-effective selection of retention patches at final harvest. *Ecological Economics* **70**(4), 771-777.

PERRY, N. (2013). The precautionary principle, uncertainty and the Noah's Ark problem. *Wildlife Research* **40**(2), 117-125.

POLASKY, S., JOHNSON, K., KEELER, B., KOVACS, K., NELSON, E., PENNINGTON, D., PLANTINGA, A. J. AND WITHEY, J. (2012). Are investments to promote biodiversity conservation and ecosystem services aligned? *Oxford Review of Economic Policy* **28**(1), 139-163.

POUTA, E. (2005). Sensitivity to scope of environmental regulation in contingent valuation of forest cutting practices in Finland. *Forest Policy and Economics* **7**(4), 539-550.

RANIUS, T., EKVALL, H., JONSSON, M. AND BOSTEDT, G. (2005). Cost-efficiency of measures to increase the amount of coarse woody debris in managed Norway spruce forests. *FOREST ECOLOGY AND MANAGEMENT* **206**, 119-133.

ROBERT, N. AND STENGER, A. (2013). Can payments solve the problem of undersupply of ecosystem services? *Forest Policy and Economics* **35**(0), 83-91.

SECRETARIAT OF THE CONVENTION ON BIOLOGICAL DIVERSITY (2005) *Handbook of the Convention on Biological Diversity Including its Cartagena Protocol on Biosafety*. 3rd edn. Montreal, Canada. Available at: http://www.biodiv.org.

SMITH, G., GITTINGS, T., WILSON, M., FRENCH, L., OXBROUGH, A., O'DONOGHUE, S., O'HALLORAN, J., KELLY, D., MITCHELL, F. G., KELLY, T., IREMONGER, S., MCKEE, A.-M. AND GILLER, P. (2008). Identifying practical indicators of biodiversity for stand-level management of plantation forests. *Biodiversity and Conservation* **17**(5), 991-1015.

SWALLOW, S. K., PARKS, P. J. AND WEAR, D. N. (1990). Policy-relevant nonconvexities in the production of multiple forest benefits. *Journal of Environmental Economics and Management* **19**(3), 264-280.

THIENE, M., MEYERHOFF, J. AND DE SALVO, M. (2012). Scale and taste heterogeneity for forest biodiversity: Models of serial nonparticipation and their effects. *Journal of Forest Economics* **18**(4), 355-369.

TYRVÄINEN, L., MÄNTYMAA, E. AND OVASKAINEN, V. (2013). Demand for enhanced forest amenities in private lands: The case of the Ruka-Kuusamo tourism area, Finland. *Forest Policy and Economics*.

UK NATIONAL ECOSYSTEM ASSESSMENT (2011). UK National Ecosystem Assessment. UNEP-WCMC. Cambridge. Available at: http://uknea.unep-wcmc.org/Resources/tabid/82/Default.aspx.

VIERIKKO, K., PELLIKKA, J., HANSKI, I. K., MYLLYVIITA, T., NIEMELA, J., VEHKAMAKI, S. AND LINDEN, H. (2010). Indicators of sustainable forestry: The association between wildlife species and forest structure in Finland. *ECOLOGICAL INDICATORS* **10**, 361-369.

VIERIKKO, K., VEHKAMAKI, S., NIEMELA, J., PELLIKKA, J. AND LINDEN, H. (2008). Meeting the ecological, social and economic needs of sustainable forest management at a regional scale. *SCANDINAVIAN JOURNAL OF FOREST RESEARCH* **23**, 431-444.

WIKBERG, S., PERHANS, K., KINDSTRAND, C., DJUPSTROM, L. B., BOMAN, M., MATTSSON, L., SCHROEDER, L. M., WESLIEN, J. AND GUSTAFSSON, L. (2009). Cost-effectiveness of conservation strategies implemented in boreal forests: The area selection process. *BIOLOGICAL CONSERVATION* **142**, 614-624.

WILLIS, K. G., GARROD, G., SCARPA, R., POWE, N., LOVETT, A., BATEMAN, I. J., HANLEY, N. AND MACMILLAN, D. C. (2003). The Social and Environmental Benefits of Forests in Great Britain. Report to Forestry Commission. Centre for Environmental Appraisal and Management, University of Newcastle upon Tyne.

Appendices

Annex I: Biodiversity: Definitions and Policy Context

Definitions

The word 'biodiversity' is strictly a contraction of the term 'biological diversity' and is commonly used to describe the number, variety and variability of living organisms. The multitude of parameters that this encompasses, and the complexities associated with its measurement, has over time perpetuated many conflicting definitions of the term. It has become widespread practice to define biodiversity in terms of genes, species and ecosystems. As such it includes the number of species, the genetic differences among them and the community structures and ecosystems in which they occur over a specified region.

Species diversity is most commonly associated with species richness i.e. the number of individual species present in an area but may also include measures of abundance. Genetic diversity represents the heritable variation within and between populations. Ecosystem diversity is less clearly defined but is likely to include the structural, functional and compositional components of a habitat.

For each attempt to measure or quantify biodiversity a definition needs to be established. These characteristically vary from study to study. They frequently assess a subset of biodiversity –for instance, species richness and community structure, but not genetic variance. Measuring multiple aspects of biodiversity – particularly those associated with ecosystems - have often led to the use of proxy indicators. Whilst these indicators might include species, they also often typically include components of a habitat such as deadwood or structural complexity that are likely to be associated with a large number of species and ecosystem functions. Biodiversity can also be considered to encompass important elements of ecological resilience and reflect a history of adaptation. Thus, for example, 'biodiversity is more than the accumulated richness and diversity of species that inhabit the earth; rather, biodiversity includes the current interactions between communities of species, as well as the evolutionary history of these interactions' (DeClerck and Salinas, 2011).

Below are some definitions and notes on biodiversity as specified by notable scientific and political documents:

The Convention on Biological Diversity (CBD, http://www.cbd.int/convention/) gives a formal definition of biodiversity. Article 2 of the Convention (http://www.cbd.int/convention/articles/default.shtml?a=cbd-02) states that: "biological diversity means the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems".

Forest biological diversity is a broad term defined in the annex to decision II/9 (https://www.cbd.int/decision/cop/default.shtml?id=7082), where the Conference of the Parties (COP) recognized that: "Forest biological diversity results from evolutionary processes over thousands and even millions of years which, in themselves, are driven by ecological forces such as climate, fire, competition and disturbance. Furthermore, the diversity of forest ecosystems (in both physical and biological features) results in high levels of adaptation, a feature of forest ecosystems which is an integral component of their biological diversity. Within specific forest ecosystems, the maintenance of ecological processes is dependent upon the maintenance of their biological diversity. Loss of biological diversity within individual ecosystems can result in lower resilience."

In UK National Ecosystem Assessment biodiversity is defined as (UK National Ecosystem Assessment, 2011, Sythesis Report, p. 19): " All ecological processes are the product of interactions between different groups of organisms and are dependent on there being a range of these present. In this sense, biodiversity – the variety and variability of living organisms – ultimately underpins the functioning of all ecosystems and thereby the delivery of all ecosystem services."

Policy context

The increasing prominence of biodiversity has in part been due to recognition of increasing species declines and extinction risks – expected to further increase under climate change (Barnosky *et al.*, 2011; Maclean and Wilson, 2011), being accorded further attention as a result of introduced pests or diseases affecting certain host trees. According to a recent report (Burns *et al.*, 2013), 60% of the UK species for which quantitative assessments are available have declined over the past 50 years. Statistics published by the Forestry Commission similarly indicate that UK populations of woodland birds have declined by about a fifth since 1970 – including a decline of around two-fifths in woodland specialists (FC, 2013, Fig 5.1).

Biodiversity targets are prominent in EU environmental policies. Enhancing and preserving forest biodiversity is a high-priority topic of FOREST EUROPE (the Ministerial Conference on the Protection of Forests in Europe),[i] with Pan-European Indicators for Sustainable Forest Management adopted in 2002 including nine biodiversity indicators (FOREST EUROPE, 2010). European Union governments in 1992 adopted legislation designed to protect the most seriously threatened habitats and species across Europe

[i] Launched in 1990, it is the European policy process for the sustainable management of the continent's forests (http://www.foresteurope.org/).

(the Habitats Directive http://ec.europa.eu/environment/nature/legislation/habitatsdirective/), complementing the Birds Directive adopted in 1979. At the heart of both Directives is the creation of a network of Natura 2000 nature protection areas to assure the long-term survival of Europe's most valued and threatened species and habitats (http://ec.europa.eu/environment/nature/natura2000/).

Similarly, the UK Biodiversity Action Plan (UK BAP) established in 1994 in response to the Convention on Biological Diversity (CBD) lists priority species and habitats identified as being the most threatened and requiring conservation (http://jncc.defra.gov.uk/page-5155). The recent Biodiversity Strategy for England (Defra, 2011), for example, includes the key aim to: "halt overall biodiversity loss, support healthy well-functioning ecosystems and establish coherent ecological networks, with more and better places for nature for the benefit of wildlife and people." The UK Forestry Standard requires woodlands to be managed in a way that conserves or enhances biodiversity taking into account factors such as rotation length, tree species, and forest design, and opportunities for enhancing biodiversity should be considered in forest management plans (http://www.forestry.gov.uk/ukfs).

References for Annex I

Barnosky, A. D., Matzke, N., Tomiya, S., Wogan, G. O. U., Swartz, B., Quental, T. B., Marshall, C., McGuire, J. L., Lindsey, E. L., Maguire, K. C., Mersey, B. and Ferrer, E. A. (2011) 'Has the Earth/'s sixth mass extinction already arrived?', *Nature*. Nature Publishing Group, a division of Macmillan Publishers Limited. All Rights Reserved., 471(7336), pp. 51–57. Available at: http://dx.doi.org/10.1038/nature09678.

Burns, F., Eaton, M. A., Gregory, R. D., Al Fulaij, N., August, T., Biggs, J., Bladwell, S., Brereton, T. M., Brooks, D. R., Clubbe, C., Dawson, J., Dunn, E., Edwards, B., Falk, S. J., Gent, T., Gibbons, D. W., Gurney, M., Haysom, K. A., Henshaw, S., Hodgetts, N. G., Isaac, N., McLaughlin, M., Musgrove, A. J., Noble, D. G., O'Mahony, E., Pacheco, M., Roy, D., Sears, J., Shardlow, M., Stringer, C., Taylor, A., Thompson, P., Walker, K. J., Walton, P., Willing, M. J., Wilson, J. and Wynde, R. (2013) *State of nature*. Report. Available at: http://nora.nerc.ac.uk/502092/.

DeClerck, F. and Salinas, A. (2011) 'Measuring biodiversity. Ch. 3', in Rapidel, B., DeClerck, F., Le Coq, J. F., and Beer, J. (eds) *Ecosystem services from agriculture and agroforestry: measurement and payment.* London: Earthscan, p. 66.

Defra (2011) 'Biodiversity 2020: A strategy for England's wildlife and ecosystem services'. Edited by Department for Environment Food and Rural Affairs (DEFRA). London. Available at: http://www.defra.gov.uk/publications/2011/08/19/pb13583-biodiversity-strategy-2020/.

FOREST EUROPE (2010) 'Enriching Forest biodiversity, FOREST EUROPE is implementing Convention on Biological Diversity provisions'. Aas, Norway: FOREST EUROPE (Ministerial Conference on the Protection of Forests in Europe). Available at: http://www.foresteurope.org/documentos/EnrichingForestBiodiversity_print.pdf.

Forestry Commission (2013) *Forestry Statistics 2013*. Edinburgh, UK. Available at: http://www.forestry.gov.uk/website/forstats2013.nsf/LUContentsTop?openview&RestrictToCategory=1.

Maclean, I. M. and Wilson, R. J. (2011) 'Recent ecological responses to climate change support predictions of high extinction risk', *Proc Natl Acad Sci U S A*. 2011/07/13, 108(30), pp. 12337–12342. doi: 1017352108 [pii]10.1073/pnas.1017352108 [doi].

UK National Ecosystem Assessment (2011) *UK National Ecosystem Assessment*. Report. Cambridge: UNEP-WCMC. Available at: http://uknea.unep-wcmc.org/Resources/tabid/82/Default.aspx.

Annex II: Evidence Review: Links between biodiversity and rotation length

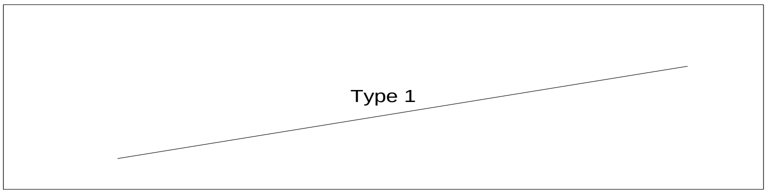
Introduction

With an aim of exploring the best approach to including biodiversity in optimal rotation length modelling, evidence was sought to confirm the broad shape of response curves between biodiversity and stand age. Drawing upon prior knowledge of some previous studies, a typology of potential relationships between biodiversity and stand age was developed. It was anticipated that this would provide a useful basis in undertaking the evidence review and be used to classify relationships for different forest types and components of biodiversity (e.g. for species that people most value). While we are primarily interested in woodland that is managed on a rotation basis for timber production, we are also interested in unmanaged stands or those with only minimal harvesting intervention as these might be included in a wider forest landscape matrix designed to enhance forest biodiversity.

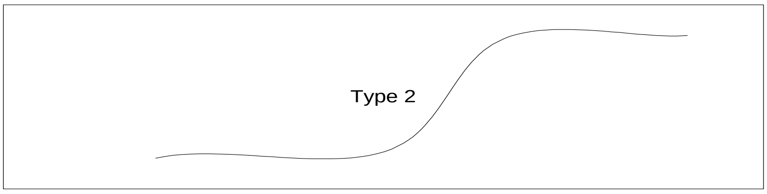
Types of biodiversity and stand age relationship

The following schematic of 8 potential relationships (by C. Quine) was initially suggested based upon prior knowledge.

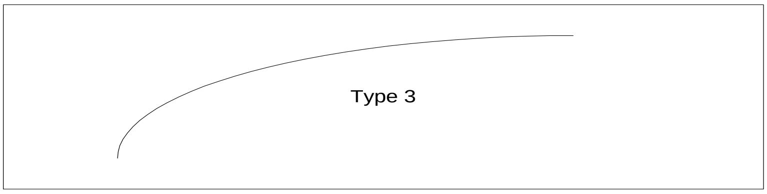
Figure 4. Schematic of some possible relationships between biodiversity (y axis) and rotation length (x axis)



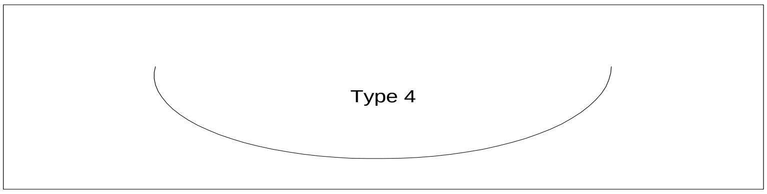
Biodiversity increases linearly with age



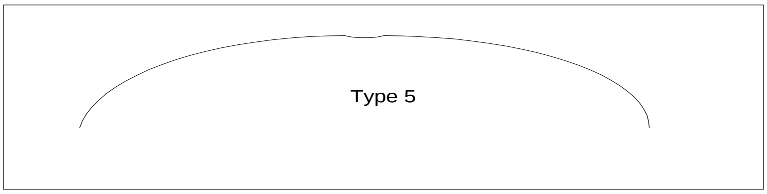
Biodiversity develops exponentially then saturates



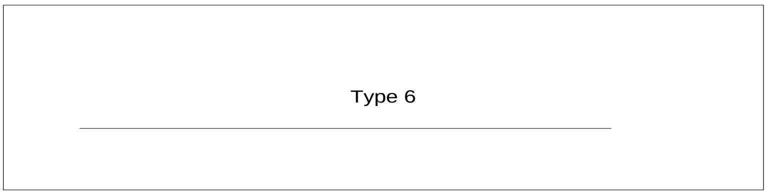
Biodiversity develops rapidly then rate of increase levels off



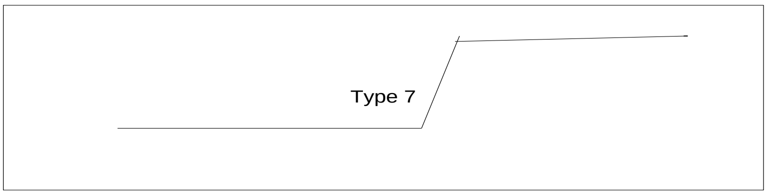
Biodiversity declines and then increases



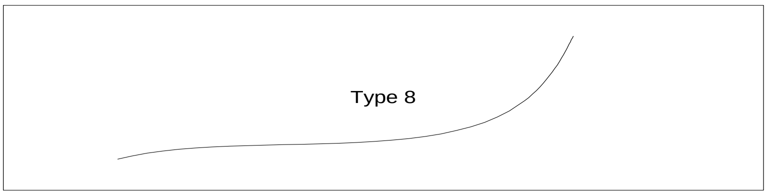
Biodiversity peaks and then declines



Biodiversity does not respond to rotation age



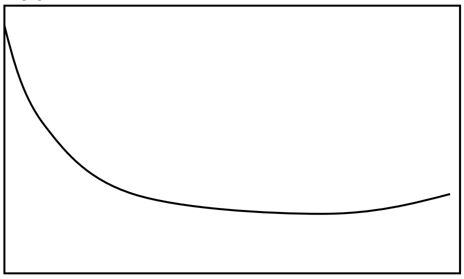
Biodiversity increases rapidly after a threshold is reached



Biodiversity increases exponentially and does not saturate within the length of rotation

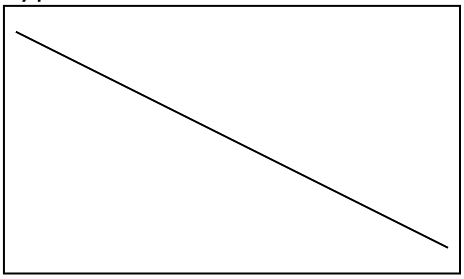
The evidence review (see below) suggested a further 5 types of relationship between biodiversity and stand age.

Type 9



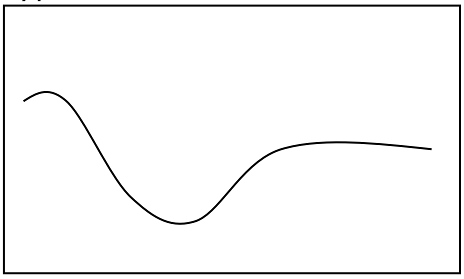
Biodiversity declines rapidly, stays stable in mid-rotation and increases slightly in late rotation.

Type 10



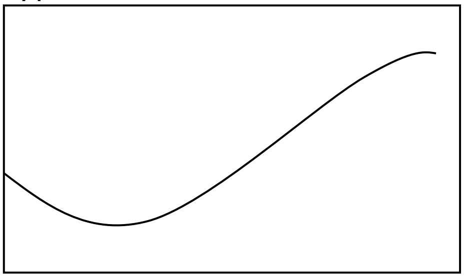
Biodiversity shows a linear decrease with rotation age.

Type 11



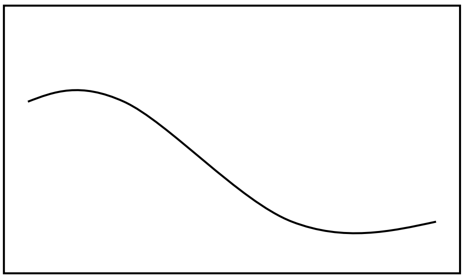
Biodiversity shows a steep slump from early to mid-rotation then increases and stabilises in late rotation stages

Type 12



Biodiversity increases in an almost linear fashion after an initial slump in early rotation

Type 13



Biodiversity shows a slight increase and then gradual decline through mid and late rotation stages

Literature Search

A literature search was undertaken using the Web of Knowledge, with all databases searched. The search was undertaken between 01/02/12 and 30/ 04/12.

The following Search String was used: Topic = (Biodiversity OR "species richness" OR "species diversity") AND Topic = (forest* OR wood* OR stand*) AND Topic = (age OR rotation OR "old growth" OR "forest management" OR felling OR harvesting*) AND Topic = (temperate OR boreal). Restricted to material published from 1992-2012, the search yielded 1,113 hits. Of these, 641 references seemed to warrant a closer look, with the Title and Abstract printed. (Others were classed as irrelevant). On further inspection, 398 references were selected as 'Possibly Relevant' (by Rachel Goldstein, intern from University of Rochester, New York) and the full articles obtained where available as a PDF (124 references). All articles that could be accessed in full, including hard copies, were then reviewed, with a table of agreed summary information collected from each article (Table 2 below), including ascribing a basic model illustrating the relationship between diversity and rotation length through time (see schematic of models 1-8 above and additional models suggested based on the literature review - see schematic of models 9-13 above).

Table 2. Summary information collected for each of the reviewed papers.

Table entry headings	Notes on entry requirements	Example of table entry
Reference:	Lead author, year Note: Studies should be no older than 20 years	Nascimbene, J., et al., 2010
Reference availability:	PDF/ HardCopy/ Abstract	Abstract only
Location:	Note latitude and longitude of study; If in the UK note as much detail as possible so that site can be related to UK climate zones. Studies conducted in temperate zones only to start	Italian Alps (no Long/ Lat available)

	with.	
Type of study	Time series (TS) = study of the same stand followed through time (e.g. through a rotation) OR Pairwise comparison (PC) = Chronosequence of similar stand types are studied at the same time	PC
Silvicultural system	Clear-cutting (CCT) Selection (SEL) Shelterwood (SW) Coppice (CP) See reference by Kerr, 1999 in Forestry - http://forestry.oxfordjournals.org/content/72/3/191.full.pdf Make a note if study is first rotation, 2nd etc. If it is first rotation, record this and make note of past landuse if this is given. Collect coppice articles - do not enter them at the moment.	SEL
Tree species	List all of the dominant canopy tree species in the stand.	Norway Spruce dominated
Stand Age/ Rotation length	Insert the ages of stands/rotation lengths ; these will be split into 3-4 categories on completion of reference searches	40-70; 80-120; 120-200; old growth >200
Biodiversity Indicator	See list of potential biodiversity indicators below that are surrogate measures of biodiversity and that should be considered when reviewing articles. Individual species or species groups are not included here as biodiversity indicators even though it is recognised that they can act as indicators in their own right. Note the units of assessment in the next entry box. This might simply be presence/absence or some method of quantifying an indicator value (e.g. frequency, volume/ha…) which might be significantly different in stands of different ages. <u>Biodiversity Indicators</u>: Deadwood, Regeneration of tree species, Shrub layer, Pits & Mounds, Veteran trees, Canopy cover & frequency/area of open areas within stand, Tree stem diameters & distribution, grazing intensity (includes damage such as bark stripping), incidence of pests& diseases, frequency of invasive species (e.g. rhododendron, Himalayan	N/A

	balsam, giant hogweed, bracken...), intensity of abiotic disturbances (e.g. windthrow, erosion, fire, damage from snow/drought).	
Assessment Methods	Describe how relationship is assessed between biodiversity and rotation age	Sampling of lichens in stands of different age categories
Measured Biodiversity	Species or species groups that were assessed (if any).	Richness of epiphytic lichens and composition in relation to tree age characteristics
Model Type	Direction of relationship with rotation age is best de=scribed by which schematic model 1-13?	Models 3 and 2
Stand-level/alpha Diversity effects (significance[a])	Include Stand-level/alpha Diversity effects (significance[a]) whether this is for Richness (S) Abund (J'), and/or Spp Diversity eg Shannon's H' [a] *n.s.* p>0.1; + p<0.1; * p<0.05; ** p<0.01; *** p<.001;	Richness (S) p<0.001

More details on the search protocol used is provided in Annex IV.

Results

The majority of studies reviewed used a pairwise comparison approach to research the effects of forest stand age on species diversity, typically comparing a chronosequence of stands of different ages at a given location. A less frequently encountered approach was a assessment of the same stands through time and in some cases this was achieved using model simulations. In many studies the silvicultural system that was practiced was not defined clearly enough to be able to distinguish between shelterwood or selection management systems that may have been in place. The composition of the stand in terms of homogeneity of stand age was also frequently not reported. Changes in species diversity for individual taxa were mostly reported in terms of the number of species present (species richness – *S*). Changes in other biodiversity indicators, such as coarse woody debris (CWD) and stand structural complexity, were also assessed in a number of cases; in these cases, relationships with biodiversity were assumed to be the following: the greater the volume of CWD or the more vertical layers of vegetation that were present, the greater the assumed species diversity. Most of the studies that were reviewed were from mainland Europe and North America. Only seven relevant British and Irish studies were reviewed.

A high proportion of the studies reviewed focussed on arthropod diversity responses to forest stand age. Among these, approximately half were studies that were investigating the responses of beetles (Table 3). Ground vegetation and lichen were also represented among the taxonomic groups that featured in many of the studies. Occurring with less frequency were studies on the responses of birds, mammals and fungi (Figure 4). Two indirect measures of species diversity included coarse woody debris and stand structure. For many of the taxa, stand age had mixed effects on species diversity, depending on the study and species group being studied. For example, there were 18 described cases of a significant positive effect of stand age on arthropod species diversity, 12 cases of a decline in species diversity through time and 6 cases where stand age had no effect. A clearer trend was observed for lichen, where the majority of studies (n=20) revealed a positive effect of stand age on species diversity with many of these following a schematic Model 1 relationship with rotation age. CWD was another biodiversity indicator that was most frequently found to be increasing through stand rotation age.

Discussion

It is apparent from the range of models that were identified in the studies reviewed, that the relationship between biodiversity and rotation length is not easily defined and may not necessarily be consistent within certain taxonomic groups (or across tree species). This is likely to be due to a hierarchy of influences on biodiversity within woodlands – from long-term shaping of assemblages (relating to past glaciation and land use history), to landscape context, woodland history, tree species composition, topography/climate, lithology and soil; stand stage; management activity. The number of references reviewed, spread across many taxonomic groups, did not yield sufficiently good sample sizes per taxonomic group to provide a clear trend in response. It is proposed that the review is focussed further to cover only a few taxonomic groups/biodiversity indicator types (e.g. lichen, CWD, ground vegetation, spiders) and also to make note of species richness responses of forest specialists through a forest rotation for different types of site as these are the species that are most in need of conservation. (This could also aim to distinguish relationships for different site types – such as for plantations on ancient woodland sites which could differ from those at other sites). There are an insufficient number of relevant studies to allow for specific selection of species or species groups that provide a clear, well described 'ecosystem service'. Finally, a meta-analysis is required to complete a more stringent review of papers and draw relationships with rotation age (e.g. see paper by Paillet *et al*., 2009). For this, inclusion criteria must be much stricter than they are at present.

Table 3. Number of studies showing temporal change in species diversity with stand age and model types that best describe these temporal changes

Taxa / Biodiversity Indicator	Model types with increase through time	Total number of significant results	Model types with decrease through time	Total number of significant results	Model types with neutral effect	Total number of significant results	Undetermined model		Total number of studies
Arthropods	1, 2, 3, 4, 7, 8	18	5, 9, 10, 11	12	6	3	14		37
Ground vegetation	1, 3, 4, 8	16	5, 9, 10, 11	9	6	1	4		26
Lichen	1, 2, 3	12	5	1	6	2	7		20
Birds	1, 3, 8	4	10, 13	3	6	2	8		14
Coarse Woody Debris	1, 4, 7, 12	7	10	1			4		11
Bryophytes	1, 8	4	5, 10	3			2		9
Fungi	1, 4	2	9, 10	2			3		7
Mammals			9	1	6	2	3		6
Vegetation Structure	1, 4	2					2		4
Amphibians			10	1					1
Molluscs	3	1							1
Bacteria							1		1

Note: In some studies more than one species or species group is studied and these may have different results. This explains why the number of studies does not add up in table as might be expected.

See Figure 1. Number of studies indicating an increase, decrease or neutral response to stand age by taxonomic group/biodiversity indicator on page 16.

Reference for Annex II:

Paillet et al. (2009) Biodiversity differences between managed and unmanaged forests: Meta-analysis of species richness in Europe. *Conservation Biology*, 24(1): 101-112.

Annex III: Forest Management Interventions to increase Biodiversity

Introduction

Internationally there is increasing demand for forests to be managed sustainably, to maintain resources and minimise biodiversity loss. This is underpinned by policy and legislation across the globe. Biodiversity is widely recognised as a critical contributor to, and product of, healthy forest ecosystems (Quine *et al.* 2013). It is significant, not only for its contribution to the functioning of our environment (nutrient cycling, water quality, natural regeneration, carbon sequestration etc.), but also for its aesthetic beauty and cultural value.

The pressures to preserve biodiversity, however, are potentially in conflict with the increasing demands for timber and wood fuel. Foresters must maintain high yields of commercially viable timber whilst also managing the woods for other ecosystem services. Although not ignored completely, as most properties of biodiversity have no direct market value they have historically not been included in forest management plans (Angelstam 1996). Trade-offs between the current costs of production and the benefits to the environment must now be made to prevent long-term economic costs associated with failing ecosystem processes such as a reduction in levels of nutrient cycling and overall levels of soil fertility where decomposer communities are negatively impacted by high intensity production forestry (Perry 1994; Lindenmayer 1999; Bengtsson *et al.* 2000; Brandtberg, Lundkvist and Bengtsson, 2000). The challenge for foresters is to establish a strategy that integrates economics and ecology.

This part of the study aims to establish if there are optimal management interventions for biodiversity under differing management regimes: clear cut, shelterwood, coppicing etc. It does not attempt to link the requirements for biodiversity to an economic framework or to comment on best silvicultural practice for other ecosystem service gains such as carbon sequestration or recreation.

Methods

In order to examine existing research on the effects of silviculture on biodiversity a literature review was completed and the results/recommendations aggregated. The outcomes were then considered in terms of potential biodiversity metrics (e.g. environment bank offsetting metrics) to ascertain if these could contribute to an optimisation model for forest biodiversity in managed woodlands.

The literature review was completed (May – August 2013) using the Web of Knowledge and Google Scholar covering published work from 1992 to the current day. Relevant studies were found using the following terms in abstracts and titles: biodiversity; species richness; species diversity; plants; birds; lepidoptera; fungi; lichen; bryophyte; invertebrate; mammal; squirrel; earthworm; ant; forest management; timber; forestry; coppice; shelterwood; silviculture; continuous cover; clear cut; thinning; felling; plantation. Biodiversity related search terms were taken or inferred from woodland species highlighted in the UK Ecosystem Assessment (Quine *et al*, 2011). Searches were supplemented by examining bibliographies of articles and books for additional references. All sourced references were stored in an EndNote Web database.

Titles or abstracts of all found papers were scanned for applicability to UK forestry and those not deemed relevant were removed from the database (e.g. in general, studies that took place in the non-temperate regions were removed as were those concerning non-European species or forestry practices).

Summary of literature

A total of 167 references were deemed worthy of further consideration. These fell into the following categories: experimental field studies; observational field studies; literature reviews; practice guidelines and reference books.

The field studies were characterised by the examination of the response of a particular species (or set of species) to woodland management regimes. This was typically assessed on the basis of species abundance, richness and/or composition.

The studies varied in their scope and it is difficult to provide an informative summary of their outcomes. This is, in part, because they analysed various silvicultural practices – for instance some compared two or three methods of management and some examined different intensities of the same management system. The majority of studies compared one management practice to an unmanaged control site (for instance clear-cut sites to semi-natural forest), with the baseline state used in making comparisons varying between studies. Predictably the research also varied in its choice of response variable: studying either single species, single taxon, species groups, guilds, indicators or mixed taxa. The three most popular study groups were birds, lichens/bryophytes and invertebrates which together accounted for over half the studies.

Species-level research and biodiversity

The measurement of biodiversity is a well-recognised problem (Ratcliffe & Peterken 1995; Kerr 1999; Angelstam *et al.* 2001) not least because the term refers to diversity at many biological levels - genes, populations, species and ecosystems - but it can also be assessed at any geographic scale - local, regional or global (Millennium Ecosystem

Assessment 2005). No single study can hope to fully evaluate biodiversity so it is tempting to aggregate species-level research to represent the bigger picture.

In respect of studies related to woodland management, however, there are many concerns with this approach. Verschuyl *et al* (2011) completed a meta-analysis on the effects of thinning treatments on biodiversity. Their paper highlights the often contradictory responses from similar studies and indicates the difficulties of pooling such research to gain comprehensive understanding.

Regardless of the management regimes being studied the concerns are widespread and can be summarised as follows:

1. Determining species richness or abundance is too limited to assess biodiversity. Intensive management often results in repeated returns of early successional species. In these cases a proliferation of generalists may cause species richness to rise but the status of woodland specialists that are more likely to be under threat would not be captured by the study (Verschuyl *et al.* 2011). This was found to be the case in several papers examined for this report (Hansson 1994; Bus de Warnaffe *et al.* 2004; Boch *et al.* 2013).
2. The time scale of studies may be too short term to determine an effect (Hartmann *et al.* 2010). Cumulative effects can be especially important in understanding risks of 'extinction debt' (Lindenmayer 1999). Seasonal and management cycles also have a large impact. In a study of plant species in clear-felled stands in Thetford Forest Eycott et al (2006) found that only 50 of 217 recorded plant species persisted through the whole management cycle. Interannual variation is also likely to vary dramatically between species (Fuller & Green 1998; Brouat *et al.* 2004) and in relation to woodland management, species richness and abundance after disturbance can increase initially and then decrease (Augenfeld *et al.* 2008).
3. The species chosen for study are often biased towards ease of research and can be inappropriately used as surrogates for higher taxonomic groups (Smith *et al.* 2008a; Halme *et al.* 2010; Dolman *et al.* 2012).
4. Results are often conflicting and what is good for one set of species will be detrimental for another (Pawson *et al.* 2006).
5. Background variation is not taken into account. Few studies assess the variations in populations or temporal trends of their study species outside the study site. There could, for instance, be an overall regional decline or increase in the species unrelated to management practices which will have confounding effects on results (Augenfeld *et al.* 2008; Hartmann *et al.* 2010).
6. The persistence of a species is dependent on its survival not just its presence on a site at a moment in time. Of the studies considered for this report only one gauged survival rates (Klenner & Sullivan 2003).
7. Most studies are focussed on a limited number of sites. As a result they could lack the statistical power to detect true impacts (Lindenmayer 1999; Halme *et al.* 2010).
8. The site conditions of the comparative study areas are rarely thoroughly examined. Site history, aspect, micro climates, topography and surrounding land

use will all affect species composition even if management regimes are comparable (Ferris *et al.* 2000b; Eycott *et al.* 2006; Boch *et al.* 2013).

9. The scale of studies are rarely equivalent and in many cases are not sufficient to predict large-scale dynamics. On small spatial scales species interactions and compositions are often variable but on a larger scale may be more stable (Brouat *et al.* 2004).

It is clear from the above that combining results from individual studies will not provide a viable account of biodiversity as a whole (Kneeshaw *et al.* 2000). The challenge is to avoid conservation that is predicated on incomplete understanding of priorities or requires multiple competing species plans (Dolman *et al.* 2012).

However, despite the difficulty in comparing individual studies, it is possible to draw from the literature reviewed particular characteristics of management regimes that are identified as having a positive or negative effect upon certain aspects of biodiversity (Table 4).

Table 4. Generalised characteristics of different management regimes as positive or negative for biodiversity.

Where particular species are mentioned these are examples found in the literature reviewed only and will not be universal on all sites or in all conditions.

Silviculture	Species identified as benefiting from silviculture method		Characteristics identified as positive for biodiversity	Characteristics identified as detrimental to biodiversity	Relevant references
Clear cut	Hemiptera, carabid beetles Early successional plants Generalist bird species Birds of prey: owls, kestrels		• Large open spaces • Refuge for grassland species in intensively managed arable landscapes • Provision of edge habitat • Providing horizontal diversity on landscape scale	• Even aged structure • Lack of horizontal and vertical stand complexity • Structure favours generalists and excludes woodland specialists • Management technique and disturbance levels preclude many species • Lack of natural regeneration • Lack of tree species diversity	**Invertebrates**: (Altegrim & Sjoberg 1996; Brouat *et al.* 2004; Bus de Warnaffe *et al.* 2004) **Plants**: (Eycott *et al.* 2006) **Birds**: (Harris and Harris 1991)
Coppice	General: Establishment phase: Canopy Closure	Butterflies and moths, birds, ground flora including bluebells Birds: Early successional species (warblers, tree pipit, whitethroat) Small Mammals: (common shrew) Vascular plants	• Permanent and temporary open space • Standard trees • Varied ground flora • Structural diversity • Deadwood in abandoned coppice	• Lack of deadwood in active coppice • Lack of tree species diversity • Lack of structural diversity associated with abandoned or over mature	**General:** (Buckley 1992, Peterken 1993) **Birds:** (Fuller & Green 1998; Deconchat & Balent 2001) **Small mammals:** (Gurnell *et al.* 1992) **Plants:** (Gondard &

	Mature coppice Standard trees	Birds: Mid successional species (nightingale, blackcaps, chiff chaff, wren) Woodland specialist mammals and plants (wood mice, wood sorrel) Ants and saproxylic beetles (esp overmature coppice) Birds in general (Chaffinches, tree pipits, tits)		coppice	Romane 2005; Bartha *et al.* 2008) **Ants:** (Schlick-Steiner *et al.* 2005) **Beetles:** (Lassauce *et al.* 2012)
Selection felling	Lichens and bryophytes Beetles Birds – including cavity nesting species (i.e. woodpeckers) Vascular plants of all successional stages Earthworms Small mammals Invertebrates Lepidoptera		• Stand continuity • Structural complexity • Standing biomass • Tree Age distribution • Gap release and open areas • Horizontal diversity	• Few refuges for species susceptible to disturbance • Open areas can be too small to benefit a full suite of open habitat species • Absence of large veteran trees	**Birds:** (Robles *et al.* 2011) **Lichen**: (Boch *et al.* 2013) **Bryophytes** (Caners *et al.* 2013) **Beetles** (Johnston & Holberton 2009; Stenbacka *et al.* 2010) **Plants**: (Wilson & Puettmann 2007; Smith *et al.* 2008b) **Earthworms** (Castin-Buchet & Andre 1998) **Small mammals:** (Wilson & Carey 2000; Sullivan *et al.* 2001) **Invertebrates**: (Altegrim & Sjoberg 1996) **Lepidoptera**: (Summerville & Crist 2002)
Shelterwood*	Birds – especially foliage gleaners and canopy songbirds Ants Generalist lichen species		• Structural Diversity in mid story • Canopy trees • Seedling regeneration	• Lack of open space, ground flora and microhabitats • Lack of horizontal diversity • Even aged structure and lack of mature	**Birds**: (King & DeGraaf 2000; Augenfeld *et al.* 2008; Goodale *et al.* 2009) **Ants**: (Johnston & Holberton 2009) **Lichens:** (Nascimbene *et al.* 2007)

			forest	

*Shelterwood is not widely practised in the UK and its methodology varies considerably. References to shelterwood sourced in this review are limited in scope and relevancy compared to other management regimes

Biodiversity Indicators and silviculture

It is notable that the characteristics of management practises identified in table 1 correspond to indicators often used as proxies for biodiversity. The use of indicators is based on the need to have a practical way of assessing biodiversity that is ecologically sensible, repeatable, economic and interpretable on multiple scales (Kneeshaw *et al.* 2000; Smith *et al.* 2008a; Coote *et al.* 2013). In general indicators capture the components of woodland that are likely to benefit the maximum number of functions and species: not just those that are dependent on trees but also the many that require open space and microhabitats associated with a naturally wooded environment (Peterken 1996). The characteristics in table 1 that could therefore serve as surrogates for biodiversity are as follows:

- Tree Species diversity
- Vertical structure
- Horizontal diversity/landscape heterogeneity
- Deadwood
- % open area and edge habitat
- Age distribution of tree species and veteran trees
- Natural regeneration

Most indicators are easily manipulated in managed forests and form the basis for optimising biodiversity whatever management regime is being employed. Whilst additional indicators are sometimes identified as appropriate for assessment such as presence of water features; invasive species, herbivory pressures etc, the literature studied for this report suggests that the above list may be the most pertinent to UK forest management.

The choice of management system will be based on many factors besides conservation. For instance, in some areas prone to windblow, clear-felling regimes may be the only viable management option for timber production (Mason & Quine 1995; Malcolm *et al.* 2001). However, there is plenty of evidence to suggest that plantation forests can enhance biodiversity particularly when structural and compositional elements are appropriately managed (Kerr 1999; Hartley 2002; Humphrey *et al.* 2002b; Brockerhoff *et al.* 2008; Quine & Humphrey 2010). Conversely, on appropriate sites, transforming clear-cut practices to continuous cover can be beneficial economically (Davies & Kerr 2011) and could have far-reaching conservation benefits, helping offset some of the risks associated with planting imported stock. The literature reviewed for this study suggests that the use of indicators may provide the most viable basis on which to establish optimal management for biodiversity.

Limitations of indicators

Indicators are, by their nature, limited in scope – as the word implies they *indicate* rather than *determine*. Critically, in terms of conservation, they do not take account of existing species compositions – most importantly rare or protected species (Smith *et al.* 2008a). Prior to any management taking place it is important to consider if rare species are known (or likely) to be present and adopt strategies to conserve them. This includes species that may have particular societal importance for instance red squirrels, dormice, bluebells etc. It could also include species of ecosystem service interest such as earthworms, ants etc. Whilst it is possible to develop specific conservation strategies for individual species like these, preserving biodiversity as a whole relies on a broader outlook. If all of the indicators identified as important for biodiversity are simultaneously well managed it is likely that a large number of species groups will have preserved habitats.

Indicators act as surrogates for habitats and microhabitats (Ferris & Humphrey 1999). Actions taken on a human scale have implications for the environment at all levels of the ecological hierarchy from landscape to microscopic niches. Dolman et al (2012) have coined the term 'management guild' - highlighting the taxonomic diversity that is affected by individual management actions. Their approach relies on a consideration of microhabitats - recognising that most species require combinations of ecological structures. Therefore biodiversity indicators need to be managed at multiple scales.

Recommendations for management

Using indicators to inform woodland management can lead to several clear recommendations as often found in literature (Ratcliffe & Peterken 1995; Kerr 1999; Lindenmayer *et al.* 2006). The approach is more or less universal whether managing under coppice, clear fell, shelterwood, continuous cover etc. This is because indicators are designed to incorporate the landscape matrix as well as stand level interventions. (The deleterious effect of clear-felling, for instance, can be reduced via retention of trees or areas set aside for old growth).

Recognising that the scale at which interventions are made will also be critical, recommendations for woodland biodiversity conservation associated with the identified indicators include the following:

- Diversify tree species
- Ensure high levels of vertical structural complexity
- Increase landscape heterogeneity
- Ensure good quantities of deadwood
- Maintain/increase areas of open space
- Include old growth and veteran trees

Greater benefit from the above practical measures will also be gained from:

- Prior establishment of existing wildlife on site now and historically - (Ratcliffe & Peterken 1995; Kerr 1999)
- Ongoing programmes of monitoring and assessment – (Lindenmayer 1999; Noss 1999)
- Thoughtful appreciation of management effects on landscape and microhabitat levels. (Humphrey *et al.* 2004; Humphrey & Watts 2004)

It is noticeable that literature on optimum management for biodiversity rarely specifies thresholds on recommendations. This stems from the fact that landscapes and wildlife are dynamic and there is insufficient supporting data to substantiate such thresholds. Management systems have to be formulated in a range of differing conditions and not viewed as a single target (Kneeshaw *et al.* 2000).

The following sections provide background to some of the recommendations associated with the specific indicators identified in the literature. Where possible threshold values are specified on the basis of the prevailing research/conditions only.

Diversify Tree Species. Despite their reputation for producing ecological wastelands – plantations of non-native species in the UK can be beneficial for wildlife (Quine & Humphrey 2010). It is, however, well known that a diversity of tree species will reduce risks associated with pests, diseases and climatic events (Perry 1994). Other wildlife can show strong associations with particular trees. In a study on macro fungi, for instance, Ferris et al (2000) found 171 species unique to pine and 90 unique to Spruce. They found a direct positive correlation between the number of fungal species and the number of host tree species. UK Forestry Standard advises that no more than 75% of a management unit is allocated to one species (Forestry Commission, 2011). A species mix is also likely to enhance the aesthetic and recreation potential of a forest. "The forest management unit is the area subject to a forest management plan or proposal. This area is selected by the owner and/or manager and will be determined by the nature of the forest, the proposed operations and management objectives." (Forestry Commission, 2011)

Vertical structural complexity. Structural diversity is associated with multiple canopy layers. It is a major contributor to biodiversity as it has a profound effect on microclimates and the abundance of plants (Ferris *et al.* 2000a; Carey & Harrington 2001); small mammals (Carey & Harrington 2001); birds (Fuller & Green 1998; Augenfeld *et al.* 2008) and invertebrates (Humphrey *et al.* 1999; Jukes *et al.* 2001; Bus de Warnaffe *et al.* 2004). Complexity has to be maintained on numerous scales and can be achieved by avoiding intensive methods such as clear cutting and giving preference to selective or group thinning or other continuous cover methods. In clear-cut areas, blocks of trees can be retained which will act as refuges for some woodland specialists and late successional species. A meta-analysis of the benefits of retention trees to biodiversity on clear-fell sites (Rosenvald & Lohmus 2007; Rosenvald & Lohmus 2008) found that no species were adversely affected by retention trees and that many benefitted. (The configuration, age and species choice for retention trees is however understudied and it is suggested that aggregated and dispersed trees have separate advantages (Hartley 2002; Rosenvald & Lohmus 2008)). Diverse retained vegetation also provides a source of seeds, spores or individuals to re-colonise the plantation. Natural regeneration can be further encouraged with targeted areas of no herbicide use. Varied planting densities can also help to encourage a diversity of shrub and herb layer plants and bryophytes that may be light or shade loving.

Landscape heterogeneity. The horizontal structure of the stand and the plantation within the landscape will both affect the connectivity of habitats. Many species have very poor dispersal abilities whilst some may require hundreds of hectares as their home range (Peterken 1996). Altering the spatial configuration and size of managed zones can create opportunities to vary ecological characteristics. If clear cutting is practised – the adjacent land use should have a good diversity of tree species and vertical structure. The clear-cut areas will ensure provision for early successional species and the adjacent woodland will provide refuge for forest specialists that could otherwise be made locally extinct. Intensively managed patches should be in proportion to wooded areas. Ratcliffe and Peterken (1995) suggest they should not exceed 30 ha. Irregularly shaped edges and margins provide a buffer between two differing landscapes and facilitate dispersal. Studies on landscape ecology suggest that a forest needs to be at least 5 ha in extent to provide a distinct set of conditions (Ferris *et al.* 2000c) and that edge effects can occur up to 50 metres into a stand (Humphrey 2005).

Deadwood. Deadwood is widely recognised as a critical component of woodland communities and is an integral part of woodland conservation. It is invariably much less prevalent in managed stands than old growth or semi-natural woodland (Hartley 2002; Humphrey 2005). It is associated with 69 priority and protected species in the UK (Humphrey & Bailey 2012) and is instrumental in decomposition and nutrient cycling functions (Ferris & Humphrey 1999). It can also encourage natural regeneration, retain water and be a sink for carbon (Perry 1994; Hartley 2002). Deadwood can take a number of forms and different species are associated with each: decayed sections in living trees, rot holes, fallen trunks and branches; stumps and coppice stools plus woody debris in water courses (Ferris-Kaan *et al.* 1993). Increasing age and vertical structure of forests can naturally enhance amounts of deadwood. Snags and large trees with dead branches should be left in situ; stems with no commercial value can be left; some fallen deadwood and brash should be aggregated and left on site. Stacks of cut timber for commercial sale should, however, be taken from site promptly before colonisation by invertebrates etc. Humphrey and Bailey (2012) provide a full summary of management practises that can enhance deadwood resources. They emphasise the importance of connecting deadwood habitats across the landscape ensuring they are represented widely but with due concern for access points, recreation and workers.

Open Space: Open space is a key element for diversity in woodlands. It can be developed on edges, rides, access routes for management/recreation and water courses – all of which can benefit wildlife (Forestry Commission 2011). Open spaces in natural woodland are created by windfall, fire, and disease which may provide a reference point for management (Kerr 1999; Quine *et al.* 1999). Canopy removal generally results in a dramatic increase in ground flora and associated fauna. Canopy closure in unthinned stands can be high – often above 90% (Wilson & Puettmann 2007) precluding the survival of all but the most shade tolerant species. Creating gaps and open spaces through thinning forms habitat for species also associated with glades and edges rather than forest interiors. In intensively farmed regions open spaces within woodland can provide a refuge from the surrounding landscape(Buckley *et al.* 1997). Some studies suggest that open space in conifer plantations should account for 15% of the area (Peterken 1999) – more than the minimum 10% specified in the UK forestry standard. An extensive experiment on open space in Irish plantation forests (BIOFOREST) provides details of the benefits to multiple taxa including plants, birds and invertebrates (Iremonger et al 2006). The study identifies that on average a forest road of width 15 m had an inter-canopy width of only 9.2 m. The authors suggest that managers should therefore focus on absolute open space rather than gap/tree height ratios – although suggested use of these is provided elsewhere (Malcolm *et al.* 2001). The rate of canopy closure will also depend on the tree species planted. In respect of glades, Wilson and Puettman (2007) indicate that gaps > 0.4 ha are necessary for early successional species to establish and alter habitat in the long term. Iremonger et al (2006) found that glades of 625-900 m^2 were sufficient to maximise solar radiation and Perry (1994) specifies that 500m^2 is the minimum to allow for shade intolerant plant species. Configuring open space to enhance connectivity, increase edge profiles and exploit existing biodiversity hotspots, such as water features, ditches and banks, are likely to increase habitat availability and possibilities for natural regeneration. Open spaces may need continual management including deer control to maximise their benefit, however.

Old growth and age distribution. The effect of rotation age on biodiversity has been the subject of a more detailed appraisal elsewhere in this report. Old growth is valued for its heterogeneous structure with relatively open canopy, patchy understorey and a mixture of tree ages (Peterken *et al.* 1992; Humphrey 2005). Wildlife favours young and old growth (Peterken *et al.* 1992; Humphrey *et al.* 2002a) so ensuring a good diversity of age structures across the landscape will be optimal. Maintaining a proportion of trees to reach maturity is desirable - extending some rotations beyond normal felling age will help achieve this. Peterken (1992) advises that this can be effected by increasing rotation age of some zones (15-25 of area %) and reducing others. Continuous cover systems can allow trees to develop beyond maturity and retention trees in clear-fell sites can be left to form old-growth habitats. Veteran trees should be preserved and others should be singled out and allowed to achieve veteran status in the future. Selective thinning in old growth stands may not be detrimental (Humphrey 2005) – particularly after the stand has started to break up naturally (Mason 2000).

Forestry and Metrics on Biodiversity

In order to facilitate a framework for optimising biodiversity it would be beneficial to formulate a biodiversity metric for woodland habitats. Such metrics are currently used for environmental offsetting in order to deliver measurable biodiversity benefits in compensation for losses caused by development. In the UK, these metrics have been developed to be 'transferable between sites and habitats' (Defra 2012). Therefore, they are not used exclusively for assessing woodlands but incorporate all habitat types.

Further to the literature review undertaken for this paper, the criteria for developing these biodiversity metrics have been examined to see if they may help to establish a tool for assessing or valuing forest biodiversity.

The metric developed by Defra employs a scoring system which takes into account the habitat's 'distinctiveness' and its condition. The scoring equates to a number of units of worth ('currency') that the habitat represents which would need to be offset in the event of loss. Additional 'value' is afforded to the habitat by means of unit multipliers based on; the time taken to reach target condition; the risks of delivery failure; spatial risks associated with offsetting in different locations etc., (Defra 2012, Treweek 2009).

The condition assessment tool that the Defra metric employs is based upon the Farm Environment Plan (FEP) from the Higher Level Agri-environment Scheme (Natural England 2010). Each habitat feature in the FEP is characterised by a list of essential attributes ('criteria') which form the basis of the condition assessment. In respect of woodland environments specifically the FEP identifies the following principal habitats: ancient trees; wood pasture and parkland; mixed woodland; plantation on ancient woodland sites; landmark woodland; native semi-natural woodland and traditional orchards. Plantation or commercially managed woodlands are not included in the plan.

The criteria for assessment of native semi-natural woodlands in the FEP are:

1. Native species are dominant. Non-native and invasive species account for less than 10% of the vegetation cover.
2. A diverse age and height structure.
3. Free from damage (in the last five years) from stock or wild mammals – there should be evidence of tree regeneration such as seedlings, saplings and young trees.
4. Standing and fallen dead trees of over 20 centimetres diameter are present.
5. The area is protected from damage by agricultural and other adjacent operations.

These individual criteria do not have thresholds and are not valued or weighted. Condition is assessed on the basis of how many criteria are met for each assessed habitat block (Table 5). In the biodiversity metric the established condition score will then be multiplied by the units deduced for other components of the metric: i.e. area, distinctiveness, time delay etc to establish a single value for the habitat.

Table 5. Feature criteria and condition assessment categories used in biodiversity offsetting metric (Natural England 2010).

'Criteria' refers to list detailed in text above

Number of missed/failed criteria	Condition assessment category (and associated metric score)	Probable management level
0	A (3)	Maintain
1	B (2)	Maintain or restore
2 or more	C (1)	Restore

Some of the criteria used for assessing semi natural woodland in the FEP correspond to the biodiversity indicators discussed in this report and are similar to the criteria that are used to assess SSSI (Site of Special Scientific Interest) woodland which has more clearly defined thresholds. The SSSI assessment criteria for woodland are as follows (Kirby *et al.* 2002):

- Area.
- Structure and natural process – canopy and shrub layer, veteran trees, open space, deadwood etc.
- Regeneration Potential – saplings and seedlings, extent of regrowth.
- Composition – proportion and number of native trees and shrubs.
- Quality Indicators – ground flora, protected species, good transition zones etc.

It is notable that both the SSSI condition assessment and the offsetting metric developed by Defra acknowledge the need to assess targets and priorities locally thereby giving scope for the recognition of species or features of conservation concern. Neither are however designed to assess commercial forestry or consider the landscape matrix.

Applying the principles of this metric to facilitate optimisation models for forest biodiversity would require confirmation of the suite of indicators to be measured across all woodland types. In addition, threshold values and weightings would need to be established. In its simplest form this could be a basic assessment of how many criteria (based on biodiversity indicators) a woodland meets (as in FEP assessments) however issues of spatial scales would need to be addressed. To create such a metric therefore would require additional research and establishment of:

- The biodiversity indicators to be included in the metric
- Thresholds within each indicator applicable across all woodland and management types – or separate metrics per management type (for instance ranges of suitable open space, ideal age distributions of trees etc)
- The scale or area over which the indicators are assessed
- Relative weightings of indicators (i.e. does the open space criteria have equal weighting for biodiversity as deadwood etc)
- An associated scoring or valuation system

A scoring system for measuring forest condition in the UK does not yet formally exist. Therefore whilst Defra's metrics could form the structural basis for scoring biodiversity indicators it does not provide enough specific information to apply to a single habitat such as woodland.

Conclusion

This review demonstrates that there is no single optimal way to manage woodlands for biodiversity. This is because woodlands have particularly dynamic and complex structures and their associated biodiversity responds to these in various ways. Furthermore a forest stand does not operate in isolation but has to be managed as part of a landscape (resembling a catchment).

It is difficult to place universal threshold values on management recommendations. There are, however, features of landscapes associated with management that are known to benefit certain species or species groups. An indicator approach provides a viable way to maximise the chance of multiple species conservation gain on many scales.

Providing multiple scales are simultaneously considered, broad management recommendations can be applicable to any silviculture method. The literature reviewed in this study suggests that the following attributes have the most relevance to biodiversity and woodland management.

- Tree species diversity
- Vertical structure
- Landscape heterogeneity
- Deadwood
- Open space and edge habitat
- Age distribution of trees and veteran trees
- Natural regeneration

Incorporating these indicators into a metric for assessment or valuation of forest biodiversity is dependent on more research and consideration. This would require agreement on which indicators should be included, their associated thresholds and how they are weighted against each other. Also whether these are applied to all woodlands universally or tailored for differing management systems. Issues of scale of assessment would also have to be resolved. However, with further research and expert consideration the indicators identified in the report could certainly form the basis of a metric along the lines established by Defra for biodiversity offsetting.

References for Annex III:

Altegrim, O. & Sjoberg, K. (1996). Comparison of invertebrate abundance and biomass between a clear-cut stand and four stands under tree cover in Swedish boreal coniferous forests. *Entomologica Fennica*, 7, 17-27.

Angelstam, P., Breuss, M. & Mikusinski, G. (2001). Toward the assessment of forest biodiversity at the scale of forest management units - a European landscape perspective. *EFI Proceedings*, 59-74.

Angelstam, P.K. (1996). Forest landscape management for maintenance of biodiversity - a Swedish perspective. *EFI Proceedings*, 69-86.

Augenfeld, K.H., Franklin, S.B. & Snyder, D.H. (2008). Breeding bird communities of upland hardwood forest 12 years after shelterwood logging. *Forest Ecology and Management*, 255, 1271-1282.

Bartha, S., Merolli, A., Campetella, G. & Canullo, R. (2008). Changes of vascular plant diversity along a chronosequence of beech coppice stands, central Apennines, Italy. *Plant Biosystems*, 142, 572-583.

Bengtsson, J., Nilsson, S.G., Franc, A. & Menozzi, P. (2000). Biodiversity, disturbances, ecosystem function and management of European forests. *Forest Ecology and Management*, 132, 39-50.

Boch, S., Prati, D., Hessenmoller, D., Schulze, E.D. & Fischer, M. (2013). Richness of lichen species, especially of threatened ones, is promoted by management methods furthering stand continuity. *PLoS ONE*, 8, e55461-e55461.

Brandtberg, P.-O., Lundkvist, H. and Bengtsson, J. (2000) 'Changes in forest-floor chemistry caused by a birch admixture in Norway spruce stands', *Forest Ecology and Management*, 130(1–3), pp. 253–264. doi: http://dx.doi.org/10.1016/S0378-1127(99)00183-8.

Brockerhoff, E.G., Jactel, H., Parrotta, J.A., Quine, C.P. & Sayer, J. (2008). Plantation forests and biodiversity: oxymoron or opportunity? *Biodiversity and Conservation*, 17, 925-951.

Brouat, C., Meusnier, S. & Rasplus, J.Y. (2004). Impact of forest management practices on carabids in European fir forests. *Forestry*, 77, 85-97.

Buckley, G.P., Howell, R., Watt, T.A., Ferris-Kaan, R. & Anderson, M.A. (1997). Vegetation succession following ride edge management in lowland plantations and woods. 1. The influence of site factors and management practices. *Biological Conservation*, 82, 289-304.

Buckley, G.P. (Ed) (1992) *Ecology and Management of Coppice Woodlands* Chapman and Hall, London

Bus de Warnaffe, G.d., Lebrun, P., de Warnaffe, G.d.B. & du Bus de Warnaffe, G. (2004). Effects of forest management on carabid beetles in Belgium: implications for biodiversity conservation. *Biological Conservation*, 118, 219-234.

Caners, R.T., Macdonald, S.E. & Belland, R.J. (2013). Bryophyte assemblage structure after partial harvesting in boreal mixedwood forest depends on residual canopy abundance and composition. *Forest Ecology and Management*, 289, 489-500.

Carey, A.B. & Harrington, C.A. (2001). Small mammals in young forests: implications for management for sustainability. *Forest Ecology and Management*, 154, 289-309.
Castin-Buchet, V. & Andre, P. (1998). The influence of intensive thinning on earthworm populations in the litters of Norway spruce and Douglas fir. *Pedobiologia*, 42, 63-70.
Coote, L., Dietzsch, A.C., Wilson, M.W., Graham, C.T., Fuller, L., Walsh, A.T. *et al.* (2013). Testing indicators of biodiversity for plantation forests. *Ecological Indicators*, 32, 107-115.
Davis, O., & Kerr, G., (2011) Cost s and Revenues of Transformation to Continuous Cover Forestry Forest Research Report
Deconchat, M. & Balent, G. (2001). Vegetation and bird community dynamics in fragmented coppice forests. *Forestry*, 74, 105-118.
Defra (2012) Biodiversity Offsetting Pilots Technical Paper: the metric for the biodiversity offsetting pilot in England Defra Publications, London
Dolman, P.M., Panter, C.J. & Mossman, H.L. (2012). The biodiversity audit approach challenges regional priorities and identifies a mismatch in conservation. *Journal of Applied Ecology*, 49, 986-997.
Eycott, A.E., Watkinson, A.R. & Dolman, P.M. (2006). Ecological patterns of plant diversity in a plantation forest managed by clearfelling. *Journal of Applied Ecology*, 43, 1160-1171.
Ferris, R. & Humphrey, J.W. (1999). A review of potential biodiversity indicators for application in British forests. *Forestry*, 72, 313-328.
Ferris, R., Peace, A.J., Humphrey, J.W. & Broome, A.C. (2000a). Relationships between vegetation, site type and stand structure in coniferous plantations in Britain. *Forest Ecology and Management*, 136, 35-51.
Ferris, R., Peace, A.J. & Newton, A.C. (2000b). Macrofungal communities of lowland Scots pine (Pinus sylvestris L.) and Norway spruce (Picea abies (L.) Karsten.) plantations in England: relationships with site factors and stand structure. *Forest Ecology and Management*, 131, 255-267.
Ferris, R., Purdy, K., Humphrey, J. & Quine, C. (2000c). An introduction to new landscape ecology research to enhance biodiversity in British forests. *Information Note - Forestry Commission*, 6 pp.-6 pp.
Ferris-Kaan, R., Lonsdale, D. & Winter, T. (1993). The conservation management of deadwood in forests. *Research Information Note - Forestry Authority Research Division*, 8 pp.-8 pp.
Forestry Commission (2011) UK Forestry Standard; The government's approach to sustainable forest management, Forstry Commission, Edinburgh
Fuller, R.J. & Green, G.H. (1998). Effects of woodland structure on breeding bird populations in stands of coppiced lime (Tilia cordata) in western England over a 10-year period. *Forestry*, 71, 199-218.
Gondard, H. & Romane, F. (2005). Long-term evolution of understorey plant species composition after logging in chestnut coppice stands - (Cevennes Mountains, southern France). *Annals of Forest Science*, 62, 333-342.
Goodale, E., Lalbhai, P., Goodale, U.M. & Ashton, P.M.S. (2009). The relationship between shelterwood cuts and crown thinnings and the abundance and distribution of birds in a southern New England forest. *Forest Ecology and Management*, 258, 314-322.

Gurnell, J., Hicks, M., Whitbread, S., (1992) 'The effects of coppice management on small mammal populations' in Buckley *Ecology and Managmenet of Coppice Woodlands*

Halme, P., Toivanen, T., Honkanen, M., Kotiaho, J.S., Monkkonen, M. & Timonen, J. (2010). Flawed Meta-Analysis of Biodiversity Effects of Forest Management. *Conservation Biology*, 24, 1154-1156.

Hansson, L. (1994). Vertebrate distributions relative to clear-cut edges in a boreal forest landscape. *Landscape Ecology*, 9, 105-115.

Hartley, M.J. (2002). Rationale and methods for conserving biodiversity in plantation forests. *Forest Ecology and Management*, 155, 81-95.

Hartmann, H., Daoust, G., Bigue, B. & Messier, C. (2010). Negative or positive effects of plantation and intensive forestry on biodiversity: A matter of scale and perspective. *Forestry Chronicle*, 86, 354-364.

Humphrey, J. & Bailey, S. (2012). *Managing deadwood in forests and woodlands*, Forestry Commission Practice Guide. Forestry Commission, Edinburgh. i–iv + 1–24 pp.

Humphrey, J., Ferris, R., Jukes, M. & Peace, A. (2002a). *Biodiversity in planted forests*.

Humphrey, J.W. (2005). Benefits to biodiversity from developing old-growth conditions in British upland spruce plantations: a review and recommendations. *Forestry*, 78, 33-53.

Humphrey, J.W., Ferris, R., Jukes, M.R. & Peace, A.J. (2002b). The potential contribution of conifer plantations to the UK Biodiversity Action Plan. *Botanical Journal of Scotland*, 54, 49-62.

Humphrey, J.W., Hawes, C., Peace, A.J., Ferris-Kaan, R. & Jukes, M.R. (1999). Relationships between insect diversity and habitat characteristics in plantation forests. *Forest Ecology and Management*, 113, 11-21.

Humphrey, J.W., Peace, A.J., Jukes, M.R. & Poulsom, E.L. (2004). Multiple-scale factors affecting the development of biodiversity in UK plantations. *Forest biodiversity: lessons from history for conservation*, 143-162.

Humphrey, J.W. & Watts, K. (2004). Biodiversity indicators for UK managed forests: development and implementation at different spatial scales. *EFI Proceedings*, 79-89.

Iremonger, S., Gittings, T., Smith, G.F., Wilson, M., Oxbrough, A., Coote, L., O'Donoghue, S., McKee, A.-M., O'Halloran, J., Kelly, D. O'Sullivan, A., Neville, P., Mitchell, F.J.G., O'Donnell, V., Kelly T. and Dowding, P. *Investigation of experimental methods to enhance biodiversity in plantation forests* BIOFOEST final report

Johnston, J.C. & Holberton, R.L. (2009). Forest management and temporal effects on food abundance for a ground-foraging bird (Catharus guttatus). *Forest Ecology and Management*, 258, 1516-1527.

Jukes, M.R., Peace, A.J. & Ferris, R. (2001). Carabid beetle communities associated with coniferous plantations in Britain: the influence of site, ground vegetation and stand structure. *Forest Ecology and Management*, 148, 271-286.

Kerr, G. (1999). The use of silvicultural systems to enhance the biological diversity of plantation forests in Britain. *Forestry (Oxford)*, 72, 191-205.

King, D.I. & DeGraaf, R.M. (2000). Bird species diversity and nesting success in mature, clearcut and shelterwood forest in northern New Hampshire, USA. *Forest Ecology and Management*, 129, 227-235.

Kirby, K., Latham, J., Holl, K., Bryce, J., Corbett, P., Watson, R. *et al.* (2002). Objective setting and condition monitoring within woodland Sites of Special Scientific Interest. *English Nature Research Reports*, 83 pp.-83 pp.
Klenner, W. & Sullivan, T.P. (2003). Partial and clear-cut harvesting of high-elevation spruce-fir forests: implications for small mammal communities. *Canadian Journal of Forest Research-Revue Canadienne De Recherche Forestiere*, 33, 2283-2296.
Kneeshaw, D.D., Leduc, A., Drapeau, P., Gauthier, S., Pare, D., Carignan, R. *et al.* (2000). Development of integrated ecological standards of sustainable forest management at an operational scale. *Forestry Chronicle*, 76, 481-493.
Lassauce, A., Anselle, P., Lieutier, F. & Bouget, C. (2012). Coppice-with-standards with an overmature coppice component enhance saproxylic beetle biodiversity: A case study in French deciduous forests. *Forest Ecology and Management*, 266, 273-285.
Lindenmayer, D.B. (1999). Future directions for biodiversity conservation in managed forests: indicator species, impact studies and monitoring programs. *Forest Ecology and Management*, 115, 277-287.
Lindenmayer, D.B., Franklin, J.F. & Fischer, J. (2006). General management principles and a checklist of strategies to guide forest biodiversity conservation. *Biological Conservation*, 131, 433-445.
Malcolm, D.C., Mason, W.L. & Clarke, G.C. (2001). The transformation of conifer forests in Britain - regeneration, gap size and silvicultural systems. *Forest Ecology and Management*, 151, 7-23.
Mason, W.L. (2000). Silviculture and stand dynamics in Scots pine forests in Great Britain; implications for biodiversity. *Investigacion Agraria, Sistemas y Recursos Forestales*, 175-197.
Mason, W.L. & Quine, C.P. (1995). Silvicultural possibilities for increasing structural diversity in British spruce forests: the case of Kielder Forest. *Forest Ecology and Management*, 79, 13-28.
Millennium Ecosystem Assessment, (2005). *Ecosystems and Human Well-being: Biodiversity Synthesis.* World Resources Institute, Washington, DC.
Natural England (2010) *Higher Level Stewardship Farm Management Plan* 3rd Edition
Nascimbene, J., Marini, L. & Nimis, P.L. (2007). Influence of forest management on epiphytic lichens in a temperate beech forest of northern Italy. *Forest Ecology and Management*, 247, 43-47.
Noss, R.F. (1999). Assessing and monitoring forest biodiversity: A suggested framework and indicators. *Forest Ecology and Management*, 115, 135-146.
Pawson, S.M., Brockerhoff, E.G., Norton, D.A. & Didham, R.K. (2006). Clear-fell harvest impacts on biodiversity: past research and the search for harvest size thresholds. *Canadian Journal of Forest Research-Revue Canadienne De Recherche Forestiere*, 36, 1035-1046.
Perry, D.A. (1994). Forest ecosystems. *Forest ecosystems.*, xvi + 649 pp.-xvi + 649 pp.
Peterken, G.F. (1993). Woodland conservation and management. *Woodland conservation and management.*, xvi + 374 pp.-xvi + 374 pp.
Peterken, G.F. (1996). Natural woodland: ecology and conservation in northern temperate regions. *Natural woodland: ecology and conservation in northern temperate regions.*, xiii + 522 pp.-xiii + 522 pp.
Peterken, G.F. (1999). Applying natural forestry concepts in an intensively managed landscape. *Global Ecology and Biogeography*, 8, 321-328.

Peterken, G.F., Ausherman, D., Buchenau, M. & Forman, R.T.T. (1992). Old-growth conservation within British upland conifer plantations. *Forestry (Oxford)*, 65, 127-144.
Quine, C.P., Bailey, S.A. & Watts, K. (2013). PRACTITIONER'S PERSPECTIVE: Sustainable forest management in a time of ecosystem services frameworks: common ground and consequences. *Journal of Applied Ecology*, 50, 863-867.
Quine, C. P., Cahalan, C., Hester, A., Humphrey, J., Kirby, K., Moffat, A. and Valatin, G. (2011). Ch 8 Woodlands. in R.Watson and S.Albon, editors. *UK National Ecosystem Assessment*. UNEP & Defra.
Quine, C.P. & Humphrey, J.W. (2010). Plantations of exotic tree species in Britain: irrelevant for biodiversity or novel habitat for native species? *Biodiversity and Conservation*, 19, 1503-1512.
Quine, C.P., Humphrey, J.W. & Ferris, R. (1999). Should the wind disturbance patterns observed in natural forests be mimicked in planted forests in the British uplands? *Forestry (Oxford)*, 72, 337-358.
Ratcliffe, P.R. & Peterken, G.F. (1995). The potential for biodiversity in British upland spruce forests. *Forest Ecology and Management*, 79, 153-160.
Robles, H., Ciudad, C. & Matthysen, E. (2011). Tree-cavity occurrence, cavity occupation and reproductive performance of secondary cavity-nesting birds in oak forests: The role of traditional management practices. *Forest Ecology and Management*, 261, 1428-1435.
Rosenvald, R. & Lohmus, A. (2007). Breeding birds in hemiboreal clear-cuts: tree-retention effects in relation to site type. *Forestry*, 80, 502-515.
Rosenvald, R. & Lohmus, A. (2008). For what, when, and where is green-tree retention better than clear-cutting? A review of the biodiversity aspects. *Forest Ecology and Management*, 255, 1-15.
Schlick-Steiner, B.C., Steiner, F.M. & Steiner, H.M. (2005). Effect of extensification of coppice management on Central European ant assemblages (Hymenoptera : Formicidae). *Entomologia Generalis*, 28, 23-37.
Smith, G.F., Gittings, T., Wilson, M., French, L., Oxbrough, A., O'Donoghue, S. *et al.* (2008a). Identifying practical indicators of biodiversity for stand-level management of plantation forests. *Biodiversity and Conservation*, 17, 991-1015.
Smith, K.J., Keeton, W.S., Twery, M.J. & Tobi, D.R. (2008b). Understory plant responses to uneven-aged forestry alternatives in northern hardwood-conifer forests. *Canadian Journal of Forest Research-Revue Canadienne De Recherche Forestiere*, 38, 1303-1318.
Stenbacka, F., Hjalten, J., Hilszczanski, J. & Dynesius, M. (2010). Saproxylic and non-saproxylic beetle assemblages in boreal spruce forests of different age and forestry intensity. *Ecological Applications*, 20, 2310-2321.
Sullivan, T.P., Sullivan, D.S. & Lindgren, P.M.F. (2001). Stand structure and small mammals in young lodgepole pine forest: 10-year results after thinning. *Ecological Applications*, 11, 1151-1173.
Summerville, K.S. & Crist, T.O. (2002). Effects of timber harvest on forest Lepidoptera: Community, guild, and species responses. *Ecological Applications*, 12, 820-835.
Treweek, J. (2009) *Scoping Study for the Design and Use of Biodiversity Offsets in an English Context* Final Report to Defra

Verschuyl, J., Riffell, S., Miller, D. & Wigley, T.B. (2011). Biodiversity response to intensive biomass production from forest thinning in North American forests - A meta-analysis. *Forest Ecology and Management*, 261, 221-232.
Wilson, D.S. & Puettmann, K.J. (2007). Density management and biodiversity in young Douglas-fir forests: Challenges of managing across scales. *Forest Ecology and Management*, 246, 123-134.
Wilson, S.M. & Carey, A.B. (2000). Legacy retention versus thinning: Influences on small mammals. *Northwest Science*, 74, 131-145.

Annex IV: Literature search protocols

The initial literature search and review followed the Government Social Research Service (GSR) Rapid Evidence Assessment (REA) guidance. An intern, a student from the US, Rachel Goldstein, conducted an initial literature search and created a bibliography of the results. The search protocol used (see Annex II) included:

- Lemmatization: On (i.e. finds alternative forms of the search term, for example, tooth and teeth)

The search string did not contain such search terms as ‘economics’ or ‘value’ to avoid unnecessary restrictions. Papers containing relevant keywords were tagged accordingly. The search string and tags were developed and agreed by the research team. Tags were developed as part of an iterative process as studies were reviewed.

Table 6. Tags used to describe papers (in alphabetic order)

1	age	18	even-aged	35	rotation length
2	ants	19	fauna	36	Scotland
3	arthropod	20	Finland	37	Scots pine
4	Australia	21	fir	38	spectral mixture analysis
5	beech	22	fungi	39	spider
6	beetles	23	indicator species	40	Spruce
7	birds	24	Ireland	41	squirrel
8	boreal	25	Italy	42	stand productivity
9	Britain	26	Japan	43	succession
10	climate change	27	lichens	44	Sweden
11	deadwood	28	mammals	45	temperate
12	disturbance	29	North America	46	TRIAD
13	economics	30	oak	47	UK
14	epiphyte	31	old-growth	48	understory
15	Estonia	32	pine	49	uneven-aged
16	eucalyptus	33	polypores	50	ungulates
17	Europe	34	retention tree groups	51	windthrow

A first selection of references was made based on the geographical location of the study and forest type. The order of geographical preference was: UK & Ireland, North Europe, Europe, North America, other. The forest type order of preference was even-aged monoculture productive forestry, other.

Titles and abstracts classed as irrelevant included studies with forest biodiversity keywords undertaken in the tropics.

Below are results for the group of 398 studies initially tagged as ‘Possibly Relevant’ and categorised using the ‘Label’ field (i.e. some studies fall under several):

Table 7. Examples of searches (through selected papers) with tags

Tags(Labels) search string	Hits
UK	1
UK OR Britain	8
UK OR Britain OR Scotland	13
UK OR Britain OR Scotland OR Ireland	16
UK OR Britain OR Scotland OR Ireland OR Europe	65
UK OR Britain OR Scotland OR Ireland OR Europe OR Sweden	98
UK OR Britain OR Scotland OR Ireland OR Europe OR Sweden OR Finland	117
North America	100
temperate	98
temperate OR boreal	345
uneven-aged	8
even-aged	31
(UK OR Britain OR Scotland OR Ireland) AND even-aged	0
(UK OR Britain OR Scotland OR Ireland OR Europe) AND even-aged	7
(UK OR Britain OR Scotland OR Ireland OR Europe OR Sweden) AND even-aged	9
(UK OR Britain OR Scotland OR Ireland OR Europe OR Sweden OR Finland) AND even-aged	10
economics	28
economics (OR Title OR Abstract OR Keywords)	28
economics AND (value in Title OR Abstract OR Keywords)	10

The table above shows, for example, that there are only 10 references for even-aged forests in Europe and only 28 references containing tag 'economics' out of the sample of 398 studies.

The Web of Knowledge database search was supplemented by adding relevant references cited in studies identified by the initial search.

Of the 238 'possibly relevant' studies available, 114 were reviewed by Rachel Goldstein (in alphabetical order before her internship ended), with the remaining 124 reviewed by Nadia Barsoum.

After the initial literature search, a few smaller searches were undertaken on specific aspects including: one focusing on ecological impacts of specific silvicultural interventions and one on biodiversity valuation. Taking advantage of free trial access to Scopus (http://www.elsevier.com/online-tools/scopus, the largest abstract and citation database of peer-reviewed literature) granted to Forest Research in 2013, a search on biodiversity valuation was run in October 2013, yielding some new results. It was run on English language publications produced after 2009 searching in titles, abstracts and keywords. The search string used was:

(biodiversity AND *valu* AND (forest* OR wood*)).

The time limit was chosen to target the most recent publications and minimise duplication of documents from the previous search. We strongly expected significant publications before 2010 would have already been identified by the previous search and the main goal of a new search with Scopus was to pick up the latest developments in the area since the first search was run. The search produced 1,984 hits, which fell to 42 after limiting results to those falling within the Economics subject area. Abstracts for the 42 hits were carefully read and some papers retained for further review work.

Data

Initial search results (from April 2012), as well as those from the subsequent searches on ecological impacts of silvicultural interventions and on biodiversity valuation are available as EndNote bibliographic databases. Further work by ecologists (including initial results) is maintained in a form of a Mendeley web based bibliography database by Robin Gill and Nadia Barsoum. This can be exported to the EndNote format if required.

Annex V: Valuing Biodiversity

This Annex briefly presents a review of economic methods currently used in valuing biodiversity. Examples of such valuations and estimated values for woodlands are then discussed.

Methods for valuing biodiversity

This summary draws upon a number of recent papers focusing in biodiversity and / or forest valuations (Atkinson *et al.*, 2012; Helm and Hepburn, 2012; Nijkamp *et al.*, 2008; Riera *et al.*, 2012), complemented by the government guidance on ecosystem valuation (DEFRA, 2007) and wider research (National Research Council, 2004).

Woodland biodiversity is largely a non-market benefit. (An exception is the developing market for biodiversity offsets).

A number of approaches have been developed to value environmental goods and services for which no market prices exist. This is either because they are not traded directly on organised markets or because they only have non-use values.

Non-use values are derived from the knowledge that environmental resources continue to exist (existence value), or are available for others to use now (altruistic value) or in the future (bequest value). Use values are associated with current or future uses of a good or service. One can distinguish: direct use values that may be 'consumptive' (e.g. edible fungi and other non-timber forest products) or 'non-consumptive' (e.g. recreational wildlife watching activities), and indirect use values that include key intermediate ecosystem services (e.g. soil formation, nutrient recycling etc.), and option values associated with retaining the option to use a resource in the future (Eftec, 2009). Major valuation methods are presented in the diagram below (Figure 5).

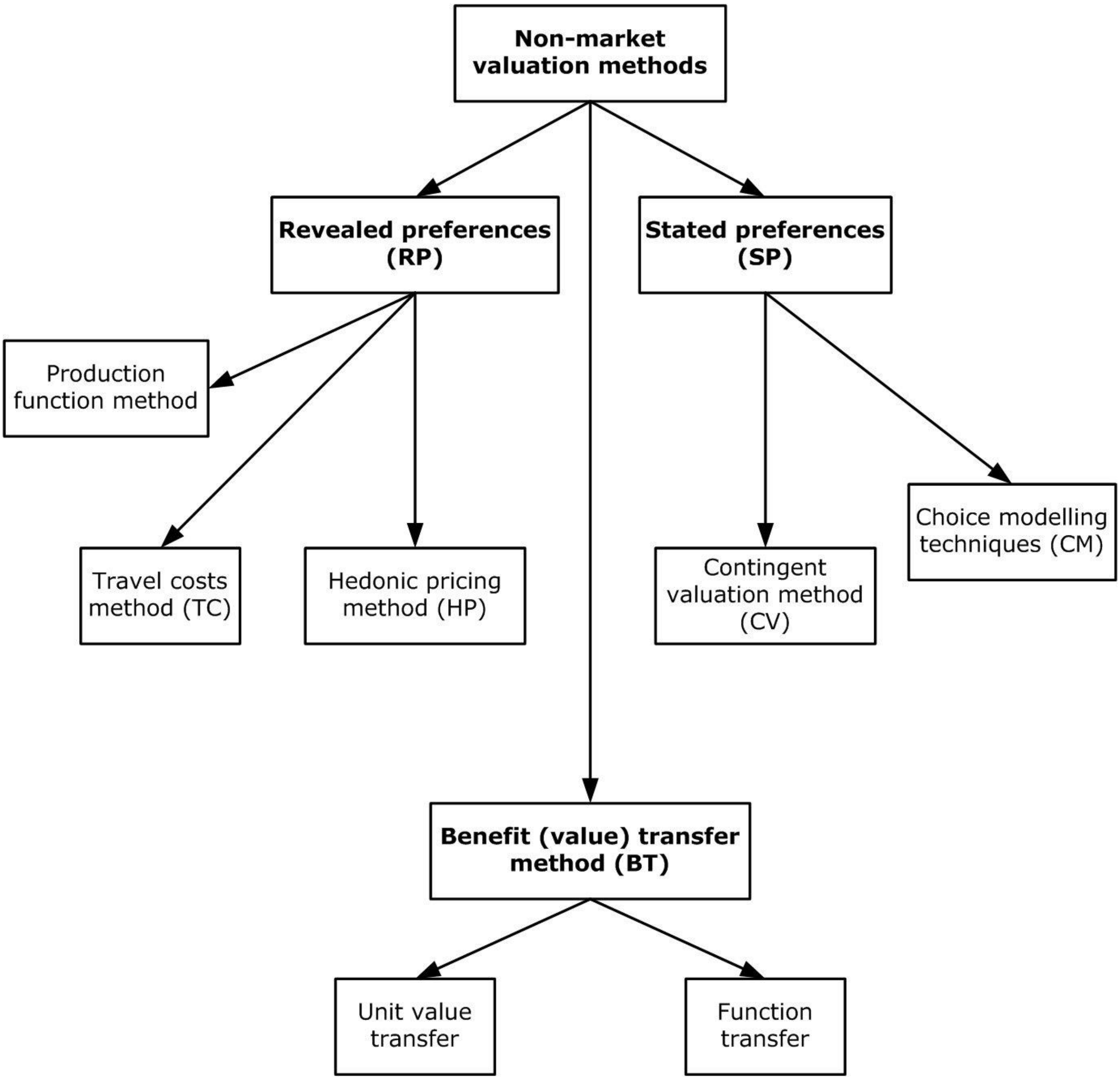


Figure 5. Non-market valuation methods

Methods to value environmental goods and services fall into two categories: revealed preference (RP) methods and stated preference (SP) methods. In the first category are production function, travel costs and hedonic pricing methods. The production function (PF) method assumes that there exists a relationship between an ecosystem service, or biodiversity, and the production of a market good or service. In this case biodiversity and ecosystem services serve as inputs to the production function and their values are deduced from the analysis of changes in production due to changes in the environment. The PF method captures indirect use values (Barbier, 2007).

The travel cost (TC) method uses estimated travel costs, including entry fees and the opportunity cost of travel time, as the implicit price of visiting recreational sites. An individual's demand (number of trips) depends on the cost of a trip to the site, the cost of visiting substitute sites, the characteristics of the site (e.g. type of forest), the characteristics of substitute sites, income and other demographic characteristics. Demand is expected to fall with an increase in the cost and rise with an improvement in site quality. The TC method is based on observed behaviour and estimates direct and indirect use value only. Major difficulties arise when trips are made to multiple destinations because it is extremely difficult to make attribution of time spent and consequently of value estimated to a specific site and associated ecosystem service. The travel cost method is often used for estimating recreational use values. The latter can encompass use values associated with biodiversity or encountering specific species.

The hedonic pricing (HP) method measures direct and indirect use value only. It values environmental (dis)amenities (e.g. visual and audio amenity) using data on market transactions for differentiated goods, e.g. houses. The value of biodiversity and ecosystem services is deduced from the relationship between the value of a property and measures of environmental (dis)amenities and other property and neighbourhood characteristics. Kevin Lancaster is credited with the idea that the consumer derives utility from various product characteristics (Lancaster, 1966; 1979). The formal theory of hedonic markets in the case of perfect competition with maximising producers and consumers, including demand function estimation was set out in Rosen (1974). The method is very data-intensive with the majority of applications related to property prices.

The second category (SP) includes contingent valuation (CV) and choice modelling (CM) techniques. CM is also often called choice experiment (CE). Both methods use surveys and involve surveys to elicit people's preferences with respect to biodiversity and ecosystem services by simulating changes in supply. In CV surveys, participants reveal their willingness-to-pay (WTP) or willingness-to-accept (WTA) compensation for a simulated change in the provision of a good or service. In CM surveys, participants are presented with a number of alternatives, including the status quo, and are asked to choose the most preferred alternative (CE), or to rate/rank (contingent rating/ranking), or to group (contingent grouping). Another version of CM surveys analyses respondents best and worst alternative choices (best-worst approach).

Some SP studies suffer from relatively small sample sizes with outcomes being strongly influenced by study design and implementation. The appropriate survey or choice experiment design is important in attempting to eliminate the common problems leading to overestimated values in SP studies: strategic behaviour, where respondents' answers depend on the perceived payment obligation and their expectations of actual programme implementation is one such problem. To obtain best estimates of WTP, careful explanation and detailed description of the goods under valuation, including the context (to avoid the embedding problem) have to be provided. The embedding problem arises when survey respondents are not valuing the specific goods in question, but reflecting some general value of a larger set of environmental resources. When asked about preserving a particular woodland, individuals may give a response that reflects their value for woodland preservation as a whole. Some of the positive responses can be explained by a "warm glow" effect of having contributed to a nature conservation cause.

Aside from specific survey/experiment design issues, other methodological issues with using SP methods include:

1. Whether to use CV or CE
2. Aggregation across individuals to obtain a total value
3. Benefits transfer problems (present in hedonic studies too)

Comparing results of CV and CE methods to identify the best for the task can be important (Hanley *et al.*, 2001). A study on valuing preservation of farmlands in Scotland argued that the CE method is best used for looking at the marginal, individual attribute values of open space resources and may therefore be best-suited for benefits transfer from one area to other areas (Hanley *et al.*, 1998, p. 4).

Aggregation issues are of particular concern in environmental cost-benefit analysis (Bateman *et al.*, 2005; Bateman *et al.*, 1999). The correct approach to aggregating individuals' benefits requires identification of the extent of the market (i.e. how broad should be geographic / spatial extent over which one aggregates individuals' marginal benefits) and how the benefit per person varies with changes in the provision of a good. Uncertainty about the individual's WTP value can have much less impact on estimates of total WTP than changes in the extent of the market associated with different approaches to aggregation. Using WTP responses to generate a spatially sensitive valuation function which incorporates both distance decay (to define the limits of the economic jurisdiction) and allows for variability in the socioeconomic characteristics of the encompassed population, is recommended.

Benefit transfer (BT), although not a valuation technique itself, is a widely used method which transfers economic values from one context (the 'study site' where they were generated) to another (the 'policy site' for which values are required) (Eftec, 2009; 2010b). It can be used with all valuation methods and allows more practical application of environmental valuations in policy-making by avoiding costly and lengthy primary valuation studies for each particular case. BT often leads to some loss in accuracy of estimates, which, however, may be acceptable for initial policy decisions. Where existing evidence is judged sufficiently robust, its potential applicability to other areas using benefit transfer can be assessed using UK government recommended guidance (Eftec, 2009; 2010b).

Comparing RP (especially HP) and SP methods, the former have an advantage of being based on market data and/or observed behaviour and, as a consequence, yield more robust and credible estimates. However, RP methods can only estimate use values.

Using SP methods, in principle, has the advantage that the total value (use and non-use) of any ecosystem service can be estimated, but their application is often difficult and expensive. Additionally SP studies can reveal the particular attributes of ecosystems that are valued by respondents. RP studies are better suited to measure the value of marginal changes in biodiversity, while the stated preference studies are capable of estimating the value of large changes. Therefore, the aspects valued and scale of changes which it is appropriate to consider are different for the two approaches.

As noted in the main body of this report, a further key consideration in valuing biodiversity is the complexity of the concept. A review of insights from behavioural economics for non-market valuation – including discussion of complexity, learning and the endogeneity of preferences and values, can be found in Moseley and Valatin (2013).

Biodiversity valuation estimates

This section presents a table with examples of UK and European valuation estimates found in the literature with a focus on woodland or related biodiversity. Priority is given to more recent studies, published after 2000. A survey of older studies – including estimates from UK woodland biodiversity valuation studies undertaken from the mid-1980s to the mid-1990s is provided by Nijkamp *et al.* (2008).

None of biodiversity valuations undertaken consider stand age explicitly, implicitly assuming fully grown forests as the basis of valuation or as alternative scenarios.

Table 8. Woodland biodiversity value estimates

Aspects of biodiversity valued	Method	Estimated values		Source and Comments
		Original study	at 2013 prices	
Native woodland biodiversity	CM (CE)	£259m per year(at 2011 prices)[1]	£268m per year	(Christie *et al.*, 2011): Estimated benefits of implementing UK BAP.
Wildlife and landscape benefits of the Environmental Stewardship scheme	CV and CE	£26.09 per household per year in England (at 2009 prices)[2]	£28.57 (per household per year)	(Boatman *et al.*, 2010) also estimated landscape and carbon benefits.
GB forest biodiversity	CM (CE)	£386m per year (at 2002 prices)[3]	£500m per year	(Willis *et al.*, 2003) based on (Garrod and Willis, 1997; Hanley *et al.*, 2002) biodiversity marginal benefit estimates for different forest types.
Biodiversity of nature reserves and countryside.	Legacies	£219/ha of National Trust countryside, £190/ha of RSBP reserve (in 2008/9 prices)	£246/ha of National Trust countryside, £213/ha of RSBP reserve (in 2013-14 prices)	(UK National Ecosystem Assessment, 2011) based on (Mourato *et al.*, 2010) estimated the non-use value of biodiversity.
Biodiversity of Sites of Special Scientific Interest (SSSIs)	CV	£0.41 to £1.14 per household per year for individual SSSIs in 2004 prices.	£0.51 to £1.41 (per household per year)	(CJC, 2004) estimated the use and non-use values of preserving or creating Sites of Special Scientific Interest (SSSI).
Diversity of Biodiversity: including species protection (familiar and unfamiliar), habitat restoration and other ES preservation.	CV and CE	Range from £42 to £59 (per household per year, in 2004 prices)[4]	£52 to £73 (per household per year)	(Christie *et al.*, 2006) examined WTP for biodiversity enhancements associated with agri-environmental and habitat re-creation policy on farmland in England.
Various aspects - including species protection, habitat restoration and other ES preservation.	CV and CE	£55 (£47) and £45 (£37) (per household per year, in 2004 prices) for Cambridgeshire and (Northumberland) respectively.[5]	£68 (£59) and £56 (£46) (per household per year) for Cambridgeshire and (Northumberland) respectively.	(Christie *et al.*, 2004) related to agri-environmental and habitat re-creation scheme in Cambridgeshire and Northumberland. Payments through increases in taxation over the next 5 years.
Biodiversity of various forest types.	Meta-analysis of various CV studies	Range £30 to £300 (per ha per year, in 2010 prices)[6]	Range £32 to £319 (per ha per year)	(Eftec, 2010a, pp.: 101-103) review of meta-analysis studies (Juutinen, 2008; Lindhjem, 2007; Nunes *et al.*, 2009) of forest biodiversity values.
Improvements in forest biodiversity attributes.	CM (CE)	€9.77 (in 2004 prices)[7]	£8.22 per person per year	(Thiene *et al.*, 2012). Study undertaken in the Solling-Harz region of Lower Saxony, Germany.
Biodiversity of increasing number of retention trees from 15 to 30 per ha.	CV	€40/ha (about 240 FIM in 1998 prices)	£38/ha	(Pouta, 2005) estimated a WTP for retention trees in Finland. Applied in (Koskela *et al.*, 2007).

Notes: 1. 'Current spend' scenario and 2011 prices (date of the report) assumed.

2. mean WTP for the landscape and wildlife scheme.

3. This study conducted eight focus groups in England, Scotland, and Wales, to evaluate the relative importance and value of biodiversity in different types of forest (e.g. upland native broadleaved woodland; lowland conifer forest; lowland ancient semi-natural broadleaved woodland). Focus was on non-use biodiversity values for different types of forest. Species (of birds, mammals, invertebrates, plants and fungi) and their richness were used for each forest type as biodiversity indicators; woodlands descriptions (size, management and composition) were presented to focus groups as well.

4. Mean WTP for the pooled scenarios. Payments through increases in taxation over the next 5 years.

5. WTP for habitat re-creation scheme and protection against biodiversity loss from development schemes.

6. WTP for non-use values of biodiversity and cultural aspects of various woodlands including old-growth boreal forests in Finland, forest protection in multiple use forestry and a general conception of biodiversity as the supporting service underpinning all other values.

7. WTP for the simplest model for species richness; assumed in 2004 prices, when data were collected. Forest biodiversity attributes, including: i) habitat for endangered species; ii) number of species (species richness); iii) forest structure; iv) landscape diversity. Euro is converted to GBP using ECB reference exchange rate, UK pound sterling/Euro average for 2004 was 0.67866 (accessed on 11/11/2013: http://sdw.ecb.europa.eu/browseSelection.do?DATASET=0&sfl1=4&FREQ=A&sfl3=4&CURRENCY=GBP&node=2018794).

References for Annex V

ATKINSON, G., BATEMAN, I. AND MOURATO, S. (2012). Recent advances in the valuation of ecosystem services and biodiversity. *Oxford Review of Economic Policy* **28**(1), 22-47.

BARBIER, E. B. (2007). Valuing ecosystem services as productive inputs. *Economic Policy* **22**(49), 177-229.

BATEMAN, I. J., GEORGIOU, S. AND LAKE, I. (2005). The aggregation of environmental benefit values: a spatially sensitive valuation function approach. *Centre for Social and Economic Research on the Global Environment (CSERGE) Working Paper EDM* **4**.

BATEMAN, I. J., LANGFORD, I. H. AND NISHIKAWA, N. (1999). The aggregation of non-user benefits: a case study illustrating different approaches. *CSERGE Working Paper GEC* **99-18**.

BOATMAN, N., WILLIS, K., GARROD, G. AND POWE, N. (2010). Estimating the Wildlife and Landscape Benefits of Environmental Stewardship. Final Report to the Department for Environment, Food and Rural Affairs (DEFRA). July 2010. York. Available at: http://archive.defra.gov.uk/evidence/economics/foodfarm/reports/documents/estimatingthewildlife.pdf.

CHRISTIE, M., HANLEY, N., WARREN, J., MURPHY, K., WRIGHT, R. AND HYDE, T. (2006). Valuing the diversity of biodiversity. *Ecological Economics* **58**(2), 304-317.

CHRISTIE, M., HYDE, T., COOPER, R., FAZEY, I., DENNIS, P., WARREN, J., COLOMBO, S. AND HANLEY, N. (2011). Economic Valuation of the Benefits of Ecosystem Services delivered by the UK Biodiversity Action Plan. Final Report to the Department for Environment, Food and Rural Affairs (DEFRA). August 2011. Aberystwyth.

CHRISTIE, M., WARREN, J., HANLEY, N., MURPHY, K., WRIGHT, R., HYDE, T. AND LYONS, N. (2004). Developing measures for valuing changes in biodiversity. Final Report to the Department for Environment, Food and Rural Affairs (DEFRA). Aberystwyth. Available at: http://jncc.defra.gov.uk/page-4020.

CJC (2004). Cost Effectiveness Study of Approaches for Delivery of PSA Target Relating to SSSIs. Final Report for Defra. April 2004. CJC Consulting. Available at: http://www.defra.gov.uk/evidence/economics/foodfarm/reports/sssi/wholerep.pdf

DEFRA (2007). An introductory guide to valuing ecosystem services. DEPARTMENT FOR ENVIRONMENT FOOD AND RURAL AFFAIRS (DEFRA). London. Available at: https://www.gov.uk/government/publications/an-introductory-guide-to-valuing-ecosystem-services--2

EFTEC (2009). Valuing Environmental Impacts: Practical Guidelines for the Use of Value Transfer in Policy and Project Appraisal. Report prepared for the Department for Environment, Food and Rural Affairs (DEFRA). December 2009. Economics for the Environment Consultancy Ltd (eftec). London. Available at: https://www.gov.uk/ecosystems-services

EFTEC (2010a). The economic contribution of the public forest estate in England. Report to the Forestry Commission England. 11 January 2010. London. Available at: http://www.forestry.gov.uk/pdf/eng-pfe-econmicresearch-final.pdf/$FILE/eng-pfe-econmicresearch-final.pdf.

EFTEC (2010b). Valuing Environmental Impacts: Practical Guidelines for the Use of Value Transfer in Policy and Project Appraisal. Technical Report prepared for the Department for Environment, Food and Rural Affairs (DEFRA). February 2010. Economics for the Environment Consultancy Ltd (eftec). London. Available at: https://www.gov.uk/ecosystems-services

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.

HANLEY, N., MACMILLAN, D., WRIGHT, R. E., BULLOCK, C., SIMPSON, I., PARSISSON, D. AND CRABTREE, B. (1998). Contingent Valuation Versus Choice Experiments: Estimating the Benefits of Environmentally Sensitive Areas in Scotland. *Journal of Agricultural Economics* **49**(1), 1-15.

HANLEY, N., MOURATO, S. AND WRIGHT, R. E. (2001). Choice Modelling Approaches: A Superior Alternative for Environmental Valuatioin? *Journal of Economic Surveys* **15**(3), 435-462.

HANLEY, N., WILLIS, K., POWE, N. AND ANDERSON, M. (2002). Social & Environmental Benefits of Forestry Phase 2: VALUING THE BENEFITS OF BIODIVERSITY IN FORESTS. Report to the Forestry Commission. April 2002. Centre for Research in Environmental Appraisal & Management, University of Newcastle.
HELM, D. AND HEPBURN, C. (2012). The economic analysis of biodiversity: an assessment. *Oxford Review of Economic Policy* **28**(1), 1-21.

JUUTINEN, A. (2008). Old-growth boreal forests: Worth protecting for biodiversity? *Journal of Forest Economics* **14**, 242-267.

KOSKELA, E., OLLIKAINEN, M. AND PUKKALA, T. (2007). Biodiversity conservation in commercial boreal forestry: The optimal rotation age and retention tree volume. *Forest Science* **53**, 443-452.

LANCASTER, K. J. (1966). A New Approach to Consumer Theory. *Journal of Political Economy* **74**(2), 132-157.

LANCASTER, K. J. (1979). *Variety, Equity, and Efficiency*. New York: Columbia University Press.

LINDHJEM, H. (2007). 20 years of stated preference valuation of non-timber benefits from Fennoscandian forests: A meta-analysis. *Journal of Forest Economics* **12**(4), 251-277.

MOSELEY, D. AND VALATIN, G. (2013). Insights from behavioural economics for ecosystem services valuation and sustainability. Forestry Commission Research Report. Forest Research. Edinburgh. Available at: http://www.forestry.gov.uk/pdf/FCRP022.pdf/$FILE/FCRP022.pdf

MOURATO, S., ATKINSON, G., COLLINS, M., GIBBONS, S., MACKERRON, G. AND RESENDE, G. (2010). Economic assessment of ecosystem related UK cultural services. UK NEA Economic Analysis Report. The Economics Team of the UK National Ecosystem Assessment, London School of Economics. London. Available at: http://uknea.unep-wcmc.org/Resources/tabid/82/Default.aspx

NATIONAL RESEARCH COUNCIL (2004). *Valuing Ecosystem Services: Toward Better Environmental Decision-Making*. Washington, D.C.: The National Academies Press.

NIJKAMP, P., VINDIGNI, G. AND NUNES, P. A. L. D. (2008). Economic valuation of biodiversity: A comparative study. *Ecological Economics* **67**(2), 217-231.

NUNES, P. A. L. D., OJEA, E. AND LOUREIRO, M. L. (2009). Mapping of Forest Biodiversity Values: A Plural Perspective. Fondazione Eni Enrico Mattei. Available at: http://EconPapers.repec.org/RePEc:fem:femwpa:2009.4.

POUTA, E. (2005). Sensitivity to scope of environmental regulation in contingent valuation of forest cutting practices in Finland. *Forest Policy and Economics* **7**(4), 539-550.

RIERA, P., SIGNORELLO, G., THIENE, M., MAHIEU, P.-A., NAVRUD, S., KAVAL, P., RULLEAU, B., MAVSAR, R., MADUREIRA, L., MEYERHOFF, J., ELSASSER, P., NOTARO, S., DE SALVO, M., GIERGICZNY, M. AND DRAGOI, S. (2012). Non-market valuation of forest goods and services: Good practice guidelines. *Journal of Forest Economics* **18**(4), 259-270.

ROSEN, S. (1974). Hedonic Prices and Implicit Markets: Product Differentiation in Pure Competition. *Journal of Political Economy* **82**(1), 34-55.

THIENE, M., MEYERHOFF, J. AND DE SALVO, M. (2012). Scale and taste heterogeneity for forest biodiversity: Models of serial nonparticipation and their effects. *Journal of Forest Economics* **18**(4), 355-369.

UK NATIONAL ECOSYSTEM ASSESSMENT (2011). UK National Ecosystem Assessment. UNEP-WCMC. Cambridge. Available at: http://uknea.unep-wcmc.org/Resources/tabid/82/Default.aspx.

WILLIS, K. G., GARROD, G., SCARPA, R., POWE, N., LOVETT, A., BATEMAN, I. J., HANLEY, N. AND MACMILLAN, D. C. (2003). The Social and Environmental Benefits of Forests in Great Britain. Report to Forestry Commission. Centre for Environmental Appraisal and Management, University of Newcastle upon Tyne.