LANDSCAPE ECOLOGY
OF TREES AND FORESTS

Proceedings of the twelfth annual IALE(UK) conference, held at the Royal Agricultural College, Cirencester, 21st-24th June 2004

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Woodland Trust, Autumn Park, Grantham, Lincolnshire, NG31 6LL

Cover design by Michelle Davies

ISBN 0 9547130 1 X

Printed in Great Britain by Colin Cross Printers Ltd, Garstang

CONTENTS

SESSION 1. RESPONSES OF PLANTS TO LANDSCAPE-SCALE PROCESSES

SESSION 2. RESPONSES OF ANIMALS TO LANDSCAPE-SCALE PROCESSES

SESSION 3. LANDSCAPE THRESHOLDS

SESSION 4. POTENTIAL OF TREES TO REDUCE IMPACTS FROM FARMING

SESSION 5. ASSESSING THE RESOURCE

SESSION 6. SPATIAL CONSERVATION STRATEGIES

SESSION 7. LANDSCAPE-SCALE ACTION

POSTER PAPERS

POSTER ABSTRACTS

PREFACE

The twelfth annual conference of the UK Chapter of the International Association for Landscape Ecology, IALE(UK), was organised in partnership with the Woodland Trust.

Landscape ecology is the study of the interactions between the temporal and spatial aspects of a landscape and its flora, fauna and cultural components.

The core objectives of IALE(UK) are to promote communication, inter-disciplinary research and the development of knowledge and interaction between scientists and those engaged in the planning and management of the landscape (www.iale.org.uk).

The Woodland Trust is the UK's leading woodland conservation charity (www.woodland-trust.org.uk). It has four main aims: no further loss of ancient woodland; restoring and improving woodland biodiversity; increasing new native woodland; and increasing people's understanding and enjoyment of woodland. It owns over 1,100 sites across the UK, covering around 19,000 hectares and has more than 300,000 members and supporters.

Habitat fragmentation and the potential impact of human-induced climate change are encouraging a move away from site-based protection to recognition of the importance of landscape-scale processes. Trees and forests play an important role in the wider countryside, providing a 'patchwork' of wildlife rich habitats as the focus for the development of an ecologically functional and sustainable landscape.

These conference proceedings will be of interest to conservation policy and decision makers, planners, land managers and researchers and explore:

- key concepts in landscape ecology, e.g., landscape thresholds, functional connectivity and physical connectedness
- spatial conservation strategies based on ideas such as forest habitat networks, focal species and generic species, functional attributes and surrogate measures, buffering to increase core area, linking to create 'wildlife corridors', and 'greenways'
- knowledge gaps.

I would like to thank the IALE(UK) committee for inviting me to organise a joint conference and for their practical support, particularly John Dover and Dan Chamberlain, whose advice on the organisation of previous IALE(UK) conferences proved invaluable. I was overwhelmed by the number and range of papers submitted and I am grateful to Kevin Watts for his suggestions on how best to structure the conference. Richard Bloomfield produced a first class website, Sandra Coleman did an excellent job of ensuring the smooth running of conference bookings and I am indebted to Gill Dennis and Rebecca Twine, whose attention to detail was vital in collating and refining the format of the book. I am very appreciative of the high standard of oral papers, poster papers and poster abstracts produced by contributors and to all their efforts in travelling from Australia, the United States and across Europe to present at the conference. Finally, I am eternally thankful to my wife, Diane, and my children, Bryony, Chloe and Rachel, for all their patience and support during an extremely busy period.

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The importance of landscape ecology for woodland policy: current issues and research priorities

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Abstract

The UK Government is committed to enhancing biodiversity at the landscape scale, and this concept is also featuring more strongly in the component country's forestry policies. The Habitat Action Plans for native woodland prescribe improvement, restoration and creation of native woodland. However, they do not indicate any relative priority between these actions, nor provide guidance on effective resource allocation. The mechanisms to deliver the HAPs, particularly incentives and associated guidance, need to prioritise very limited resources in order to achieve greatest ecological value for money. The development and delivery of policy requires us to address a series of landscape ecological questions. These range from the landscape scale through to the regional and national scale. Questions include: should ancient woodland restoration and improvement of semi-natural woodland be targeted at large and well-connected woods, or small and isolated woods? Should de-fragmenting (linking up isolated ancient woodland remnants) be approached with more caution? How can we evaluate (and therefore prioritise) the matrix or context in which woodland is created or restored? We also need to resolve how we should allocate resources between woodland improvement, restoration, creation and wider changes in land use. We frame these questions in terms of their landscape ecological research areas and suggest a series of research questions, which need to be addressed to inform policy development, with particular reference to England.

Introduction

International commitments and national policy context

The Forestry Commission (FC) has made commitments at the Global and European level to conserve and enhance forest biodiversity at the landscape scale through forest habitat networks, landscape planning and reducing fragmentation (MCPFE, 2003; PEBLDS 2002). The UK has also made a commitment at the Global and European level to develop monitoring frameworks and indicators of landscape ecological restoration. For example, there has been a call for development of indicators and targets to be integrated in all programmes of work within the Convention on Biological Diversity (CBD COP7, 2004).

The Global Partnership on Forest Landscape Restoration was launched last year by the Forestry Commission, WWF and IUCN the World Conservation Union (IUCN, 2004). It provides a focus for governments, communities, organisations and others the world over who are engaged in restoration activities to share on-the-ground examples of successful forest landscape restoration projects. The partnership is attracting a growing range of partners and has already provided support for a number of national and regional events to inspire and facilitate the positive exchange of experience for those active in landscape-scale restoration.

The UK Biodiversity Action Plan (UK BAP), produced in 1994 as a response to the Convention on Biodiversity, still provides the overall framework for conserving both habitats and species. This has been given legal weight in England and Wales through the Countryside and Rights of Way Act, 2000, which places a duty on Defra 'to further the conservation' of 'listed' habitats and species (HMSO, 2000). The England Biodiversity Strategy, produced in 2002 (Defra, 2002) is an expression of this duty and provides a policy commitment to help deliver the UK BAP. Progress with delivery of the Habitat and Species Action Plans is a headline indicator of the achievements of this Strategy. In Scotland there is parallel legislation and a Biodiversity Strategy is about to be published.

The England Biodiversity Strategy stresses the need for a landscape scale approach. The first aim given for the woodland sector, "to protect biodiversity-rich woodland", stresses the importance of addressing the impacts of adjoining land-uses, and addressing pests, such as deer, at the landscape scale. One of the other stated aims for woodland is, "to ensure that forestry, woodland management and creation...contributes to the conservation of biodiversity at a wider landscape scale".

Prioritising conservation activity

The Habitat Action Plans (HAPs) for native woodland (UK Biodiversity Partnership, 1994 - 1998) prescribe four basic activities:

- Protection of the existing area of native woodland
- Improvement in the condition or quality of the existing woodland
- Restoration of degraded woodlands, particularly those converted to plantations
- Creation of new native woodland

There is no relative prioritisation between these different activities, and the only targeting indicated in the plans is to achieve better progress with improving the condition of SSSIs. Local Biodiversity Action Plans, and emerging regional biodiversity strategies provide an opportunity for geographic prioritisation, but neither are currently closely linked to allocation of resources. On the other hand, the incentives available to promote all these activities are strictly limited, and considerably short of what is necessary to achieve even the existing targets. In England, our current prioritisation of activities needs to be strongly influenced by degree of threat and this suggests the following hierarchy for the HAP actions:

1. protect threatened high value ancient and semi-natural woodland (ASNW)
2. restore plantations on ancient woodland sites (PAWS) (since these are often threatened by the continuing impacts of exotic species)
3. improve the condition and quality of other ASNW
4. improve other secondary semi-natural woodland
5. create secondary woodland

A new package of woodland incentives is currently being developed, the *English Woodland Grant Scheme*, and this will facilitate regional and local targeting of incentives (Forestry Commission England, 2004). Such targeting will be directed by regional woodland strategies (sometimes called Regional Forestry Frameworks), which are also currently under development. If we are to deliver the native woodland HAPs we need to provide clear guidance on relative priorities, as funding will always be limited. The above priorities could help, but to confirm and refine them we really need answers to questions such as:

- Which threats are the most serious, urgent and irreversible?
- Should we prioritise improving the large and most valuable ancient woodlands, or protecting the more vulnerable small and isolated woods?
- How far should the presence of 'priority species' skew this prioritisation?
- Do some good quality secondary woodlands have greater potential than some degraded ancient woodland sites?

Improving the landscape context of woodland

One action is lacking from the Habitat Action Plans and that is:

- Improving the landscape context of native woodland.

One could argue that it is implicit in reducing threats and improving condition, but it needs to be made explicit that some aspects of this need to be addressed at a landscape scale. Woodland creation can also be considered as a means by which this might be achieved. Such creation appears to have been conceived within the HAPs primarily as a means of expanding the woodland resource but we are increasingly seeing it as a way of protecting the surviving fragments of ancient woodland. Almost by definition such recent secondary woodland cannot hope to match the ecological quality of ancient woodland, at least for many centuries. An early attempt at landscape targeting of woodland creation was made through the 'Jigsaw' Challenge under the Woodland Grant Scheme. This was a direct attempt to 'defragment' (in fact, expand, link and buffer) isolated ancient woodlands. There is now general agreement that buffering is almost always valuable, expanding is probably useful but physically linking ancient woodlands through woodland creation may be of limited benefit to the sedentary ancient woodland communities themselves.

Faced with dramatic predictions of the movement in 'climate space' (Harrison *et al.*, 2001) the common presumption is that we must enable habitats and species to move with it. But this seems to be at odds with the sedentary nature of ancient woodland specialists (Honnay *et al.*, 1999). And yet ancient woodland communities established themselves throughout the UK in a few thousand years following the end of the last glaciation. We clearly need a better understanding of the conditions that will allow such migration of sedentary species. This need leads to many questions around the landscape ecology of ancient woodland communities, in particular:

- What proportion of the species found in woodland, remembering that woodland glades are often the last refuges of non-woodland species, will benefit from newly created woodland corridors?
- How vulnerable are the meta-populations, which have survived for c.1,000 years in a semi-isolated state?
- How urgent is it to 'defragment', and what are the dangers of doing so?
- Can we provide guidance on the distances and intervening land uses which will allow interaction between existing and nearby newly-created woodland?

- How can we re-create the conditions under which ancient woodland species can move with their climate space, both incrementally and through chance long-distance dispersal?

Landscape approaches to incentives and delivery

The whole structure of agricultural support is being radically changed through reform of the Common Agricultural Policy (CAP). Maintaining 'Good Agricultural and Environmental Condition' is becoming a 'cross-compliance' requirement for receipt of Pillar 1 payments (Defra, 2004a). At the same time, the Pillar 2 agri-environment schemes are being re-designed (Defra, 2004b), this includes a new Entry Level Scheme. Both of these measures have huge potential to reduce threats at a landscape scale, to create buffers around woodland and to make farmland more 'wildlife friendly'. Such measures, dubbed 'broad and shallow', will achieve modest but near universal improvements in the landscape. The complementary Higher Level Scheme will provide more targeted payments for more substantial habitat management and enhancement activities ('narrow and deep'). These will take a 'whole farm approach' and some woodland management and creation actions are being incorporated within the Higher Level Scheme, with the aim of achieving a more integrated approach. More fundamentally, the 'rural funding streams' and the institutional arrangements for rural delivery in England are currently being reviewed, with the emphasis on achieving greater integration of the advice and incentives currently provided by different departments and agencies. Finally, there is growing interest in collaboration between adjoining landowners, in order to achieve landscape-scale benefits. For example, the area around Dunwich Heath and Minsmere in Suffolk, where half a dozen owners are collaborating to produce outcomes which are larger than the sum of the parts.

These changes prompt and facilitate a reconsideration of how changes in wider land use can help conserve woodland. The questions which underlie these new schemes and reviews include:

- Are changes in land use around woodland more or less important than changes in management of the woodland itself?
- Will 'broad and shallow' schemes deliver more or less benefit to woodland communities than 'narrow and deep' ones?
- What changes in surrounding land use will be most beneficial to woodland communities? (Buffers around woodland and/or hedges? Organic agriculture? Conserving field trees?)
- Conversely, what changes in woodland management will contribute most to the conservation of wider biodiversity outside the woodland?

Woodland management operations

Finally, the actual techniques used in woodland management, creation and regeneration can be critical. FC England is currently revising its guidance ('good practice') and conditions of grant aid paid for operations in ancient woodland. Even operations within woodland often need a landscape ecology perspective and raise questions such as:

- After felling, is it better to achieve woodland canopy more rapidly, through planting, or to wait for 10 or even 20 years for natural colonisation to occur?
- Re-creation of existing open space in ancient woodland is generally welcomed, but should we allow or even encourage creation of new open space?

- What parameters do we use to produce guidance on 'locally native species' and determine future natural ranges of our native species (e.g., beech, *Fagus sylvatica*)?

Landscape ecological research areas

Some of the questions raised above are now considered in the language of landscape ecology, looking at the lines of research that are likely to provide the answers.

'Single large or several small'?

The single large or several small (SLOSS) problem has long been addressed in a large number of empirical and theoretical studies as to whether a single large or several small patches offer the best 'value' (Simberloff, 1974; Diamond, 1975; 1980; Margules *et al.*, 1982). That is, the most cost-effective way of ensuring the survival of ancient woodland remnants, and increasing their resilience to future threats. Yet, still no coherent solution has yet been reached. However, recent research suggests that the optimal number of patches is largely affected by the relationship between habitat patch area and rates of immigration, emigration and local extinction (Ovaskainen, 2002). These rates clearly depend on the species in question, suggesting that we cannot expect one single solution to maximise biodiversity gain for ancient woodlands. Different taxa have different requirements. Interior species (which are more sensitive to fragmentation) need big blocks, whereas edge species are often assumed to be ubiquitous and well catered for by our fragmented resource. But some still require ancient woodland edge, and the quality or structure of that edge, and the intensity of the surrounding land use, may be critical to survival of such species. Which solution maximises biodiversity gain in woodland? We can begin to explore solutions through model development to find the most parsimonious solution for species with a range of dispersal, colonisation and area requirements. This will inevitably require a better understanding of the needs of different species, particularly those which are most demanding or challenged.

Defragmentation of isolated communities

Recent studies for plants, especially ancient woodland species, suggest that even when new woodlands are located adjacent to source woodlands time required for colonisation is lengthy (Honnay *et al.*, 2002). Evidence for colonisation of more mobile species is also sparse. However, real concern now exists about the movement of invasive species and their role as vectors for disease transmission (the competitive interaction of grey squirrels, *Sciurus carolinensis*, with red squirrels, *Sciurus vulgaris*, and the possible link to spread of the parapox virus, being well-known examples).

Landscape context and permeability

Ecologically functioning landscapes, functional connectivity between woodlands, will vary according to taxa/species dispersal abilities. This in turn will depend upon the permeability of the matrix and in any event is only important if the species are able to colonise (or utilise) woodlands once reached. Again, exploration of movement through modelling may increase our understanding of matrix permeability according to landscape type, species dispersal abilities etc. At the regional/national scale information is required to resolve how we should allocate resources between woodland improvement, restoration, creation and wider changes in land use. Also, which options deliver the most benefit for ancient woodland communities, for least cost?

Landscape models

The potential for modelling to inform our research questions has been made reference to several times above. Many examples exist that have gone a long way to increase our understanding of the processes involved. Attempts have been made to target limited resources to reduce the effects of fragmentation for BAP species (Bailey *et al.*, 2002), though based on large-scale models. However, there is still a dearth of empirical data with which to populate models (Bailey, 1998). Without this validation it is becoming increasingly difficult, and there is a need for evidence of species movement through restored landscapes (Bailey *et al.*, in prep). The lack empirical research on the effect of fragmentation on the abundance and diversity of woodland species in real landscapes needs to be addressed to ensure that appropriate policy can be developed to guide the management of our woodland resource for optimal biodiversity protection.

Microsatellite work

The discovery of new molecular markers such as microsatellites may offer a new approach to the study of species dispersal in fragmented landscapes (Arnaud, 2003). Different ecological and evolutionary processes can leave distinct genetic signatures in current spatial structuring of neutral genetic variation (McCauley, 1995; Bohonak, 1999). Differentiation of these signatures is possible even at very fine spatial and temporal scales, facilitating a greater understanding of the interactions between interconnected populations, as well as the reasons for the cessation of these interactions following fragmentation (Arnaud *et al.*, 2001). It should therefore be possible to track the colonisation of woodlands by individuals from established populations and relate movements to permeability of the matrices between woodlands (eg., presence of corridors, proportion of semi-natural habitat). This would provide evidence of species movement (or not!) through landscapes containing recent woodland. In addition, it would provide much needed information of species dispersal abilities through real landscapes.

As with all experimental work, results can only ever reflect the conditions in which they have been gathered. Species dispersal and colonisation abilities will be defined according to: the character of the landscape in which analysis has occurred; history of woodland management etc. Therefore, the transfer of information to different landscapes must be done with caution. We must also be aware of the inherent limitations in the techniques used, and in particular the failure of many techniques to provide good information on chance long-distance dispersal. The Forestry Commission is assessing the evidence for the application of hypervariable molecular markers such as microsatellites in studies of past species dispersal and colonisation rates (speed, distance and frequency) through non-wooded, wooded and newly wooded landscapes.

Conclusions

There is clearly a rapidly increasing focus on landscape-scale approaches to woodland conservation. This is being expressed at various levels in the UK:

- International policy commitments
- National and regional woodland and biodiversity strategies

- Policy implementation measures, incentive schemes and institutional structures.
- Multi-ownership partnership projects to achieve landscape-scale collaboration on the ground.

This paper has given a UK and England perspective, but it is suggested that the questions raised will be equally relevant to any country with a highly-fragmented woodland resource lying in an intensively farmed landscape.

Growing interest has been catalysed and fostered by pioneering research by landscape ecologists, but the resulting policy discussions are now producing far more questions for researchers to address. The need for further research in this area is being flagged up in the current review of the Forestry Commission's GB Research Strategy.

Everyone agrees that policy needs to be 'evidence-based' and delivery needs to based on well-founded 'best practice'. But we have to be realistic and accept that there is such a high level of interest in landscape-scale approaches that neither policy nor practice can wait for all the questions raised in this paper to be answered. This means that in the short-term we will have to adopt two tactics. Firstly, we will need to rely on 'first principles' to guide and underpin policy development. Secondly, we will need to take an empirical approach to implementation, and be ready and willing to change our approach if we are not achieving the outcomes we desire.

References

Arnaud, J-F., Madec, L., and Daguzan, J. (1999) Spatial differentiation of allozyme frequencies in a sub-divided population the land snail *Helix aspersa. Journal of Molluscan Studies* **65:** 267 – 271.

Arnaud, J-F. (2003) Metapopulation genetic structure and migration pathways in the land snail *Helix aspersa*: Influence of landscape heterogeneity. *Landscape Ecology* **18:** 333-346.

Bailey, S-A., (1998) *Habitat fragmentation in England's ancient woods.* Unpublished Phd Thesis, University of Nottingham.

Bailey, S-S., R. H. Haines-Young and C. Watkins (2002) Species presence in fragmented landscapes: modelling of species requirements at the national level. *Biological Conservation*, **108:** 307-316.

Bailey, S.-A. and Watkins, C. (in prep) Research, policy and habitat fragmentation, an exploration through British woodland.

Bohonak, A.J. (1999) Dispersal, gene flow and population structure. *Quarterly Review of Biology* **74:** 21 – 45.

CBD COP7 9-20 February 2004: Detailed report.

Defra (2002) *Working with the grain of nature: A biodiversity strategy for England.* Department for food and rural affairs, London. (www.defra.gov.uk)

Defra (2004) Consultation on cross-compliance measures under CAP reform.

Defra (2004) Agri-environment schemes review. http://www.defra.gov.uk/erdp/reviews/agrienv

Diamond, J.M. (1975) The island dilemma: lessons of modern biogeographic studies for the design of nature reserves. *Biological Conservation* **7:** 129 – 146.

Diamond, J.M. (1980) Patchy distributions of tropical birds. M.E. Soule and B.A. Wilcox (Eds.) *Conservation biology: an evolutionary ecological perspective.* Sinauer Association Incorporated, Massachusetts.

Forestry Commission England (2004). *English Woodland Grant Scheme.* http://www.forestry.gov.uk/forestry/HCOU-4UCEVV

Harrison, P.A., Berry, P.M. & Dawson, T.P. (Eds.) (2001) *Climate change and nature conservation in Britain and Ireland: modelling natural resource responses to climate change (the MONARCH project).* UK CIP Technical Report.

HMSO (2002) The Countryside and Rights of Way Act. UK Acts of Parliament. *http://www.uk-legislation.hmso.gov.uk/acts.htm*

Honnay O., Hermy, M., Coppic, P. (1999) Effects of area, age and diversity of forest patches in Belgium on plant species richness, and implications for conservation and restoration. *Biological Conservation*, **87**: 73-84.

Honnay O., Bossuyt B., Verheyen K., Butaye J., Jacquemyn H. & Hermy M.(2002) Ecological perspectives for the restoration of plant communities in European temperate forests. *Biodiversity and Conservation* **11**: 213-242.

IUCN 2003 http://www.unep-wcmc.org/forest/restoration/homepage.htm

Margules, C., Higgs, A.J. & Rafe, R.W. (1982) Modern biogeographic theory: are there any lessons for nature reserve design? *Biological Conservation* 24: 115 – 128.

McCaulay, D.E. (1995) Effects of population dynamics on genetics in mosaic landscapes. L. Hansson *et al.* (Eds.) *Mosaic landscapes and ecological processes*, Chapman and Hall, London, UK. 178 – 198.

MCPFE (2003) Vienna Declaration and Vienna Resolutions: adopted at the Forth Ministerial Conference on the protection of forests in Europe. 28 – 30th April, 2003, Vienna, Austria.

Ovaskainen, O. (2002) Long-Term Persistence of Species and the SLOSS Problem. *Journal of Theoretical Biology* 28: 419 – 433.

PEBLDS, (2002) Pan European Biological and Landscape Strategy: rolling work programme. Council of Europe, and hosted by European Center for Nature Conservation. *http://www.strategyguide.org/rolpr.html*

Simberloff, D. (1974) Equilibrium theory of island biogeography and ecology. *Annual review of Ecology and Systematics* 5, 161 – 182.

UK Biodiversity Partnership (1994 – 1998) *Habitat Action Plans*. www.ukbap.org.uk

Session 1

Responses of plants to landscape-scale processes

Meso-scale floristic richness of ancient woodland indicator species: woodland type and landscape effects

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Abstract

Species richness of 'ancient woodland indicator' (AWI) plants was analysed at the meso-scale, using 10km square units in England. Broadleaved woodland was the most favourable type promoting richness, followed by mixed woodland, coppice, scrub and coniferous cover. Isolation between ancient woodlands had a negative effect that was not compensated for by the presence of secondary woodlands. The number of ancient woodland patches per unit area, independent of overall woodland area, was an important positive factor increasing species richness, and had a markedly greater influence than that of mean patch size. Replanted ancient woodlands were not as favourable habitats as semi-natural ancient woodlands in terms of their floristic richness. The results confirm that optimum plant species richness is best maintained by preventing further erosion of ancient woodlands and restoring replanted examples to favourable condition, rather than pursuing extensive planting programmes of new secondary woodland.

Introduction

Ancient woodlands (woodlands originating before 1600AD) typically contain a number of plant species whose ability to colonise new woodland sites appears to be limited. However, the faithfulness of these so-called 'ancient woodland indicator' (AWI) species to long-established woodland sites varies considerably throughout Britain. Recognising this, in the 1970s and 80s regional lists of indicator plants were drawn up by the Nature Conservancy Council, wildlife trusts and other conservation bodies in order to identify and evaluate ancient woodlands for conservation purposes. The poor dispersal and colonising ability of AWI species makes them potentially useful for identifying management practices that sustain forest biodiversity, a central tenet of ecologically sustainable forest management.

Variance partitioning by Canonical Correspondence Analyses of the floristic composition of 10km squares has shown that regional AWI preferences are predominantly environmentally-determined, for example by the mean 10km values of variables such as accumulated temperature, moisture deficit, Ellenberg soil reaction, nitrogen and continentality (Mouflis, 2004). However, woodland landscape variables were more important in explaining the 10km diversity of AWI species. The aim of our investigation was to identify which of these woodland landscape factors were responsible for the variation in the observed AWI species richness across England.

Data and methods

Geographical distributions of about 150 'indicator' species, based on their recorded presence or absence in 10km squares forming the British National Grid, were obtained from the Biological Records Centre, Abbots Ripton (Arnold 2000) and imported into a Geographical Information System. Species richness was represented by the number of AWI species per 10km square (AWS). For England, the Ancient Woodland Inventory (English Nature 1998-2003) was used to derive patch and inter-patch metrics of semi-natural and replanted ancient woodlands using the Patch Analyst extension in ArcView (Elkie *et al.*, 1999). Contiguous polygons of each type were also merged to derive patch metrics of (combined) ancient woodland. These included the area of ancient woodland, the number of patches, the mean patch size, median patch size and mean patch edge. Other metrics calculated per 10km square were; the total edge and edge density of ancient woodland, the mean distance from ancient to the nearest secondary woodland, the mean nearest neighbour or second nearest neighbour distance between ancient woodlands, the total area of all woodland cover, and the total area of secondary woodland.

The National Inventory of Woodland and Trees (NIWT) (Forest Research - Woodland Surveys, 2001) was also used to derive the total area of 8 woodland types in each 10km square. The NIWT categorises woodland into **Broadleaved, Coniferous, Mixed, Young trees, Felled, Scrub** or shrub, **Coppice** (and coppice with standards) and 'Ground prepared for planting' (coded **'Converted'** here). The total area of each woodland type in a 10km square was used as the bases of these analyses. The two G.I.S. layers of the Ancient Woodland Inventory and the NIWT were intersected to derive the areas of woodland types that were ancient (overlapping polygons) or secondary in origin.

Pearson and Spearman correlations were calculated to examine the pairwise relationships of the number of AWIs with successive woodland landscape variables, followed by a joint analysis of a selection of these variables using Principal Component Analysis. A Kruskal-Wallis ANOVA was then used to compare the mean 10km values of woodland landscape variables with **high** AWI species richness (AWS>82 species), **intermediate** species richness (interquartile of AWS, 54-82 species), and **low** species richness (AWS<54). Finally, partial correlation of AWS was used to identify the conditional effect of woodland landscape variables while controlling for the area of ancient woodland habitat.

Results

Inspection of pairwise scatterplots of all woodland landscape variables against AWS revealed a monotonic relationship with the exception of shape complexity / sinuosity metrics (mean shape index, perimeter-area ratio and patch fractal dimension) of ancient woodlands. These had a unimodal relationship with AWS and therefore their relationship cannot be properly quantified by linear correlations or PCA. Environmental variables also had a unimodal relationship with AWI species richness.

Pearson pairwise correlations showed that the class area as well as the other woodland landscape variables of combined and semi-natural ancient woodland were more highly positively correlated with AWS than those of replanted ancient woodland, suggesting that replanting may lead to a decrease of AWI species richness (Table 1). The correlation of AWS with replanted ancient woodland area was similar to that with total woodland cover (NIWT), but the total area of secondary woodland cover was only very weakly positively correlated with AWS.

Table 1. List of pairwise Pearson correlations of woodland patch metrics with AWI species richness (AWS). Very highly significant correlations are highlighted. Mean NN= mean nearest neighbour distance between ancient woodlands; Mean 2NN= mean nearest neighbour distance from ancient to the second nearest ancient woodland neighbour; MNNSec= mean distance from ancient to the nearest secondary woodland.

	Ancient woodland				**Woodland area and type (NIWT)**		
	Total	**Semi-natural**	**Re-planted**		**Total**	**Secondary**	**Ancient**
Class Area	**0.50 p=0.00**	**0.49 p=0.00**	**0.35 p=0.00**	**Broad-leaved**	**0.49 p=0.00**	**0.22 p=0.00**	**0.48 p=0.00**
Number of patches	**0.56 p=0.00**	**0.55 p=0.00**	**0.45 p=0.00**	**Conifer**	**0.13 p=0.00**	0.04 p=0.20	**0.15 p=0.00**
Mean patch size	**0.10 p=0.00**	**0.21 p=0.00**	0.05 p=0.15	**Mixed**	**0.34 p=0.00**	**0.19 p=0.00**	**0.37 p=0.00**
Mean patch edge	**0.10 p=0.00**	**0.16 p=0.00**	0.04 p=0.27	**Young trees**	**0.09 p=0.00**	-0.02 p=0.48	**0.25 p=0.00**
Total edge	**0.55 p=0.00**	**0.53 p=0.00**	**0.42 p=0.00**	**Felled**	**0.25 p=0.00**	**0.11 p=0.00**	**0.34 p=0.00**
Mean shape index	0.04 p=0.25	0.02 p=0.56	0.03 p=0.41	**Scrub**	**0.22 p=0.00**	**0.17 p=0.00**	**0.24 p=0.00**
Mean NN	**- 0.34 p=0.00**			**Coppice**	**0.24 p=0.00**	**0.16 p=0.00**	**0.23 p=0.00**
Mean 2NN	**- 0.41 p=0.00**			**Converted**	-0.06 p=0.06	-0.08 p=0.01	0.03 p=0.32
Mean NNSec	-0.03 p=0.30			**All cover types**	**0.37 p=0.00**	**0.13 p=0.00**	

The number of woodland patches had a stronger positive correlation with AWS than any other variable, indicating that many separate woodlands were more likely to host a higher number of species. Mean patch size was weakly positively correlated with AWS, but this was greater for semi-natural, ancient woodland than for either the total or replanted categories, the last being not significant. Median patch size was not significantly related to AWS and is probably an inferior descriptor of the patch size effect. Mean patch edge was weakly positively correlated with AWS except in the case of replanted ancient woodland. In contrast, the total woodland edge was strongly positively correlated with AWS, reflecting the positive influence of a spread of habitat availability rather than an increase in edge conditions which is better described by variables such as the perimeter ÷ area ratio, measuring the increase of edge conditions per unit area of woodland patch.

The areas of all woodland types (per 10km square), except the converted ('ground prepared for planting') category were positively correlated with AWS, particularly broadleaved woodland, followed by mixed woodland, felled, coppice, scrub, coniferous and young trees. When the areas were broken down into ancient and secondary components, however, the latter had relatively lower correlations, with secondary coniferous woodland and secondary new planting (young trees) being unrelated to AWI species richness. The implication is that if AWIs are organised into metapopulations with ancient woodlands representing the 'source' (preferred) patches, broadleaved secondary woodlands function only weakly as 'sink' habitats.

On the other hand, when only the ancient portion of each woodland type was considered, most showed a positive relationship with AWS. In particular, better correlations of species richness were found in 'young trees' and 'felled' woodland categories compared with their 'secondary' counterparts, emphasising the importance of site history over canopy type. Ancient 'coniferous' woodland (effectively conifer-replanted woodland in England) was less hospitable for AWIs than mixed or broadleaved canopy types, while young trees (resulting from replanting or natural regeneration) were only positively correlated with AWS if present on ancient woodland sites. Young growth stages, represented by felled, scrub and coppice categories all had weaker positive correlations with AWI richness than mixed and broadleaved woodland.

The mean nearest neighbour distance between ancient woods had a highly significant negative correlation with AWI species richness and this relationship was even more pronounced for the second nearest neighbour distance. The latter finding provides some support for the hypothesis that AWI plants are organised into metapopulations that allow at least limited dispersal and (re)colonisation between patches. Interestingly, the mean nearest neighbour distance from an ancient to a secondary wood had no significant relationship with species richness of AWIs. This indicates that while proximity of ancient woodlands allows an established diverse ancient woodland flora to be maintained and inter-migrate between AW patches, the same facility is not provided by more recent woods. The implication is that new planting *per se* may not increase AWI species richness at least in the short term, although strategic planting (e.g., by enlarging or connecting together ancient woods) may facilitate this. However, common woodland plants with more efficient dispersal abilities are likely to take advantage of improved connectivity, probably resulting in increased overall species richness at the scale of the individual woodland patch, but probably not at the mesoscale of the 10km square.

PCA analysis (Figure 1) confirmed that 10km AWI species richness was promoted most by the number of AW patches, the AW area and the area of broadleaved woodland, while separation between AW patches had a negative effect. Mixed woodland, coppice and felled woodland and the total area of woodland were also positively correlated with species richness to a lesser extent, and there was a weak positive correlation between AWS and mean patch size of AW and scrub area. On the other hand the area of secondary woodland, young trees, ground prepared for planting and coniferous woodland, as well as the isolation of AW from secondary woodland were little related to the AWI species richness, which again suggests that they are inferior or unutilised habitats for AWI species.

When groups of 10km samples containing the lowest, intermediate and highest AWI species richness were compared, large differences were found between the mean values of different woodland landscape variables using a Kruskal Wallis ANOVA (Table 2). While all variables showed differences between these groups, the area of ancient and broadleaved woodland, the number of AW patches, the area of semi-natural AW and isolation between AW patches were particularly effective in discriminating between species-rich and species-poor samples. Once again, the mean distance between ancient woodlands and secondary woodlands was not significant between sample groups.

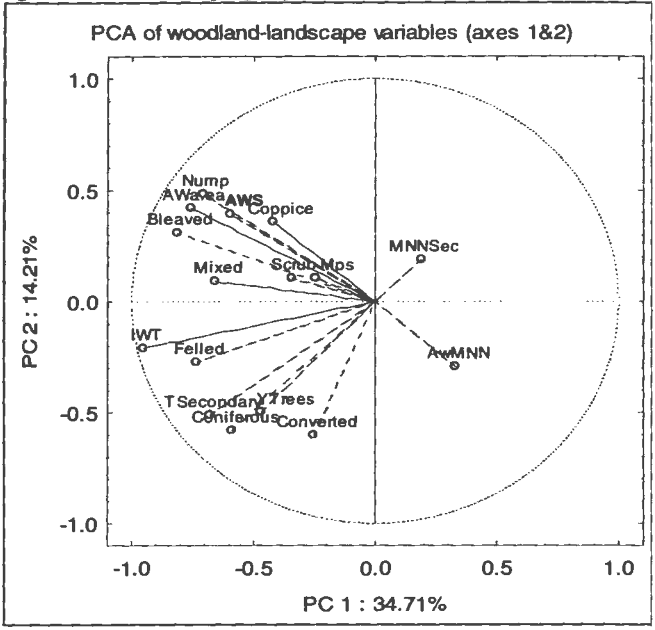

Figure 1. PCA of woodland landscape variables (axes 1&2), 10km squares weighted by their area. Nump: Number of AW patches, AWS: Number of AWI species/10km square, Mps: Mean patch size of AW, MNNSec: Mean distance from ancient to nearest secondary woodland, AwMNN: Mean nearest neighbour distance between ancient woodlands, IWT: Total area of woodland, Tsecondary: Total area of secondary woodland.

As many of the landscape metrics examined above were directly influenced by the amount of ancient woodland available within a 10km square, it was necessary to investigate whether the examined correlations represented a causal effect or were due to variable inter-correlations. For this purpose a partial correlation analysis was performed to identify the effect of woodland landscape variables with AWS (number of AWI species per 10km square) after controlling for the area of ancient woodland and the area of the 10km square (Table 3). This again indicated that the number of ancient or semi-natural woodland patches were key factors contributing to species richness, reflecting variation in site and soil conditions within the 10km square. Species richness was in turn related to the increased woodland perimeter and, in the case of ancient woodland only, negatively to the mean patch size. Relative isolation of ancient woods remained an important negative factor, independent of the amount of ancient woodland cover. The importance canopy type was again signalled by positive correlations between species richness and broadleaved, mixed and scrub canopy types in either ancient or secondary categories. In contrast, coppice, felled and converted categories showed no correlation with species richness.

Table 2. Mean values (± standard errors) per 10km square of woodland landscape variables, grouped according to AWI species richness. The Kruskal-Wallis ANOVA (by ranks) indicates significant ($p<0.01$) differences for all but the last variable (mean nearest neighbour distance between ancient and secondary woods, $p=0.76$). Variables are ranked by their ability to discriminate among the sample groups of species richness as measured by the test statistic H. SW= secondary wood.

Woodland landscape variable	N	**Low** species richness **<54 AWS**	**Medium** species richness **54-82 AWS**	**High** species richness **AWS>=82**	**All samples in England**	**H** (Kruskal -Wallis by **Ranks)**
Ancient woodland area (ha)	1441	**47.8** ±4.7	**227.6** ±10.1	**649.7** ±33.7	**289.2** ±11.6	**620.2**
Broadleaved area (ha)	1417	**132.6** ±7.5	**301.9** ±9.6	**613.0** ±21.8	**340.5** ±8.9	**474.8**
Number of AW patches	1235	**6.72** ±0.45	**14.68** ±0.50	**35.96** ±1.38	**19.69** ±0.59	**395.7**
Total woodland area (ha)	1417	**430.6** ±67.2	**659.7** ±24.2	**1202.3** ±47.4	**743.6** ±25.1	**392.2**
Semi-natural AW area (ha)	1212	**44.9** ±4.2	**119.8** ±5.8	**372.1** ±21.2	**184.2** ±8.0	**371.3**
MNN distance to 2nd AW (m)	1235	**2382** ±153	**1456** ±57	**703** ±29	**1384** ±44	**259.9**
Mixed woodland area (ha)	1417	**46.9** ± 4.4	**78.9** ±4.1	**188.2** ±12.1	**99.3** ±4.2	**221.7**
Coniferous woodland (ha)	1417	**191.6** ±60.8	**197.4** ±15.0	**260.9** ±22.1	**212.4** ±17.8	**204.2**
Replanted AW area (ha)	1066	**42.2** ±5.1	**91.1** ±5.2	**178.5** ±10.1	**113.7** ±4.6	**173.8**
MNN distance AW – AW (m)	1235	**1374** ±121	**759** ±40	**350** ±21	**739** ±31.4	**172.0**
Felled area (ha)	1417	**13.8** ±2.7	**23.5** ±1.9	**45.5** ±4.1	**26.8** ±1.6	**167.1**
Coppice area (ha)	1417	**0.63** ±0.48	**3.81** ±0.89	**27.41** ±4.28	**9.15** ±1.23	**156.7**
Secondary wood area (ha)	1417	**370.3** ±64.7	**422.7** ±17.8	**522.7** ±28.1	**435.6** ±19.8	**129.2**
Scrub area (ha)	1417	**5.99** ±0.99	**9.09** ±0.75	**17.33** ±1.37	**10.46** ±0.58	**116.9**
Area of 10km square (ha)	1441	**8065** ±169	**9562** ±52	**9773** ±49	**9228** ±55	**107.6**
Young trees (ha)	1417	**34.9** ±4.8	**42.3** ±2.8	**48.2** ±2.5	**42.0** ±1.9	**98.8**
Mean patch size, AW (ha)	1235	**11.16**± 0.90	**14.06** ±0.56	**16.4** ±0.69	14.29 ±0.40	**68.8**
Converted area (ha)	1417	**4.14** ±2.13	**2.83** ±1.38	**1.70** ±0.38	**2.86** ±0.87	**31.6**
MNN distance, AW – SW (m)	1238	**579.8** ±44.6	**498.7** ±11.5	**496.5** ±12.2	**511.4** ±10.3	0.6

Johnson and Simberloff (1974) suggested that increased species richness is due mainly to habitat heterogeneity rather than habitat area. In this context, the number of patches of ancient woodland also appears to reflect environmental heterogeneity (since a higher number of woodland patches is likely to include a wide range of soil types, topographic and environmental conditions), giving a high correlation ($r=0.32$, $p<0.05$) with AWS. An additional analysis revealed that the partial correlation of AWS with AW area (after controlling for the area of the 10km square and the number of patches of ancient woodland) was smaller (0.19; $p<0.05$), indicating that the positive partial effect of patch number (= environmental heterogeneity) is stronger than that of habitat area for ancient woodland indicator species in England.

Table 3. Partial correlations of AWS with selected woodland landscape variables, controlling for the area of ancient woodland and the size of 10km square, extracted from 1012 squares of England.

Partial correlations significant at α=0.05		**Partial correlations NOT significant at α=0.05**	
Number of AW patches	**0.32**	MNN distance, AW - secondary wood	-0.06
Number of semi-natural patches	**0.30**	Mean patch size, replanted AW	-0.06
Total edge of AW	**0.26**	Mean patch edge, replanted AW	-0.06
MNN distance to 2nd AW neighbour	**-0.25**	Converted secondary woodland	-0.06
Edge density, semi-natural woodland	**0.23**	Coniferous secondary woodland area	0.05
Total edge of semi-natural woodland	**0.22**	Converted woodland area	-0.05
Broadleaved woodland area	**0.22**	Felled secondary woodland area	0.05
Number of patches of replanted AW	**0.20**	Ancient scrub area	0.04
MNN distance between AWs	**-0.20**	Felled woodland area	0.04
Total woodland area (IWT)	**0.16**	Mean patch size, semi-natural woods	-0.04
Scrub area	**0.16**	Ancient coppice area	-0.03
Secondary scrub area	**0.16**	Coniferous area	0.03
Secondary broadleaved woodland area	**0.16**	Coppice area	-0.03
Mixed woodland area	**0.15**	Ancient converted woodland area	0.02
Ancient broadleaved woodland area	**0.15**	Secondary coppice area	0.01
Total secondary woodland area	**0.13**	Replanted ancient woodland area	-0.01
Secondary mixed woodland area	**0.13**	Mean patch edge, semi-natural woods	-0.01
Semi-natural AW area	**0.11**	AW with young trees	0.01
Mean patch size, AW	**-0.11**	AW coniferous area	-0.01
Ancient mixed woodland area	**0.10**	Young trees area	0.01
Edge density, replanted AW	**0.10**		
Mean patch edge, AW	**-0.06**		

Discussion

These results show that to maximise AWI species richness would rely heavily on maintaining large areas of existing ancient woodland, by definition a resource that cannot be artificially recreated. In a conservation planning context, since the destruction or degradation of even small woodland patches can affect the floristic richness at the 10km scale, preventing further losses of such woodland is therefore a clear priority. The evidence presented here also suggests that that for plants, at least, replanting or restoring woodland in 'several small' parcels as opposed to 'fewer large' blocks is the better strategy. To be most effective, therefore, these 'several small' woodlands should be allocated to locations with varying soil pH, texture, topographic and drainage conditions so that complementary floras can develop. Such an approach is compatible with planting to consolidate 'forest districts' or in conjunction with existing ancient woods.

As increasing isolation of ancient woodlands results in reduced species richness, inter patch isolation may not simply be a corollary of AW availability but an indication that som AWI metapopulations can function effectively when the woods are in close proximity Conversely, the strong positive influence of total edge and edge density variables on AWS was most probably related to their correlation with the number of woodland patches (0.88 an 0.73 respectively), and does not necessarily indicate a positive edge effect. However the tota secondary woodland area did have a significant positive partial effect that was due mainly t the broadleaved, scrub and mixed component.

Canopy type is another important consideration, shown by the positive correlation of AWS with the class area of semi-natural ancient woodland, whereas that of replanted ancient woodland showed no correlation. This indicates that semi-natural ancient woodlands have qualities additional to their area that have declined in replanted ancient woodlands. Other woodland types such as broadleaved, mixed and scrub woodland also showed positive correlations with AWS. Whether or not the lower species numbers found in replanted woodlands is indeed real, or because they have been surveyed with the same thoroughness as semi-natural sites, their inherent species richness compared with secondary woods means that their restoration to broadleaved cover remains a high priority (Thompson *et al.*, 2003). The positive partial correlation of mixed woodlands with AWS further suggests that some conifers, intermixed with broadleaved trees, could be maintained in such restoration projects. The strong association of coniferous woodland cover in England with plantations, base-poor, acid soils, or less favourable replanted ancient woodland probably explains the poor correlation of this canopy type with AWS.

Finally, it must be remembered that this study relied on coarse distributional recording at the mesoscale, a level intermediate between α-diversity and β-diversity which tends to 'average out' internal spatial patterns of different woods or other suitable habitats present within the same 10km square. However, for the large England geographical dataset' species-environment and species-woodland are relationships are relatively robust, such that between-sample variation exceeded between-sample variation. These relationships now need to be examined further at a finer scale using more detailed data sets, for example using tetrad (2x2km) or Phase 2 recording available for individual regions.

References

Arnold, H.R. (2000) (personal communication). Biological Records Centre, Centre for Hydrology and Ecology, Monk's Wood, Abbots Ripton.

Elkie, P., R.; Rempel & A. Carr (1999) Patch Analyst User's manual: A tool for quantifying landscape structure. Ontario Ministry of Natural Resources & Northwest Science and Technology, Ontario.

English_Nature (1998-2003) Ancient Woodland Inventory (Provisional) for England - Digital Boundaries., Vol. 2001. English Nature, Peterborough, England.

Forest Research - Woodland Surveys, F.C. (2001) National Inventory of Woodland and Trees. Forestry Commission, Crown Copyright.

Johnson, M.P. & D.S. Simberloff (1974) Environmental determinants of island species numbers in the British Isles. *Journal of Biogeography* **1:**, 149-154.

Thompson R.N., Humphrey, J.W., Harmer, R. & Ferris R (2003) *Restoration of native woodland on ancient woodland sites.* Forestry Commission, Edinburgh.

Metapopulation dynamics in changing landscapes: a new spatially-realistic model applied to forest plants in central Lincolnshire, United Kingdom

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Abstract

In fragmented landscapes, habitat patches are often destroyed and created through time, though most metapopulation models treat patch networks as static. Here we present a generally-applicable, modified version of Hanski's Incidence Function Model (IFM) that incorporates landscape dynamics (i.e., habitat patch turnover). We parameterised the model with data on patch occupancy patterns for forest plants in central Lincolnshire, UK. The modified IFM provided a better, or equally good, fit to species' patch-occupancy patterns than logistic regression. Estimated colonisation and extinction rates, and the results of logistic regression analyses, varied significantly among species with different life-history traits. For example, species with low seed production and predominantly short distance seed dispersal showed lower rates of colonisation and extinction, and were more likely to show effects of patch age and connectivity on patch level presence than species with the opposite set of traits. Model simulations demonstrated a profound negative influence of habitat turnover rate on metapopulation dynamics and persistence, particularly for slow-colonising species.

Introduction

The metapopulation concept provides the dominant theoretical paradigm for understanding species dynamics in highly fragmented landscapes (Hanski, 1999). Levin's (1969) classic model of colonisation and extinction in an infinite number of equally-connected habitat patches has since been elaborated on to include spatially-explicit colonisation (e.g., Durret and Levin, 1994; Bascompte and Solé, 1996), variation among patches in area and isolation (e.g., Hanski, 1994), rescue from extinction via continued immigration (e.g., Brown and Kodric-Brown, 1977), and within-patch population dynamics. However, all of the above-described models take no account of destruction and creation of habitat patches (i.e., landscape dynamics), processes known to characterise many human-dominated landscapes. Incorporating landscape dynamics into the metapopulation paradigm has become an important goal for metapopulation research, in addition to being of potentially great importance for conservation efforts in many parts of the world. Some recent models have revealed important potential effects of landscape dynamics on metapopulation persistence (e.g., Keymer *et al.*, 2000; Amarasekare and Possingham, 2001; Johst *et al.*, 2002;

Ellner and Fussmann, 2003; Hastings, 2003), though parameterisation of these models presents a significant challenge. The "spatially realistic" Incidence Function Model (IFM, Hanski, 1994) represented a major advance in connecting metapopulation models with data in that its parameters can be estimated with patch-occupancy data of the type often collected by empirical ecologists, but the IFM has yet to be modified to incorporate landscape dynamics.

In this study, we introduce a generally applicable modified version of the IFM that accounts for landscape dynamics and fit it to Peterken and Game's (1984) data on patterns of forest plant patch-occupancy in central Lincolnshire (UK), where the history of forest fragmentation and dynamics is well-documented (Peterken, 1976; Peterken and Game, 1984). The Lincolnshire data set (155 species and 326 patches) holds great potential for studying the effects of patch turnover rate on metapopulation dynamics for a number of reasons: (1) forest cover has been approximately constant at 5% for at least 1000 years; (2) continuous forest reclamation and reforestation has taken place (Peterken, 1976), and consequently, present-day forests have variable and known ages; (3) it has been shown that the presence of many forest plants in central Lincolnshire (Peterken and Game, 1984) and elsewhere is related to forest age (for a review see Verheyen et al., 2003a). Both observational (e.g., Matlack, 1994; Bossuyt *et al.*, 1999; Verheyen *et al.*, 2003b) and experimental (e.g., Ehrlén and Eriksson, 2000; Verheyen and Hermy, 2004) research, has demonstrated that the limited colonising capacity of many forest species in Europe is caused mainly by dispersal limitation and not by covariation between habitat suitability and forest age. Hence, we can safely assume that colonisation and extinction probabilities do not change with patch age, and that the overall spatio-temporal patch configuration has been more or less stable over time, satisfying the assumption of the IFM that patch-specific colonisation and extinction rates are approximately constant.

Methods

Study area and data collection

The study area covered 930 km² in central Lincolnshire (UK) of which only 4950 ha, or 5.3% of the total land surface, was forested in 1980. Lincolnshire's forests were already highly fragmented during the Roman era and by 1086 AD (the Domesday Book) forests covered only 4% of the land (Peterken, 1976; Rackham, 2003). Since then, the overall forest cover has remained more or less constant, indicating approximately equal rates of forest creation and destruction. According to Peterken (1976), average forest creation and destruction rates during the past two centuries were respectively 7.8 ha/yr and 10.0 ha/yr (i.e., a yearly turnover of ~0.2%). In 1980, 326 forest patches were present, of which 5% originated after 1946, and 8%, 31% and 34% between 1887-1946, 1820-1887 and 1600-1820, respectively (dates based on regional forest maps). The remaining patches (22%) were considered 'ancient forests', originating before 1600. Between 1972 and 1981, a presence-absence list of 155 vascular plant species was compiled for all 326 patches. For statistical reasons only the 63 species with a frequency between 20% and 80% were used for further analyses. We refer readers to Peterken (1976) and Peterken and Game (1984) for more detailed descriptions of the study area and the data collection methods.

Logistic regression analyses

For each species, presence or absence across the 326 patches was first modeled using multiple logistic regression with four explanatory variables: patch area; connectivity; patch age; and habitat-unsuitability based on Ellenberg's indicator values revised for the UK (Hill *et al.*, *1999*). Patch connectivity was calculated according to Moilanen and Nieminen (2002).

The modified Incidence Function Model (IFM)

Hanski's (1994) IFM can be rewritten to incorporate landscape dynamics as follows:

$$J_i(age) = \frac{C_i - C_i(1 - C_i - E_i)^{age}}{C_i + E_i}$$

where J_i is the probability that a patch is occupied at a certain age. We followed Hanski (1994, 1999) for the relationships of patch area and connectivity to the extinction and colonisation rates (E_i and C_i,, respectively) and incorporated habitat unsuitability as a potential reduction in the "effective" area of a habitat patch with respect to extinction probability and the strength of a patch as a potential source of colonists (formulas not shown). To fit the modified IFM, five parameters needed to be estimated for each species using maximum likelihood techniques.

For *Mercurialis perennis* and *Taraxacum officinale*, species representing slow and fast colonisers respectively, we then used the model to assess the influence of habitat turnover rate on patch occupancy and extinction probability at the landscape scale. Both species had similar observed patch occupancy in the real data (0.32 for *M. perennis*; 0.31 for *T. officinale*).

Results

For all but two species, reasonable fits to the logistic regression models were obtained. These two species (*Arrhenatherum elatius* and *Stellaria media)* were omitted from further analysis. Significant effects of the habitat-unsuitability index were obtained for 50 of the remaining 61 species (82%). Significant area, connectivity and age-effects were found for 35 (57%), 17 (28%) and 18 (30%) species, respectively. For all species, the signs of the relationships were either consistently negative (habitat-unsuitability index) or positive (area, connectivity and age). The only exception was a negative age-effect for *Primula veris*.

The procedure for fitting parameters to the IFM was successful. Furthermore, similar relationships between the plant traits and results of the two kinds of model were found (Figure 1). Species for which a significant age effect was found in the logistic regressions were characterised by heavy seeds, low seed production, no adaptations for long-distance dispersal, a transient seed bank and a delayed age of first reproduction. These species also tended to flower earlier and were more likely to be evergreen and biennial. Similar, but weaker, relationships were found for connectivity. Although even weaker relationships were found with area effects, their sign was generally opposite to the relationships with age and connectivity effects. From the IFM, the colonisation-extinction axis was strongly positively correlated with seed number per plant, and negatively correlated with seed mass, age of first reproduction, and strictness of germination requirements (Figure 1). The colonisation-extinction variable was greatest for species with wind-dispersed seeds, intermediate for species with vertebrate-dispersed seeds, and lowest for species with seeds dispersed by ants, by gravity, or ballistic projection (Figure 1).

Model simulations indicate that habitat destruction has a much greater effect on a poor coloniser, *M. perennis*, than on a good coloniser, *T. officinale*. The observed patch occupancy for both species (~0.3) corresponded quite closely to the estimated rate of habitat turnover in Lincolnshire of 0.002 (Figure 2A). However, with no habitat turnover, *M. perennis* should benefit greatly, with patch occupancy almost double that observed in Lincolnshire (Figure 2A). In contrast, reducing habitat destruction to zero has no effect on patch occupancy of *T. officinale* (Figure 2B). With increasing habitat destruction, *M. perennis* patch occupancy declines dramatically, and extinction probability rises from zero to almost one across a narrow range of habitat destruction rates (0.01 – 0.015; Figure 2). *T. officinale* can persist in the landscape at much higher rates of habitat destruction, though extinction probability exceeds 0.5 when habitat destruction rate >0.03 (Figure 2B).

Discussion

The forest history in Lincolnshire is broadly representative of northwestern Europe, and to a lesser extent eastern North America, where human land use has resulted in a patchwork of forests varying in area, isolation, and age (Whitney, 1994; Kirby and Watkins, 1998). Applied to the Lincolnshire data set, the modified IFM has provided novel insights into the landscape-scale population dynamics of (forest) plants in highly fragmented and dynamic landscapes. A substantial body of research has focused on the colonisation of restored recent forests from source populations in remnant ancient forests in Europe (e.g., Bossuyt *et al.*, 1999; Grashof- Bokdam, 1997; Verheyen *et al.*, 2003b) and in North America (e.g., Matlack, 1994; Bellemare *et al.*, 2002; Flinn and Marks, 2004). In a formal metapopulation context, ancient and recent forests have been previously treated as a mainland-island system (Vellend, 2003). However, forest destruction (and creation) has been an ongoing process since at least the Roman era (e.g., Dupouey, *et al.* 2002) and consequently, a more realistic framework relaxing the formal distinction between ancient and recent forest was warranted (see also Vellend, 2004). This is particularly true for the northwestern European landscape. Our results indicate that, due to continuous patch turnover, species such as *M. perennis* that are known to be slow colonisers in central Lincolnshire (cf. Peterken and Game, 1981) have much lower patch occupancies than those expected if the landscape were static (Figure 2). Furthermore, the maximum rate of patch turnover that allows persistence is several times lower for slow colonising species than for species known to be rapid colonisers, such as *T. officinale*.

Species' sensitivity to landscape structure and dynamics is expected to depend on their life-history traits. Simulations by Johst *et al.* (2002) predicted that species characterised by small local population growth rates and short-range dispersal were particularly vulnerable in highly dynamic and fragmented landscapes, and that the upper-limit of patch turnover rate allowing metapopulation persistence was greater for long-distance dispersers. By analysing slow colonising forest plant species from across Europe and eastern North America, Verheyen *et al.*

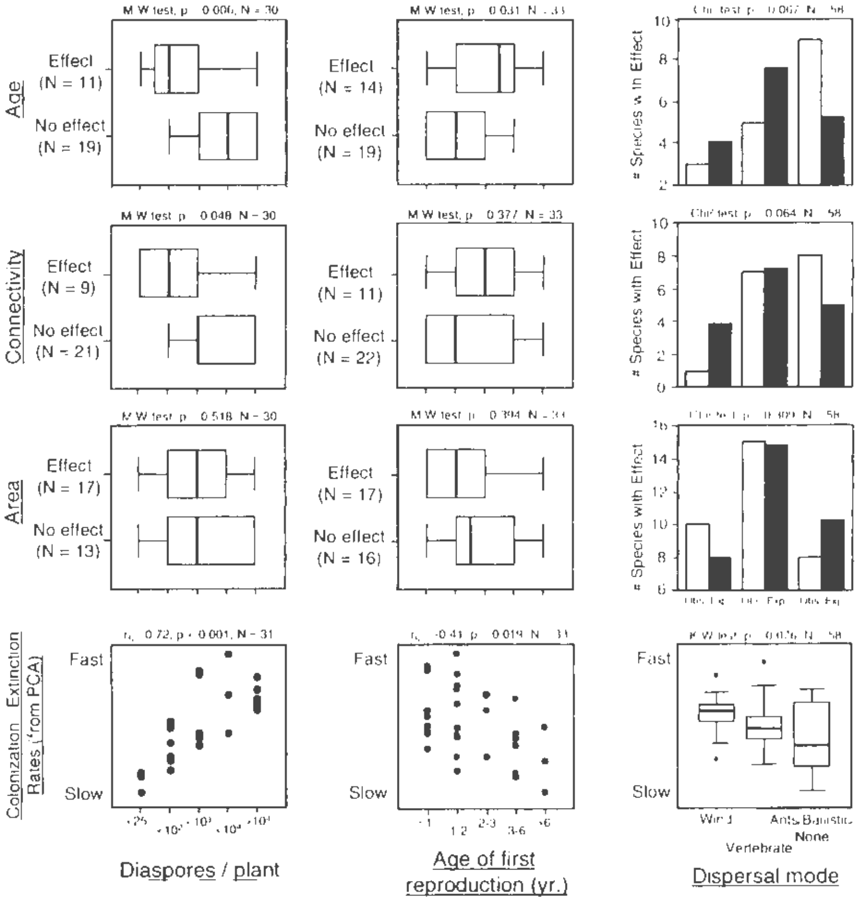

Figure 1. Relationships between three selected life-history traits (diaspore production, age of first reproduction and dispersal mode) and the presence/absence of age, connectivity and area effects in the logistic regressions, and the colonisation-extinction PCA axis derived from the modified IFM. Boxplots are used for diaspore production and age of first reproduction while the observed (white bars) vs. the expected (black bars) number of cases exhibiting age, area and connectivity effects are depicted for agency of dispersal.

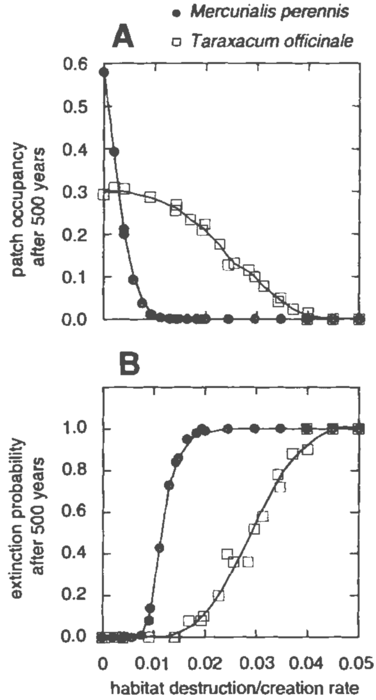

Figure 2. The effect of habitat destruction rate on patch occupancy (A) and extinction probability (B) after 500 years for Mercurialis perennis (filled circles) and Taraxacum officinale (open squares). Each data point in A represents the mean of 50 simulations; in panel B, the y-axis is the proportion of simulations in which the species went extinct.

(2003a) found that these species were generally characterised by long generation times and that they produce relatively few, heavy seeds which lack adaptations for long-distance dispersal and do not form a persistent seed bank. Here we found similar traits to be correlated with colonisation and extinction rates (Figure 1). The nature of the trait correlations, together with the infrequent co-occurrence of both age- and area-effects in the logistic regression models (see also Ouborg, 1993; Grashof-Bokdam, 1997), therefore suggest that life-history trade offs (e.g., many small vs. few large seeds) result in a colonisation-extinction trade-off as well. Species that are capable of rapid colonisation tend to be extinction prone at the patch scale, and vice versa.

Finally, our findings also have implications for management and conservation. Most conservation efforts have been oriented towards the establishment and protection of ancient forest reserves (Peterken, 1996), so that a fraction of the patches become static. Our results emphasise the importance of this strategy, in that some species (e.g., *M. perennis*) are extremely sensitive to habitat turnover, and may benefit greatly from habitat protection. Also, knowing a species' traits is surprisingly informative regarding the rate of metapopulation dynamics (Figure 2), and since species with slow dynamics are particularly sensitive to landscape change, knowledge of species' traits can also help determine which species to monitor or use as indicators in a conservation context. Considerable attention has been paid to the effect of landscape structure on metapopulation dynamics and persistence (e.g., Bascompte and Solé, 1996). Our results provide both theoretical and empirical evidence of an equally important influence of temporal landscape dynamics on the viability of metapopulations. Destroying and creating patches of habitat is profoundly different than leaving habitat patches alone.

References

Amarasekare, P. & H. Possingham (2001) Patch dynamics and metapopulation theory: the case of successional species. *Journal of Theoretical Biology* **209**:333-344.
Bascompte, J. & R. Solé (1996) Habitat fragmentation and extinction thresholds in explicit models. *Journal of Animal Ecology* **65**:465-473.
Bellemare, J.; Motzkin, G. & Foster, D. R. (2002) Legacies of the agricultural past in the forested present: an assessment of historical land-use effects on rich mesic forests. Journal *of Biogeography* **29**:1401-1420.
Bossuyt, B.; Hermy, M. & Deckers, J. (1999) Migration of herbaceous plant species across ancient-recent forest ecotones in central Belgium. *Journal of Ecology* **87**:628-638.
Brown, J. H. & Kodric-Brown, A. (1977) Turnover rates in insular biogeography: effects of immigration on extinction. *Ecology* **58**:445-449.
Dupouey, J. L.; Dambrine, E; Laffite, J. D. & Moares, C. (2002) Irreversible impact of past land use on forest soils and biodiversity. *Ecology* **83**:2978-2984.
Durrett, R. & Levin, S. A. (1994) The importance of being discrete (and spatial). *Theoretical Population Biology* **46**:363-394.
Ehrlén, J., & Eriksson, O. (2000) Dispersal limitation and patch occupancy in forest herbs. *Ecology* **81**:1667-1674.
Ehrlén, J., & Eriksson, O. (2003) Large-scale spatial dynamics of plants: a response to Freckleton and Watkinson. *Journal of Ecology* **91**:316-320.
Ellenberg, H.; Weber, H. E., Düll, R.; Wirth, V.; Werner, W. & Paulissen, D. (1992) *Zeigerwerte von Pflanzen in Mitteleuropa*. Verlag Erich Goltze, Göttingen.
Ellner, S. P. & Fussmann, G. (2003). Effects of successional dynamics on metapopulation persistence. *Ecology* **84**:882-889.
Flinn, K. M & Marks, P. L. (2004). Land-use history and forest herb diversity in Tompkins County, New York, USA. O. Honnay, K. Verheyen, B. Bossuyt, and M. Hermy (eds). Forest biodiversity: Lessons from history for conservation, CAB International, Wallingford 81-96.
Freckleton, R. P. & Watkinson A. R. (2002) Large-scale spatial dynamics of plants: metapopulations, regional ensembles and patchy populations. *Journal of Ecology* **90**:419-434.
Grashof-Bokdam, C. (1997). Forest species in an agricultural landscape in the Netherlands : Effects of habitat fragmentation. Journal *Vegetation Science* **8**:21-28.
Hanski, I. (1994) A practical model of metapopulation dynamics. *Journal of Animal Ecology* **63**:151-162.
Hanski, I. (1999) *Metapopulation ecology*. Oxford University Press, Oxford.
Hastings, A. (2003) Metapopulation persistence with age-dependent disturbance or succession. *Science* **301**:1525-1526.

Hill, M. O.; Mountford, J. O.; Roy, D. B. & Bunce, R. G. H. (1999). *Ellenberg's indicator values for British plants.* Institute for Terrestrial Ecology, Huntingdon, United Kingdom.

Johst, K., Brandl, R. & Eber, S. (2002) Metapopulation persistence in dynamic landscapes: the role of dispersal distance. *Oikos* **98**:263-270.

Keymer, J. E., Marquet, P. A.; Velasco-Hernández, J. X., and Levin, S. A. (2000). Extinction thresholds and metapopulation persistence in dynamic landscapes. *American Naturalist* **156**:478-494.

Kirby, K. J. & Watkins, C. (1998) *The ecological history of European forests.* CAB International, Wallingford.

Levins, R. (1969) Some demographic and genetic consequences of environmental heterogeneity for biological control. *Bulletin of the Entomological Society of America* **15**:237-240.

MacArthur, R.H., & Wilson, E. O. (1967) *The theory of island biogeography.* Princeton University Press, Princeton.

Matlack, G. R. (1994) Plant species migration in a mixed-history forest landscape in eastern North America. *Ecology* **75**:1491-1502.

Moilanen, A., & Nieminen, M. (2002) Simple connectivity measures in spatial ecology. *Ecology* **84**:1131-1145.

Ouborg, N. J. (1993) Isolation, population-size and extinction. The classical metapopulation approaches applied to vascular plants along the Dutch Rhine-system. *Oikos* **66**:298-308.

Peterken, G. F. (1976) Long-term changes in the woodlands of Rockingham forest and other areas. *Journal of Ecology* **64**:123-146.

Peterken, G. F. (1996) Natural *woodland. Ecology and conservation in northern temperate regions.* Cambridge University Press, Cambridge.

Peterken, G. F. & Game, M. (1981) Historical factors affecting the distribution of *Mercurialis perennis* in central Lincolnshire. *Journal of Ecology* **69**:781-796.

Peterken, G. F., & Game, M. (1984) Historical factors affecting the number and distribution of vascular plant species in the woodlands of central Lincolnshire. *Journal of Ecology* **72**:155-182.

Rackham, O. (2003) Ancient woodland, its history and uses in England. Castlepoint Press.

Vellend, M. (2003) Habitat loss inhibits recovery of plant diversity as forests regrow. *Ecology* **84**:1158-1164.

Vellend, M. (2004) Metapopulation dynamics following habitat loss and recovery: forest herbs in ancient and recent forests. O. Honnay, K. Verheyen, B. Bossuyt, and M. Hermy (eds). Forest biodiversity: Lessons from history for conservation. CAB International, Wallingford 163-178.

Verheyen, K, Honnay, O, Motzkin, G., Hermy, M & Foster, D. R. (2003a). Response of forest plant species to land-use change: a life-history based approach. *Journal of Ecol*ogy **91**:563-577.

Verheyen, K., Guntenspergen, G, Biesbrouck, B & Hermy, M. (2003b). An integrated analysis of the effects of past land-use on forest plant species colonization at the landscape scale. *Journal of Ecology* **91**:731-742.

Verheyen, K. & M. Hermy (2004) Recruitment and growth of herb-layer species with different colonizing capacities in ancient and recent forest. *Journal of Vegetation Science* **15**:125-134.

Whitney, G. G. (1994) From coastal wilderness to fruited plain. Cambridge University Press, New York

Does pesticide spray drift and fertiliser over-spread have impacts on the ground flora of ancient woodland margins?

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Abstract

We recorded the distance and concentration of pesticide drift into woodland edges under a variety of wind and edge density conditions. The highest drift concentrations in woodland are generally confined to within 5m of the spray boom, although drift may be detectable at higher windspeeds at least up to 10m.

We assessed the sensitivity of a variety of ancient woodland plants to herbicide (glyphosate) and fertiliser at drift and run-off concentrations. Species' responses were variable, ranging from minor symptoms to complete mortality. A single application of pesticide at drift concentration had long term effects on growth that persisted through 16 months.

Finally, we conducted woodland margin surveys to investigate the impacts of agri-chemicals on ground flora communities *in situ*. Fewer species and lower individual abundance were recorded in woodland margins alongside farmland with high agri-chemical inputs compared to land with lower inputs. Species identified as highly sensitive in greenhouse tests occurred less frequently in margins alongside high-input farmland.

Our results provide much needed empirical information on which decisions on no-spray buffer zones may be based.

Introduction

The expansion of agriculture in Britain over the course of centuries has led to the fragmentation of woodland habitats and their relative isolation within the wider agricultural matrix (Rackham, 1986). Most British native woodland plant species are best represented within ancient semi-natural woodlands. The current restricted distribution of many of these species reflects historical woodland loss and fragmentation, emphasised by relatively poor dispersal and colonising abilities (Rose, 1999). Consequently, remaining fragments of ancient semi-natural woodland are critical sites for the conservation of many woodland plants.

Native woodland plant communities are under threat from conifers planted on ancient woodland sites (Curtis *et al.*, 2002), increased shade due to loss of traditional management (Kirby & Solly, 2000), competition with introduced (Gilbert & Bevan, 1999) or ruderal species (Boutin & Jobin, 1998), over grazing by rising deer populations (Kirby & Solly, 2000), as well as changes in climate and/or disease (Barkham, 1992). In addition to these factors agricultural intensification particularly over the past few decades has led to increased pressure on the remaining fragments of semi-natural woodland, through direct and indirect impacts of agri-chemicals on their flora and fauna. Off-target deposition of pesticides has been estimated at around 10% of the amount applied (Elliot & Wilson, 1983) and fertiliser overspread in field boundaries may be up to 195% (Rew *et al.*, 1992). Tsiouris & Marshall (1998) have shown that peak deposition at the field edge can reach up to 150 kg N/ha. There is also evidence that forest edges may concentrate airborne pollutants (Weathers *et al.*, 2001), but there has been very little direct research into drift and the impacts of agri-chemicals on woodlands.

We therefore undertook to firstly quantify the extent of spray drift into woodland margins, and secondly to assess the impact of spray drift on selected woodland species in controlled experiments and by way of field surveys.

Measurement of spray drift

The pattern of spray drift into woodland margins is likely to be affected by several biotic and abiotic factors. In particular, the influence of wind speed and margin density on spray drift and deposition were investigated.

Methodology

Thirty replicated trials measuring spray drift into woodland margins, which differed in aspect, edge density and composition, were carried out over a three year period, representing the full range of weather conditions likely to be encountered by farmers during the application of pesticides. Vertical line samplers (2 mm x 1 m polythene tubing) and horizontal deposition samplers (petri dishes) were placed in woodland margins, along transects perpendicular to the field edge. The samplers were positioned at 0, 1, 2, 3, 4, 6, 8 and 10 m from the margin in order to capture spray drift and deposition, and the set-up was replicated 5 times with 10 m between the transects. Control samplers were positioned in the field edge, 2 m from the margin, directly under the path of the spray boom. A tracer (sodium fluorescein) was applied in solution to the field edge, only when the wind was blowing directly into the margin. A typical herbicide application regime was followed, simulating standard agricultural practice, using a wheelbarrow sprayer. Spray was applied through 6 Lurmark-orange nozzles (BCPC code F110/0.8/3) on a 2.5 m boom, 60 cm high. Ten passes were made with the sprayer, in order to even out any inconsistencies in individual spray events and to concentrate the tracer. Wind speed was measured using a hand-held anemometer, and the density of the margin vegetation was estimated for each transect by contrast analysis of digital photographs. Immediately after collection the samplers were bagged and stored in the dark to prevent degradation of the tracer. The samples were eluted in the bag, with 25 ml distilled water, and the solution decanted into glass vials. The solutions were then measured using a fluorimeter, and the concentration of fluorescein was calculated as a percentage of the applied dose.

Results

Up to 2 m distance into the woodland margin spray drift was, at higher windspeeds (4+ km/hr), around 5-10% of full application and occasionally as much as 25%. At 3-4 m spray drift concentration dropped to 2-3% (Figure 1i). Measurable (but low), amounts of drift were recorded up to 10 m from all applications at higher windspeeds (4+ km/hr). At lower windspeeds (0-4 km/hr) drift concentrations became negligible beyond 1-2 m. The effect of wind speed on spray drift (tested by multiple regression analysis on arcsine transformed data) was highly significant at 2, 6 and 10 m distances into the margin ($p < 0.002$, df = 3, 26). Spray deposition (i.e. the sedimentation of mostly larger droplets) tailed off very quickly from the edge of the sprayer boom (Figure 1ii). The influence of wind speed on deposition was also very marked close to the field edge (2 and 6 m) ($p = 0.004$ and 0.005 respectively, df = 3, 26), but became not significant further into the wood margin (10 m). Indeed, only under the higher wind speeds (4+ km/hr), within the recommended range for pesticide applications (2-9.6 km/hr), was detectable spray deposition carried beyond 4 m.

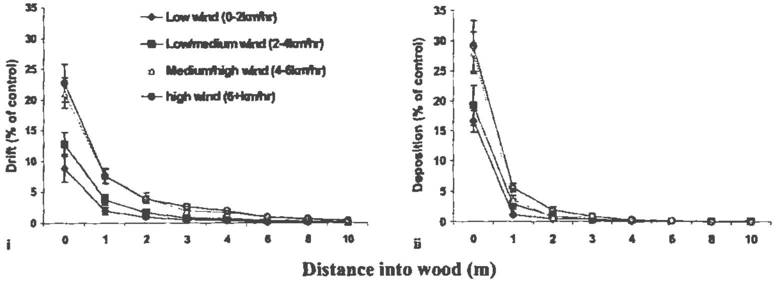

Figure 1. Influence of wind speed on the movement of i. spray drift and ii. spray deposition into woodland margins. Thirty tests sub-divided into four wind speed classes from low (0-2 km/hr), through medium (2-4 km/hr and 4-6 km/hr) to high (6+ km/hr). Shown as mean percent of control (sampler under boom), with standard error bars.

The density of woodland margin vegetation reduced spray drift significantly at 2 and 6 m ($p = 0.004$ and 0.006 respectively, df = 3, 26) but not at 10 m. Margin density also significantly reduced deposition at 2 m ($p = 0.002$, df = 3, 26), but had no apparent affect further into the wood. There was an interaction effect between wind speed and margin density at 2 m ($p = 0.003$, df = 3,26), which may be explained by increased capture efficiency of margin vegetation at higher windspeeds. This was supported by further studies, which showed that as wind speed increased so did the effectiveness of margin vegetation at reducing the amount of drift and deposition.

Woodland plant sensitivity

Three screens were undertaken to examine the impacts of herbicide and fertiliser at spray drift and over-spread concentrations on a variety of ancient woodland ground flora. Two greenhouse studies were conducted which considered short-term impacts, and one field trial to assess long-term impacts of herbicide drift and fertiliser overspread, under more realistic conditions.

Methodology

Fourteen plant species were selected for the first screen, representing either ancient woodland indicators (Rose 1999), or species of high conservation interest. These were exposed to glyphosate applied at 100%, 50% and in some cases 10% of median field application rate (equivalent to 6, 3 and 0.6 l a.i./ha). Glyphosate was chosen as the herbicide model as it is broad spectrum, has been used in previous experiments and is being increasingly used in agriculture. From these plants six species spanning the range of sensitivities were selected for further study (see Table 1). These were exposed to glyphosate concentrations of 0, 1, 5, 10 and 25% of the median field application. A fertiliser treatment was applied to half of these plants, to investigate interactions between this and the herbicide, since woodland plant communities may be exposed to the two factors simultaneously. The fertiliser treatment was 50% of average field application rate (100% = 140 kg N/ha).

Table 1. Sensitivity of screened species to herbicide treatments

High sensitivity	Intermediate sensitivity	Low sensitivity
Festuca gigantea	*Carex sylvatica*	*Adoxa moschatellina*
*Geranium robertianum**	*Galium odoratum**	*Anemone nemorosa*
*Primula vulgaris**	*Hyacinthoides non-scripta*	*Carex remota**
Sanicula europaea	*Lamiastrum galeobdolon*	*Viola riviniana**
Veronica montana	*Mercurialis perennis**	

Herbicide model: glyphosate. Relative sensitivity of species tested in the three plant screens.
*Plants selected for further study.

Results

Herbicide applied at typical drift concentrations (up to 50 percent) produced reductions in plant biomass and in some cases caused mortality, while fertiliser overspread was found to alter resource allocation in some species, by promoting shoot growth. Most species tested showed reduced growth rates and tissue damage even at glyphosate concentrations as low as 5% of the field application rate (Figure 2).

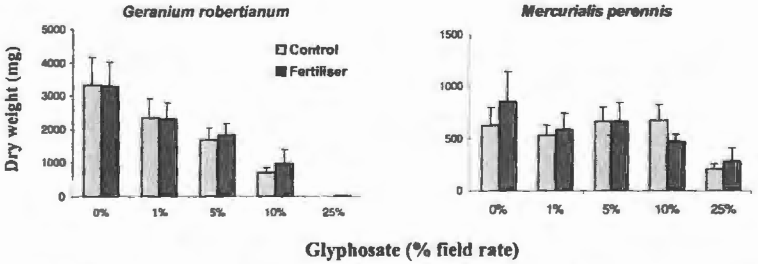

Figure 2. Field screen results for two species, harvest weight 16 months after treatment. Above ground biomass shown as mean dry weight, with standard error bars, 10 replicates for each treatment. Control (no fertiliser) shown in light grey, fertiliser treatment (70 kg N/ha) dark grey, herbicide treatment shown as percent of median field rate (6 l a.i./ha = 100%).

The severity of damage response by different species varied widely in all three screens. For example, *Geranium robertianum* biomass in the field screen was reduced by all herbicide treatments down to 1% of field rate ($p < 0.000$, df = 4), while *Mercurialis perennis* only showed biomass reduction at the highest dose rate - 25% ($p = 0.001$, df = 4) (Figure 2). The short and long-term impacts of herbicide at drift concentrations were different in some species, for instance *Mercurialis perennis* appeared to be highly sensitive immediately after treatment but recovered well, and showed fewer long-term effects than many other species. The tested species were divided into three sensitivity bands on the basis of their response to the herbicide treatments in the three experiments (Table 1).

Woodland margin surveys

Field surveys were carried out in woodland margins to investigate the potential impacts of herbicide spray drift and fertiliser run-off on the distribution of ground flora species (higher plants only). The margins were subject to potentially contrasting agri-chemical impacts, as represented by the adjacent agricultural matrix and land use histories.

Methodology

A total of 90 woodland sites were selected on the North Downs of Kent, all within woods on chalk substrate. Thirty margins were surveyed alongside each of three different land-use types: unimproved grassland, improved grassland and arable land, thought to represent low, medium and high agri-chemical input regimes. At each site, a total of 6 transects were surveyed, each approximately 10 m apart. Each transect consisted of three 2 x 4 m quadrats (Q), at distances of 0-2 m, 2-4 m and 10-12 m, termed outer margin, inner margin and interior for the purpose of analysis. Each quadrat was subdivided into eight 1 x 1 m quadrats (q). Presence/absence of all species within each smaller quadrat was noted allowing for a crude estimate of frequency and abundance. Associated biological and site variables, including soil type, slope, aspect, and vegetation density of the margin were recorded, as well as anthropogenic variables such as woodland management.

Results

Statistical analyses were performed using parametric (ANOVA) or non-parametric (Kruskal-Wallis) tests in the statistical package S-plus. Data for individual species were arcsine transformed to make them normally distributed, allowing parametric statistical analysis. An error term was included to account for the nesting of quadrat (Q) within each margin. Initially all three levels of Q were included, but no single or group of species showed significant differences between margin types at the 0.01 level, so subsequently the data for the interior quadrats were removed. This can be justified as the spray drift measurements indicate that only under consistently higher windspeeds would drift be carried that far (10-12 m).

Whole data-set analysis

Species richness, diversity and abundance were extracted from the whole data-set for analysis. The Shannon-Wiener diversity index was used for diversity scores. Analyses of the composition of vegetation of just the outer quadrat (0-2 m) of woodland margins alongside the three different land-uses, indicated that there were significant differences between outer margins. Changes in species richness and abundance between the outer margins were both highly significant ($p = 0.001$ and $p = 0.002$ respectively; df = 2), but differences in diversity were less significant (p = 0.030; df = 2). There were fewer species and lower abundance in the outer margins alongside arable land compared to the other two land uses, and both parameters were highest alongside unimproved grassland (Figure 3).

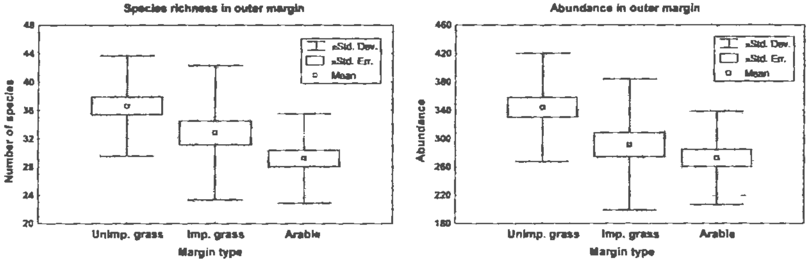

Figure 3. Changes in species richness (number of species) and abundance (occurrences of all species) in the outer margin (0-2 m) alongside different adjacent land uses. Shown as mean with intervals for standard error and standard deviation, for each of the three adjacent land-use types, unimproved grassland (left), improved grassland (middle) and arable land (right).

Individual species analysis

Common species were considered to be those which were observed in more than 100 of the smaller quadrats (q). Each species was analysed individually for changes in its distribution within the different woodland margins. Even with the inner quadrat data removed only one species showed significant differences at the level of margin type (*Geranium robertianum* – p = 0.005, df = 2). However, a much larger number (29 out of 51 common species) showed significant differences at the quadrat (Q) level. Two species showed a significant interaction effect at the $p < 0.01$ level (*Hyacinthoides non-scripta* and *Glechoma hederacea*), but this is probably attributable to the highly significant quadrat effect.

Multiple species analysis

The common species were assembled into groups with shared attributes, based on criteria such as rarity, distribution, growth habit and status as indicator species in the SE region (Rose, 1999). There were significant differences in the frequency of both all common species and common ancient woodland indicators (p = 0.003 and p = 0.002 respectively, df = 2), when only the outer margin data (0-2 m) were included in the analysis. In both cases the highest mean frequency was in woodland margins alongside unimproved grassland. The lowest frequency of common species was beside arable margins, the lowest frequency of common indicator species was beside improved grassland margins. Common non-indicator species showed only a marginal difference in frequency alongside the different margin types (p = 0.019, df = 2).

Data for the frequency of the 14 species in the plants screen were extracted from the survey data as a whole, and divided into three groups of high, medium and low sensitivity using Table 1. The frequency of high sensitivity screened species was significantly related to margin type (p = 0.002 – rising to 0.005 with all three levels of Q, df = 2). There was a much higher frequency of these sensitive species in margins alongside unimproved land compared with the other two types of land use, the lowest frequency being alongside arable margins. The medium and low sensitivity groups did not show significant responses to the different margin types.

Conclusions and recommendations

The highest drift concentrations in woodland are generally confined to within 5 m of the spray boom. In conditions with open margins and higher wind speeds, drift may be detectable at least up to 10 m. At lower wind speeds (0-4 km/hr) the physical structure of the woodland margin has little impact on the amount of drift measured. However, as the wind speed increases (>4 km/hr), so the attenuating effect of margin density increases, suggesting that margin structure may be important in reducing the impacts of the most extreme drift events. To put these figures in perspective the Code of Practice for the Safe Use of Pesticides on Farms and Holdings (MAFF, 1998) recommends spraying when the wind speed is between Force 1 and 3, i.e., 2 to 9.6 km/hr.

Spray drift, mainly within the first 4 m, has been measured at concentrations, which have been shown to have impacts on certain woodland plants. The 14 species tested here showed a range of sensitivity to the herbicide used (glyphosate), however, it remains to be seen how many other species would fall into the sensitive category if tested. The survey results confirmed that the distribution/abundance of those species, which were identified as sensitive to the herbicide treatment in controlled experiments, were also affected in the field to a significant degree. Providing evidence that land use can have an impact on the community composition of adjacent woodland habitats. The distributions of ancient woodland vascular plants are limited, so populations of those species which are found to be particularly sensitive to herbicide drift, may be threatened. If the changes in margin flora seen in these surveys in Kent are repeated across the country, then there could be serious implication for the conservation of particular species. The results presented here are similar to work describing woodland margins bordering farmland in North America (Jobin *et al.*, 1997), which showed that species diversity was higher adjacent to fields where herbicides had not been used recently compared to those where they had. However, surveys of field boundaries in the Netherlands by Kleijn & Verbeek (2000) indicated that fertiliser inputs influenced boundary vegetation, whilst herbicides did not. Although, pot experiments by the same author suggested that both fertilisers and herbicides act to decrease boundary vegetation diversity (Kleijn & Snoeijing, 1997).

Buffer zones of 5 m would protect woodland margin plants from the most damaging effects of drift. Whilst the most obvious impacts are limited to 0-4 m into the margin, particularly sensitive species may be affected over a greater area. Wider buffer zones may be required to protect sensitive species, particularly where these occur at wood edges, to minimise cumulative impacts, or to facilitate recruitment. The results presented relate primarily to the impacts of herbicides on woodland ground flora. Care should be taken with respect to the application of buffer zone recommendations to mitigate the potential impacts of other groups of pesticides, which may display different toxicities and risk of drift. For example, both insecticides and fungicides are often applied in much finer sprays than herbicides and may carry greater environmental risks, leading to the need for more extreme measures, and wider buffer zones to limit their potential impacts.

Acknowledgements

This work was supported by the John Stanley Foundation. Additional funds provided by English Nature and the Woodland Trust allowed us to expand the work and to carry out some of the research detailed in this paper. We would like to thank Graham Matthews, Keith Kirby and Richard Smithers for their advice and comments on different aspects of the work.

References

Barkham, J.P. (1992) The effects of management on the ground flora of ancient woodland, Brigsteer Park Wood, Cumbria, England. *Biological Conservation* 60: 167-187.

Boutin, C. & Jobin, B. (1998) Intensity of agricultural practices and effects on adjacent habitats. *Ecological Applications* 8(2): 544-557.

Curtis, T; Pryor, S.N. & Peterken, G.F. (2002) *Restoring plantations on ancient woodland sites: Conversion to native woodland or conservation of ancient woodland communities?* The Woodland Trust.

Elliot, J.G. & Wilson, B.J. (1983) The influence of weather on the efficiency and safety of pesticide application: the drift of herbicides. *BCPC occasional publication no.3*, BCPC, London, UK.

Gilbert, O. & Bevan, D. (1999) The effect of urbanisation on ancient woodlands. *British Wildlife* 8(4): 213-218.

Jobin, B; Boutin, C. & DesGranges, J.-L. (1997) Effects of agricultural practices on the flora of hedgerows and woodland edges in southern Quebec. *Canadian Journal of Plant Science* 77: 293-299.

Kirby, K. & Solly, L. (2000) Assessing the condition of woodland SSSIs in England. *British Wildlife* 11(5): 305-311.

Kleijn, D. & Snoeijing, G.I.J. (1997) Field boundary vegetation and the effects of agrochemical drift: Botanical changes caused by low levels of herbicide and fertilizer. *Journal of Applied Ecology* 34: 1413-1425.

Kleijn, D. & Verbeek, M. (2000) Factors affecting the species composition of arable field boundary vegetation. *Journal of Applied Ecology* 37: 256-266.

MAFF (1998) *Green Code - Code of practice for the safe use of pesticides on farms and holdings*, HMSO, London, UK.

Rackham, O. (1986) Woodland. *The history of the countryside*, Phoenix Press, London, UK 62-118.

Rew, L.J; Theaker, A.J. & Froud-Williams, R.J. (1992) Nitrogen fertilizer misplacement and field boundaries. *Aspects of Applied Biology* 30: 203-206.

Rose, F. (1999) Indicators of ancient woodland - The use of vascular plants in evaluating ancient woods for nature conservation. *British Wildlife* 8(4): 241-251.

Tsiouris, S. & Marshall, E.J.P. (1998) Observations on patterns of granular fertiliser deposition beside hedges and its likely effects on the botanical composition of field margins. *Annals of Applied Biology* 132: 115-127.

Weathers, K.C; Cadenasso, M.L. & Pickett, S.T.A. (2001) Forest edges as nutrient and pollution concentrators: potential synergisms between fragmentation, forest canopies, and the atmosphere. *Conservation Biology* 15(6): 1506-1514.

Ecological continuity and spatio-temporal connectivity: modelled response of the lichen *Bryoria furcellata* to landscape dynamics

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Abstract

In recent years there has been greater appreciation amongst landscape ecologists and land-use planners of the importance of the spatial arrangement of habitats in the landscape. However, especially for non-vagile species, the temporal arrangement of habitats through history may be an equally important factor. For instance, certain old forest lichen species are said to indicate 'ecological continuity' of woodland habitat. In dynamic systems such as Scottish upland woodlands, historical presence of habitats may be discontinuous at the site level, however, ecological continuity may persist given sufficient functional spatio-temporal connectivity of suitable habitat in the whole landscape.

To investigate the effects of habitat dynamics on forest dwelling species, a spatially explicit forest dynamics model for Glen Affric, northern Scotland was constructed, using the landscape modelling tool SELES (Spatially Explicit Landscape Event Simulator). This model is a stochastic, cohort-based model of succession and wind disturbance. Future spatio-temporal connectivity (and thus ecological continuity) for the pinewood epiphytic lichen *Bryoria furcellata* was assessed by linking the forest dynamics model with a habitat suitability model and spatially explicit population model. Results of the simulations suggest that assessment of spatial connectivity considered without temporal context may be insufficient to explain landscape utilisation.

Introduction

In north-western Europe there has been a long history of clearance and fragmentation of natural forest (Angelstam *et al.*, 2001). Woodland biodiversity is severely affected by the loss and fragmentation of forest habitats (Fahrig, 1997; Kouki *et al.*, 2001; Olff and Ritchie, 2002). For example, many species associated with old-growth forests have become extinct or exist as small populations at risk from extinction in the future (Alexander, 1999; Berg *et al.*, 1994; Humphrey *et al.*, 2002; Ranius *et al.*, 2003; Schiegg, 2000).

By the beginning of the 20th century only 4-5% of the land area of Britain was occupied by ancient semi-natural woodland, the modified remnants of the original 'wildwood' (Peterken, 1996; Spencer and Kirby, 1992; Roberts *et al.*, 1992). There has been increasing interest in restoring wooded landscapes, by expanding existing semi-natural woodland area, rehabilitating damaged woodland habitats, and reconnecting isolated fragments (Humphrey *et* *al.*, 2003). These measures are being implemented strategically through the development of forest habitat networks (*sensu* Peterken *et al.*, 1995).

The network idea is based on two well-established ecological concepts. Firstly, there is wide acceptance that temporal continuity of habitat conditions is essential to maintain populations of many woodland species (Peterken and Game, 1984; Honnay *et al.*, 2002). Rose (1974) coined the term 'ecological continuity' to refer to continuity of habitat suitable for epiphytic lichens; the term has since been extended to other taxa (Nordén and Appelqvist, 2001). High ecological continuity is indicated by presence of a large number of species with presumed poor dispersal capabilities that are rarely found on sites without a long historical record of woodland cover. However, in boreal and oceanic temperate forest landscapes, the importance of ecological continuity at the site scale for sustaining woodland biodiversity is being increasingly questioned (Groven *et al.*, 2002; Gustafsson and Weslien, 2003; Nordén and Appelqvist, 2001; Ohlson *et al.*, 1997). Ecosystems that are subject to large-scale disturbance experience temporal disjunction of habitat continuity at the site scale, but may retain temporal continuity at the landscape scale (Lindenmayer and Franklin, 2002). For example, many boreal forest ecosystems experience regular stand replacement disturbance through fire and there is increasing evidence that many species are adapted to such regimes and can persist in the landscape provided that there is sufficient spatio-temporal connectivity (Fries *et al.*, 1997).

Secondly, there is recognition of the importance of spatial connectivity of suitable habitat in allowing woodland species to move within and between landscapes (Beier and Noss, 1998; Brooker *et al.*, 1999). For example, Humphrey *et al.* (2004) found that development of biodiversity in UK plantations is influenced by the proximity and amount of semi-natural vegetation within the landscape. Counts of bryophyte species on deadwood were positively correlated with the amount of semi-natural woodland within 1 km of the stand.

However, there has been a tendency in the past to look at these concepts in relative isolation; ecological continuity research usually focuses at the site scale with little attention given to spatial arrangement of habitats, whilst studies of spatial connectivity generally regard the landscape as a static entity. Exceptions (e.g., Verheyen *et al.*, 2003) have usually been for intensively managed landscapes.

A consideration of the importance of spatio-temporal habitat connectivity in biodiversity maintenance marks a significant departure from recent work that focuses largely on quantifying the importance of spatial connectivity. One difficulty lies in appropriate quantification of spatio-temporal connectivity. Techniques based on landscape indices do not present viable spatio-temporal analogues because of the lack of explicit inclusion of the time dimension. In this paper we demonstrate how a linked forest landscape dynamics model, habitat suitability model and spatially explicit population model can be used to generate spatio-temporal habitat patterns and analyse their connectivity for a selected species, the epiphytic lichen, *Bryoria furcellata*.

Study area

Glen Affric is situated in the northern highlands of Scotland, at a latitude of 57° 17' N and longitude 5° W. It is notable for its large extent of native 'Caledonian' pinewood and important biological diversity. Palynological evidence suggests that the Holocene composition of the forest has been highly dynamic, with cyclical relationships occurring at the stand scale between pine, birch and open ground (Wolff & Tipping, 1999).

Forest landscape dynamics modelling

Forest dynamics were simulated at the landscape scale using the model GALDR (Glen Affric Landscape Dynamics Reconstruction; see Hope, 2003). This model may be described as a stochastic, cohort-based model of natural disturbance and succession. The operational aim of the GALDR model is to depict change in tree species composition and forest structure over large spatial extent and long timescales. GALDR represents forest structure in terms of even-aged cohorts of trees. Each cohort may be of either Scots pine (*Pinus sylvestris*) or birch (*Betula pubescens* and *Betula pendula*, aggregated) and may be further described in terms of age, height and stem density. Dynamic behaviour is governed by five submodels representing key processes: growth and autogenic mortality; seed dispersal; establishment and recruitment; herbivore impacts; and wind disturbance. Fire was not included as it was considered that the pinewoods in Glen Affric experience an oceanic climate in which fire would not be an important determinant of structure or dynamics.

The simulated area consists of 9,142 ha of land below 750 m elevation, within the River Affric catchment and contained within a rectangle defined by the Ordnance Survey grid-squares NH12 and NH22. The spatial resolution of the state data is 50 m and the temporal resolution of the dynamic simulation is ten years. Initial state data is derived from the Forest Enterprise subcompartment database.

GALDR is designed in, and implemented by, the modelling support tool SELES (Spatially Explicit Landscape Event Simulator; see Fall and Fall, 2001). It inherits concepts from forest landscape dynamics models such as LANDIS (Mladenoff *et al.*, 1996) and from community dynamics models such as the gap models originating with Botkin *et al.* (1972)

Modelling lichen abundance

Spatial outputs from GALDR were used as inputs to a simple population dynamics model for the epiphytic lichen *Bryoria furcellata* (Glen Affric Lichen Abundance Model; GALAM), also implemented using SELES. This model comprises two parts: a habitat suitability model (HSM) and a spatially explicit population model (SEPM). The elements of the whole modelling framework are shown in Figure 1. This model shows many similarities with (but was developed independently from) a lichen abundance model for *Bryoria* species in Scandinavian woodland developed by Dettki (2000).

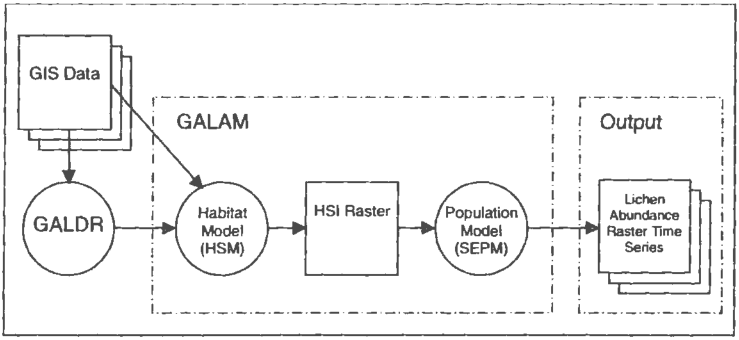

Figure 1. Diagrammatic representation of linkages in GALDR and GALAM. Squares represent datasets; circles represent models. Solid arrows show data flow.

Habitat suitability model

The HSM processes spatial output from GALDR to produce a raster map of Habitat Suitability Index (HSI). The HSI is calculated as a function of cohort variables, representing substrate availability, and cell variables, representing local environmental conditions. Cohort variables are species, age and number of stems. It is assumed that the species prefers bark of old pine trees. Cell variables comprise indicator values for light and wind regime, calculated from topographic factors and structural attributes combined from all cohorts in the stand. The assumption used here is that the lichen may be limited by low light levels or high windspeeds (and hence reduced humidity). Suitable habitat thus occurs in a balance between dense stands, which may be too dark, and open stands, which may be too windy.

Population model

The GALAM population model simulates changes in the number of thalli of *Bryoria furcellata* predicted to occur in the cells that contain suitable habitat. The model represents population change at two scales: intra-cell population dynamics and inter-cell dynamics (i.e., colonisation of new cells). Intra-cell dynamics are represented by a simple logistic model, but where carrying capacity is determined by the HSI provided by the HSM. Thus, lichen abundance will respond to changes in habitat quality but with a time lag of some decades.

A stochastic model has been used to simulate dispersal and subsequent colonisation of new cells. The range of possible dispersal loci has been limited to cells adjacent to the source cell because the species has been assumed to be a poor disperser. The number of propagules arriving at the destination cell will be proportional to the number of propagules arising from the source cell, which in turn can be assumed to be proportional to the number of thalli. Furthermore, the probability of a propagule establishing successfully in the destination cell will be proportional to the abundance of suitable substrate, which is taken to be directly related to the HSI in the destination cell. The initial distribution pattern for the lichen was taken as a 20% random sample of suitable habitat.

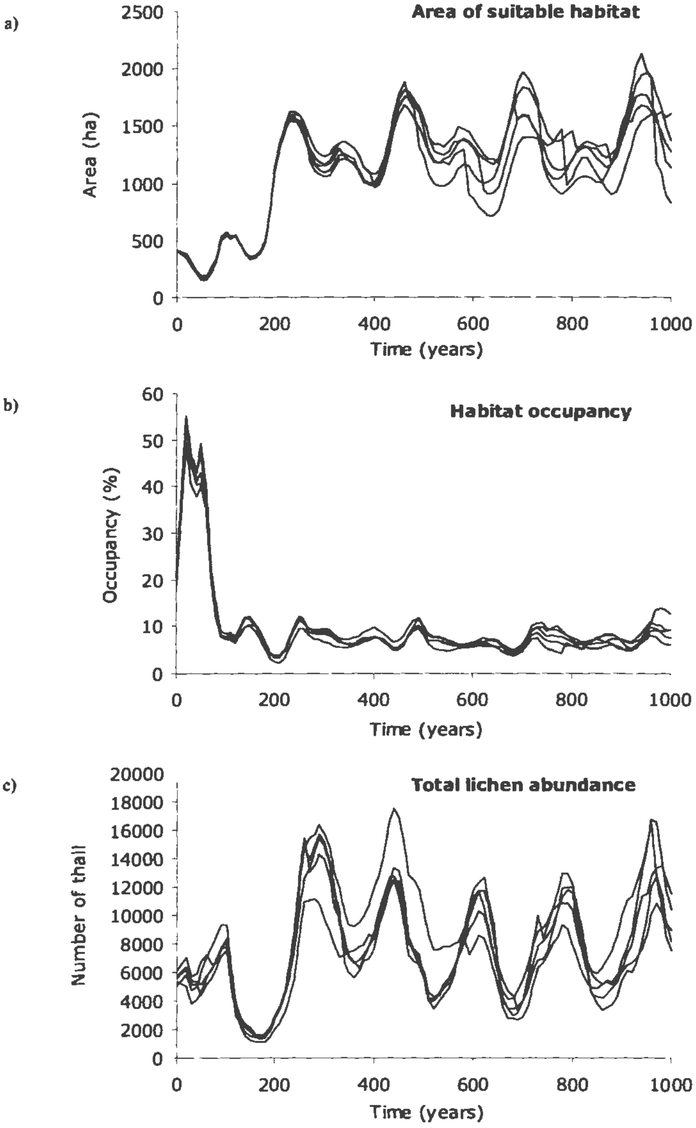

Figure 2. GALAM output (ten replications) showing predictions of (a) total area of suitable habitat, (b) habitat occupancy, and (c) total abundance over whole landscape, for *Bryoria furcellata*

Model results

Figure 2 shows an example of output from ten runs of GALDR and GALAM for a scenario of reduced herbivore pressure and woodland expansion. The first plot (a) shows initial habitat decline as many patches of habitat initially contain dense cohorts of young trees that rapidly create dense shade. The subsequent periodic pattern is a consequence of the initial age structure of the woodlands. The second plot (b) shows the percentage of cells containing suitable habitat that are actually colonised. This plot rises initially, as 'seed' cells colonise neighbouring cells, and reduction in suitable habitat leads to convergence in colonised area and habitat area. Later, new habitat arises at distance from already colonised areas and so remains unutilised; the low values of habitat occupancy that persist for the rest of the simulation reflect the functional spatio-temporal connectivity in the landscape. The last plot (c) further illustrates the decoupling of the organism's dynamics from those of its habitat. The total landscape abundance of the lichen shows a similar periodic pattern to that of habitat area but note that the dynamics are asynchronous and the frequencies differ.

Interestingly, the plot of total area colonised by the lichen (not shown in Figure 2) follows that of habitat area rather than total lichen abundance. Inspection of the spatial output reveals that this discrepancy is due to concentration of lichen density in a few specific regions with particular habitat attributes. These regions represent areas where conditions are marginal for tree growth and so regeneration is never dense enough to completely exclude the lichen through shading. Tree regeneration may commence earlier because stand densities are consistently lower in such areas, giving rise to the shorter period for total lichen abundance observed in the simulation. Such patches act as refugia for *Bryoria furcellata* when conditions become unsuitable in the wider forest; as such, they form a keystone of landscape spatio-temporal connectivity for the lichen, providing the vital ecological continuity for its survival.

Uncertainty is a universal feature of all models, but levels tend to be particularly high in ecological models dealing with large areas and long timescales. In the present application the main sources of uncertainty stem from: (a) quality and adequacy of input data; (b) limited understanding of the ecological processes governing habitat dynamics and (c) limited autecological understanding of the target species. Levels of uncertainty are very high for each of these factors and compound each other in final outputs. As a result, circumspection is required in use of the models to aid planning decisions. However, despite high uncertainties, the model may serve to illustrate general points about the response of poorly vagile organisms in dynamic habitats.

Conclusions

The results of this study indicate that spatio-temporal connectivity of habitats may be a crucial factor in determining ecological continuity in dynamic landscapes. Assessment of spatial connectivity considered without temporal context may be insufficient to explain landscape utilisation. It is suggested that actual or predicted proportion of suitable habitat occupied by a species may provide a good measure of functional spatio-temporal connectivity for that species. However, *high* spatio-temporal connectivity may not always be a necessary condition for species persistence, so long as such connectivity as does exist allows sufficient provision of ecological continuity. Furthermore, high historical spatio-temporal connectivity is neither a necessary or sufficient condition for high spatial connectivity in the present.

Acknowledgements

The authors are grateful to the following individuals and organisations: Brian and Sandy Coppins, who gave valuable advice on the ecology of *Bryoria furcellata*; Forest Enterprise for allowing access to Glen Affric and providing the sub-compartment data; and to Chris Quine for valuable comments on earlier drafts of this paper. The work described in this paper was carried out by JCH as part of a PhD project funded by the Forestry Commission.

References

Alexander, K.N.A. (1999) The invertebrates of Britain's wood pastures. *British Wildlife* **11:** 108-117.

Angelstam, P., Breuss, M. & Mikusinski, G. (2001) Toward the assessment of forest biodiversity at the scale of forest management units - a European landscape perspective. A. Frank, O. Laroussinie & T. Karjalainen (eds). *Criteria and indicators for sustainableforest management at the forest management unit level,* European Forest Institute, Joensuu 59-74.

Beier, P. & Noss, R.F. (1998) Do habitat corridors provide connectivity? *Conservation Biology* **12:** 1241-1252.

Berg, A., Ehnstrom, B., Gustafsson, L., Hallingback, T., Jonsell, M. & Weslien, J. (1994) Threatened plant, animal and fungus species in Swedish forests: distribution and habitat associations. *Conservation Biology* **8:** 718-731.

Botkin, D.B., Janak, J.F. & Wallis, J.R. (1972) Some ecological consequences of a model of forest growth. *Journal of Ecology* **60:** 849-873.

Brooker, L., Brooker, M., & Cale, P. (1999) Animal dispersal in fragmented habitat: measuring habitat connectivity, corridor use, and dispersal mortality. *Conservation ecology* **3**(1). URL: http://www.consecol.org/vol3/iss1/art4

Dettki, H. (2000) Epiphytic lichens in boreal forest landscapes: influence of forestry and spatial structure. Unpublished PhD Thesis. Umeå University, Umeå, Sweden.

Fahrig, L. (1997) Relative effects of habitat loss and fragmentation on population extinction. *Journal of Wildlife Management* **61:** 603-610.

Fall, A. & Fall, J. (2001) A domain-specific language for models of landscape dynamics. *Ecological Modelling* **141:** 1-18.

Fries, C., Johansson, O., Pettersson, B. & Simonsson, P. (1997) Silvicultural models to maintain and restore natural stand structures in Swedish boreal forests. *Forest Ecology and Management* **94:** 89 - 103.

Groven, R., Rolstad, J., Storaunet, K.O. & Rolstad, E. (2002) Using forest stand reconstructions to assess the role of structural continuity for late-successional species. *Forest Ecology and Management* **164:** 39-55.

Gustafsson, L. & Weslien, J. (2003) High occurrence of red-listed species in mature boreal forest in Sweden. Skogforsk Resultat 17, Skogforsk, Uppsala.

Honnay, O., Bossuyt, B., Verheyen, K., Butaye, J., Jacquemyn, H. & Hermy, M. (2002) Ecological perspectives for the restoration of plant communities in European temperate forests. *Biodiversity and Conservation* **11:** 213-242.

Hope, J.C.E. (2003) Modelling forest landscape dynamics in Glen Affric, northern Scotland. PhD Thesis. University of Stirling, Stirling.

Humphrey, J.W., Peace, A.J., Jukes, M.R. & Poulsom, E.L. (2004) Multiple-scale factors affecting the development of biodiversity in UK plantations. O. Honnay, K. Verheyen, B.

Bossuyt & M. Hermy (eds). *Forest Biodiversity: Lessons from History for Conservation,* CABI, Leuven 143-162.

Humphrey, J.W., Stevenson, A. & Swailes, J. (2002) *Life in the deadwood: a guide to the management of deadwood in Forestry Commission forests. Forest Enterprise Living Forests Series. Forest Enterprise, Edinburgh.*

Kouki, J., Lofman, S., Martikainen, P., Rouvinen, S. & Uotila, A. (2001) Forest fragmentation in Fennoscandia: linking habitat requirements of wood-associated threatened species to landscape and habitat changes. *Scand.J.For.Res.Suppl.* **3:** 27-37.

Lindenmayer, D.B. & Franklin, J.F. (2002) *Conserving forest biodiversity: a comprehensive multiscaled approach*, Island Press, Washington DC.

Mladenoff, D.J., Host, G.E., Boeder, J. & Crow, T.R. (1996) LANDIS: a spatial model of forest landscape disturbance, succession and management. M.F. Goodchild, L.T. Steyaert & B.O. Parks (eds). *GIS and environmental modeling: progress and research issues,* GIS World Books, Fort Collins Colorado 175-180.

Nordén, B. & Appelqvist, T. (2001) Conceptual problems of ecological community and its bioindicators. *Biodiversity and Conservation* **10:** 779-791.

Ohlson, M., Söderström, L., Hörnberg, G., Zackrisson, O. & Hermansson, J. (1997) Habitat qualities versus long-term continuity as determinants of biodiversity in boreal old-growth swamp forests. *Biological Conservation* **81:** 221 - 231.

Olff, H. & Ritchie, M.E. (2002) Fragmented nature: consequences for biodiversity. *Landscape and urban planning* **58:** 83-92.

Peterken, G.F. & Game, M. (1984) Historical factors affecting the number and distribution of vascular plant species in the woodlands of central Lincolnshire. *Journal of Ecology* **72:** 155-182.

Peterken, G.F. (1996) *Natural woodland. Ecology and conservation in northern temperate regions*, Cambridge University Press, Cambridge.

Peterken, G.F., Baldock, D. & Hampson, A. (1995) A forest habitat network for Scotland. Research Survey and Monitoring Report 44, Scottish Natural Heritage, Edinburgh.

Ranius, T., Kindvalla, Kruys, N. & Jonsson, B.G. (2003) Modelling dead wood in Norway spruce stands subject to different management regimes. *Forest Ecology and Management* **182:** 13-29.

Roberts, A.J., Russell, C., Walker, G.J. & Kirby, K.J. (1992) Regional variation in the origin, extent and composition of Scottish woodland. *Botanical Journal of Scotland* **46:** 167-189.

Rose, F. (1974) The epiphytes of oak. M.G. Morris & F.H. Perring (eds). *The British oak: its history and natural history,* Classey, Faringdon, Oxfordshire 250-273.

Schiegg, K. (2000) Effects of dead wood volume and connectivity on saproxylic insect species diversity. *Ecoscience* **7:** 290-298.

Spencer, J.W. & Kirby, K.J. (1992) An inventory of ancient woodland for England and Wales. *Biological Conservation* **62:** 77-93.

Verheyen, K., Guntenspergen, G.R., Biesbrouck, B. & Hermy, M. (2003) An integrated analysis of the effects of past land-use on forest herb colonisation at the landscape scale. *Journal of Ecology* **91:** 731-742.

Wolff, H. & Tipping, R. (1999) Recent vegetation change within the pinewoods of east Glen Affric, and implications for their conservation management. Unpublished report to Forest Enterprise, Fort Augustus.

Modelling long-distance seed dispersal across fragmented landscapes

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Abstract

The potential for plants to disperse rapidly over large distances is a question of critical conservation importance as populations are increasingly threatened by climate change and the fragmentation of habitats. The ability of species to track their future climate space and to locate suitable habitat will depend on the dispersal mechanisms by which migrations occur. Recent evidence suggests that rare long-distance dispersal events may play a crucial role in driving such large-scale migrations. Hereby, a small percentage of seeds released travel significantly beyond the mean dispersal distance. Using an example from the MONARCH project of beech woodland species in Hampshire the importance of species' dispersal characteristics and the implications for landscape management are explored.

Introduction

Climate change remains high on the global research agenda with recent Intergovernmental Panel on Climate Change (IPCC) scenarios predicting that global temperatures could increase by between 1.4 and 5.8°C by 2100 relative to 1990 levels (IPCC, 2001a) with a potentially significant effect on biodiversity (IPCC, 2001b; McCarty, 2001; Thomas *et al.*, 2004). Observational evidence indicates that species are already being impacted by climate change, as well as by other human activities (e.g., Parmesan and Yohe, 2003).

At the continental to global scales, climate is expected to be the dominant factor affecting the distribution of species and, therefore, the components of habitats (Pearson *et al.*, 2002). However, other factors, such as habitat availability and dispersal ability, may be more important at local to regional scales as these affect the ability of species to establish themselves within their potential climate range.

The British and Irish governments have a number of commitments to international, European and national conservation objectives relating to species and habitats but current approaches to conservation management are being challenged due to the potential displacement of species resulting from climate change effects.

The MONARCH (Modelling Natural Resource Responses to Climate Change) project is an on-going phased investigation into the impacts of climate change on the conservation of the natural resources of Britain and Ireland. This study uses a case study area, Hampshire, from this project to evaluate the combined effects of climate change and habitat fragmentation on a selection of woodland species' distributions using a stochastic spatially-explicit simulation of plant dispersal which is coupled with habitat suitability surfaces for individual species, so as to investigate the potential role of different dispersal characteristics in mediating the ability of species to migrate rapidly through fragmented landscapes. This approach provides a theoretical framework for local landscape level guidance on setting nature conservation objectives, taking account of climate change impacts in areas of conservation importance.

Methodology

SPECIES (Spatial Estimator of the Climate Impacts on the Envelope of Species) uses a complex computer simulation program (neural network) to characterise the current distribution of species in Europe and to estimate their potential re-distribution under alternative climate change scenarios in Britain and Ireland (Pearson *et al.*, 2002).

A number of integrated algorithms in SPECIES, including a climate-hydrological process model, are used to pre-process climate (temperature, precipitation, solar radiation, vapour pressure and wind speed) and soils (AWC: available water holding capacity) data to derive relevant bioclimatic variables for input to the neural network. The model integrates these data to predict the distribution of species through the characterisation of bioclimatic envelopes. Those variables found to be most successful for plant distributions are given in Table 1 (Berry *et al.*, 2003).

Table 1 Bioclimatic input variables used in the SPECIES model.

Variable	
GDD5	Growing degree days > 5°C
AbminT	Absolute minimum temperature expected over a 20-year period
MaxT	Annual maximum monthly temperature
SWD	Accumulated annual soil water deficit
SWS	Accumulated annual soil water surplus

Baseline climate data for all years from 1961 to 2000 at a 5km x 5km spatial resolution was obtained from UKCIP (Hulme *et al.*, 2002). Six variables were directly extracted from the UKCIP baseline climatology: mean, minimum and maximum temperature; precipitation; sunshine hours; and wind speed. Monthly values of solar radiation were derived from the sunshine hours data using the method of Rietveld (1978). Monthly values of potential evapotranspiration were calculated using the method of Penman (1948). Future climate change scenarios were available from UKCIP (Hulme *et al.*, 2002; the UKCIP02 scenarios) and were based on the Hadley Centre high resolution regional climate model (HadRM3). Four scenarios have been constructed from HadRM3, which reflect differences in greenhouse gas emissions (low emissions, medium-low emissions, medium-high emission and high emissions) based on the IPCC Special Report on Emissions Scenarios (B1, B2, A2 and A1FI, respectively). Soils data and AWC were obtained from the Soil Survey and Land Research Centre (SSLRC) and the Macaulay Land Use Research Institute (MLURI) for England and Wales and Scotland respectively.

The model was trained using existing empirical data on the European distributions of species to enable a wide climate space to be characterised that captures the climatic range of future scenarios. Once a network is trained and validated at the European scale, it is then applied at a finer spatial resolution in Britain and Ireland.

The SPECIES output, in the form of a climate suitability surface was used as an input to a second neural network, along with the presence/absence of land cover types based on the Land Cover Map 2000 (provided by the Centre for Ecology and Hydrology), to generate regional scale combined climate-habitat suitability surfaces (Pearson *et al.*, 2004). The downscaled SPECIES model defines the relationship between climate, land cover and species' distributions at a 10km^2 spatial resolution before applying these relationships at a 1km^2 resolution.

The ability of species to track changes in the regional suitability surfaces simulated by the downscaled SPECIES model will be dependent on their dispersal characteristics. Thus, the modelled suitability surfaces for the case study areas were coupled with a spatially-explicit cellular automata simulation of species dispersal in changing environments. The model simulated the stochastic dispersal of species in three basic steps: (i) survival; (ii) within-cell population dynamics; and (iii) dispersal (Pearson and Dawson, 2004). Long distance dispersal was incorporated within the model enabling investigation of the potential for species to disperse rapidly under future climate change. Since we were simulating dispersal as a non-deterministic process, the model was run using a Monte Carlo approach whereby the dispersal process is run many times, so as to build up a probability surface, identifying those cells more/less likely to be populated under certain dispersal assumptions. Output from this hierarchy of models was a map of the probability of occurrence for each species, based on dispersal characteristics, climatic suitability and future land cover scenarios.

The models were tested for species in selected habitats in four contrasting case study areas, including beech woodland in Hampshire. In this paper the preliminary results are presented for *Fagus sylvatica* (beech), *Fraxinus excelsior* (ash) and *Mercurialis perennis* (dog's mercury), all dominant or characteristic species of beech hangers (hillside woodlands on chalk substrate) in Hampshire.

Results

Changes in species' suitability surfaces

The climate change scenarios show that while all of Britain and Ireland represent suitable climate space for *F. excelsior* and *M. perennis* (Figure 1a) with only the highest areas of the Grampians and north west Highlands as unsuitable climate space, *F. sylvatica* (using its native distribution) is largely restricted to lowland England and Wales. Under the 2020s low scenario much of England and Wales, except the Pennines and eastwards would become suitable for *F. sylvatica*, as would parts of the central valley in Scotland and lower areas around the Solway Firth and scattered areas in Northern Ireland. All these areas show some expansion under the 2020s high and the 2050s low scenarios. The latter also shows an increase in the unsuitable area in parts of the East Midlands and East of England and this expands further under the 2050s high scenario. This also occurs for *F. excelsior* (Figure 1b),

Figure 1. Model outputs for *Fraxinus excelsior*

a) Simulated British distribution

b) 2050s high climate change scenario suitability surface

c) Simulated British current distribution (incorporating climate and land cover)

d) 2050s high scenario suitability surface based on climate change and land cover

e) Dispersal in Hampshire under 2020s high scenario. Black dots represent current presences

f) Dispersal in Hampshire under 2050s high scenario. Black dots represent current presences and to a lesser extent *M. perennis*, with part of the lower Thames valley and Essex coast also becoming unsuitable for the former.

The down-scaled SPECIES model is trained against the presence/absence data provided by the Biological Records Centre, which for *F. sylvatica* does not distinguish between its native and planted status. The model simulates almost the whole of the country as representing suitable space with only some of the higher parts of the Grampians and north west Highlands of Scotland as unsuitable. This picture is maintained through the climate change scenarios. This shows that beech is able to grow in a wide variety of circumstances and it is only altitudinal related variables that constrain it. This is shown more clearly when the outputs are shown on the 1 km grid where suitability in highland areas is confined to the lower elevations.

The SPECIES model shows *F. excelsior* to be absent from the higher land in northern Scotland, but the down-scaled SPECIES model suggests that a slightly wider area, based on climate and land cover is unsuitable (Figures 1c and d). At the 1km grid scale, as for *F. sylvatica*, it is the higher parts of the country and eastern England that are shown as unsuitable. The down-scaled model for *M. perennis* shows a good match between the observed and simulated distribution, although part of the Cairngorms are simulated as unsuitable. More importantly for this study some squares in Hampshire are unsuitable under the combined climate and land cover scenario.

Dispersal

Mechanisms by which long-distance dispersal occur include transportation of seed in updrafts, dispersal by birds in nest material and movement of seeds attached to the fur of mammals and by human vectors. The dispersal model shows very limited dispersal for *F. sylvatica*, due to a combination of the time scale being considered (50 years) and the long time to reproductive maturity of the species, which will affect when dispersal can occur from the newly dispersed propagules. The dispersal model shows a limited range expansion for *F. excelsior*, for similar reasons as *F. sylvatica*, although under the 2050s scenario, much of Hampshire could be colonised (Figures 1e and f), *M. perennis*, however, has the potential to spread throughout much of the eastern part of Hampshire under the 2020s scenarios and that under the 2050s scenarios large parts of the New Forest could be colonised. A number of factors, however, could adversely affect the realisation of this, such as soils, herbivore predation and increase in forest canopy cover.

Discussion

A number of limitations to the modelling approach should be highlighted, including the restricted explanatory power of neural networks and the reliance on correlations between observed distributions and environmental variables. Further limitations are inherent in the availability and accuracy of datasets. Data can rarely be generated for all resolutions and for all spatial extents, but rather tends to be available for large extents at coarse resolutions, or small extents at fine resolutions. Thus, in the MONARCH study species distributions were obtained at 0.5° resolution for Europe, 10 km^2 resolution for Britain and Ireland, and 1 km^2 resolution for the local case studies. It has been necessary to design the modelling framework to take best advantage of the available data. Questions regarding the accuracy of the data also arise, in particular regarding the assumption that observed species absences are true absences, and not a result of insufficient sampling (Griffiths *et al.*, 1999). Base errors arising from data limitations are unavoidable. However, the level of success that has been achieved in simulating current species distributions has demonstrated that biogeographical trends can be identified regardless of the imperfect data that is so often all that is available in ecological studies.

In the downscaled SPECIES model the presence/absence of land cover types at a 10 km resolution does not always provide a good correlate with actual species' distribution. This is due to the fact that at this resolution nearly all 10 km^2 cells incorporate at least a small patch of suitable land cover (i.e., a 'presence'), leading to blanket coverage throughout the study region. In order to better identify correlations between land cover type and distribution of these species it would be necessary to adopt a finer resolution of analysis at which patterns in the distribution of suitable land cover are apparent in the dataset. This was not possible in this study because 10km^2 was the finest consistent resolution at which observed species' distributions are available for Britain and Ireland. Despite these limitations, this is the first time that linking such a combination of models has been attempted and the preliminary findings discussed here show the importance of such an approach.

The results for *F. sylvatica*, *F. excelsior* and *M. perennis* have shown how, for species with a comparatively low mean (<20m) and maximum dispersal distance (1km), there is some potential for them to expand their distribution in Hampshire in response to changing climate and land cover suitability. Hampshire is well within the geographical suitability range for these species, but their ability to track changes in climate space and cross unsuitable habitats at their range margins will be restricted. Thus, developing greater physical connectedness of habitats for these woodland species at their range margins is of limited relevance, as only long distance dispersal will enable them to fulfil their potential future range. There is a need to consider the possibility of: increasing the frequency of such dispersal events; enhancing the permeability of the landscape matrix surrounding suitable habitats; and ensuring that the habitats themselves are favourable for colonisation and establishment. The very different abilities of species to keep track of changing climate regimes as a result of dispersal will also have important consequences for the future composition and functioning of ecological communities.

The interaction between climate and habitat availability plays an important role in determining the biogeography of species. The SPECIES and downscaled SPECIES models have shown good potential for studying the combined effects of climate and land cover change on individual species. The modelling approach also has the potential to help uncouple effects of climate and habitat change in the interpretation and prediction of species distribution. Integration of the dispersal model enables investigation of the potential for species to migrate under future climate change and preliminary results from a selection of beech woodland species in Hampshire have shown that promoting the frequency and success of chance long distance dispersal events will be important in facilitating species' dispersal to track changes in suitable climate space.

Acknowledgements

This paper is based on work carried out in the MONARCH project (Modelling Natural Resource Responses to Climate Change). This was funded by English Nature (lead partner), Countryside Council for Wales, Environment Agency, Environment and Heritage Service, Forestry Commission, Joint Nature Conservation Committee, National Parks and Wildlife (Republic of Ireland), National Trust, Royal Society for the Protection Birds, Scottish Executive, Scottish Natural Heritage, UK Climate Impacts Programme, Welsh Assembly Government, Woodland Trust. We would like to thank these organisations for their intellectual input and practical support. The research was carried out collaboratively by the Environmental Change Institute, University of Oxford (lead partner), ADAS, CABI Bioscience and BTO.

References

Berry, P.M., Dawson, T.P., Harrison, P.A, Pearson, R.G. and Butt, N. (2003) The sensitivity and vulnerability of terrestrial habitats and species in Britain and Ireland to climate change. *Journal for Nature Conservation* **11**: 15-23.

Griffiths, G. H., Eversham, B. C. and Roy, D. B. (1999) Integrating species and habitat data for nature conservation in Great Britain: data sources and methods. *Global Ecology and Biogeography* **8**: 329-345.

Hulme, M., Jenkins, G.J., Lu, X., Turnpenny, J. R., Mitchell, T.D., Jones, R.G., Lowe, J., Murphy, J.M., Hassell, D., Boorman P., McDonald, R. & Hill, S. (2002) *Climate Change Scenarios for the United Kingdom: The UKCIP02 Scientific Report.* Tyndall Centre for Climate Change Research, School of Environmental Sciences, University of East Anglia, Norwich, UK, 120 pp.

Intergovernmental Panel on Climate Change (2001a) *Climate Change 2001: The Scientific Basis.* Intergovernmental Panel on Climate Change, Cambridge University Press, Cambridge.

Intergovernmental Panel on Climate Change (2001b) *Climate Change 2001: Impacts Adaptation and Vulnerability. Summary for Policymakers.* Intergovernmental Panel on Climate Change, Cambridge University Press, Cambridge.

McCarty, J.P. (2001) Ecological consequences of recent climate change. *Conservation Biology* **15**: 320-326.

Parmesan, C. and Yohe, G. (2003) A globally coherent fingerprint of climate change impacts across natural systems. *Nature* **421**: 37-42.

Pearson, R.G. and Dawson, T.P. (2004) Modelling long-distance plant dispersal across fragmented landscapes. In preparation.

Pearson, R.G., Dawson, T.P. and Lui, C. (2004) Modelling species distributions in Britain: a hierarchical integration of climate and land-cover data. *Ecography:* in press.

Pearson, R.G., Dawson, T.P., Berry, P.M. and Harrison, P.A. (2002) SPECIES: a spatial evaluation of climate impact on the envelope of species. *Ecological Modelling* **154**: 289–300.

Penman, H.L. (1948) Natural evaporation from open water, bare soil, and grass. *Proceedings of the Royal Society, London Series A* **193**: 120-146.

Rietveld, M.R. (1978) A new method for estimating the regression coefficients in the formula relating solar radiation to sunshine. *Agricultural Meteorology* **19**: 243-252.

Thomas, C.D., Cameron, A., Green, R.E., Bakkenes, M., Beaumont, L.J., Collingham, Y.C., Erasmus, B.F.N., Ferreira de Siqueira, M., Grainger, A., Hannah, L., Hughes, L., Huntley, B., van Jaarsveld, A.S., Midgley, G.F., Miles, L., Ortega-Huerta, M.A., Peterson, A.T., Phillips, O.L. and Williams, S.E. (2004) Extinction risk from climate change. *Nature* **427**: 145-148.

Deer as vectors of plant dispersal in woodland networks

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Abstract

Dispersal is a key process in fragmented landscapes. Deer and lagomorph species may have a significant role in the creation and development of plant assemblages of a range of forest habitats, as they deposit viable seeds of a number of species in their faecal pellet groups. We present data from a large study of the viable seed content of herbivore faecal matter. 84 vascular plant species were recorded, including seeds with a variety of physical dispersal mechanisms. Herbivore functional groups may differ in their role in the dynamics of forest plant assemblages. Large-bodied grazing deer had a greater seed and species content per pellet group, and dispersed a greater number of species, than browsing deer or lagomorphs. Faecal deposition rates and population density estimates are used to estimate the seed rain from the different deer functional groups; roe deer (*Capreolus capreolus*) deposit the most seeds per square metre in our study area.

Introduction

Dispersal has an applied importance in forest ecosystem restoration. Negative impacts of forest loss and fragmentation on biodiversity are well established (Harrison and Bruna, 1999), however less well understood are the processes by which species composition responds to the restoration of forest cover. Persistence, extirpation, colonisation and dispersal all play important roles in the restructuring of local species assemblages upon the addition or alteration of wooded landscape components (Dolman & Fuller, 2003). Dispersal allows the flow of individuals and genes between suitable patches in mosaic landscapes.

In temperate Europe, much research has focused on plant species' colonisation ability and the re-creation of species assemblages of deciduous semi-natural woodland, both in the UK (e.g., Peterken and Game, 1984; Ratcliffe and Peterken, 1995; Dolman and Fuller, 2003) and continental Europe (e.g., Dzwonko, 2001; Brunet and von Oheimb, 1998; Verheyen and Hermy, 2001). The main conclusion of this body of work is that woodland species are poor dispersers and colonisers.

Plant species exhibit a wide range of mechanisms for seed dispersal, a number taking advantage of the movement of animals and of air. The existence of such a mechanism in a given species is often inferred from the structure of the diaspore (the seed and any attached part), for example seeds with plumes or wings, hooks or awns attached, or with fleshy pulp as the reward for swallowing the seeds inside. However, a number of species have seeds with no physical adaptation for dispersal.

Many plants have seeds that could be swallowed accidentally by grazing herbivores, the foliage of the plant taking place of fleshy pulp (Janzen, 1984). The seeds may then pass through the digestive system and be deposited viable in faecal matter. It is less obvious which seeds have this potential dispersal mechanism, and not much is known about seeds which can survive ingestion and passage through the digestive system of herbivores (Pakeman *et al.*, 2002). However, Janzen (1984) suggested that small, rounded seeds are more likely to avoid being crushed between the teeth, and seeds need a resistant coat to avoid being destroyed by digestive enzymes.

Deer are the main species of wide-ranging, large herbivores in the UK and are therefore potentially major vectors for dispersal and colonisation of a wide range of plant species into newly created woodlands. Deer populations are increasing through temperate Europe (Fuller and Gill, 2001) and North America (Rooney, 2001). Muntjac (*Muntiacus reevesi*) are of particular interest as they are introduced in the UK (Chapman *et al.*, 1994). Over half the current wooded area of the UK has been planted in the last 200 years, and it is current UK government policy to continue to increase woodland area (Forestry Commission, 1998). New woodlands will include those managed primarily for conservation and amenity, and also economic forestry. We present results from a large study of viable plant seeds in deer faeces, in pellet groups taken from a commercial conifer forest in Eastern England, UK.

Methods

Study area

Thetford Forest (0° 40' E, 52° 27' N) is the largest lowland conifer forest in the UK, being 200 km^2. The forest occupies approximately one third of Breckland, an area of East Anglia characterised by poor, chalk-sand drift based soils, and a more continental climate compared to the rest of the British Isles (Sheail, 1979). The forest is managed primarily for timber production, although recreation and amenity, biodiversity and conservation are all considered important and included in the management plan (Forest Enterprise East Anglia, unpublished).

Thetford Forest is not one contiguous area but seven separate blocks (mean area 1,024 ha ± 586 s.d.) and a main central section (10,563 ha). Most of the forest is managed as a *Pinus* spp. clearfell rotation, creating a successional mosaic with the highest plant species richness in recently replanted stands (mean species richness of stands 1-5 years is 14.6 ± 6.1 s.d., Eycott *et al.* unpublished data). The ride network covers approximately 15 % of the total forest area, and a further 2 % is other open space. (Richard Brook, personal communication).

Thetford Forest was planted in the early 20th century, on grazed grass heath and low-intensity arable land, during a period of agricultural depression and low land values (Sussams, 1996). Extensive sheep grazing, and intensive rabbit warrening from the thirteenth century (Sheail, 1971) led to the development in many areas of lichen- and bryophyte-rich lowland heath of considerable historical and biodiversity importance (Watt, 1971; Dolman and Sutherland, 1992). The forest retains a range of species characteristic of these open ground habitats in both managed stands (particularly recently replanted areas) and ride margins (Eycott *et al.*, in prep.). The area of heath remaining in Breckland is small (Ratcliffe, 1977), and farming is characterised by extensive permanent arable fields. Other than coniferous plantations, there is very little woodland in the Breckland. The nearest ancient semi-natural woodland is Wayland wood, which is outside the boundary of the Breckland Environmentally Sensitive Area and 20 km from Thetford Forest.

There are six species of mammalian herbivore (excluding rodents) in Thetford Forest, and pellet groups were collected from each of these. Deer are present in high numbers: approximately 300 red deer (*Cervus elaphus*) and 400 fallow deer (*Dama dama*) (Trevor Banham personal communication), 5,200 roe deer and 11,900 muntjac (Hemami 2003). Hare (*Lepus europaeus*) are also present throughout the forest, and rabbit (*Oryctolagus cuniculus*) are abundant in localised populations. We classified the six species into three functional groups: Grazing deer (red and fallow) and browsing deer (roe and muntjac) after Hoffmann (1985), and lagomorphs (hare and rabbit).

Faecal pellet survey

One thousand, one hundred and twenty one pellet groups were collected, from *Pinus* stands of a range of ages and areas of the forest, in June, August and October 2002. Only fresh droppings were collected, to reduce decay and contamination by seed rain. A deer dropping pile was considered to be fresh if it was moist and green inside with no hollowing caused by desiccation. Rabbit and hare droppings were considered fresh if they were brown or green not grey, did not disintegrate easily and had no visible fungal growth.

Each pellet group sample was air dried for ten days in a paper packet, weighed to the nearest 0.01 g, and then stratified in the dark at 0-1°C for one month. Each sample was crushed carefully, mixed thoroughly with approximately the same amount of Levington multi-purpose compost, and laid on top of more compost in pots or trays according to the size of the sample (grazer samples in 15 cm x 20 cm trays, browser samples in 8 cm diameter pots, and lagomorph samples in 6 cm diameter pots). Samples were kept for 10 months in a greenhouse, at 20°C with a 16-hour daylength provided by artificial lighting. Seedlings were identified, counted and removed, as soon as they were large enough. Twelve controls, 30 cm x 20 cm seed trays containing the same compost, were laid out in amongst the samples and treated in the same manner.

Results

We recorded 10,950 seedlings, of 112 different plant species. Of these, 2,350 seedlings from 23 species had plumes on their seeds and could have contaminated the samples via the greenhouse ventilation system. Another five species were recorded as compost contaminants in the control trays. Our subsequent analysis in this paper considers only those 84 remaining species that were not recorded in the controls and did not have plumes on their seeds.

Approximately half of all seedlings recorded were of three species: fat hen (*Chenopodium album)* is an arable weed uncommon in Thetford Forest, stinging nettle (*Urtica dioica)* is very abundant in the forest, and creeping bent grass (*Agrostis stolonifera)* is one of the most dominant grasses in the forest (Eycott, unpublished data). In contrast, 78 % of species recorded occurred in fewer than 1 % of samples, and 48 % of plant species recorded produced ten or fewer seedlings.

Of the 84 species recorded from pellet groups, 72 have been recorded in surveys of the ground vegetation of coniferous stands and ride margins in Thetford Forest, out of a total of

247 vascular plant species recorded in those habitats (Eycott, unpublished data). Nineteen of the 84 species are present in Wayland Wood, the nearest ancient semi-natural woodland; of these 19, 17 are already present in Thetford Forest. Forty-three of the 84 plant species have been recorded on Breckland heaths by Dolman and Sutherland (1992) or Dolman and Lake (unpublished report to English Nature, 1995). Forty-seven of the 84 plant species have been recorded on arable headlands and wildlife strips in the region (ADAS, 1997).

Species not recorded in Thetford Forest included 2 common crops (*Zea mays* and *Solanum tuberosum*), a number of common arable weeds and escapes, a few heath species (including *Erica tetralix*), and a woodland shrub, *Rhamnus catharticus*.

Seeds with a range of different physical dispersal mechanisms were recorded (Fig. 1). Lagomorph samples contained the greatest proportion of seeds with no physical dispersal mechanism ($\chi^2_1 = 11.02$, $P = 0.001$), and very few seeds with wings, berries or nuts, or an alternative dispersal mechanism. Browser samples contained the greatest proportion of seeds from species with berries or nuts (non-significant). Thirty-four species recorded had seeds with no physical adaptation for dispersal.

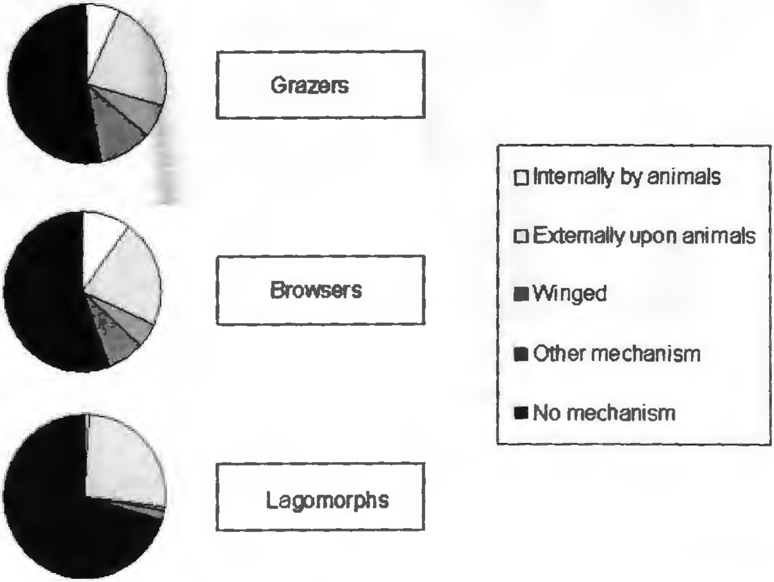

Figure 1. The proportion of the total number of seeds recorded in samples from each herbivore functional group that had different physical dispersal mechanisms. 'Other mechanism' includes seeds in an explosive capsule, seeds with an elaiosome to attract ants, and seeds in capsules which are scattered in the wind.

Seed content from different herbivore functional groups

Samples from grazers (red and fallow deer) had the highest mean number of species and seedlings per pellet group (Fig. 2a), the greatest overall diversity and total species richness (Fig. 2b), and contained 82 of the 84 species recorded. Lagomorphs had the lowest mean number of seedlings per group and species per group, species richness and diversity. In each of the herbivore groups, a few plant species were very abundant, and many plant species were infrequent.

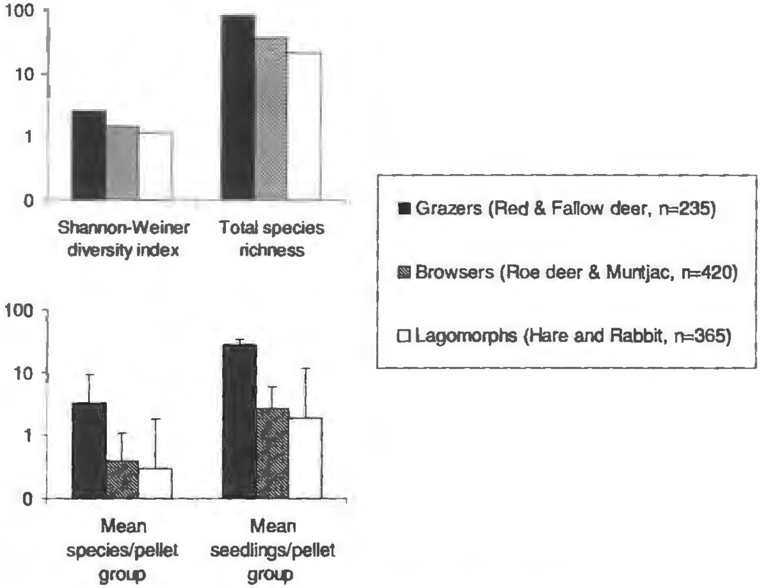

Figure 2. **a)** Overall species diversity and richness, and **b)** mean (± 1 s.d.) species and seedlings per pellet group, of pellet groups from different herbivore functional groups.

Deposition rates

Measures of deer population density and faecal deposition rate (Mayle *et al.*, 1999) were used to estimate seed rain (Table 1). Despite having the highest number of seeds per pellet group, grazers deposit fewer seeds in the forest overall due to their lower population size. Roe deer deposit more seeds per square metre than muntjac, as they defecate more often and have a higher seed number per pellet group.

Table 1. Estimated seed input per square metre by different deer species. [1] Banham, personal communication, [2] Hemami 2003.

Species	**population size**	**mean seeds/ m^2/ June-October**
Red and fallow deer	700[1]	**0.15**
Roe deer	5200[2]	**0.38**
Muntjac	11,900[2]	**0.11**

Discussion

At least 84 plant species are dispersed internally by herbivores in Thetford Forest. Species dispersed in this manner have the potential to travel long distances and colonise habitats that they may not have reached by other means of dispersal. A large number of the total plant species recorded have no physical mechanism for dispersal, and large herbivores may therefore play a vital role in the meso-scale dispersal of seeds of woodland species. The mean seed input from deer pellet material amounts to approximately 6,500 seeds per hectare in Thetford Forest, and this should be an adequate input to allow plant species to colonise newly planted woodland stands.

Differences between the composition of viable seeds of pellet groups from different herbivore species may be related to herbivore behaviour and physiology. The grazing deer species (red and fallow), which are larger, feed mostly on ground vegetation and consume a large amount of material. They also range over quite a large area. Browsing deer (roe and muntjac) are smaller bodied, concentrate-selective feeders, browsing on small amounts of high-quality food items such as shoots of shrubs like bramble (*Rubus* spp.) (Harris & Forde, 1986; Hemami *et al.*, in press). They are territorial; muntjac home ranges are in the order of 25 ha for bucks and 12 ha for does, roe home ranges vary throughout the year peaking at around 80 ha for bucks and 120 ha for does (Chapman *et al.*, 1993). This, combined with the low seed content of browsing deer pellet groups, means that grazing deer may have a more significant role in the long-distance dispersal of species. The lagomorph species browse ground vegetation; unfortunately no estimates of population size were available.

Comparison to other studies suggests that the plant species pool available has an effect on both plant species richness and seedling density of pellet groups. Red deer and rabbit pellet groups analysed from Mediterranean dehésa had a higher species richness and seedling density than in this study (Malo & Suarez, 1995). In contrast, pellet groups from Scottish grazed highland, which has a more species poor flora than Breckland conifer forest or dehésa, have a lower seedling density (Welch, 1985). The only other study in the Breckland was of rabbit pellets, and these pellets contained a greater overall number of plant species than from our study, despite being from a smaller area with lower overall ground flora species richness (Pakeman *et al.*, 1999).

Deposition of seeds by herbivores may affect vegetation dynamics on a number of scales (Malo & Suarez, 1995). Seeds could be moved between stands within one forest block, from rides to stands, between forest blocks or in or out of the forest. The different species of deer are not evenly distributed in Thetford Forest; for example some areas have little or no red or fallow deer. On a smaller scale, deposition is likely to introduce heterogeneity to the vegetation of individual stands, as the seeds are introduced in concentrated amounts, in discrete piles. We have shown that deer potentially introduce species into recently felled stands. To confirm that colonisation of stands after felling is affected by dispersal of species in faecal pellet material, one would have to remove pellets from newly felled stands and assess whether this affected stand species richness.

Deer are commonly seen as a constraint on conservation aspects of forestry (Fuller and Gill, 2001). In British broadleaved woodland they inhibit regeneration (Fuller and Gill, 2001), and browsing deer species reduce understorey structural diversity by selecting *Rubus* spp. and non-bracken ferns (Kirby, 2001). Deer may also affect the growing tree crop by bark-stripping (Putman and Moore, 1998). In contrast, the effects of seed dispersal by deer upon vegetation dynamics and plant species' regional population structure are poorly understood in British deciduous woodlands. We have shown that deer in coniferous forest landscapes disperse a large number of plant species including many of woodland habitats. Deer are therefore potentially important in the colonisation of new patches of a variety of habitat types, and may play a role in meso-scale and even large-scale, long-distance dispersal of vascular plant species.

Acknowledgments

We thank Forest Enterprise (East Anglia District) for access to Thetford Forest, extensive assistance and advice. We also thank NWT for Wayland Wood data, Dave Alden for technical support, and Mahmoud-Reza Hemami for advice on identifying deer pellet groups. AEE was funded by NERC, and is grateful to a number of people for assistance in the field.

References

ADAS (1997) *Biological monitoring of arable field margins in the Breckland ESA 1989-1996.* Unpublished report to MAFF.

Brunet, J. & von Oheimb, G. (1998) Migration of vascular plants to secondary woodlands in southern Sweden. *Journal of Ecology* **86:** 429-438.

Chapman, N.G.; Claydon, K.; Claydon, M.; Forde, P.G. & Harris, S. (1993) Sympatric populations of muntjac (*Muntiacus reevesi*) and roe deer (*Capreolus capreolus*): a comparative analysis of their ranging behaviour, social organization and activity. *Journal of Zoology* **229:** 623-640.

Chapman, N.; Harris. S. & Stanford, A. (1994) Reeves' muntjac *Muntiacus reevesi* in Britain: their history, spread, habitat selection, and the role of human intervention in accelerating their dispersal. *Mammal Review* **3:** 113-160.

Dolman, P.M. & Fuller, R.J. (2003) The processes of species colonisation in wooded landscapes: a review of principles. Eds. J. Humphrey, A. Newton, J. Latham, H. Gray, K. Kirby, E. Poulsom, & C. Quince, *Restoration of wooded landscapes.* 25-36.

Dolman, P.M. & Sutherland, W.J. (1992) Ecological changes of Breckland grass heaths and consequences of management. *Journal of Applied Ecology* **20:** 402-413.

Dzwonko, Z. (2001) Effect of proximity to ancient deciduous woodland on restoration of the field layer vegetation in a pine plantation. *Ecography* **24:** 198-204.

Forestry Commission (1998) *The UK Forestry Standard.* Forestry Commission, Edinburgh.

Fuller, R.J. & Gill, R.M.A. (2001) Ecological impacts of increasing numbers of deer in British woodland. *Forestry* **74:** 193-200.

Harris, S. & Forde, P. (1986) The annual diet of muntjac (Muntiacus reevesi) in the King's Forest, Suffolk. *Bulletin of the British Ecological Society* **17:** 19-22.
Harrison, S. & Bruna, E. (1999) Habitat fragmentation and large-scale conservation: what do we know for sure? *Ecography* **22:** 255-232.
Hemami, M.-R. (2003) *The ecology of roe deer (Capreolus capreolus) and muntjac (Muntiacus reevesi) in a forested landscape in eastern England.* Unpublished PhD Thesis, University of East Anglia, Norwich.
Hemami, M.-R.; Watkinson, A.R. & Dolman, P.M.; (In Press) Habitat selection by sympatric muntjac (Muntiacus reevesi) and roe deer (Capreolus capreolus) in a lowland commercial pine forest. *Forest Ecology and Management.*
Hoffmann, R.R. (1985) Digestive physiology of deer - their morphophysiological specialisation and adaptation. *The Royal Society of New Zealand Bulletin* **22:** 393-407.
Janzen, D.H. (1984) Dispersal of small seeds by big herbivores: foliage is the fruit. *American Naturalist* **123:** 338-353.
Kirby, K.J. (2001) The impact of deer on the ground flora of British broadleaved woodland. *Forestry* **74:** 219-230.
Malo, J.E. & Suarez, F. (1995) Herbivorous mammals as seed dispersers in a Mediterranean dehésa. *Oecologia* **104:** 246-255.
Mayle, B.A.; Peace, A.J. & Gill, R.M.A. (1999) *How many deer? A field guide to estimating population size.* Forestry Commission, Edinburgh.
Pakeman, R.J.; Digneffe, G. & Small, J.L. (2002) Ecological correlates of endozoochory by herbivores. *Functional Ecology* **16:** 296-304.
Pakeman, R.J.; Engelen, J. & Attwood, J.P. (1999) Rabbit endozoochory and seedbank build-up in an acidic grassland. *Plant Ecology* **145:** 83-90.
Peterken, G.F. & Game, M. (1984) Historical factors affecting the number and distribution of vascular plant species in the woodlands of central Lincolnshire. *Journal of Ecology* **72:** 155-182.
Putman, R.J. & Moore, N.P. (1998) Impact of deer in lowland Britain on Agriculture, Forestry and Conservation Habitats. *Mammal Review* **28:** 141-164.
Ratcliffe, D. (1977) *A Nature Conservation Review.* Cambridge University Press, Cambridge.
Ratcliffe, R.R. & Peterken, G.F. (1995) The potential for biodiversity in upland spruce forests. *Forest Ecology and Management* **79:** 153-160.
Rooney, T.P. (2001) Deer impacts on forest ecosystems: a North American perspective. *Forestry* **74:** 201-228.
Sheail, J. (1979) Documentary evidence of the changes in the use, management, and appreciation of the grass-heaths of Breckland. *Journal of Biogeography* **6:** 277-292.
Sheail, J. (1971) *Rabbits and their history.* David & Charles, Newton Abbot.
Sussams, K. (1996) *The Breckland Archaeological Survey.* Suffolk Archaeological Service, Bury St. Edmunds.
Verheyen, K. & Hermy, M. (2001) The relative importance of dispersal limitation of vascular plants in secondary forest succession in Muizen Forest, Belgium. *Journal of Ecology* **89:** 829-840.
Watt, A.S. (1971) Rare species in Breckland - Their management for survival. *Journal of Applied Ecology* **8:** 593-609.
Welch, D. (1985) Studies in the grazing of heather moorland in north-east Scotland. IV. Seed dispersal and plant establishment in dung. *Journal of Applied Ecology* **22:** 461-472.

Session 2

Responses of animals to landscape-scale processes

The bird communities of small woods: the influence of landscape context and changes in woodland structure

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Abstract

Complete bird censuses of 56 small woods, in an arable landscape, over 13 years have provided an insight into how woodland structure and the nature of the surrounding landscape affect changes in breeding bird species over time. Increasing woodland area by planting new trees adjacent to small woods led to an increase in the numbers of breeding pairs in the original woods and to a smaller increase in the numbers of breeding species. The size of these increases was more strongly related to the length of edge between the original and newly planted woods, than to the change in area. Vegetation changes in woods due to the maturation of shrubs and young trees, along with the nature of the surrounding landscape and increases in regional bird populations, influenced changes in species richness in individual woods. Thus, optimising the value of new farm woodland for birds requires different planting strategies in different landscapes.

Introduction

In many studies, woodland area was the main factor influencing the number of breeding bird species found in a wood, as reviewed by Andrén (1994). It has also been recognised that small woods support fewer woodland dependent bird species and have a greater proportion of woodland edge that is more likely to attract species using the surrounding landscape (e.g., Hinsley *et al.*, 1995; Bellamy *et al.*, 1996). Thus, larger woods are often seen as more desirable and the planting of new woodland adjacent to existing woods has been proposed to aid colonisation of new sites by woodland wildlife (Woodland Trust, 2001; defra, 2003), and to increase abundance of game species (Game Conservancy Trust, 2003). Young farm woods provide habitat for a number of open country and generalist woodland bird species which are also common in hedges and gardens (Fuller *et al.*, 2001; Vanhinsbergh *et al.*, 2002). However, it may take 50-100 years for woodland to provide the requirements of more specialist woodland bird species. Between these two extremes, we can ask the question, does newly planted woodland adjacent to an existing wood provide any shorter term benefits for woodland birds, by adding new habitats or by acting as a buffer between the wood and surrounding habitat? If this is the case, then we might expect the number of breeding pairs to increase in the original wood, in addition to increases due to new pairs or new species found in the new woodland.

Woodland area is not the only factor affecting breeding species in small woods (Hinsley *et al.*, 1995; Bellamy *et al.*, 1996). The structure of a wood within the context of its surrounding landscape will influence its suitability for species with different ecological requirements (Bennett *et al.,* 2004), and will interact with regional and national changes in bird species distributions. At low population levels, the most suitable woods are likely to be occupied preferentially, but as populations increase, these woods will become saturated and the occupation of poorer quality woods will increase (e.g., Bellamy *et al.,* 2000). Nationally, many common woodland and garden bird species, which comprise most of the avifauna of small woods, have been increasing in recent years, whereas more than half of the open-country species have been declining (Raven *et al.,* 2003). A similar pattern was found across all of our study woods; for 15 of the most frequently occurring species, the populations of six were stable, eight were increasing and only one was declining. By combining information about how birds use new woodland planting and what characteristics of the wood and the landscape are optimal for different ecological groups of birds, we can start to provide guidelines to maximise the potential of new woodland plantings for birds.

Methods

The study area, approximately 55 km from north to south and 35 km east to west, was mainly in the county of Cambridgeshire in eastern England, centred on CEH Monks Wood (co-ordinates 52°24'N 0°14'W). The area is poorly wooded, with 3% woodland cover recorded in the whole of Cambridgeshire (Forestry Commission, 2002) and the land-use is dominated by intensive arable farming (73% of area). Within this area, we censused 56 small woods (size range 0.1-9.0 ha) annually, between 1990 and 2002.

The breeding bird populations of the study woods were censused using a territory mapping technique with four visits to each wood in each breeding season (see Hinsley *et al.,* 1995, for details). To look at factors influencing species richness and changes in it, we excluded game birds and most corvids because their populations were strongly influenced by game management. Nocturnal species and water birds were not censused efficiently by this method and were also omitted. The remaining 47 species which bred in at least one year were split into three ecological groups: (i) woodland dependent species - reliant on woodland for breeding (18 species); (ii) ubiquitous species - common woodland species, which also breed in other habitats such as hedges and gardens (9 species); and (iii) open-country species, which mostly use open habitats such as field boundaries, grassland, and arable fields, but often nest in small woods (20 species).

Seven small woods (0.1-1.2 ha, mean 0.5 ha) had extensive areas of trees (small saplings) planted adjacent to them in one year in the period 1990 to 1992. This increased the area of these woods to 0.6-2.5 ha (mean 1.4 ha). By 2002 most of this additional woodland had grown sufficiently to produce a closed canopy at 4-8 m. We paired each of these woods with another in which no major change had occurred over the same time period. Woods were paired to be similar in size and structure and to have a similar amount of woodland in the surrounding landscape. Bird abundance was measured using the sum of the numbers of pairs of the ten most frequent territorial species in these woods: wren, *Troglodytes troglodytes*; blackbird, *Turdus merula*; chaffinch, *Fringilla coelebs*; robin, *Erithacus rubecula*; dunnock, *Prunella modularis*; blue tit, *Parus caeruleus*; great tit, *Parus major*; blackcap, *Sylvia atricapilla*; whitethroat, *Sylvia communis*; and willow warbler, *Phylloscopus trochilus* (in decreasing order of abundance). In woods with new plantings in 2002, territories which spanned the boundary between the old and new area were counted as 0.5. We compared bird abundance in the years 1990 with 2002 in the seven woods with additional plantings and also in the seven woods with no change in area. We used the full list of 47 species to compare species richness in the same way, i.e., between years for the two sets of seven woods. Comparisons were made using paired t tests.

Bird species richness within woods may also change over time in response to more subtle changes in structure and landscape context. Thus, changes in species richness from 1990 to 2002 were also investigated using the 49 of the 56 censused woods in which no significant alteration in the extent of woodland cover had occurred, e.g., no large areas of new planting adjacent to a wood or the clearing of a relatively large area of scrub or trees. For this analysis, we used the mean number of breeding species for the first three years of the study (1990-92) and for the last three years (1999-2002, a full census was not possible in 2001 due to foot and mouth disease restrictions and hence this year was omitted) to even out short term changes between individual years. The influence of landscape and woodland structure on species richness, and on changes in species richness for the different ecological groups, was explored using stepwise multiple regression and the variables summarised below. In the models for change in species richness, explanatory variables which changed between the two recording periods were included as the initial value for 1990-92 and the change in value by 1999-2002.

Variables describing the attributes of each wood and the landscape surrounding each wood are described in full by Hinsley *et al.* (1995). Briefly, the woodland attributes were: area (ha); perimeter (m); shape; mean canopy; shrub and field layer densities for 1990-92; and the number of structural features (structural diversity). Canopy, shrub and field layer densities were assessed annually and the structural features present were also assessed in the first three years and again in 2000. The variables used for landscape context were in three groups: (i) distances to the nearest wood of any size (dist. nearest), the nearest woods > 2 ha (dist. 2 ha) and > 10 ha (dist. 10 ha) and the nearest village (dist. village); (ii) the area of woodland within 0.5 km (wood 0.5 km) and 1 km (wood 1 km) and the length of hedgerow within 0.5 km (hedge 0.5 km) and 1 km (hedge 1 km); (iii) the numbers of hedges (hedge), ditches (ditch), tracks (track) and total linear features (linear connect.) connected to a wood.

Results

For the seven pairs of small woods, woods with adjacent new trees had on average four more pairs breeding in 2002 (mean = 10.9 pairs) than they had in 1990 (mean = 6.9 pairs) (t = 4.26, p = 0.005, n = 7); the woods with no change in area showed no change in numbers of pairs (1990 mean = 7.6 pairs, 2002 mean = 7.7 pairs, t = -0.76, p >0.10, n = 7) (Figure 1). This comparison was for the birds breeding in the original part of the wood only, i.e., excluding the new planting, but included territories shared between the old and new woodland with a value of 0.5.

There was no relationship between the increase in the numbers of pairs with any measures of change in the size of the wood or the size of the original wood. However, the relationship between the increase in numbers of breeding pairs and length of common edge, showed a clear outlier (Figure 2). This wood in addition to the new planting, had an organic waste recycling plant, including large scale compost heaps, established next to it in the mid 1990's. When this outlier was removed there was a significant relationship between the increase in numbers of breeding pairs in the original wood and the length of common edge (log pairs 2002 = 0.09 + 5.51 log pairs 1990 + 0.60 log edge length, r^2 = 0.94, p = 0.007, n =

6). The increase in the numbers of breeding pairs continued to show no relationship with either the area or the perimeter of the original wood or the area of newly planted wood.

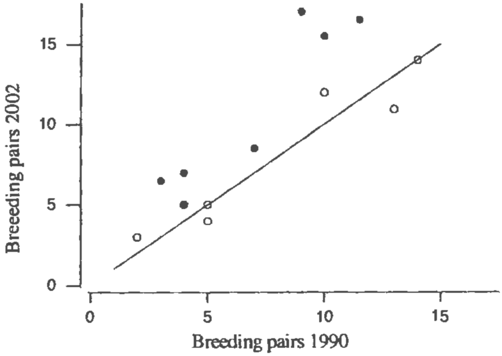

Figure 1. Numbers of breeding pairs in 2002 plotted against the numbers of breeding pairs in 1990. Sites with no change in woodland area are shown by open symbols and sites with new plantings by solid symbols. The line shows the relationship of no change between years.

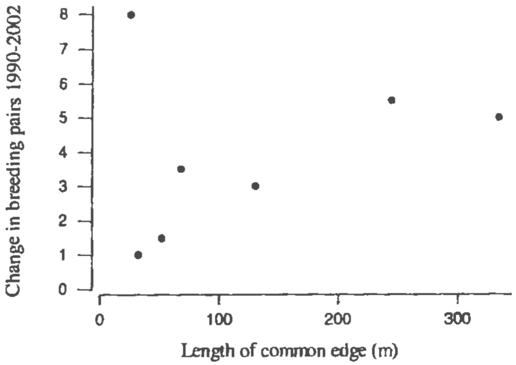

Figure 2. The change in the numbers of breeding pairs between 1990 and 2002 in relation to the length of common edge between the original wood and the new planting.

As with the numbers of pairs, there were larger increases in the numbers of species between 1990 and 2002 for the seven small woods with adjacent new plantings (1990 mean = 6.3 species; 2002 mean = 8.0 species; $t = 2.30$, $p = 0.061$, $n = 7$) than those with no change in area (1990 mean = 7.6 species; 2002 mean = 8.4 species; $t = 1.55$, $p > 0.10$, $n = 7$).

In the 49 woods without major changes in woodland area between 1990 and 2002, wood attributes were dominant in explaining the numbers of breeding woodland and ubiquitous species, whereas landscape characteristics were dominant in explaining the numbers of open-country species. There were more woodland species in larger woods and in woods with denser shrub layers; however more linear connections reduced the numbers of species (Table 1). Numbers of ubiquitous species were also strongly related to woodland area, with more species in larger woods and in woods with more hedges in the surrounding landscape (Table 1). There were more open-country species in more isolated woods, and in woods with a higher structural diversity, with denser field layers and more linear connections having smaller influences on numbers of species (Table 1).

Table 1. Wood attributes and characteristics of the surrounding landscape which significantly influenced bird species richness and changes in species richness in stepwise multiple regression models. Directions of relationships are indicated by +/-.

	Wood attributes	Surrounding landscape	Total variance explained by the model (r^2)
Numbers of woodland species 1990-92	+ log area *** + shrub density ***	- linear connect. *	0.87
Change in numbers of woodland species between 1990-92 and 1999-2002	- shrub density ***	- dist. 10 ha*	0.20
Numbers of ubiquitous species 1990-92	+ log area ***	+ hedge 0.5 km**	0.63
Change in numbers of ubiquitous species between 1990-92 and 1999-2002	- log area **	- hedge 1 km *	0.27
Numbers of open-country species 1990-92	+ structural diversity *** + field density *	+ dist. 10 ha *** - wood 0.5 km * + linear connect.*	0.57
Change in numbers of open-country species between 1990-92 and 1999-2002	+ perimeter*	- dist. 10 ha *** + dist. nearest** - wood 1 km *	0.31

Statistical significance: *** $p \leq 0.001$; ** $p \leq 0.01$; * $p \leq 0.05$.

There were significant changes in the mean numbers of breeding species between the two periods with all species groups increasing, but ubiquitous species showed the largest increases (Table 2). The features of the woods recorded during both time periods showed some changes. Structural diversity increased and there was a small shift in the vegetation densities in different layers, as young trees and scrub matured to provide greater canopy closure, which reduced field layer densities as shading increased (Table 2). There were few changes in numbers of connections to the woods.

Table 2. Changes in the mean numbers of species and in woodland structure between 1990-92 and 1999-2002. The statistical significance of comparisons between years was assessed by paired t tests, $n = 49$.

	1990-92	1999-2002	t	p
Woodland species	4.8	5.2	1.96	0.056
Ubiquitous species	6.3	7.0	5.27	< 0.001
Open species	2.0	2.4	2.70	0.009
Canopy density	1.52	1.65	1.98	0.053
Shrub density	2.43	2.35	-1.47	> 0.10
Field density	2.93	2.67	-4.36	< 0.001
Structural diversity	4.6	5.1	2.99	0.004

The different ecological groups showed different responses to population change over the study period in relation to wood attributes and characteristics of the surrounding landscape. The strongest increase in woodland species numbers occurred in woods with a thinner shrub layer, but there was also an increase in woods closest to woods larger than 10 ha (Table 1). Although the initial density of the shrub layer gave the strongest relationship with the change in the numbers of woodland species, the change in shrub density also gave a weak relationship on its own ($r^2 = 0.06$, $p = 0.045$). However, these relationships depended on two woods; one initially had dense shrub which then thinned out naturally, and one had very little shrub layer in the early 1990's, but scrubbed up later. Removal of these two woods leaves no relationship of change with any variable. Woodland species also showed the weakest increase of all of the groups. It appeared that in most cases, species were occupying woods that were initially vacant but similar in character to those already occupied. The change in the numbers of ubiquitous species led to the increased occupation of woods of less than 1 ha, but increases also occurred in woods with fewer hedgerows in the landscape (Table 1). Changes in open-country species were largely influenced by the nature of the surrounding landscape, with greatest increases in woods closer to woods larger than 10 ha, and with few smaller woods in the surroundings (Table 1). Woods with longer perimeters also showed an increase in open-country species.

Discussion

Young woodland during the first few years of growth provides habitat for a variety of open-country bird species and woodland generalists (Vanhinsbergh *et al.*, 2001). However, once succession leads to canopy closure, habitat structure becomes unsuitable for open-country species, whilst not yet providing adequately for woodland specialists requiring features such as mature trees with nest holes, rotten wood, more varied bark surfaces and a greater abundance of canopy foliage. At this earlier stage of growth, the new woodland provides breeding habitat for some generalist woodland birds. In our small woods, it provided additional habitat for species originally breeding in the older wood. The extra habitat allowed for a greater density of breeding pairs in the older wood, with territories on the edge of the original wood also encompassing some of the new planting.

The increase in the numbers of species in woods with new plantings adjacent to them was largely due to the addition of ubiquitous species, particularly dunnock, robin and chaffinch, but also a few woodland species. In contrast, any increase in the numbers of species in the control woods, which was due to ubiquitous species only, was not dominated by any particular species. The new woodland also added further species not found in the original wood. These were mostly woodland species associated with early successional woodland or dense shrub cover, i.e., willow warbler, blackcap, song thrush *(Turdus philomelos)*, and long-tailed tit *(Aegithalos caudatus)*. Therefore adding new woodland to existing small woods increased the abundance and species richness of ubiquitous breeding birds in the existing wood and also added new woodland species indicative of young woodland. We did not find any increase in the numbers of open-country species, e.g., whitethroat, linnet *(Carduelis cannabina)* and yellowhammer *(Emberiza citrinella*, which were found to be a major component of the avifauna in other young farm woods (Vanhinsberg *et al.*, 2002). This may have been due to the generally low population densities of these species in our study area.

As has been discussed elsewhere, smaller isolated woods were more attractive to open-country species, but less so for woodland species (Hinsley *et al.*, 1995; Bellamy *et al.*, 1996). Within the wider survey of 49 woods with no change in area, the numbers of ubiquitous species were highest in larger woods, but the change in the number of species was dominated by increasing numbers in small woods of less than 1 ha. Similarly, but less pronounced, more woodland species were found in woods with a denser shrub layer, yet numbers increased the most in woods with a thinner shrub layer. Thus there was an increase in the occupation of less favoured woods as bird population sizes have increased. The models for open-country species are more difficult to interpret. Changes in numbers appeared to be more influenced by the surrounding landscape (as might be expected) than by attributes of the woods, although the positive relationship with perimeter may be related to the use of woodland edge for nesting.

As well as the size and shape of a wood, both local and regional landscape context can affect numbers of breeding woodland and open-country species (Bennett *et al.*, 2004). Woodland species are more likely to be found in both larger woods and in woods in more wooded landscapes. A more wooded landscape should support larger regional populations and thus also larger numbers of potential colonists. A higher density of woodland in the landscape should also aid dispersal between woods, facilitating colonisation. Although isolation did not influence the numbers of breeding woodland species in this data set, there was an indication that more isolated woods were being colonised as populations increased. There are indications that dispersal may be a problem for some individual species in our study area (e.g., Hinsley *et al.*, 1995; Bellamy *et al.*, 2003).

Conclusions

New woodland planting adjacent to existing woods clearly benefits ubiquitous species by increasing abundance and the numbers of breeding species. Planting to produce a long edge between the new wood and existing wood will maximise the benefits for these common and widespread species. New plantings adjacent to existing small woods also slightly increased the numbers of woodland species in the original wood and added new woodland species in the recently planted area. However, all of these were small woods of less than 2 ha and therefore, in the absence of effective dispersal, were unlikely to provide habitat for sustainable populations of woodland dependent species in the long term. New plantings such as this are most likely to benefit woodland species if they occur in well wooded landscapes or where there is a large wood to act as a stable population source. These small woods can then supplement existing populations and increase functional connectivity between woods. New plantings adjacent to existing small woods did not provide any appreciable differences in the numbers of open-country species. In very open arable landscapes where existing woods are mostly small (<1 ha), specialist woodland species populations will be sparse or absent and these small woods will have fewer ubiquitous species. In this situation, more benefit may be derived from planting separate small woods which may attract more open-country species, especially if woods are designed to provide shrubby and more open areas within the wood to provide foraging and nesting opportunities.

References

Andrén, H. (1994) Effects of habitat fragmentation on birds and mammals in landscapes with different proportions of suitable habitat: A review. *Oikos* **71**: 355-366.

Bellamy, P.E.; Hinsley, S.A. & Newton, I. (1996) Factors influencing bird species numbers in small woods in south-east England. *Journal of Applied Ecology* **33**: 249-262.

Bellamy, P.E.; Rothery, P.; Hinsley, S.A. & Newton, I. (2000) Variation in the relationship between numbers of breeding pairs and woodland area for passerines in fragmented habitat. *Ecography* **23**: 130-138.

Bellamy, P.E.; Rothery, P. & Hinsley, S.A. (2003) Synchrony of woodland bird populations: the effect of landscape structure. *Ecography* **26**: 338-348.

Bennett, A.F.; Hinsley, S.A.; Bellamy, P.E.; Swetnam R.D. & MacNally R. (2004) Do regional gradients in land-use influence richness, composition and turnover of bird assemblages in small woods? *Biological Conservation* In Press.

defra (2003) *Farm woodland premium scheme – rules and procedures.* defra. (http://www.defra.gov.uk/corporate/regulat/forms/erdp/fwps/fwpsrules.pdf)

Forestry Commission (2002) *National inventory of woodland and trees: county report for Cambridgeshire.* Forestry Commission, Edinburgh. (http://www.forestry.gov.uk/pdf/cambridgeshire.pdf/$FILE/cambridgeshire.pdf)

Fuller, R.J.; Chamberlain, D.E.; Burton, N.H.K. & Gough, S.J. (2001) Distribution of birds in lowland agricultural landscapes of England and Wales; how distinctive are bird communities of hedgerows and woodland? *Agriculture, Ecosystems and Environment* **84**: 79-92.

Game Conservancy Trust (2003) *Woodland conservation and pheasants.* The Game Conservancy Trust, Fordingbridge. (http://www.gct.org.uk/pheasantconservation/index.html)

Hinsley, S.A.; Bellamy, P.E.; Newton, I. & Sparks, T.H. (1995) Habitat and landscape factors influencing the presence of individual breeding bird species in woodland fragments. *Journal of Avian Biology* **26**: 94-104.

Raven, M.J.; Noble, D.G. & Baillie, S.R. (2003) *The Breeding Birds Survey 2002.* BTO Research Report No. 334. BTO, Thetford. (http://www.bto.org/bbs/results/bbsreport.htm)

Vanhinsbergh, D.; Gough, S.; Fuller, R.J. & Brierly, E.D.P. (2002) Summer and winter bird communities in recently established farm woodlands in lowland England. *Agriculture, Ecosystems & Environment* **92**:123-136.

Woodland Trust (2001) New woods for people. The Woodland Trust, Grantham. (http://www.woodland-trust.org.uk/publications/publicationsmore/newwoodsforpeople.pdf)

Spatial arrangement of woody habitats and the distribution of birds and butterflies in agricultural landscapes in the Netherlands

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Abstract

In this study we analysed the relationship between the spatial arrangement of woody habitat and the presence of bird and butterfly species in agricultural landscapes in the Netherlands. The occupation probabilities of 32 bird species and 8 butterfly species in km-grids were analysed with logistic regression analysis. First parameters describing the internal quality of the km-grids and next parameters describing the spatial arrangement within and outside the km-grids were added to the regression model.

The regression analysis showed the importance of the spatial arrangement both within and outside the km-grids. We found a significant effect of the internal spatial arrangement for 27 bird species and of the external arrangement for 26 bird species. We found for 6 butterfly species an effect of the external spatial arrangement on the occupation probabilities. The effect of the landscape within 2 km had a stronger effect on the butterflies than the landscape within 5 km around the km-grids.

The results show that measures to increase biodiversity in agricultural landscapes should not only take the amount and quality of habitat into account, but also with the spatial arrangement of the habitat, both at small (1 km^2) and larger scales (several km^2).

Introduction

Landscape elements with semi-natural vegetation in intensively used agricultural landscapes are of the utmost importance for the survival of many plant and animal species in these landscapes (Fry, 1994; Smart *et al.*, 2002). These elements can consist of woody vegetation (forest fragments, tree rows etc.), but also of herbaceous or aquatic vegetation (road verges, ditches, etc). Together the elements constitute a habitat network on the landscape scale that is more or less continuous in space. We call the network of mostly linear and small patchy elements the 'green veining', because this fine-meshed network permeates the agricultural landscape with habitat and corridors for flora and fauna. Density and spatial arrangement of suitable habitat in the green veins are assumed to be crucial for the survival of individual species and the diversity of species groups in a landscape. In small and isolated patches the extinction probability of local populations will be high and the colonisation probability will be low. The resulting diversity of species groups in agricultural landscapes is, therefore, expected to be positively related with the density and spatial cohesion of habitat in the green veins.

The relationship between the spatial arrangement and biodiversity has more often been studied for nature reserves than for agricultural landscapes (but see, for example, Le Coeur *et al.*, 1997; Quin and Burel, 2002; Geertsema, 2002). The aim of this study is to find out how the presence of bird and butterfly species is influenced by the spatial arrangement of habitat in woody landscape elements in intensively used agricultural landscapes. It is a follow up of studies that showed the importance of green veining as habitat for butterflies (e.g., Dover *et al.*, 2000) and birds (e.g., Hinsley *et al.*, 1994) but here we focus on the importance of spatial arrangement of habitat. We are interested in the effect of the spatial arrangement both within and outside km-grids. The knowledge developed in this study is needed for the development of more effective measures for nature conservation in agricultural landscapes.

Methods

Selection of species

In this study we related data of presence and absence of bird and butterfly species in km-grids to characteristics of the green veins network that could be derived from digital geographical information. Woody vegetation makes up a characteristic part of the agricultural landscape and can be easily recognised on digital maps; we therefore focused in this study on species that are limited to woody vegetation. The bird species had to be dependent on trees, but species were excluded that were: too common (present in > 95% of selected km-grids); species living in colonies; species restricted to farmyards or coniferous trees; and species that are limited to the forest interior. The butterflies were species of woodland edges, not forest interiors, which were neither too common nor too rare in agricultural landscapes. Using these criteria we selected 32 bird and 8 butterfly species.

Selection of km-grids

The distribution data of birds were derived from the Atlas of Dutch Breeding Birds (SOVON 2002). Fieldwork for this atlas was carried out in 1998-2000. 11,254 1-km squares (covering one third of the Netherlands) were systematically selected and were visited between 1 April - 15 May and 16 May - 30 June. All bird species observed were recorded according to a standardised procedure. From this dataset km-grids were selected which consisted of at least 67% agricultural habitats, and without large forest patches (patches more than 75 m wide). This resulted in a dataset of 4,005 km-grids distributed over the entire country.

The distribution data of butterflies was collected by the Dutch Butterfly Conservation in the period 1992-2002. The monitored km-grids are distributed over the Netherlands. The quality of the data depends on the number of monitoring visits that were made in the km-grids. From the dataset we selected km-grids with high quality data where sufficient visits were made. The km-grids with high quality data were not randomly distributed over the Netherlands but concentrated in a number of regions. Those with few high quality grids were omitted from the selection. Next km-grids were selected where agricultural land use covered at least 67%. A further random selection was made from this dataset, resulting in 988 km-grids distributed over a limited number of regions in the Netherlands.

Explanatory variables

The presence and absence data of the species were related to variables that describe: the quality of the km-grids; the spatial arrangement of woody habitats within the km-grid (internal arrangement); and the spatial arrangement of woody habitats outside the km-grid (external arrangement).

The bird species were related to region, area of habitat, disturbance by traffic and the area of farmyards. The regions were defined in such a way that variation in soil, moisture and land use is covered by that variable. All woody elements were considered habitat; forest patches, hedgerows, tree lines and single trees. Both deciduous and coniferous forest patches were included but, within our selection of km-grids with at least 67% agricultural land use, the amount of coniferous forest patches was very limited. We defined two alternative variables for the internal arrangement of habitat; ***habitat density*** and ***habitat cohesion***. Habitat density is high when it is mostly arranged in patchy forest fragments and low when it is arranged in linear elements. Habitat cohesion is a measure for the amount of habitat around each habitat element in the km-grid; the contribution of patches to this amount is weighted by the area and the distance (Verboom *et al.*, 1991; Hanski, 1994). We also defined two alternative variables for the external arrangement; distance to nearest forests larger than 0.5, 5, 50 and 300 ha, and habitat cohesion. External habitat cohesion in this case was measured over larger distances (2, 5, 11, 20 and 30 km, distances based on dispersal distances of birds) than internal habitat cohesion (0.5 km).

The butterfly species were related to: region; area of habitat in edges of patchy forest fragments; area of habitat in linear elements (tree rows and hedges); and single trees. The edges of deciduous forests were included in the analysis, as well as the edges along paths wider than 8 m within forests. With a GIS package, ArcView, the area of edges was determined after the forest pattern was rasterised to grids of 25 x 25 m. The information in the 25 x 25 m grids included the amount of habitat in those grid cells. Grid cells that consisted of a maximum of 67% forest, were considered 'forest edge-grids' and in that way we could calculate the area of habitat in edges of patchy forest fragments. A comparison of the original topographical map with the grid-map showed that this gave a good selection of grids with forest edges. The regions were defined in such a way that variation in soil, moisture and land use is covered by that variable. We also added two variables that describe the quality of the woody habitat. The first was the percentage of habitat that is located within nature reserves. The second was the percentage of habitat that was located on moist soils, which accounts for variation of moisture within the km-grid. We only calculated one variable for the internal spatial arrangement for the butterflies, internal spatial cohesion, which was calculated with a similar method as for the birds. As we suggested that the quality of habitat in edges of forest patches differs from the quality of linear elements, these types of habitats were added separately to the model. Thus a variable similar to the habitat density for the bird species was not relevant for butterflies. The external spatial arrangement was calculated with two measures for habitat cohesion; within a distance of 2 km or 11km around the km-grids.

Statistical analysis

The analysis of the species distribution data was done with multiple logistic regression analysis for each species separately. The analysis for the birds was done in two steps. First the variables describing the quality of the habitat within the grid was added, next the model was extended with one variable for internal habitat arrangement as well as one variable for external habitat arrangement that together gave the best model fit. In that way we could see whether and which variables for internal and external habitat arrangement had statistically significant effects on the presence of the species, given the effect of other habitat variables and the external or internal habitat arrangement respectively.

A comparable method was used for the butterfly species. However, it appeared that the measure for internal habitat arrangement was strongly correlated with the amount of habitat in the km-grid. We, therefore, only added the variables for external habitat arrangement and checked which of the two led to the largest improvement (based on lowest Mean Deviation) of the model with variables for quality of the habitat within the km-grid. This model with variables for quality of habitat within the grid was limited to those showing a significant relation with occupation probability ($p < 0.05$).

Results

Bird species

The region in which the grids were located and the amount of habitat (area of woody vegetation) in the grids influenced significantly the occupation probability of almost all 32 bird species (Table 1). As expected the effect of the amount of habitat was positive for the species. For about half of the species the area of farmyards, which always contain some woody vegetation, had a positive relation with the occupation probability of the species. Five species were negatively related with the area of farmyards. Thirteen species were negatively related with disturbance by traffic and, surprisingly, one positively.

Table 1. Effect of landscape characteristics on presence of bird species. Given is the number of bird species (total = 32 species) showing a significant effect ($p < 0.05$) of the variable on the occupation probability in multiple logistic regression analysis.

Variable	No. of species	sign regression coefficient
Region	32	
Habitat (m^2)	30	+ (all 30)
Farmyard (area)	21	- (5) and + (16)
Disturbance by traffic	14	- (13) and + (1)

After correction for the above mentioned variables, the effect of the internal spatial arrangement of habitat within the km^2 grid was related significantly to the presence of 27 of the 32 bird species and the external arrangement outside the grid to the presence of 26 of the species (Table 2). The habitat density was the more important parameter for internal cohesion. For many species there was a negative relationship between habitat density and occupation probability, so higher occupation probabilities were found when habitat was arranged in linear elements (tree lines, hedgerows) than in patchy shaped elements (forest fragments). The effect of external spatial arrangement was significant for 26 species. For 21 of these species the spatial cohesion of all woody habitats around the km-grid was more important than the presence of a single large forest at a certain distance. For 12 out of those 21 species the spatial cohesion had a positive effect on the occupation probability, but for 9 of the species the effect was negative. Figure 1 shows the relation between the predicted occupation probability of one of the bird species, the tree pipit (*Anthus trivialis*), and the spatial cohesion within 5 km around the km-grids.

Table 2. Effect of spatial arrangement of landscape elements on presence of bird species: number of species for which either of the two internal and two external parameters gave the best model fit ($p < 0.05$).

internal or external	parameter	No. of species	sign
internal	habitat density	16	- (15) and + (1)
	spatial cohesion	11	+ (all 11)
external	distance to large forest	5	- (all 5)
	spatial cohesion	21	- (9) and + (12)

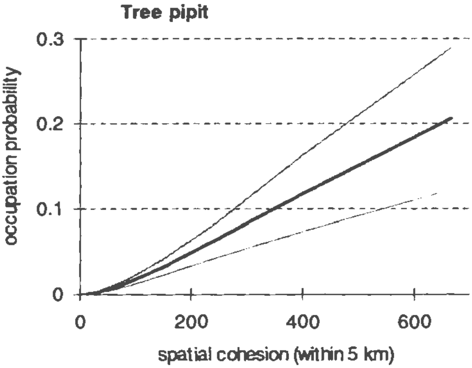

Figure 1. Relation between external spatial cohesion of the landscape within 5 km and the predicted occupation probability (with 95% confidence intervals) shown for the tree pipit.

Butterfly species

Table 3 shows the results of the first part of the multiple logistic regression analysis for the butterfly species. We found significant effects of the region for all species. The amount of habitat that is located in forest edges is a significant factor for all but one species. Additionally the amount of habitat that is located in linear structures such as tree rows is important for two species; map butterfly (*Araschnia levana)* and speckled wood (*Pararge aegeria*). The percentage of habitat that is located on moist soils had a positive effect on all species, but it was statistically significant effect for only two; speckled wood and orange-tip (*Anthocharis cardamines*). The percentage of habitat that in nature reserves had a positive effect as well, but it was only for one species statistically significant, the white admiral (*Ladoga camilla*).

The effect of the spatial cohesion of woody habitats around the km-grids were positively related with the occupation probability of the species. For six of the eight species this relation was statistically significant ($p < 0.05$). Five species showed a significant effect of both spatial cohesion within 2 km and within 11 km around the grid cell. For two of those species the spatial cohesion within 11 km gave the best model fit, for the other six species the spatial cohesion within 2 km gave the best model fit.

Table 3. Effect of landscape characteristics on presence of butterfly species. Listed is the number of butterfly species (total = 8 species) showing a significant effect ($p < 0.05$) of the variable on the occupation probability in multiple logistic regression analysis. The variables had a positive relation with the occupation probability.

Variable	No. of species
Region	8
Habitat in edges of patchy forest fragments	7
Habitat in linear elements (tree rows and hedge rows)	3
% moist habitat	2
% habitat in nature reserves	1

Discussion

In this study we were able to show that the presence of bird and butterfly species in intensively used agricultural landscapes is positively related to the amount of habitat in landscape elements such as forest fragments, tree rows and hedges. In addition, not only the amount of habitat but also the spatial arrangement within km-grids and in the landscape surrounding the km-grids is a significant factor influencing the occupation probability of species.

About half of the studied bird species preferred landscapes where habitat was arranged in narrow landscape elements, instead of patchy elements with a higher density of habitat. A possible explanation may be that these species prefer the gradient of forest to open landscape. Nine bird species showed a negative relation with external habitat cohesion. Of these nine, seven species also preferred grid-cells with landscapes where habitat was arranged in narrow landscape elements. This might indicate that those species really prefer smaller narrow elements above large forest patches, both within the km-grid and outside the km-grid. Twelve species showed a positive relation with external habitat cohesion. The better results for spatial cohesion, as compared to the distance to a large forest, indicates that it is important that the birds can find habitat around the km-grid, but that it is not important that this habitat is concentrated in a single large forest.

For most butterfly species the edges of forest patches were more important than tree rows and hedges. An important difference between forest patches and tree rows is the presence of shrubs and herbs in the undergrowth. These are common in the edges of forest patches, whilst especially shrubs are rarely present under tree rows. The hedges consist of shrubs but these are not very common in the km-grids, so their effect will not be very strong. This difference between forest patches and tree rows results in more favourable microclimates and more food sources in the edges of forest patches, as compared to the tree rows. Groenendijk *et al.* (2002) have shown that larvae of butterflies in the Netherlands are extremely sensitive to spraying of insecticides used in the Dutch agricultural systems. It should be mentioned that on the digital maps wooded banks were not represented by lines but by patches with a high edge to interior ratio. So, despite their linear shape, wooded banks are represented in the analysis as edges of forest patches. In order to know more about the effect of spatial arrangement of habitat within a km-grid on the occupation probability of butterfly species, other variables will be needed than the ones we calculated in this study.

We were able to show the relationship between the spatial arrangement of habitat in the surrounding landscape and the occupation probability for six butterfly species. This confirms our assumption that not only the amount but also the spatial arrangement of habitat influences the survival of species. This applies not only to nature reserves but also to habitat in the network of green veining in intensively used agricultural landscapes.

References

Dover, J.; Spartks, T.; Clarke, S.; Gobbet, K. & Glossop, S. (2000) Linear features and butterflies: the importance of green lanes. *Agriculture, Ecosystems and Environment 80: 227-242.*

Fry, G.L.A. (1994) The role of field margins in the landscape. *BCPC Monograph No 58: Field Margins: Integrating Agriculture and Conservation: 31-40.*

Geertsema, W (2002) Plant survival in dynamic habitat networks in agricultural landscapes. *PhD thesis Wageningen University.*

Groenendijk, D.; Van Mannekes, M.; Vaal, M. & Van den Berg, M. (2002) Butterflies and insecticides: a review and risk analysis of modern Dutch practice. *Proc. Exper. Appl. Entomol. NEV Amsterdam 13: 29-34.*

Hanski, I. (1994) A practical model of metapopulation dynamics. *Journal of Animal Ecology 63: 151-162.*

Hinsley, S.A.; Bellamy, P.E., Enoksson, B.; Fry, G., Gabrielsen, L.; McCollin, D. & Schotman, S. (1998). Geographical and land-use influences on bird species richness in small woods in agricultural landscapes. *Global Ecology and Biogeography Letters 7: 125-135.*

Le Coeur, D.; J. Baudry & F. Burel (1997) Field margins plant assemblages: variation partitioning between local and landscape factors. *Landscape and Urban Planning 37: 57-71.*

Quin, A. & Burel. F. (2002) Influence of herbaceous elements on butterfly diversity in hedgerow agricultural landscapes. *Agriculture, Ecosystems and Environment 93: 45-53*

Smart, S.; Bunce, R.G.H.; Firbank, L.G. & Coward, P. (2002). Do field boundaries act as refugia for grassland plant species diversity in intensively managed agricultural landscapes in Britain? *Agriculture, Ecosystems and Environment 91: 73-87.*

SOVON (2002) Atlas of Dutch Breeding Birds 1998-2000. Nederlandse fauna 5. *Nationaal Natuurhistorisch Museum Naturalis, KNNV Uitgeverij & European Invertebrate Survey-Nederland, Leiden.* in Dutch, with English summary.

Verboom, J.; Schotman, A.; Opdam, P. & Metz, J.A.J. (1991) European nuthatch metapopulations in a fragmented agricultural landscape. *Oikos 61: 149-156.*

Simulating population viability in fragmented woodland: nuthatch (*Sitta europaea* L.) population survival in a poorly wooded landscape in eastern England

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Abstract

An individual-based spatially explicit population model of the nuthatch was developed and used to explore reasons behind the scarcity of nuthatches in apparently suitable woodland habitat in an area of eastern England. Predictions were compared with observed population sizes in the study area and with the previously published results of an analytical model applied to the same problem. Within our model, based on individual behaviour, nuthatch life cycle and immigration (direction and rate) parameters were user selectable. A novel aspect is user-selection of habitat perceptual range, which was found to influence the study area population size. The number of breeding pairs predicted by the model gave a closer match to observed levels than those achieved by the analytical model. According to model simulations, the main cause of nuthatch scarcity in the study area was the inability of patches to support viable populations. Predicted population sizes were found to be a function of immigration direction, immigration rate and habitat perceptual range and it is recommended that these essential elements be considered in population viability studies, especially when habitat is fragmented and patchily distributed.

Introduction

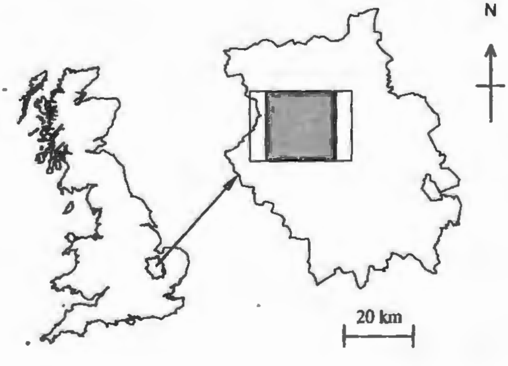
The nuthatch, *Sitta europaea* L., is a small (c.22-26g) cavity-nesting woodland passerine (Nilsson, 1976) resident in the UK, whose scarcity from an area in eastern England, centred on Monks Wood (52°24'N, 0°14'W) in Cambridgeshire, has been the subject of several studies (Hinsley *et al.*, 1995; Bellamy *et al.*, 1998).

Bellamy *et al.* (1998) compared observed numbers of breeding pairs in woods in the Monks Wood study area with the results from an analytical model developed in the Netherlands to predict nuthatch abundance in fragmented woodland (Schotman and Meeuwsen, 1994). Bellamy *et al.* (1998) found that the numbers of breeding pairs predicted by the Dutch model were lower than that expected from habitat area alone and that observed numbers were even lower than the model predictions. The nuthatch is noted as being sensitive to isolation (e.g., Enoksson *et al.*, 1995; Matthysen *et al.*, 1995), but despite taking this into account, the analytical model still predicted too high an occupancy level. To investigate whether an alternative modelling technique could more closely match the observed occupancy and abundance values, an individual-based spatially explicit population model (SEPM), named PatchMapper, was developed. This paper describes the model structure, followed by its application to the problem of nuthatch scarcity in the Monks Wood area. The aims of this study were to investigate whether an individual-based SEPM could produce more realistic predictions than the analytical model (Schotman and Meeuwsen, 1994) and whether factors other than habitat isolation influenced the numbers of breeding nuthatches in the study area

Study Area

The main study in Cambridgeshire, in eastern England (Figure 1), was based on that used by Bellamy *et al.* (1998), but was trimmed to 20 x 20 km (40,000 ha) to match the model input requirements (Figure 2). This 'main' map was used for comparison with the analytical model results of Bellamy *et al.* (1998). Approximately 4% (1570 ha) of this reduced area was wooded and 28 woods (totalling 512 ha) were deemed suitable for nuthatch occupancy. The total area of good quality nuthatch territory was approximately 261 ha (0.65% of the total landscape area) (Bellamy *et al.*, 1998). To avoid any directional bias in the results, the main map was adjusted approximately 1 km westwards and 5 km southwards to place Monks Wood at the centre. This second 'centralised' map (also 20 x 20 km), was used for the initial tests and had similar woodland cover (*c.* 4%) and number of woods (35) to the main study area. South Rockingham Forest, about 12 km west of the main study area, was the most suitable area within reasonable dispersal distance capable of acting as a source of immigrant nuthatches.

Figure 1. Location of the 20 x 20 km main study area (darker shading) in relation to the 20 x 25 km study area used by Bellamy *et al.* (1998), within the county of Cambridgeshire in eastern England, UK (redrawn from Bellamy *et al.*, 1998).



Methods

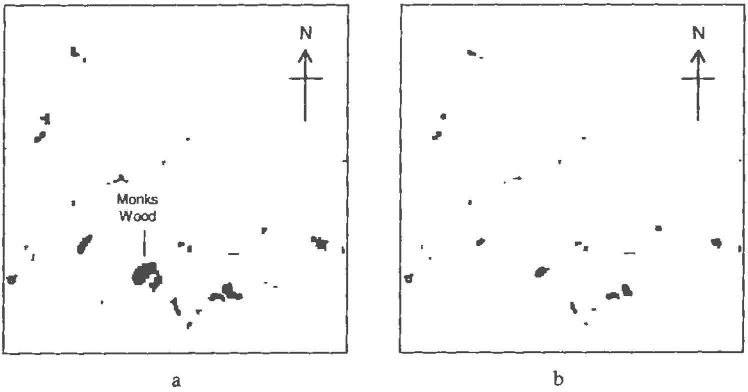
Model description

PatchMapper combines a nuthatch population simulator with a grid-based representation of the landscape. Nuthatch life history, dispersal and territorial parameters are user selectable, as are initial population sizes and immigration rates into the study area. Different habitat configurations and landscape management scenarios can be tested with a choice of landscapes, either input from a GIS or user generated. During a run, nuthatch locations are superimposed on the landscape, providing a real-time visual output.

Landscape

The landscape was modelled with three elements: target habitat, non-target habitat and matrix. Non-target habitat is a wooded area likely to be perceived by a dispersing nuthatch as containing good quality habitat, but which in fact does not. Target habitat represents good quality habitat and the matrix represents non-wooded areas. Two maps were used to accommodate this idea. The first map (the habitat map) was a cartographic view, representing all woods in the study area. A second map (the quality map) represented habitat quality, which determines nuthatch territory size (Wesolowski and Stawarczyk, 1991) (Figure 2). Maps were represented using a grid of square cells. For this study, the grid size was 200 x 200, which for the study area of 20 x 20 km gives a cell area of 1ha. Nuthatch territory size for the area was taken as 2 ha (Bellamy *et al.*, 1998).

Figure 2. The two main maps used in PatchMapper. The first map is a binary representation of the landscape comprising wooded areas (potential habitat) and matrix. For illustration, woods containing suitable habitat are shown in black, with all other woods shaded light grey. Monks Wood, the focal wood for this study is labelled (Figure 2a). The second map represents areas of habitat of sufficient quality to contain nuthatch territories (Figure 2b). Data extracted from survey results by Bellamy *et al.* (1998).

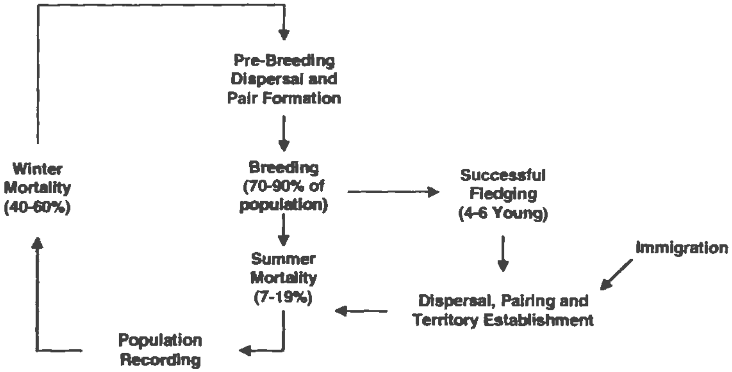


Nuthatch simulator

Nuthatches were modelled by applying behavioural rules to each individual bird, with reference to the structure of the habitat and quality maps, an approach in which both landscape structure and species' characteristics govern population sizes and distribution.

Figure 3 shows the nuthatch modelling cycle. Life cycle parameters and territorial establishment behaviour for this study were taken from the literature (Nilsson, 1987; Wesolowski and Stawarczyk, 1991; Pravosudov, 1993; Matthysen *et al.*, 1995; Bellamy *et al.*, 1998; Matthysen and Adriaensen, 1998) and implemented as a set of rules applied to each individual. Population size, dispersal mortality, average age and other demographic statistics are recorded after each cycle of the model.

Figure 3. Nuthatch lifecycle modelled by PatchMapper. The immigration input allows individuals to disperse from distant sources into the study area. The parameters are taken from the literature, as cited in the text.



Dispersing nuthatches search for habitat, traversing the matrix with a correlated random walk. When a habitat patch is encountered, it is searched for good quality habitat. If none is found, the search for a new patch is resumed. If good quality habitat is found, then depending on its occupancy status, a new territory is established, a pair formed or if no vacancies exist, the disperser is forced to search for further suitable habitat within the wood or to search the matrix for another wood. A dispersal mortality probability is applied at each step.

To incorporate the possible effects of landscape structure on dispersal patterns, a habitat perceptual range mechanism was added to the basic dispersal rules, allowing the disperser to move directly to the nearest wood. The distance at which the disperser perceives a wood should represent the actual habitat perceptual range, but such data is not known for the nuthatch (or for many other species). A visual landscape survey was carried out to determine at what distance a wood could be clearly identified. The resulting mean distance of 2 km was used for this study.

Tests

To test the self-supporting ability of the main landscape, the model was populated and run without immigration. With immigration into the centralised map, the effects of immigration direction, immigration rate and habitat perceptual range of dispersers were tested for their effects on population size in Monks Wood. To test the effects of variations in immigration rate and direction on the nuthatch population size in the study area, the model was run with immigration into the area from an external source located at each of the cardinal and quadrantal points. If habitat perceptual range is a factor in guiding dispersers towards distant woods and reducing dispersal mortality, then varying the awareness distance should have an effect on population size. To test this, a series of runs with a perceptual range of 0 to 10 km and westerly immigration levels of 10, 20 and 30 individuals were undertaken. The results were used with the main map to judge PatchMapper's performance against the analytical model and study observations (Bellamy *et al.*, 1998). Each run was repeated for 10, 20 and 30 immigrating individuals. All tests used 100 runs of 100 years per run.

Results

The self-supporting test revealed that without immigration, the main study area population reduced to zero after a mean of 8 years ± 0.5 years (95% C.I. over 100 runs). Immigration was therefore included in the model.

Figure 4 shows that both the direction and the numbers of immigrants were potential factors governing the population size in Monks Wood. Annual immigration into the centralised study area of between 10 and 20 birds, from the north east and westerly directions matched the observed level of 2 pairs in Monks Wood (Bellamy *et al.*, 1998), as did immigration from the south east at approximately 10 individuals per annum. Rockingham Forest, some 12 km to the west, is the most likely source of nuthatches, so a westerly immigration rate into the study area of 10 – 20 individuals per annum seemed the most realistic. The habitat perceptual range tests (Figure 5) showed that the distance at which a dispersing nuthatch can perceive habitat influenced the population size in Monks Wood, suggesting that perceptual range should also be modelled.

Discussion

The fragmented structure of suitable habitat in the study area was found to be incapable of supporting a nuthatch population without immigration. This was seen as the main reason for the scarcity of nuthatches in the study area. When using viable, but unvalidated parameters, PatchMapper's predictions gave a closer match to the observed population size in Monks Wood than the analytical model's prediction of 63 pairs (Bellamy *et al.*, 1998). Individual-based SEPMs have been criticised for their reliance on unverified parameters, particularly with respect to dispersal and movement rules (Conroy *et al.*, 1995; Wennergren *et al.*, 1995). Nevertheless, the results from this study suggest that the technique of modelling population viability by simulating the behaviour of individuals in a spatially explicit grid-based landscape has the potential for higher accuracy than analytical models, with the added advantage of increased generality. Immigration direction and rate can have a large influence on population sizes (Figure 4), indicating that these inputs should be separate and controllable instead of using wrap-around or reflective immigration models such as BACHMAP (Pulliam *et al.*, 1992).

Figure 4. The effects of immigration direction on the mean number of pairs of nuthatches in Monks Wood using the centralised study area (error bars give 95% C.I.). Results are given for 10, 20 and 30 immigrating individuals per annum, shaded as per the key. Habitat perceptual range was set to 2 km.

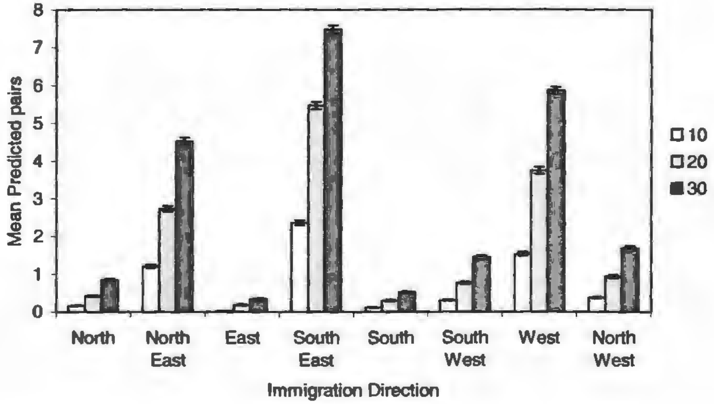
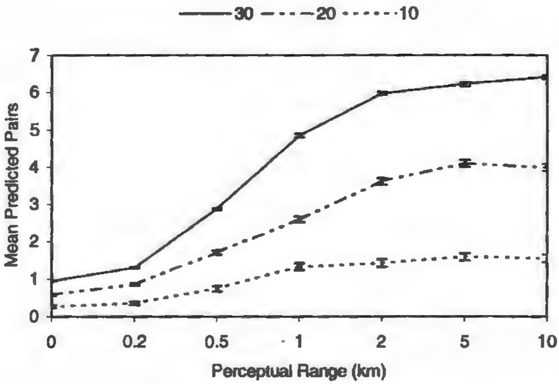


Figure 5. The effect on population size in Monks Wood of habitat perceptual range from 0 to 10 km, with westerly annual immigration rates into the centralised study area of 10, 20 and 30 individuals, as per the key. Error bars give 95% confidence levels.



South Rockingham Forest is thought to be the main source of immigrating nuthatches into the study area, but another possible source is a wood near Peterborough, which lies to the northwest of the study area (Enoksson, 1993). With immigration direction and rate affecting predicted population sizes within the study area (Figures 4 and 5), identification of these sources and their likely emigration rate are necessary to increase the accuracy of PatchMapper's predictions. The immigration level from the nearby Rockingham forest is therefore an untested input, but is the subject of a separate study (Alderman *et al.*, in prep).

The thresholds revealed by the habitat perceptual range test (Figure 5) warrant further investigation. They suggest that habitat perceptual range enables dispersing individuals to overcome the effects of fragmentation, to some extent. The results suggest that perceptual range has a distinct spatial threshold: above a minimum critical level, a bird will always be able to detect an adjacent habitat patch, at least within the limits defined by the spatial pattern of a particular landscape.

Conclusions

The analytical study model was developed for use in the Netherlands, but when applied to a different, although superficially similar, landscape gave erroneous results. Such models take the effects of landscape structure, immigration and habitat perceptual range into account by default. They may, therefore, be prone to error if any of these parameters change, which is likely when landscape configuration changes. Caution should therefore be applied when using models outside their target landscapes.

Immigration direction, immigration rate and habitat perceptual range were all found to have an important effect on population size. It is recommended that these inputs are included in population viability models, but the values chosen must be verifiable within the landscape in question and should result in realistic effects. For example, dispersal characteristics can vary with the degree of habitat fragmentation (Matthysen *et al.*, 1995). Landscape structure affected results and demonstrated the importance of including all suitable habitat in the model and not just good quality breeding habitat. The effects of habitat perceptual range invite further investigation, in particular the reasons behind any thresholds (Figure 5). The results of this study suggest that individual-based simulation modelling is a viable technique for the future.

Acknowledgements

Thanks to Ruth Swetnam for supplying and updating the digital maps.

References

Bellamy, P.E.; Brown, N.J.; Enoksson, B.; Firbank, L.G.; Fuller, R.J.; Hinsley, S.A. & Schotman, A.G.M. (1998) The influences of habitat, landscape structure and climate on local distribution patterns of the nuthatch (*Sitta europaea* L.). *Oecologia* **115**: 127-136.
Conroy, M.; Cohen, Y.; James, F.; Matsinos, Y. & Maurer, B. (1995) Parameter estimation, reliability and model improvement for spatially explicit models of animal populations. *Ecological Applications* **5**: 17-19.

Enoksson, B. (1993) Nuthatch. D.W. Gibbons; J.B. Reid & R.A. Chapman (Eds.) *The New Atlas of Breeding Birds in Britain and Ireland: 1988-1991*. British Trust for Ornithology, Thetford, 376-377.

Enoksson, B.; Angelstam. P. & Larsson, K. (1995) Deciduous forest and resident birds: the problem of fragmentation in a coniferous forest landscape. *Landscape Ecology* **10**: 267-275.

Hinsley, S.A.; Bellamy, P.E.; Newton, I. & Sparks, T.H. (1995) Habitat and landscape factors influencing the presence of individual bird species in woodland fragments. *Journal of Avian Biology* **26**: 94-104.

Matthysen, E. & Adriaensen, F. (1998) Forest size and isolation have no effect on reproductive success of eurasian nuthatches (*Sitta europaea*). *The Auk* **115**: 955-963.

Matthysen, E.; Adriaensen, F. & Dhondt, A.A. (1995) Dispersal distances of nuthatches, *Sitta europaea*, in a highly fragmented forest habitat. *Oikos* **72**: 375-381.

Nilsson, S.G. (1976) Habitat, territory size and reproductive success in the nuthatch *Sitta europaea*. *Orins Scandinavica* **7**: 179-184.

Nilsson, S.G. (1987) Limitation and regulation of population density in the nuthatch *Sitta europaea* (Aves) breeding in natural cavities. *Journal of Animal Ecology* **56**: 921-937.

Pravosudov, V.V. (1993) Breeding biology of the eurasian nuthatch in northeastern Siberia. *Wilson Bulletin* **105**: 475-482.

Pulliam, H.; Dunning, J. & Liu, J. (1992) Population dynamics in complex landscapes: a case study. *Ecological Applications* **2**: 165-177.

Schotman, A.G.M. & Meeruwsen, H.A.M. (1994) *Predicting the effects of defragmentation measures in the Rolde/Gieten area.* (In Dutch). IBN report 118, Instituut voor Bos en Natuuronderzoek (IBN-DLO), Wageningen, Netherlands.

Wennergren, U.; Ruckelshaus, M. & Kareiva, P. (1995) The promise and limitations of spatial models in conservation biology. *Oikos* **74**: 349-356.

Wesolowski, T. & Stawarczyk, T. (1991) Survival and population dynamics of nuthatches *Sitta europaea* breeding in natural cavities in a primeval temperate forest. *Ornis Scandinavica* **22**: 143-154.

Using a multi-scale model for identifying priority areas in capercaillie (*Tetrao urogallus*) conservation

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Abstract

The capercaillie (*Tetrao urogallus*) is a forest grouse species with large spatial requirements and narrow habitat preferences. A large amount of research and conservation efforts, on mostly small spatial scales, could not slow down the decline of capercaillie in Central Europe. Apparently, many population processes operate on larger scales than usually considered, and require large-scale and multi-scale analyses to understand them.

We developed a multivariate habitat suitability model for capercaillie in the Swiss Prealps and Alps. This model, in which we included variables expressed on different spatial scales, can explain capercaillie occurrence with a high degree of precision. As it predicts presence in almost all occupied areas and identifies ecologically similar areas, it can be interpreted as simulating potential capercaillie habitat. As such, it is an important tool in the national capercaillie Species Action Plan and helps to strengthen capercaillie populations more efficiently.

Introduction

The role of scale in ecology has been discussed extensively over the past three decades and is now widely recognised as being of vital importance for understanding ecological processes (Wiens, 1989; Levin, 1992). Although there has been much research exploring and confirming the scale dependence of species-habitat relationships (Fuhlendorf *et al.*, 2002; Lawler and Edwards, 2002; Thompson and McGarigal, 2002), many investigations are still conducted at arbitrary and often small spatial scales. This, however, can lead to wrong conclusions about the importance of different habitat factors and thus to less efficient tools in conservation practice.

The capercaillie is a charismatic forest grouse species with narrow habitat preferences (e.g., Schroth, 1992; Sjöberg, 1996) and large spatial requirements (Rolstad *et al.*, 1988; Storch, 1995). This makes it highly susceptible to habitat and landscape changes. Capercaillie populations are declining in most of their central European range (Klaus *et al.*, 1986; Storch, 2000), as habitat loss and fragmentation have resulted in small populations with different degrees of connectivity or isolation. In Switzerland, the remaining population of 900 -1000 individuals (Mollet et al., 2003) faces a high extinction risk.

In capercaillie, most research as well as conservation measures have hitherto focused on small spatial scales (e.g., Klaus *et al.*, 1985; Schroth, 1992). These efforts did little to stop the ongoing decline of the grouse species. Recent work mainly in Scandinavia has shown that capercaillie populations are in fact substantially driven by processes operating on the landscape-scale (Rolstad and Wegge, 1989; Helle *et al.*, 1994; Kurki *et al.*, 2000; Storch, 2002). Presently available habitat models at the landscape scale do not address spatial scale explicitly (Sachot *et al.*, 2003) or do not include different spatial scales in a single model (Storch, 2002; Suchant, 2002). Thus, larger-scale habitat relationships are still insufficiently known in those regions where the species is endangered (Storch, 2000). Analyses should be conducted at multiple scales (Keppie and Kierstead, 2003).

We analysed species-habitat relationships of capercaillie on different landscape scales in the Swiss Prealps and Alps. We hypothesised that the predictive power of single variables would vary among the spatial scales. Therefore, we calculated univariate models for all variables on several spatial scales and thus identified the optimal scale for every variable. This information was used to calculate a multivariate model where all variables were entered on their optimal scale.

Methods

Based on our own field records and additional data sets (from regional recorders and the Swiss Ornithological Institute), we allocated presence/absence information to a boolean grid covering the Swiss Prealps and Alps (Graf *et al.*, in prep.). A set of environmental data available area-wide for the whole of Switzerland was used as independent variables. All data were processed in a GIS (ARC/INFO 8.3) as grids with a cell size of 1 ha. With a moving window analysis, we calculated neighbourhood means for different circular areas for every variable. Circular area was increased stepwise from 5 ha up to 1100 ha, which is about two times the size of the bird's home range (Storch, 1995).

In a first step, we calculated univariate logistic regression models (in SPSS, Menard, 2002) for each variable on every spatial scale (Graf *et al.*, in prep.). By comparing the explained variance of all these models, we defined the optimal scale for every predictor variable as the one with the highest amount of variability explained; in other words, we searched for the scale at which the variable best explained the spatial pattern of presence-absence in capercaillie. In a second step, we calculated a multivariate logistic regression model (stepwise backwards) where we entered every variable on its optimal scale. If two variables were highly correlated (Spearman's rho > 0.7, Hosmer and Lemeshow, 1989), we omitted the one that we felt had less biological significance. The resulting multi-scale model was validated on data from the region that had provided the data for model calibration and on independent data from a separate area. To measure accuracy, we used Kappa-statistics (Monserud and Leemans, 1992) and calculated the area under the ROC-function (AUC; Fielding and Bell, 1997).

By applying the model equation in a GIS (ARC/INFO 8.3) we produced a grid-based map of predicted probability of capercaillie occurrence (1 = Presence, 0 = Absence). The grid cells were then reclassified according to an accurate threshold value (0.5). The result was a boolean map representing predicted presence and absence areas (Figure 1a).

Results and discussion

The variables tested univariately performed best on scales ranging from 1 - 530 ha (Table 1). Some variables were almost indifferent to the spatial scale of analysis (e.g., average temperature), others did best on small scales (e.g., index of topography, slope) and some variables had the highest predictive power on large scales (e.g., proportion of forest, proportion of moors). All variables were then used on their optimal scale for building a multivariate, multi-scale model.

The multi-scale model performed excellently when tested in the region that had provided the data for model calibration ($\text{Kappa}_{opt} = 0.89$, AUC = 0.96). The validation of the model on data from the independent area of the Jura Mountains ($\text{Kappa}_{opt} = 0.60$, AUC = 0.88) indicates that the model has a high degree of generality and could also be applied elsewhere. Two very important variables in the model are included on a large spatial scale (proportion of forest, proportion of moors; Table 1). They account for the large spatial requirements of capercaillie (average home range c. 500 ha; Storch, 1995). By containing several variables on the 1ha-scale, the model identifies the areas that are suitable with respect to local topography and forest type.

As the multi-scale model predicts presence for almost all occupied areas and ecologically similar zones (Figure 1), it can be interpreted simulating potential capercaillie habitat. In fact, the predicted presence areas are to a large degree identical with the areas that were occupied by capercaillie in the past decades (1950-1970), when the populations were about twice as large as today (Mollet *et al.*, 2003). As we had to rely on environmental data-sets that were available area-wide, the predictive power of our models is limited. Predicted presence areas that did not hold capercaillie may have been abandoned because of other reasons working on smaller scales. For instance, forest structure may be locally unsuitable or human activities may cause frequent disturbances.

Table 1. Variables included in the multivariate logistic regression model. Scale: spatial scale at which a variable was included. Beta-value: coefficients from the regression equation. P-value: significance of single variables. ^{2} squared variable included because of a unimodal response.

Variable	Scale [ha]	Beta-value	P-value
Proportion of forest	250	0.39483	< 0.001
Proportion of moors	530	0.55717	< 0.001
Slope	13	- 0.00174	< 0.001
Average temperature	1	0.03458	< 0.001
Average temperature^2	1	- 0.00001	< 0.001
Index of topography	1	0.00586	< 0.001
Cost distance to large rivers	1	0.00003	0.001
Forest type coniferous	1	0.72786	0.029

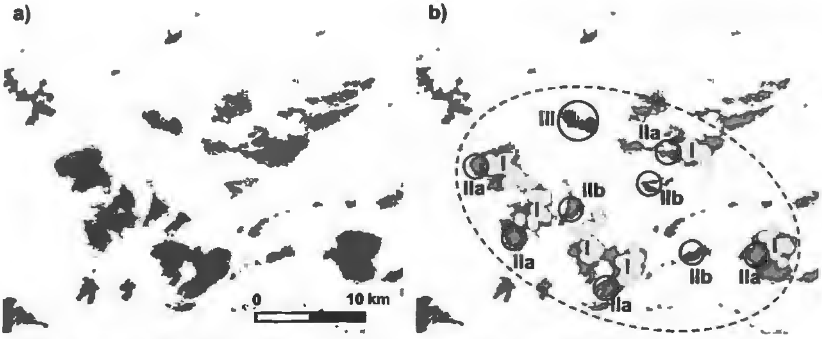

Figure 1. Distribution of predicted presence areas for the multi-scale model (a) and priority setting in the Species Action Plan (b); light grey: forest areas; dark grey: predicted presence, streaked: actual populations (within a focus area), I: Priority I (actual presence), IIa: Priority II (close to presence), IIb: Priority II (stepping stone), III: Priority III (potential expanding areas).

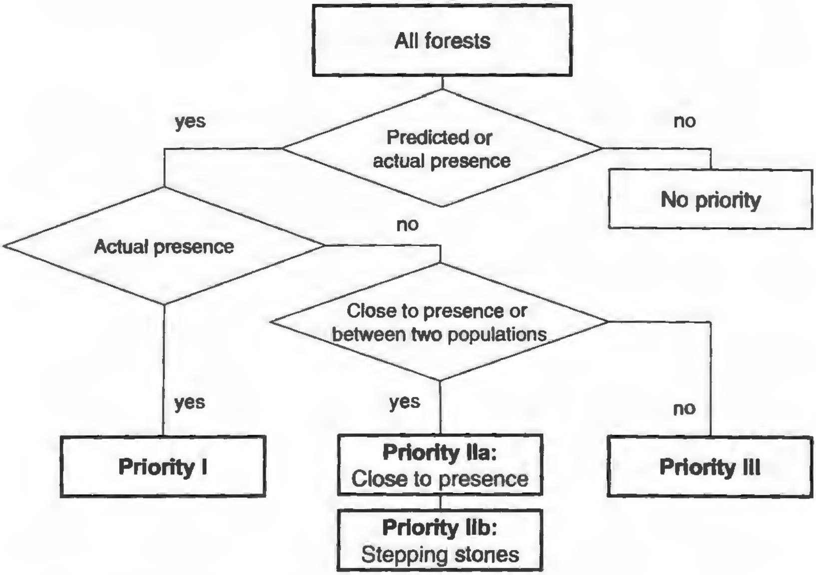

Figure 2. Priority setting in the Species Action Plan.

Implications for conservation

In combination with the inventory of the actual populations, our model provides a basis for the national Species Action Plan. It allows us to define priority areas for conservation measures in a spatially explicit way (Figure 1b, Figure 2). The areas with actual populations (Priority I) are crucial for the conservation of the species. They are the fragmented remnants of a formerly much larger and contiguous distribution and are often inhabited by only few individuals. In this situation, every small population can be essential for the survival of a metapopulation (Segelbacher, 2002). Thus, all areas where capercaillie are present have to be conserved by all means.

In the long term, it should be possible to restore capercaillie populations in the Swiss Prealps to a viable level, if the actual distribution can be enlarged, better connected and if abundance can be increased. This can most probably be attained in areas with suitable landscape ecological characteristics that are close to extant populations (Priority IIa). Some areas with predicted presence but actually without capercaillie can act as linkages between existing subpopulations (Priority IIb), if they are situated at reasonable distances. Such "stepping-stones" can significantly increase the viability of the whole capercaillie metapopulation (Segelbacher, 2002).

Until only a few decades ago, additional areas were occupied by capercaillie (Mollet *et al.*, 2003). At present, these areas are not likely to be colonised again soon and can not act as stepping stones because they are situated peripherally (Priority III). However, if we are able to increase the Swiss capercaillie population onto a sustainable level, these areas may eventually be populated again.

In all priority categories, the same conservation measures are necessary to strengthen the capercaillie populations: habitat improvement and reduction of human disturbance. However, conservation measures should be brought forward in areas of priority I and II. Forest structure has to be improved for capercaillie where the stands are presently too dense and uniform. At least two thirds of the forested area should provide an open and diverse forest structure with a lush field layer. Wherever possible, measures to improve the habitat should be included in normal forestry practice. Suitable habitats for hens with chicks need special attention because, in most cases, summer habitat is supposed to be more limited than winter habitat (Storch, 1999). Open and diversely-structured forests should be furthered also in lower priority regions (Priority III). As capercaillie has proven to be a suitable umbrella species (Suter *et al.*, 2002) a large number of rare mountain birds with similar habitat requirements will benefit from conservation actions in favour of this single forest grouse species. Additionally, disturbance by human activities should be minimised. At least in areas with actual populations, tourists (hikers, snow shoe hikers, etc.) should be advised to stay on the trails and forestry roads should be closed.

Our habitat suitability model provides a tool for decision-making in the planning of conservation measures on a large scale. However, additional information with smaller resolution is needed for planning conservation actions in detail (Storch, 1997). For instance, to optimise the measures to improve habitat in a certain area we need information about actual stand structure and tree species composition, cover of field layer, soil characteristics and potential natural vegetation. Information about existing or planned forest reserves should also be included in the planning process. In such areas, it can be much more promising to implement conservation measures than in areas with regular management mainly focusing on wood production.

By combining the potential habitat map with data on the actual capercaillie distribution, forest structure and site characteristics, an effective instrument for the spatially explicit planning of conservation measures in favour of this threatened species can be developed. In fact, our potential habitat map will promptly be included in the national Species Action Plan that will be completed in 2004.

References

Fielding, A.H. & Bell, J.F. (1997) A review of methods for the assessment of prediction errors in conservation presence/absence models. *Environmental Conservation* **24**: 38-49.
Fuhlendorf, S.D.; Woodward, A.J.W.; Leslie, D.M., Jr. & Shackford, J.S. (2002) Multi-scale effects of habitat loss and fragmentation on lesser prairie-chicken populations of the US Southern Great Plains. *Landscape Ecology* **17**: 617-628.
Graf, R.F.; Bollmann, K.; Suter, W. & Bugmann, H. (in prep.) Analysis of capercaillie habitat on different spatial scales – comparing single- and multi-scale models.
Helle, P.; Helle, T. & Lindén, H. (1994) Capercaillie (*Tetrao urogallus*) lekking sites in fragmented Finnish forest landscape. *Scandinavian Journal of Forest Research* **9**: 386-396.
Hosmer, D.W. & Lemeshow, S. (1989) *Applied logistic regression,* John Wiley & Sons, New York.
Keppie, D.M. & Kierstead, J.M. (2003) The need to improve our attention to scale of resolution in grouse research. *Wildlife Biology* **9**: 385-391.
Klaus, S.; Andreev, A.V.; Bergmann, H.-H.; Müller, F.; Porkert, J. & Wiesner, J. (1986) *Die Auerhühner*, A. Ziemsen Verlag, Wittenberg Lutherstadt.
Klaus, S.; Boock, W.; Görner, M. & Seibt, E. (1985) Zur Ökologie des Auerhuhns (*Tetrao urogallus* L.) in Thüringen. *Acta Ornithoecologica* **1**: 3-46.
Kurki, S.; Nikula, A.; Helle, P. & Linden, H. (2000) Landscape fragmentation and forest composition effects on grouse breeding success in boreal forests. *Ecology* **81**: 1985-1997.
Lawler, J.J. & Edwards, T.C., Jr. (2002) Landscape patterns as habitat predictors: building and testing models for cavity-nesting birds in the Uinta Mountains of Utah, USA. *Landscape Ecology* **17**: 233-245.
Levin, S.A. (1992) The problem of pattern and scale in ecology. *Ecology* **73**: 1943-1967.
Menard, S. (2002) *Applied logistic regression analysis,* Sage Publications, London.
Mollet, P.; Badilatti, B.; Bollmann, K.; Graf, R.F.; Hess, R.; Jenny, H.; Mulhauser, B.; Perrenoud, A.; Rudmann, F.; Sachot, S. & Studer, J. (2003) Verbreitung und Bestand des Auerhuhns *Tetrao urogallus* in der Schweiz 2001 und ihre Veränderungen im 19. und 20. Jahrhundert. *Der Ornithologische Beobachter* **100**: 67-86.
Monserud, R.A. & Leemans, R. (1992) Comparing global vegetation maps with the Kappa statistics. *Ecological Modelling* **62**: 275-293.
Rolstad, J. & Wegge, P. (1989) Capercaillie populations and modern forestry - a case for landscape ecological studies. *Finnish Game Research* **46**: 43-52.
Rolstad, J.; Wegge, P. & Larsen, B.B. (1988) Spacing and habitat use of capercaillie during summer. *Canadian Journal of Zoology* **66**: 670-679.
Sachot, S.; Perrin, N. & Neet, C. (2003) Winter habitat selection by two sympatric forest grouse in western Switzerland: implications for conservation. *Biological Conservation* **112**: 373-382.

Schroth, K.-E. (1992) *Zum Lebensraum des Auerhuhns (Tetrao urogallus L.) im Nordschwarzwald,* PhD-thesis, Forstwissenschaftliche Fakultät der Universität München, München.

Segelbacher, G. (2002) *Genetic structure of capercaillie populations: a non-invasive approach at multiple spatial scales,* PhD-thesis, Wissenschaftszentrum Weihenstephan für Ernährung, Landnutzung und Umwelt. Technische Universität München, München.

Sjöberg, K. (1996) Modern forestry and the capercaillie. M. DeGraaf & R.I. Miller (Eds.). *Conservation of Faunal Diversity in Forested Landscapes,* Chapman & Hall, London 111-135.

Storch, I. (1995) Annual home ranges and spacing patterns of capercaillie in Central Europe. *Journal of Wildlife Management* **59**: 392-400.

Storch, I. (1997) The importance of scale in habitat conservation for an endangered species: the capercaillie in Central Europe. J.A. Bissonette (Ed.). *Wildlife and landscape ecology: effects of pattern and scale*, Springer-Verlag, New York 310-330.

Storch, I. (1999) Auerhuhnschutz - Aber wie? Ein Leitfaden. Page 43. Wildbiologische Gesellschaft München e.V., Ettal.

Storch, I., (Ed.). (2000) *Grouse status survey and Conservation Action Plan 2000-2004.* WPA/BirdLife/SSC Grouse Specialist Group. IUCN, Gland. Switzerland and Cambridge, UK and the World Pheasant Association, Reading, UK.

Storch, I. (2002) On spatial resolution in habitat models: can small-scale forest structure explain Capercaillie numbers? *Conservation Ecology* **6** (1) [online].

Suchant, R. (2002) *Die Entwicklung eines mehrdimensionalen Habitatmodells für Auerhuhnareale (Tetrao urogallus L.) als Grundlage für die Integration von Diversität in die Waldbaupraxis,* PhD-thesis, Forstwissenschaftliche Fakultät. Universität Freiburg, Freiburg.

Suter, W.; Graf, R.F. & Hess, R. (2002) Capercaillie (*Tetrao urogallus*) and avian biodiversity: testing the umbrella-species concept. *Conservation Biology* **16**: 778-788.

Thompson, C.M. & McGarigal, K. (2002) The influence of research scale on bald eagle habitat selection along the lower Hudson River, New York (USA). *Landscape Ecology* **17**: 569-586.

Wiens, J.A. (1989) Spatial scaling in ecology. *Functional Ecology* **3**: 385-397.

The persistence of a key butterfly species *Boloria selene* (small pearl-bordered fritillary) in Clocaenog Forest and the impact of landscape-scale processes.

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Abstract

The heterogeneity of the matrix and the spatial and temporal patterns of habitat patches are both important when considering butterfly species persistence in a forest network. In Clocaenog Forest dispersal potential and changes in patch occupancy of the small pearl-bordered fritillary have been assessed through fieldwork. Ecological parameters derived from the fieldwork have been used to model functional patch networks within the forest, and also to assess least cost movement routes for butterfly dispersal. Models will also be produced to assess the likely impact of continuous cover forestry on the small pearl-bordered fritillary patch networks in the future and to influence the future management.

Introduction

Species research has often focused on small-scale site-based approaches, but more research emphasis on the response of species to landscape-scale processes is needed. One area that has been the subject of research within population ecology is metapopulation theory, the idea that a series of populations are connected by occasional dispersal. Each occupied site has a probability of becoming extinct and the unoccupied ones of being re-colonised. The long-term viability of a metapopulation is dependent on these occasional colonisation events and any processes affecting colonisation will play an important role. One of these factors is the nature of the landscape, and in particular the surrounding matrix between patches of suitable habitat. The permeability of the matrix for dispersal will differ by habitat from species to species, but it is important to understand the nature of landscape permeability and connectivity within forests if species are to persist.

The small pearl-bordered fritillary and Clocaenog Forest

This project was undertaken collaboratively by Butterfly Conservation and Forest Research (FR). It constitutes part of FR's Landscape Ecology Research Project, which aims to improve understanding of how biodiversity responds to forest management at a landscape scale, and to translate this into practical management guidance. A key principle is that particular spatial and temporal patterns of habitat patches will influence the ability of species to survive in and disperse through the landscape.

We studied *Boloria selene* (small pearl-bordered fritillary) in Clocaenog Forest, Denbighshire, UK. The butterfly occurs in discrete patches, where the larval foodplant, violets, grow in damp grassy vegetation and is an ideal model to examine the impact of landscape-scale management and functionality. The habitat patches occur in a heterogeneous matrix, which has differing resistance to butterfly dispersal across the whole landscape. An indication of the potential of the small pearl-bordered fritillary to move through this heterogeneous matrix is modelled and shown as least-cost movement routes.

The small pearl-bordered fritillary is a species of conservation concern in the UK BAP, and it has undergone a severe decline, most noticeably in England. It is locally abundant and widespread in Scotland and Wales although there is some evidence to suggest that it is declining here as well. These declines have coincided with widespread land-use changes such as afforestation, over-grazing, improvement of grassland and cessation of coppicing which all threaten habitat suitability for the butterfly. In woodland habitats the small pearl-bordered fritillary occurs in open rides and glades where the larval foodplant, violets are abundant in damp, grassy vegetation. These habitats typically occur in discrete patches and within most forest systems the butterfly probably exists as a single metapopulation. The species therefore provides a good model to examine the impact of landscape scale management within an extensive forestry system.

Clocaenog Forest is situated in north-east Wales close to Ruthin in Denbighshire. It covers an area of approx 5,500 ha and is dominated by managed stands planted from the mid 1950s. These stands are dominated by conifers (sitka spruce, Norway Spruce and Scots pine). The non-forest habitats include small areas of open grassland and managed riparian areas (areas associated with stream catchments).

The aims of the small pearl-bordered fritillary fieldwork were:

- To determine locations and status of existing colonies in Clocaenog Forest.
- To undertake mark-recapture experiments to determine mobility.
- To assess key habitat factors and identify suitable but unoccupied habitat patches.

The fieldwork was conducted over three field seasons, with over 90 sites surveyed, timed counts were undertaken on each suitable site each year, to build up a picture of occupancy within the forest. Mark-release-recapture (MRR) was undertaken 2001/02, to determine mobility and to accurately assess population size and structure. The MRR showed that within the forest the species occurs in discrete 'colonies', with most individuals remaining at their natal patch. Habitat assessments were also undertaken each year, to determine key factors affecting occupancy and dispersal routes.

Occupancy changes and mobility: evidence for a metapopulation

Suitable sites for the butterfly were identified within the forest during 2001-2003. These sites were either; continuously occupied for the whole study period, occupied for part of the time or not occupied at all during the three years. Likely colonisation events were also identified. This dynamic structure indicates that the system is displaying classic metapopulation attributes, characterised by re-colonisation and extinction events over time.

Timed counts were undertaken on all suitable habitat identified in 2001/02 and on the new sites found in 2003. There are 30 sites that are suitable for the small pearl-bordered fritillary in Clocaenog Forest, of which 28 were occupied at some point during the study period. Changes in site occupancy are summarised in Table 1 and Figure 1. Two likely colonisation events have also been identified. In 2003 there were four medium (101-1,000 individuals) populations and the rest were small (1-100). This contrasts to 2002 where all the populations were small, and 2001 where all the populations were small, apart from one that was medium in size. The small size of populations (relative to populations in the UK generally) suggests that extinction rates maybe higher in this complex, highlighting the need for high connectivity and hence re-colonisation ability within the system.

Table 1. Summary of site occupancy

Attribute	**No. of Sites**
Sites continuously occupied	16
Sites occupied in two years	3
Likely colonisations	2
New sites in 2003	7
Suitable but unoccupied sites	2
Sites no longer suitable	4
Total	34

Figure 1. Changes in small pearl-bordered fritillary occupancy in Clocaenog Forest

(a) 2001 (b) 2002

(c) 2003

In 2001 and 2002 MRR experiments showed that the small pearl-bordered fritillaries were moving between sites; nine individuals had travelled minimum distances varying from between 0.8-3.4km. The possible colonisation events that have been recorded (one colonisation in 2003, one colonisation in 2002 and two recolonisations in 2003) would have required minimum distances to be travelled of between 350m-1,000m from the nearest occupied site. In 2001 the mean distance moved by the small pearl-bordered fritillary was 90m, with 65% of movements less than 50m. In 2002 the mean was 48m, with 76% movements less than 50m (recapture rates were similar in both years).

Functional networks of patches

The matrix is often considered as a "sea of unsuitable uniform habitat" surrounding the habitat patches. We hypothesise that the varying nature of the matrix affects this forest system in the following ways:
(1) Barrier effects from the matrix affect occupancy through shading and the ability of the butterfly to find/leave patches.
(2) The "permeability" of the matrix defines the most likely, least cost routes that butterflies will use to disperse through the landscape.

The functional networks of patches determine the persistence of this butterfly in the forest. The effective area and effective isolation of the patches can be measured and an idea gained about the current health of the metapopulation in the forest. The effective isolation is made up of the distance to the nearest patches and also the permeability of the matrix in between the patches. The effective area of the patch is a function of the size and quality of the patch. Small pearl-bordered fritillary has a relatively low dispersal and needs good quality habitat; their needs can be ranked on a scale against the tolerance of effective isolation and the requirements for effective area. We have measured the size of all patches, have some qualitative information on the habitat condition/quality and have measured the distance between patches. From these parameters, functional networks within the forest can be assessed.

Figure 2. Connected and isolated functional networks

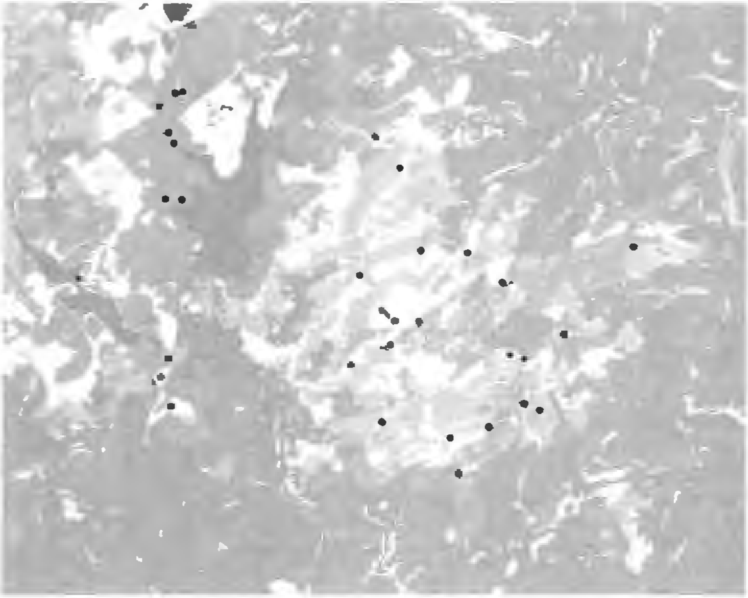
Connected

Isolated

The well-connected functional network in Figure 2 shows a group of small pearl-bordered fritillary sites situated in open riparian areas. The sites have good quality habitat and the butterflies can disperse relatively easily between them. The isolated functional network contains sites where dispersal in between is blocked by mature conifer. The habitat suitability of the sites for the small pearl-bordered fritillary is lower as well.

The whole forest can be assessed in terms of resistance to movement through the heterogeneous nature of the matrix in between the small pearl-bordered fritillary sites. In Figure 3 the darker areas are more resistant to movement than the lighter areas.

Figure 3 Movement resistance based on land cover and the location of riparian areas



Least cost movement routes

From these analyses of resistance to movement of the landscape matrix, least cost movement routes for the butterflies can be modelled. These give us an idea of the likely movement routes that the butterflies may use when they disperse through the forest. Figure 4 shows the straight line routes between sites, where movements have been recorded from mark-release-recapture in the field. It is unrealistic to assume that the dispersing butterfly would take these routes e.g. over the top of hills and through thick forest cover. The more likely routes (Figure 5) have been produced using spatial data, taking into account factors such as the topography, tree cover, wind direction etc. From this the least cost movement routes for the butterflies are calculated. It is likely that the butterflies will follow rides, lower contours, open areas and along watercourses in habitats that have a lower cost for movement associated with them.

Figure 4. Shortest routes travelled by the butterflies (euclidean distance)

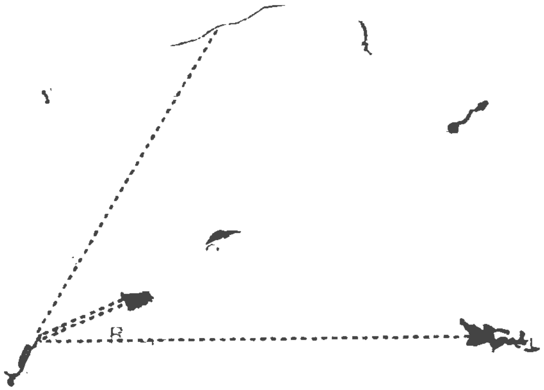
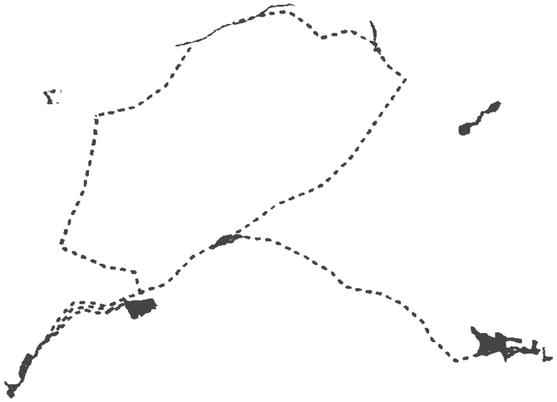


Figure 5. Least cost movement routes: theoretical routes taken by the butterflies based on analysis of topography and tree cover

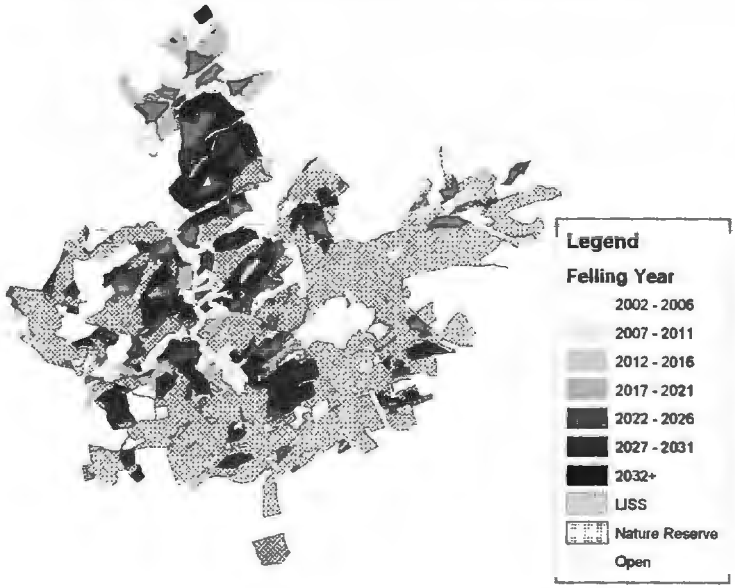


Temporal landscape changes (continuous cover): effects on the butterfly

A target of 50% of State woodlands in Wales has been agreed to go into continuous cover (Lower Impact Silvicultural Systems: LISS) over the next 20 years, and Clocaenog Forest has been selected as a pilot area. This could be a threat to the small pearl-bordered fritillary as a number of the good sites identified are in recently clear-felled areas.

Figure 6 shows the areas to be targeted in Clocaenog for continuous cover forestry. Using this data, maps showing the change in suitability for the butterfly as the forest cover changes can also be produced. This time series, predicting changes in occupancy, will be able to be tested in the field in the future. We will be able to tell whether the proposed plans for LISS and the other changes in forest cover will have a negative impact on the small pearl-bordered fritillary in Clocaenog Forest. Some functional networks may become unsuitable, but other areas may be opened up.

Figure 6. Felling plan showing areas highlighted for LISS in the future



Concluding remarks

The small pearl-bordered fritillary is being maintained within the Clocaenog Forest complex. The species is operating as a metapopulation, with all sites being important for overall persistence. The butterfly is relatively sedentary, with most individuals staying at their natal patch, but is capable of dispersing at least 3km. In the forest network if this distance is calculated as a straight line, then no patch is too far away for colonisation. Analyses based on the heterogeneity of the landscape, and the resistance of the matrix to dispersal, show that the butterflies are unlikely to take straight routes between sites. The effective isolation of a site can be measured by the actual distance away from the nearest patches, but also by the permeability (for dispersal) of the matrix. The effective area can also be assessed; this is a function of the area of the patch and also the habitat quality. These parameters can be used to highlight effective functional networks of patches, areas that are crucial for the long term persistence of a metapopulation. These types of landscape analyses are the kind of results needed from the project to determine how biodiversity responds to the landscape, the matrix and the habitat patches.

Acknowledgements

Thanks to the local Forestry Commission staff at Llanrwst (in particular Iolo Lloyd for his help and interest), our field assistants (Jenny Joy, Andrea Wilcockson, Hannah Woods and Estella St-Pierre), Butterfly Conservation volunteers (Rob Whitehead in particular) and also to Butterfly Conservation staff who gave up some of their time to assist in the field and provided photographs (Tom Brereton and Russel Hobson).

References

Stewart, K.E.J. & Bourn, N.A.D. (2004) *The status, mobility and habitat requirements of the small pearl-bordered fritillary (Boloria selene) in Clocaenog Forest: an overview of three years work*, Butterfly Conservation.

Ancient woodland in the developing landscape: the effects of fragmentation and isolation on carabid assemblages

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Abstract

The ancient woodlands of Greater Manchester are, and have been for some time, fragmented and scattered across the county. The matrix around them is in a constant state of flux and it is theorised that the dynamic nature of the surrounding landcover and landuse will have a direct effect on the biodiversity of these woodlands.

Eleven woodlands covering a range of patch characteristics and locations were examined for their ground beetle (*Coleoptera; Carabidae*) assemblages at both the woodland edge and core. Carabid abundance and diversity was analysed in conjunction with 46 different environmental variables using a correlation matrix. Both carabid abundance and diversity were significantly affected by the external and internal woodland characteristics and the neighbouring landcover border type.

Introduction

Many works have looked at the effect of fragmentation and isolation on habitats and populations at the landscape scale; few however have looked at fragmentation at the patch scale. At this level, limited dispersal abilities coupled with constant disturbance may isolate populations to compartments within a site. McArthur and Wilson (1963) suggested that with isolation there is a reduction in connectivity of the habitats and therefore a genetic isolation of the populations in those habitat islands. Throughout recent years these theories have been refined (Hubbell, 2001) and applied to the terrestrial environment looking at isolation by differing habitats (Saunders, *et al.* 1991; Picket and White, 1985).

Once fragmented, geographical characteristics may have a serious impact on a habitat's ability to act as a reserve for species that would have been present in the original contiguous habitat. The contrasting effects of fragmentation are well documented, being seen as both beneficial in terms of increased edge habitat and also detrimental for the same reason (Parker and MacNally, 2002).

Landscape spatial structure is being altered over increasingly shorter time scales (Fahrig and Merriam, 1994) and at an accelerating rate (Picket and White, 1985). The spatial arrangement of the remaining habitat patches may negatively affect species richness in those remnants (Kareiva and Wennergren, 1995; Franklin and Forman, 1987) by causing an increase in both isolation and the amount of edge habitat (Andren, 1994).

At the landscape scale habitat buffering especially that of semi-natural habitats can significantly increase the habitat core area and has the potential to mitigate landscape scale effects (Woodland Trust, 2000; 2002). Here the Woodland Trust refers specifically to the buffering of semi-natural patches with new native woodland, which may not be feasible within a highly urbanised landscape such as that of Greater Manchester.

Many of Manchester's woodlands are bordered by landcovers that are non-native, artificial (e.g., golf courses) and often urban in nature. However not all of these may be detrimental to the biodiversity of the particular fragment which they surround. In the rural landscape buffering by scrub woodland as a transitional zone between grassland and mature forest may be favoured to maximise biodiversity on the landscape scale. In the absence of native woodland or even natural landcover types to surround the patches the best alternative must be considered in order to maintain biodiversity at the patch scale even if this means using an urban landcover over a more natural one.

Internal woodland characteristics may also play an important role in the habitat dynamics of what are often considered continuous patches. Disturbance patterns within a patch from path networks and desire lines may limit the core area present at that site and create hard boundaries within the woodland. Standard edge zones may be artificially increased by internal disturbance in relation to individual species, as these internal edges may facilitate movement for other generalist species.

This study looks at the effects on woodlands of external characteristics at the landscape scale, and the internal characteristics and landcover borders at the patch scale. These effects are identified through the study of the component ground beetle fauna.

Materials and methods

Site choice

A GIS was used in conjunction with the modified ancient woodland inventory for Greater Manchester (mAWI) (Allen, 2004), based on English Nature's inventory (1994), to identify candidate woodlands out of the present 152 sites for study across the county.

Natural clustering was used to assemble the woodlands into three main groups, those located in Stockport (south east), those in the Bury area (mid county) and those of the Wigan area (north west). Within each of these areas, three woodlands were chosen based on their shape characteristics as defined by the Mean Shape Index (MSI) (McGarigal and Marks, 1994), one wood to represent a small compact shape (low MSI), one to represent an irregular elongated shape (high MSI) and one intermediate shape (scores were analysed relative to the county average and not just the cluster). Throughout the county as a whole the nine woods selected were chosen to represent a range of habitat sizes (again from the mAWI), the smallest being 3.3 ha and the largest 43.2 ha.

In addition to these nine woodlands two other woods were chosen due to their isolation within two different landcover types; one almost entirely surrounded by agricultural land and the other completely surrounded by urban and residential land. Landcover characteristics were defined by the interpretation of aerial photography and classification of the most recent cloud free satellite remote sensing data (Allen, 2004). For comparative reasons the woodlands chosen were of similar size and shape characteristics.

Invertebrate sampling

Once identified, characteristics of the woodlands pertinent to the investigation of biodiversity were derived from the mAWI. These included edge perimeter, core area estimates and possible sites for invertebrate sampling for each woodland.

The edge width of the woodland was set at 10 m. This arbitrary distance was in part determined from the ranging ability of some carabid species. It is likely that some species will be able to cross this threshold. However this examination aims to determine potential effects on general biodiversity by looking at disturbance gradients, therefore, the edge and core zones are merely guides to the general site characteristics.

Core area extents were calculated using Patch Analyst (Elkie, *et al.*, 1999) with the edge dimension of 10 m used as an internal buffer. When sampling, core sample sites were set well within the estimated core area boundary to limit the possibility of an underestimation with regards to the edge zone.

Two sites within each woodland were identified for the collection of the carabid samples. Site visits were necessary for the identification of the exact locations of these collection zones. Most of the woodlands under test are subject to heavy recreational use and therefore these collection sites had to be located away from direct human influence, although apart from this all attempts were made to limit locational bias when laying the traps. Topography in the form of extreme slope gradients often left many areas difficult to sample with vegetation cover (particularly *Rubus* sp.) posing additional problems.

A number of factors highlighted pitfall trapping as the best method for collecting the carabid species. These considerations were time, consistency and conspicuousness of the traps themselves. Time was a major factor due to the frequency of the collection period as the area covered by the study is approximately 128,584 ha. Initial site visits suggested that only three sites could be sampled within one day. With there being five locations across the county this would take four days to complete, with the two isolated woodlands being sampled on the same day. For a full county collection to take place the minimum collection return would be fourteen days (excluding weekends from potential collection days).

The layout of the pitfall trapping was designed to obtain a qualitative representation of the carabids at any one site as an assessment of biodiversity rather than just number of individuals present. For this reason a grid pattern trap layout was employed with two metre spacing between the pots in a grid of nine (3 x 3). If a grid could not be created due to unavoidable conditions such as topography, tree presence or surface bedrock, individual pots were placed as close to the grid layout as possible.

Due to the time scale of the collection rotation, 550 ml pitfall traps were used in the grid to ensure that they would not fill and enable some individuals to escape. The double trap method whereby two pots are inserted to allow ease of removal during collection was also used (Wheater and Cook, 2003). A 50 % methanol solution at a depth of two inches was used as a preservative in the traps. Methanol was chosen over ethylene glycol due to the environmental sensitivity of the areas and the size of the traps involved.

Using these specifications two grids were laid at each site, one at the edge and another in the core and sampled every fortnight for the period of one year beginning in August 2001.

Defining environmental characteristics

Three groups of environmental characteristic were recorded for the eleven woodland sites: the woodlands' external characteristics (EC), looking at its physical dimensions (size, shape etc.); the woodlands' internal characteristics (IC), looking at the effects of internal fragmentation and path networks; and finally the landcover borders surrounding the woodland (LC). The specific details of these variables can be found in Allen (2004).

The woodlands' external characteristics were calculated from the ancient woodland polygons (mAWI) using the Patch Analyst and X-Tools (Delaune, 2004) extension scripts within the GIS ArcView (ESRI, 1992).

Each woodland's internal characteristics were defined from the internal path network, which was measured (using a pedometer, compass and GPS) by plotting the internal paths and desire lines relative to the woodland boundary. These were then digitised onto the ancient woodland polygons and patch characteristics were calculated for each new compartment to provide new statistics for the woodland as a whole.

Finally the landcover border characteristics were identified using the ancient woodland polygons overlaid onto AD2000 digital aerial photography (CRV, 2000). The border length was measured using onscreen digitisation tools; a percentage was derived by comparing the border length to the woodland perimeter.

Results

Over the twelve month study period 13,548 individual carabids were collected from 22 genera and 47 species. There were on average 24 species per woodland with a slight difference in the number of species in the different woodland zones, with on average 19 species at the woodland edge and 18 species in the woodland core. A correlation matrix constructed using the Pearson's product moment correlation coefficient (*r*) and a two tailed significance value (Table 1) was used to compare the carabid data with the three major environmental characteristic groups.

The strongest correlation within the matrix was between the carabid number in the core (CNC) and the percentage of core area loss. This was further supported by the strong positive correlation between the CNC and the number of paths within the woodland, a potential causal factor of core area reduction. Consequently there are similar correlations between the total carabid number (CN) and these two variables. Both CN and CNC are correlated to the number of patches created within the woodlands as a result of internal fragmentation, again a potential consequence of the number of paths. The other correlation for CNC shown in the matrix is with the residential landcover border.

Table 1. Correlation matrix using Pearson's product moment correlation coefficients of relationships between carabid abundance and diversity with environmental variables.

Environmental Variable	CN	CNE	CNC	SN	SNE	SNC
ER (m)		0.619*				
MSI					0.615*	
AWMSI					0.615*	
SCA:IFCA		0.676*			0.612*	
IFCA % of SCA		0.676*			0.612*	
CA % Loss	0.795**		0.805**		0.596*	
NumPa	0.676*		0.704**			
NumPT	0.691**		0.656*			
DW (%)				0.619*		
Res (%)			0.653*			
RD (%)					-0.624*	
RW (%)					-0.698**	

CN = Carabid number; SN = Species number, E and C denote whether the individuals were caught at the woodland edge or core respectively; ER = Elevation range; MSI = Mean Shape Index; AWMSI = Area Weighted Mean Shape Index; SCA:IFCA = Standard Core Area to Internally Fragmented Core Area ratio; IFCA% of SCA = Internally Fragmented Core Area as a percentage of the Standard Core Area; CA% Loss = Percentage of core area loss after internal fragmentation; NumPa = Number of Paths; NumPT = Number of internal woodland patches; DW = Deciduous woodland; Res = Residential; RD = Road; and RW = Railway (these last four represent the percentage of the woodland boundary that is bordered by that particular landcover).

* Significant at $P \leq 0.05$ ** Significant at $P \leq 0.01$

The number of carabids recorded at the woodland edge (CNE) shows some correlation with the elevation range. CNE also shows a strong positive correlation to the ratio of standard core area (SCA) and the internally fragmented core area (IFCA) (SCA:IFCA). The same positive correlation is seen between CNE and the IFCA as a percentage of the SCA (IFCA % of SCA).

The number of species (SN) within the woodlands shows only one significant positive correlation to the deciduous woodland landcover border. The number of species at the woodland core (SNC) did not appear to be correlated to any of the environmental variables on test.

The most correlations between the carabid and environmental data were found to be with the number of carabid species recorded in the woodland edge (SNE). In total there were seven (five positive and two negative) significant correlations. The first two correlations were found between SNE and the mean shape index of the woodland (MSI) and the area weighted MSI. In species terms these two variables are in essence the same and will be considered as one for the rest of this section.

As with the CNE the SNE shows a strong positive correlation with the SCA:IFCA as well as the IFCA % of SCA. SNE also shows a strong correlation to the percentage of core area loss (CA%L) from the woodland as a result of internal fragmentation. CN does not show this correlation therefore it is likely that these species are specific to the woodland edge.

The final correlations between the SNE and the environmental variables are to two landcover classes bordering the woodlands; road and railway embankment. Both these correlations are negative indicating that these landcovers are having a detrimental effect on the species number caught at the woodland edge.

When investigating correlations between species and environmental variables it became apparent that there were a large proportion of significant correlations between the different woodland characteristics. Therefore PCA was employed as a data reduction technique to extract the main characteristics of the environmental data as orthogonal factor scores. A second matrix (Table 2) was constructed using '*r*' values with the factor scores replacing the environmental data. Only correlations between the abundance and diversity data and the environmental variables was considered and not between the same variables.

Table 2. Correlation matrix using Pearson's product moment correlation coefficient to show the significant relationships between carabid abundance and diversity with environmental variables reduced to factor scores.

Environmental Variable	CN	CNE	CNC	SN	SNE	SNC
Factor 1 (IC)	0.707**		0.699**			
Factor 2 (IC)						-0.594*
Factor 2 (EC)					0.683*	
Factor 3 (EC)					0.608*	

Factor 1 (IC) = NumPa (Table 1), NumPt (Table 1), Total number of internal Core Patches (TCP). Factor 2 (IC) = Mean Patch Size (MPS), Maximum Internal Patch Size (MaxPS), Mean Core Area (MCA). Factor 2 (EC) = Woodland shape. Factor 3 (EC) = Woodland elevation

* Significant at $P \leq 0.05$ ** Significant at $P \leq 0.01$

In the second matrix there were five significant correlations ($P = 0.05$) two of which were significant to $P = 0.01$. These were the same correlations as within Table 1; the factor scores relating to a grouping of similar environmental variables. The strongest correlations were found between the CN and CNC and factor 1 for the internal characteristics (IC). Factor 1 of the internal characteristics relates to the number of patches, number of paths and the total core patches. These show some similarities with the correlations found with the un-factorised environmental data and the CN and CNC.

The other correlations were between species number and not carabid abundance. Firstly there was a strong negative correlation between SNC and the second IC factor (relating to the mean internal patch size, the maximum internal patch size and the mean core area). The other two significant correlations were for the species number recorded in the woodland edge zone relating to the external characteristics of the woodlands. The stronger of the two correlations was to woodland shape complexity (EC Factor 2). The second correlation was to factor 3 of the external characteristics relating to woodland elevation. Both of these correlations further support those found in the original un-factorised correlation matrix.

Discussion and conclusions

It appears from the correlation matrix that carabid abundance at the site scale is related to the internal path networks and thus the internal fragmentation and patch dynamics of the individual woodlands. This is highlighted by the number of correlations between the abundance data and the woodland internal characteristics. These effects may further enhance the site by site differences seen in both carabid abundance and diversity.

Carabid abundance at the woodland edge increases with elevation throughout the study sites; however this is a range of only 131.4 m. There was no apparent increase in carabid diversity with elevation but a noticeable increase was seen with regards to increasing shape complexity and thus habitat heterogeneity at the woodland edge. Higher woodlands within the county may correlate with increasingly rural areas due to the topography. Increased elevation may therefore increase abundance through lack of disturbance. Carabid diversity in relation to woodland shape, and not elevation, indicates that with more edge habitat and contrasting landcover borders, increased habitat heterogeneity results in more species (Parker and MacNally, 2002). These species may be generalist species and not woodland specialists (Kareiva and Wennergren, 1995).

Carabid abundance for the sites in general and those individuals caught in the woodland cores increases with the number of internal path networks present at the site. There is also a resultant increase in the carabid diversity of those individuals found at the woodland edge in relation to the same variable, probably as a result of increased edge habitat within the woodland.

The number of species recorded within the woodland in general increases with a surrounding landcover of more deciduous woodland. This was the only landcover border to increase the number of species present within any of the sites and may indicate the benefits of woodland buffering (Woodland Trust, 2000; 2002), assuming those species are found to be woodland specialists as opposed to site generalists. There is however a significant increase in carabid abundance at the woodland core in relation to the presence of residential land at the woodland edge. It is hypothesised that the height of residential buildings and relative lack of disturbance and pollution associated with this landcover simulate to some degree the presence of mature woodland limiting abiotic influences such as temperature differentials and the effect of wind at the woodland edge. Work relating to this is ongoing.

The only major negative association for the carabids in general was a significant decrease in the number of species recorded at the woodland edge with close proximity to transport routes (roads and railways). Other diversities and abundances show negative associations to transport routes as a neighbouring landcover type, however they were not significant in this study. Landcover borders such as these present hard boundaries with high levels of usage and thus disturbance to the neighbouring sites.

The PCA correlation matrix did not highlight an association between carabid abundance or diversity and the landcover border variables. This is in part due to the extraction of the factors for the landcover variables from the PCA and the close correlation of some factors, e.g., grassland and golf courses. Individual species response to the environmental characteristics is currently being examined using ordination techniques to determine small scale disturbance to woodland specialists. This may also highlight correlations not seen within the grouped carabid data or factorised environmental variables.

References

Allen, B.R. (2004) *The ancient woodlands of Greater Manchester distribution and characteristics in a dynamic matrix.* Unpublished PhD thesis. Manchester Metropolitan University.

Andren, H. (1994) Effects of habitat fragmentation on birds and mammals in landscapes with different proportions of suitable habitat. A review. *Oikos* **71:(3)** 355 – 366

Cities Revealed. (2000) *CD 1: Areas 1 – 3. Embedded meta statement.* www.crworld.co.uk

Delaune, M. (2004) X-Tools for ArcView 3.x. http://www.odf.state.or.us/DIVISIONS/management/state_forests/GIS/Documents/xtools.htm

Elkie, P. C., Rempel, R. S. & Carr, A. P. (1999) *Patch Analyst user's manual: A tool for quantifying landscape structure. NWST Technical manual TM - 002.* Boreal Science, Ontario. http://flash.lakeheadu.ca/~rrempel/patch/paregform4.html

English Nature (1994) Digital ancient woodland boundary data for 100km tiles, SD, SE, SJ & SK (www.english-nature.org/pubs/gis/GIS_selection.asp)

ESRI (1992) ArcView GIS 3.2. Environmental Systems Research Institute, Inc. http://www.esri.com/

Fahrig, L. & Merriam, G. (1994) Conservation of fragmented populations. *Conservation Biology.* **8** 50 - 59

Franklin, J.F. & Forman, R.T.T. (1987) Creating landscape patterns by forest cutting: ecological consequences and principles. *Landscape Ecology* **1:** 5 – 18

Hubbell, S.P. (2001) The unified neutral theory of biodiversity and biogeography. *Monographs in Population Biology 32.* Princeton University Press

Kareiva, P., & Wennergren, U., (1995) Connecting landscape patterns to ecosystem and population processes. *Nature* **373:** 299 – 302

McArthur, R.H. & Wilson, E.O. (1963) An equilibrium theory of insular zoogeography. *Evolution – International journal of organic evolution.* **17:4** 373 – 387.

McGarigal, K. & Marks, B.J. (1994) Fragstats: spatial pattern analysis program for quantifying landscape structure (*Version 2*)

Parker, M. & MacNally, R.M. (2002) Habitat loss and the habitat fragmentation threshold: an experimental evaluation of impacts on richness and total abundances using grassland invertebrates. *Biological Conservation* **105:** 217 - 229

Pickett, S.T.A & White, P.S. (Eds.) (1985) *The ecology of natural disturbance and patch dynamics.* Academic Press, New York.

Saunders, D.A., Hobbs, R.J. & Margules, C.R. (1991) Biological consequences of ecosystem fragmentation: a review. *Conservation Biology* **5:1** 18 – 32

The Woodland Trust (2000) *Woodland biodiversity: expanding our horizons.* (www.woodland-trust.org.uk/policy/publications.htm)

The Woodland Trust (2002) *Space for nature: landscape-scale action for woodland biodiversity.* (www.woodland-trust.org.uk/policy/publications.htm)

Wheater, C. P. & Cook, P.A. (2003) *Studying invertebrates.* Naturalists' handbook 28, Richmond Publishing Co. Ltd, Slough.

Landscapes with ancient trees: invertebrate mobility and population viability

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Abstract

There is a widespread misconception that Britain has lost all of its "old growth" forest and, with it, its old forest communities. However, the cultural landscape preserves many areas of relatively unmodified old growth habitat in the guise of ancient wood pastures and historic parklands. Relatively species-rich assemblages of the key old growth species assemblages (epiphytic, saproxylic, and mycorrhizal species) are centred on these modern concentrations of ancient open-grown trees.

A proportion of these core sites have some limited protection through legal and other designations; many are in relatively sympathetic ownership. But as our understanding of landscape ecology and population dynamics develops, and with the new impetus provided by man-made climate change, it is becoming increasing apparent that conservation of these old growth communities needs a wider vision that: recognises the need for renewed linkages between fragmented and isolated populations; acknowledges the importance of ancient trees in the wider landscape; and thereby promotes population viability through enabling species mobility. These issues will be illustrated by a series of case studies of saproxylic beetle assemblages within the cultural landscape, based on specialist invertebrate surveys in historic parklands, traditional orchards, and conventional farmland.

Introduction

"Old growth" is a term familiar to most Europeans, primarily from the Pacific North West of America where it was used by conservationists to promote the protection of primary coastal conifer forests. But its usage in Europe has until recently been very limited; it has been assumed that it only refers to primary forest and therefore does not apply in Europe. This does raise difficulties; however, questions of definitions arise. There is no doubt that the particular forests in America have been used by native Americans for centuries, so why should European forests used by native Europeans for centuries automatically be excluded? How does one define the degree of human activity that precludes the use of the term?

The applicability of the term "old growth" in Britain and elsewhere in Europe has recently been reviewed in a report commissioned by English Nature (Alexander *et al.*, 2002). The report recommended the adoption of the definition provided by the Convention on Biological Diversity (CBD) web-site: '*old growth forest stands are stands in primary or secondary forests that have developed the structures and species normally associated with old primary woodland of that type that have sufficiently accumulated to act as a woodland ecosystem distinct from any younger age class.*' The report recommended that wood pastures and parklands are considered to be included within the term "forest" in the European context, and identified these particular habitat types as the key areas in Britain and elsewhere in Europe for those "species normally associated with old primary woodland".

The Woodland Trust has been ahead of the pack in adopting old growth as a useful term in the UK (Woodland Trust, 2000) and more recently Forest Research have similarly identified wood pastures and parklands as Europe's old growth (Humphrey, in press).

The CBD definition relies on the ability to recognise "structures and species associated with old primary woodland". There is considerable debate at the moment about the structures and it will be a while before this is resolved. Vera (2000) has thrown down a very heavy gauntlet that challenges the long-established belief that primary forest in Europe had a closed canopy high forest structure. As Rackham (2003) says, "human activities are such a long-standing and continuous factor that it is now hardly profitable to speculate on what woods would be like without them" and "we know next to nothing about numbers and sizes of trees in the wildwood", so primary woodland structures are really something for the academic world to worry about and are of no help to practising conservationists.

However, primary woodland species are something we can work with. And beetles have the huge advantage of being tough creatures which preserve notably well as sub-fossils in recent sediments. We are steadily building up a useful picture of the species composition of the post-glacial forests (e.g., Dinnin & Sadler, 1999). Buckland and Dinnin (1993) have concluded that the remaining refuges for these insects are the ancient wood pastures, and that these species are hanging on in areas of parkland, "a managed habitat which is perhaps the oldest in Britain". We know that many of the species present in undisturbed primary forest are still present in the modern countryside, but now concentrated into those surviving examples of ancient wood pastures and historic parklands, species like: the weevil (*Dryophthorus corticalis*), the woodworm beetle (*Gastrallus immarginatus*) and the lime bark beetle (*Ernoporus caucasicus*). Of our 700 odd native beetle which depend on decaying wood 180, more than one quarter are regarded as characteristic of ancient wood pastures and historic parklands (Alexander, 2004).

These species are clearly indicators of ecological continuity with old growth forest. They are sometimes referred to as indicators of "ancient woodland", a term that tends to be used much more specifically for woodlands which were enclosed many centuries ago and which have developed particularly rich ground vegetation through centuries of protection from grazing. These woodlands were enclosed, so that timber and wood could be systematically exploited and, so, these are not the places rich in old growth beetles (Alexander, 1996). Ancient wood pastures and historic parklands are something of a Cinderella when it comes to the term ancient woodland, sometimes included in the guest list but more often forgotten. Very few feature in the Ancient Woodland Inventory and there is no obvious pattern behind their inclusion or exclusion. The term "old growth" provides them with their rightful prominence.

So, we now have our ancient beetles recognised as indicators of old growth forest. But why are they confined to these areas? What is it about these relict areas that are so good for these beetles? Habitat quality is part of the answer but a key consideration is their mobility, their lack of ability to spread out from these islands of old growth habitat and colonise new areas. Large old trees apparently suitable for these species have developed in many areas of the modern countryside but they are all too often far removed from the isolated relict old growth areas and the beetles are unable to reach them and establish new colonies.

Species mobility is notoriously difficult to study. A strong correlation between relict old growth species and relatively low mobility values has long been suspected and there is much circumstantial evidence. Old growth invertebrates have been demonstrated colonising suitable trees *within* old wood pastures and parklands, and extending short distances outside of the fragment of their habitat in which they live. However, evidence for movement *between* sites is extremely rare, and evidence for colonisation of *new* sites is non-existent. Alexander (2003b) has recently shown that one saproxylic beetle species (*Gastrallus immarginatus*) has not been able to cross a distance of just 500m between suitable field maple (*Acer campestre*) host trees in about 400 years. Brunet (2003) presents further evidence from Sweden that certain wood-decay beetles demonstrate severe dispersal limitation at distances farther than about 200m.

A proportion of the core old growth sites have some limited protection through legal and other designations; many are in relatively sympathetic ownership. But as our understanding of landscape ecology and population dynamics develops, and with the new impetus provided by man-made climate change, it is becoming increasing apparent that conservation of these old growth communities needs a wider vision that: recognises the need for renewed linkages between fragmented and isolated populations; acknowledges the importance of ancient trees in the wider landscape; and thereby promotes population viability through enabling species mobility.

These ideas will now be illustrated by a series of case studies of work carried out by the author, partly from private studies and partly from project work carried out under contract for a variety of organisations.

Case studies 1: Medieval forest country

Many of the former medieval forests of Britain are detectable in today's landscape as concentrations of enclosed woodlands in a matrix of intensive agricultural land. These were originally a patchwork of enclosed coppice-with-standards woodlands within open rough pastures with scattered pollarded trees, as can be seen today at Hatfield Forest in Essex, but all too often the open pastures have been converted over the years to increasingly intensive agricultural use. Some of the old open forest trees may have survived in hedgerows and a few within pasture fields and even occasionally within arable land.

Mottisfont is a large lowland estate in the Test Valley of Hampshire. It is of very high nature conservation value, partly for its many old field and hedgerow trees which lie within the farmed matrix. There are also small enclosed ancient woodlands scattered throughout. Older generation trees throughout the estate have been found to support a nationally important wood-decay invertebrate fauna characteristic of relict old forest, and especially the open-grown trees of the fields, hedgerows and wood edges (Alexander, 2001).

The discovery of an important relict old forest wood-decay fauna at Mottisfont should be no real surprise. The estate lies on the south-eastern fringes of the former medieval forest of Buckholt which was part of Clarendon Forest until the early 13th century. Additionally the medieval Forest of Bere lies immediately to the east, the New Forest lies to the south, and large old trees are widespread on the estate.

A key feature of the estate is the association of the rarest old forest beetles with the large open-grown trees and not the enclosed woodlands. Where a relict old forest beetle species was found in association with an enclosed wood it was invariably on a boundary oak and not in the shady heart of the woodland. The best find has been the British Red Data Book beetle (Hyman, 1992) *Scraptia testacea*, the only modern record for Hampshire. The Index of Ecological Continuity (IEC; Alexander, 2004) has reached a value of 28, making Mottisfont of national importance for this fauna.

Sherborne Park, Gloucestershire was developed in the 16th century from the extensive wood-pastures which surrounded Wychwood Forest (A. Mayled, pers. comm.) and is outstanding in its wood-decay beetle fauna for a Cotswold dip-slope location. The IEC is 17, of regional importance (Alexander, 1993). Examples of similar situations include Stourhead Park, Wiltshire (Alexander, 1996) and Forthampton Manor, Gloucestershire (Alexander, 2002).

These examples all show how some of the old growth fauna of the postglacial forest of Britain has survived changing human land-use patterns through medieval Forest declarations into historic parklands and even former wood pasture trees being incorporated into modern intensive farmland. They are examples where the wood-decay beetle species have been able to move about within the ancient tree matrix but not spread out into the surrounding landscape.

Case studies 2: the old orchard landscapes of the Three Counties

The Three Counties area (Gloucestershire, Herefordshire and Worcestershire) is a part of England which has had a high proportion of medieval forest in the past, some still remaining in rather parlous state as the Forests of Dean and Wyre, others now largely gone and forgotten such as the Chase of Malvern, already referred to above. It is an area well known amongst entomologists for its rich and interesting wood-decay invertebrate fauna, especially for relict old forest species. Areas of special interest for wood-decay beetles include not only the old parklands, notably Moccas Park (Harding & Wall, 2000), but also the old rough pastures of the Cotswold escarpment with their scattered ancient pollard trees and, perhaps more surprisingly, old orchards.

Parts of the Three Counties have long been famous for fruit growing and old orchards are present at high concentration across the landscapes of northwest Gloucestershire and extending into the neighbouring counties of Herefordshire and Worcestershire. These orchards are very much at landscape scale and have in the past linked the old medieval forests and chases together and extended the old open-grown tree resource across enormous sweeps of countryside. This has enabled some of the relict old forest species to spread out of the historic forest boundaries. Old orchards which survive in these areas today are a major reservoir of relict old growth wood-decay beetles. The old literature often mentions records of interesting wood-decay beetles from fruit trees but knowledge of the rich orchard fauna largely died out in recent decades until Roger Key of English Nature put one key species forward for the Priority List of the UK Biodiversity Action Plan: noble chafer (*Gnorimus nobilis*). This large and colourful beetle has mainly been known in England from old orchards, although elsewhere in Europe is better known as a relict old forest species. It was thought to be on the verge of extinction in Britain, with very few recent sightings.

Knowledge of the habitat requirements of noble chafer has subsequently blossomed through research commissioned by the Lead Partner for the Species Action Plan, the Peoples Trust for Endangered Species (PTES). We now appreciate that the species needs large populations of old open-grown trees which have been hollowed by heartwood decay fungi and that, when the adult beetles are actively flying between trees seeking mates and fresh larval habitat, they are attracted to the large white flowers of plants such as elder, meadowsweet and umbellifers such as hogweed (Alexander, 2003a). Such conditions are no longer found in the old forests, due to changing management priorities, but the old orchards are ideal habitat.

Work organised by the PTES has now shown that there is one major population of this rare beetle left in Britain and that it is scattered across the old orchards of the Three Counties, with a few local hotspots where the species is active in almost every suitable tree. Away from this major noble chafer landscape the species has been seen, in very low numbers only, in one area of the New Forest where hogweed is able to flower protected from grazing by a fenced roadside section, and also one small area in the south Chilterns. It formerly occurred much more widely, in suitable areas virtually throughout lowland England, from Devon to Cumbria.

The discovery of such a major population of noble chafer, sprawling at landscape scale, was amazing, but even more amazing has been the increasing realisation (rediscovery) that these old orchards are incredibly rich in wood-decay invertebrates more generally. These old orchards are proving to be a major refuge for relict old forest insects generally (Alexander, in prep.), species driven out from other land-use systems through the increased disappearance of ancient trees.

Case studies 3: old wood pasture landscapes

Remaining briefly in this part of the country, the important Cotswold escarpment was mentioned earlier. This is another important landscape of ancient trees, this time scattered along the rough pastures of the lower landslip slopes along the foot of the Cotswold Escarpment. The land has always been too rough to plough and, even for modern agriculture, much is too difficult to get a tractor and trailer onto to fertilise. Rough pasture has been the main land use throughout history, although afforestation has accounted for some sections in recent years. Inaccessibility has also protected the trees from use for timber and the older trees are predominantly pollards, the trees having been exploited for wood and fodder. These old trees form a strip of open wood pasture which stretches from near Bath up to beyond Chipping Campden, although it is mainly the section north from Cheltenham that is the richest in ancient trees and wood-decay invertebrates. Here there are populations of rare wood-decay beetles of European significance, notably violet click beetle (*Limoniscus violaceus*) (Whitehead, 1996), listed for special protection under the EU Habitats and Species Directive and full protection under the Wildlife & Countryside Act 1981.

A smaller scale version of this situation has recently come to light in Broadclyst Parish, Devon, where old common wood pasture trees, mainly ancient oaks, survive widely on the National Trust's Killerton Estate (Spalding Associates, 2003). The estate as a whole includes about 100 ancient trees, making it exceptional in the context of Devon and Cornwall. The trees are older than the mansion of Killerton House and its designed landscape. These trees are relicts of the older agricultural landscape of this area. The ancient pollards are concentrated around the higher ground and especially along old route-ways up onto what was then the common grazings of the parish; the trees are effectively former wood-pasture trees. The old deer park of Killerton was almost certainly located there because of the existing large old trees which created an instant parkland effect.

Overall, the wood-decay beetle fauna of the Killerton Estate is one of the richest anywhere in Devon, only Whiddon Park and possibly Shute Parks have a richer fauna. Although the fauna is concentrated in Killerton Park and around Ashclyst Forest, the foci of the ancient wood-pasture trees, the rare and uncommon species also occur widely within the surrounding agricultural landscape. Its successful conservation is dependent on active measures throughout the estate, not just in the core areas.

Case studies 4: other examples

While most of the preceding studies link the fauna with the landscapes in and around former medieval forest, the Cotswold escarpment pastures appear to be an exception, with no obvious direct links with old forest landscapes and perhaps have links with the original postglacial forest without an intermediary hunting forest stage. There are other examples and some detail will be provided of two, which illustrate two different ways in which the wood-decay fauna and landscape history of trees are linked together: in Shapwick parish, Somerset, and on the Buscot & Coleshill Estates, Oxfordshire.

The Shapwick study (Clements & Alexander, 2002) set out to compare the invertebrate faunas of a range of hedgerows in the parish of Shapwick in Somerset. These hedgerows were chosen on the basis of historical evidence so as to represent a wide range of presumed dates of origin, varying from ancient parish boundary and medieval hedgerows through to more recent enclosure hedges. Shapwick parish benefits from a very extensive and detailed, long-term programme of research into all aspects of its history and archaeology, via the 'Shapwick Project' coordinated by researchers at the University of Bristol and elsewhere (see summary article by Aston & Gerrard 1999). This project began in 1989 and has continued to the present-day and, as a result, there is excellent documentary, cartographic, historical, topographical and archaeological evidence available relating to the hedgerows of the parish.

The hedgerows were found to support an unexpectedly good range of wood-decay invertebrates, including an interesting range of species which have been associated with sites with ecological continuity, and there were positive correlations between both the presence of these indicator species and total numbers of saproxylic species with hedgerow age, as demonstrated by historical evidence. The study indicated that consideration of the wood-decay invertebrate fauna can usefully inform historical and archaeological assumptions about the age and origin of individual hedgerows, especially when considered alongside other ecological data. It was concluded that old hedgerows are an important habitat resource for relict old growth and wood-decay invertebrates, and can support good numbers of nationally and regionally scarce and local species. The IEC value for the whole network reached 8, a figure comparable with, perhaps even better than, that which might be expected from a moderate-sized enclosed ancient semi-natural woodland (see also Alexander 1996).

The Coleshill and Buscot Estates provide a different situation where old forest wood-decay faunas have survived in a modern agricultural landscape. Alexander & Foster (1999) found that on this estate enclosed and ungrazed ancient semi-natural woodland, and even historic parklands, are relatively poor in wood-decay fauna, while the conventional farmland is relatively rich. The key feature of the estate is that it coincides with low-lying ground along the Thames and its minor tributary the River Cole. The two river floodplains include networks of field drains and hedgerows lined by mature and overmature trees, including old pollards as well as standards. Only a few hundred years ago this would have been grazing marsh country and effectively an open wood pasture system. Once again we have an example of rich wood-decay fauna as a survival from an earlier landscape. Similarly interesting faunas have been noted along other river floodplains (Alexander & Foster, 1999).

Conclusions

The case studies demonstrate that, while many of the old growth beetles, these links with the post-glacial forests, are trapped in isolated and fragmented areas with old growth characteristics, they are capable of spreading into the adjoining countryside to some extent, provided there are old trees to support them. They are also capable of maintaining viable populations there even where the old growth refuge area appears to have disappeared (the Shapwick hedgerow example) or is no longer suitable (the Buscot and Coleshill example).

Suitable habitat can come in a variety of shapes and forms. The ancient wood pastures and historic parklands are the classic examples, and the Sherborne and Stourhead examples show how historic parklands can maintain viable populations of some of these species once the old forest matrix has become largely unsuitable. The Mottisfont, Forthampton and Broadclyst examples show how many of the species can survive in modern farmland provided the old wood pasture trees survive as field and hedgerow trees. The Cotswold escarpment and old orchard examples demonstrate how these rich faunas can extend over very large areas of countryside, at a landscape scale, provided the ancient trees are there to support them. While the Cotswold landscape is actually a very old one, dating back into the medieval period at least, the history of the old orchards is less clear.

In conclusion then, the available evidence suggests that conservation of many of these presently rare and threatened species is possible if:

- old and ancient trees are cherished and protected at a landscape level
- new generations of open-grown trees are established at landscape scale
- priority is given to linking and extending known areas of importance.

These measures should promote population viability through enabling species mobility, possibly even in the face of global climate change.

Acknowledgements

Permission to present unpublished data was kindly given by Valerie Keeble (PTES) and John Channon (National Trust).

References

Alexander, K.N.A. (1993) *Anitys rubens* (Hoffmann, JJ) (Coleoptera: Anobiidae) new to Gloucestershire, and other deadwood beetles from Sherborne Park. *British Journal of Entomology and Natural History* 6: 16.

Alexander, K.N.A. (1996a) The dead-wood beetles of Stourhead, including *Thymalus limbatus* (Fabricius) (Peltidae) new to Wiltshire. *The Coleopterist* 4: 90.

Alexander, K.N.A. (1996b) The value of invertebrates as indicators of ancient woodland and especially pasture woodland. *Suffolk Natural History* (*Trans. Suffolk Nat. Soc.*) **32**: 129-137.

Alexander, K.N.A. (2001) An important wood decay fauna from the Mottisfont Abbey Estate, south Hampshire. *The Coleopterist* **10**:85-86.

Alexander, K.N.A. (2002) Forthampton Oaks, Gloucestershire: a site of major importance for saproxylic invertebrates. *British Journal of Entomology and Natural History* **15**: 63-64.

Alexander, K.N.A. (2003a) Prince amongst beetles. *Mammals UK* **Spring 2003**: 12.

Alexander, K.N.A. (2003b) *Gastrallus immarginatus* (Müller) (Anobiidae) at Hatfield Forest, Essex. *The Coleopterist* **12**: 23.

Alexander, K.N.A. (2004) Revision of the Index of Ecological Continuity as used for saproxylic beetles. *English Nature Research Report* No. **574**.

Alexander, K.N.A. (in prep.) The importance of old orchards for Noble Chafer *Gnorimus nobilis* and other wildlife – ecology, condition assessment and conservation management issues. *English Nature Research Report.*

Alexander, K.N.A & Foster, A.F. (1999) Scarce wood-decay beetles in a river floodplain farmed landscape in the Upper Thames Valley. *British Journal of Entomology and Natural History* **12**: 213-214.

Alexander, K.N.A., Smith, M., Stiven, R. & Sanderson, N. (2003) Defining 'old growth' in the UK context. *English Nature Research Report* **No 494**: 1-54.

Aston, M. & Gerrard, C. (1999) 'Unique, traditional and charming': The Shapwick Project, Somerset. *Antiquaries Journal* **79**: 1-58.

Brunet, J. (in press) Colonization of forest species – patterns in time and space. In: Honnay, O. et al (Eds) *Forest biodiversity: lessons from history for conservation.* CABI Publishing.

Buckland, P.C., & Dinnin, M.H. (1993) Holocene woodlands, the fossil insect evidence. *In:* Kirby, K.J., & Drake, C.M. (Eds) Dead wood matters: the ecology and conservation of saproxylic invertebrates in Britain. *English Nature Science* No. 7, Peterborough, pp. 6-20.

Clements, D.K., & Alexander, K.N.A. (in press) The Shapwick Project: a comparative study of the invertebrate faunas of hedgerows of differing ages. *English Heritage Survey Grant for Publication, coordinated through the University of Bristol.*

Dinnin, M.H., & Sadler, J.P. (1999) 10,000 years of change: the Holocene entomofauna of the British Isles. *Quaternary Proceedings* **No. 7**: 545-562.

Harding, P.T., & Wall, T. (2000) *Moccas: an English deer park.* English Nature, Peterborough.

Humphrey, J. (2002) Poster at *Forest biodiversity: lessons from history for conservation* IUFRO Conference, Leuven

Hyman, P.S. (revised Parsons, M.S.) (1992) *A Review of the scarce and threatened Coleoptera of Great Britain.* Part 1. UK Nature Conservation: 3. Joint Nature Conservation Committee.

Rackham, O. (2003) *Ancient woodland.* Castlepoint Press, Dalbeattie.

Spalding Associates (2003) *Biological survey of the Killerton farmed estate.* Unpublished report to the National Trust.

Vera, F.W.M. (2000) *Grazing ecology and forest history.* CABI Publishing, Wallingford.

Whitehead, P.F. (1996) The notable arboreal Coleoptera of Bredon Hill, Worcestershire, England. *The Coleopterist* **5**: 45-53.

Woodland Trust (2000) *Woodland biodiversity: expanding our horizons.* Woodland Trust, Grantham. www.woodland-trust.org.uk/policy/publications.htm

Session 3

Landscape thresholds

Landscape-level requirements for the conservation of woodland birds: are there critical thresholds in habitat cover?

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Abstract

Patch-level studies of populations and faunal assemblages have provided many insights for nature conservation, but there is a need for a broader 'landscape' perspective. Key issues at the landscape level include the amount of habitat required and the influence of its configuration, and whether there are thresholds of habitat cover below which rapid loss of species occurs? Theory suggests a disproportionate loss of species when total habitat cover decreases to 10-30% of the landscape. However, little empirical evidence has been collected to test for such thresholds. Here, we review results from two studies of the responses of woodland birds to landscape pattern in Victoria, Australia. The first, based on records from the Atlas of Australian Birds, collated data for 'landscapes' of 10' latitude by 10' longitude (~ 270 km^2). The second involved systematic surveys in 24 landscapes, each 10 x 10 km (100 km^2), selected to represent a gradient from 2% to 60% wooded cover. In both studies, species richness of woodland birds declined as wooded cover decreased. There was evidence for a threshold response, with a steep decline in richness in landscapes with less than 10% tree cover. While these studies can be used to guide conservation planning, we caution that they are based on presence or absence of species, with responses averaged across the overall assemblage. There is a critical need for knowledge of the demographic status of populations in heterogeneous landscapes, to assist in planning for the long-term persistence of species in modified rural environments.

Introduction

Concern at the effects of ongoing loss and modification of habitats has stimulated numerous studies into the status of species and assemblages of birds in fragmented landscapes. Much of this research has been carried out at the 'patch-level', i.e., the unit of study is individual patches of habitat and their use by the fauna. This has provided a wealth of new knowledge of how the occurrence of species, and the richness and composition of assemblages, are related to attributes such as the size and isolation of habitat patches, land management and habitat quality, and position in the landscape (e.g., Barrett *et al.*, 1994; Hinsley *et al.*, 1996; Mac Nally *et al.*, 2000).

A consistent theme from such studies is the need to expand the spatial scale of investigation in order to understand processes and population dynamics at the landscape and regional scale (Bellamy *et al.*, 1998; Mac Nally and Horrocks, 2000). First, for most species, single patches of habitat are not sufficient to support self-sustaining populations in the long-term. The status of bird populations in isolated habitats is frequently associated with the amount and location of other suitable habitat in the surrounding landscape. Second, many bird species use resources that are patchy in space and time, which requires them to move among different habitats. This may involve concurrent use of different habitats (e.g., for breeding and foraging resources), or sequential use of habitats to 'track' the availability of resources such as nectar or seeds (Law and Dickman, 1998; Mac Nally and Horrocks, 2000). Third, the landscape context of habitats is an important influence on the avifauna because land-uses and biota in the environment surrounding habitat patches can have a significant impact on species living within the patch. Similarly, ecological processes such as rising water tables and associated salinisation, spread of pest plants and animals, and wildfire, all operate at broad scales. Finally, from a practical perspective, there is general recognition among land management agencies that planning must occur at broad scales, but knowledge to implement biodiversity restoration in landscape planning is limited.

Among the important issues at the landscape level are: 1. How do species respond to the *amount* of habitat present in the landscape? 2. What is the influence of *spatial configuration* of habitat? 3. Are there *thresholds* of habitat cover below which rapid loss of species occurs? To date, these issues have been addressed mainly by computer models (e.g., With and Crist, 1995; Fahrig, 1997), generally based on simple landscape patterns and concerned with extinction thresholds of individual species. There have been few field-based studies in which whole landscapes (rather than patches) have been the unit of study (Fuller *et al.*, 1997; Trczinski *et al.*, 1999; Villard *et al.*, 1999).

The concept of critical thresholds is a topical issue of direct relevance to conservation and land management. Interest in this concept stems from two types of observation (With and Crist, 1995; Wiens *et al.*, 2002): first, that many ecological relationships are non-linear in form; and, second that often such relationships have ranges in which an abrupt or rapid change in an ecological response variable occurs with only minor change in a habitat variable (e.g., Jansson and Angelstam, 1999; Radford and Bennett, 2004). Andrén (1994) suggested that for many species, thresholds may occur when habitat cover declines to approximately 10-30% of the landscape. Above this level, population decline or loss of species is likely to be linearly related to habitat loss (a random sampling phenomenon). The threshold represents a stage at which there is a shift in underlying processes; fragmentation effects such as reduced connectivity exacerbate the effect of habitat loss, and result in rapid decline.

Response of woodland birds to landscape pattern in southern Australia

In Australia, there is much concern about the conservation status of the biota of dry forests and woodlands of the south, due to the extent of habitat destruction and degradation, and consequent changes to ecological processes (Hobbs and Yates, 2000). While most groups of plants and animals have been detrimentally affected, the decline in bird species is most clearly documented (Robinson and Traill, 1996; Ford *et al.*, 2001). Major community initiatives to restore rural landscapes are underway, both to enhance biodiversity outcomes and the sustainability of the land for agriculture. Management agencies are seeking quantitative guidelines for the extent and pattern of vegetation to be restored in the future.

Here, we briefly review results from two studies in Victoria, Australia, into the relationship between species richness of woodland birds and landscape pattern. The main question we address is whether there is evidence for a threshold response by woodland birds to the amount of wooded habitat at the landscape level. Critical thresholds have been defined as "*transition ranges across which small changes in spatial pattern produce abrupt shifts in ecological responses*" (With and Crist, 1995). Consequently, if woodland birds show a threshold response to habitat cover, we would expect to observe an abrupt change in species richness with only small change in the amount of habitat in the landscape.

The main topographic features in the study region are the flat alluvial plains in the north, rarely exceeding 150 m elevation, giving way to gentle hills of sedimentary origin in the north-central area (up to 400 m). Mean annual rainfall ranges from approximately 650 mm in the east to 450 mm in the north-west. Agriculture is the major land-use, mainly grazing of livestock (sheep, cattle) and cereal cropping, although in some locations the availability of water has allowed dairy farming and horticulture (fruit orchards). The natural vegetation was primarily *Eucalyptus*-dominated forests and woodlands, with 'box' species such as grey box (*E. microcarpa*) and yellow box (*E. melliodora*) being common tree species on the plains, and river red gum (*E. camaldulensis*) dominant along streams; while on the slopes with shallow stony soils, the main tree species included red ironbark (*E. tricarpa*), red stringybark (*E. macrorhyncha*) and red box (*E. polyanthemos*). More than 80% of the natural vegetation across the study region has been cleared, with remaining stands disproportionately located on the infertile soils of the hills. Less than 6% tree cover remains on the more-fertile soils of the alluvial plains, where sampling was focused in these studies.

Insights from Atlas data

Data bases compiled during Atlas projects provide a valuable source of data for carrying out landscape-level studies, as the effort required to sample whole landscapes is great. In Australia, volunteer observers collecting data for the first Atlas of Australian Birds (Blakers *et al.*, 1984) recorded the presence of bird species within grid cells of 10' latitude by 10' longitude. In northern Victoria this is a mean area of 277 km^2 (approximately 15.0 by 18.5 km). Bennett and Ford (1997) collated data on the occurrence of bird species recorded from 75 such grid cells (here termed 'landscapes') in the Northern Plains of Victoria, and used this data set to examine factors that influence the richness of bird species.

Woodland species were recognised as those birds primarily associated with remnant woodland or forest, and which are seldom observed in highly modified environments, such as open farmland, scattered trees in pasture, or towns. Vagrants and species not characteristic of the region were excluded (Bennett and Ford, 1997). Potential predictor variables for each landscape included; measures of the physical environment (temperature, rainfall, elevation, number of streams), tree cover (total tree cover, % cover in blocks >100 ha, % cover in blocks <20 ha), and intensity of human settlement (indices of the number of towns, density of roads, irrigation channels and ditches). The number of bird 'lists' submitted per landscape was included as a measure of survey effort. This latter variable was important because species richness was highly correlated with survey effort ($r = 0.66$, $p<0.001$).

The number of woodland species reported from the study landscapes ranged from 7 to 72 (mean 34.2 species). After correcting for variation in survey effort, species richness was significantly positively correlated with; mean annual rainfall, number of streams, total tree cover, % tree cover in blocks >100 ha, and significantly negatively correlated with % tree cover in blocks <20 ha, number of roads, and number of channels (Bennett and Ford, 1997). Extent of tree cover was the most important predictor of species richness (other than survey effort), highlighting the dependence of woodland birds on availability of suitable habitat and their sensitivity to habitat loss. Landscapes with few woodland birds typically were those dominated by treeless open farmland or intensive irrigated agriculture. A multiple regression model relating richness of woodland birds to landscape attributes included three variables; number of lists, total tree cover, and number of streams ($r^2 = 0.728$, $F = 52.68$, $P < 0.001$). Thus, for a given survey effort, the greatest richness of woodland birds was predicted to be in landscapes with extensive tree cover and numerous streams (Bennett and Ford, 1997).

To generate predictions of species richness for landscapes with differing amounts of tree cover, survey effort and number of streams were held at their mean value (Figure 1). The model predicted that species richness declines as tree cover in the landscape declines. For landscapes with relatively high tree cover (>25%), the decline in species richness is gradual. However, a rapid decline in richness was predicted for landscapes with less than 10% tree cover, signaling a marked shift in the structure of the avifauna. Notably, 80% of landscapes are below this level, suggesting that throughout much of the region woodland bird populations are in a precarious position (Bennett and Ford, 1997).

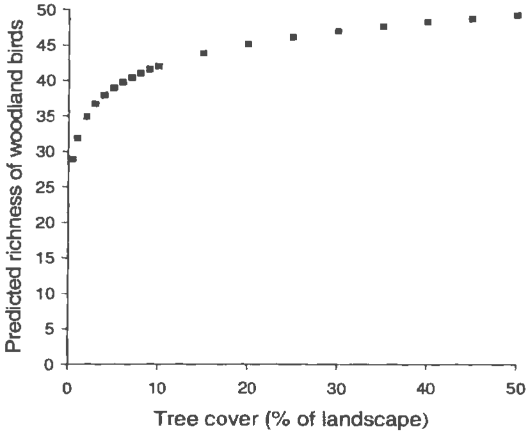


Figure 1. Predicted richness of woodland bird species at the landscape scale (277 km^2) in the Northern Plains, Victoria, in relation to tree cover (from Bennett and Ford 1997).

Insights from empirical field research

Data sets from Atlas projects have a number of limitations, mainly associated with the way data is collected. Observer effort varies between locations, the skill of observers may be highly variable and there is little control on where observations are made. To overcome some of these issues, Radford *et al.* (submitted) designed a project in which avifaunal data was collected in a systematic manner from a series of 'landscapes', each 10 x 10 km (100 km^2) in northern Victoria. The set of 24 landscapes was selected to represent a gradient in tree cover from 2% to 60% of the landscape. Along this gradient, landscapes were further chosen to represent those in which remaining tree cover was strongly aggregated compared with those that had a dispersed configuration. All landscapes were within a defined range in elevation and mean annual rainfall.

Bird censuses were carried out at 10 sites in each landscape, with each site sampled on four occasions by recording the occurrence and abundance of species on a 2 ha census transect. Species outside transects, but within the landscape, were also noted. The location of sites in each landscape was stratified in relation to the type and amount of remaining vegetation in the landscape. Thus, for landscapes in which most wooded vegetation occurred in small blocks or along roadsides, sites were primarily distributed in these landscape elements (Radford *et al.*, submitted).

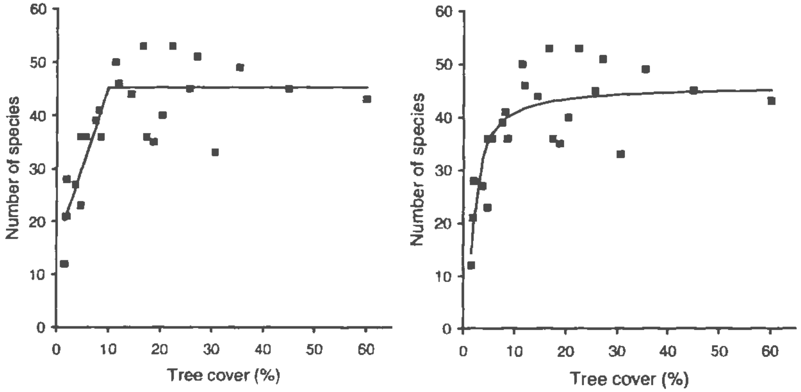
Variables measured for each 100 km^2 landscape included; total tree cover, extent of riparian vegetation, an index of habitat condition, several indices of configuration of tree cover, range in elevation, mean rainfall, mean temperature and extent of land uses. Species richness was tallied for three groups of bird species; those dependent on woodland or forest for most resource requirements (woodland birds), those tolerant of open country (open tolerant) and those that regularly occur in open farmland (open country). Other sub-groups were also defined based on food types and foraging substrates, but attention here is given to the assemblage of woodland species.

From a total of 140 species 'typical' of the region (excluding water birds) that were detected during censuses (on and off site), 80 were classed as woodland species, 34 as open tolerant, and 26 as open country species. The number of woodland species in each landscape ranged from 12 to 53, with a mean of 38.4. Total tree cover in the landscape was clearly the strongest correlate of the richness of woodland birds. A multivariate model of factors influencing the number of woodland birds in 100 km^2 landscapes included four variables; tree cover, range in elevation, a configuration index, and easting (geographic location), which accounted for 75.3 % of variance in woodland bird richness. Total tree cover was the most influential explanatory variable, accounting for 55% of the variance on its own.

The univariate relationship between woodland bird richness and % tree cover was clearly non-linear. It was modeled in several ways, including a broken-stick regression (Figure 2a) and an asymptotic curve based on an inverse relationship with tree cover (Figure 2b). The broken-stick model consisted of a linear regression fitted to landscapes with less than 10% cover (RICHNESS = 15.59 + 0.029 TREE COVER, $R^2 = 0.69$), and then a constant value equal to the regression estimate at 10% cover for landscapes with >10% tree cover. This model highlighted the discontinuity in the response of woodland bird species. Below 10% cover there was a rapid loss of species as tree cover decreased (Figure 2a) (Radford *et al.*, submitted).

Is a 'threshold response' a useful concept for biodiversity conservation?

There is a striking similarity between these studies in the response of woodland birds to landscape-level change in vegetation cover. The two studies largely overlap in geographic area and describe the same avifauna, but are independent data sets. The model based on Atlas data signals a rapid loss of woodland species in landscapes with <10% tree cover, while the use of a broken-stick regression model with empirical field data highlighted 10% cover as a zone of sharp discontinuity in the relationship between species richness and tree cover.



Figures 2a and b. Richness of woodland-dependent bird species in 100 km^2 landscapes in northern Victoria, Australia, in relation to the extent of native tree cover. a) A broken-stick regression with the discontinuity at 10% tree cover; b) An asymptotic model based on an inverse relationship between richness and tree cover (Radford *et al.*, submitted).

Together, they suggest that a major ecological change is occurring in response to extent of native vegetation around this level. These observations are consistent with the concept of a critical threshold (With and Crist, 1995). Clearly, when a threshold of approximately 10% wooded cover in the landscape is crossed, the processes that lead to local extinction of woodland species precipitate a rapid sequence of extinctions of individual species. Such processes may include loss of specialised habitats, disrupted dispersal or reduced colonisation due to increasing isolation, and increases in edge effects such as changes in interspecific competition and predation regimes (Ford *et al.*, 2001).

What does this mean for landscape management and the conservation of woodland birds? Threshold responses can be a valuable tool for land management because they assist managers to identify quantitative goals that are directly linked to ecological outcomes. In this instance, a management goal in excess of 10% tree cover is required to prevent the collapse of the woodland avifauna in rural landscapes. The rapid decline in species richness means that the end point of a protracted extinction process has been reached for many species. Thus, the management goal should be well on the 'safe' side of the critical threshold at which abrupt ecological changes occur. Threshold responses may also be used to guide efficient use of resources for habitat restoration. For example, landscape restoration of sufficient magnitude to shift a landscape across the threshold zone is likely to have greater conservation benefits than scattered restoration efforts that fall well below the threshold level.

Not all non-linear ecological relationships represent threshold responses. Further work is required to clarify what is, and what is not, usefully described as a critical threshold. Sharp thresholds, corresponding with particularly abrupt changes in ecological responses, will more readily translate to management guidelines than broad zones of change. It is also important when considering land management, that the habitat variable with which a threshold is associated actually represents a causal relationship. Here, there is a sound basis for believing that tree cover has a causal relationship with richness of woodland birds; forest and woodland vegetation is the habitat of these species. Further, in both studies tree cover was the most influential variable explaining richness of woodland birds; it accounted for a much greater proportion of variance than other significant correlates.

Different species respond to the environment in different ways, and so identifying a single threshold for a diverse group such as 'woodland birds' may conceal other important responses. It is likely that some species have threshold responses at higher levels of tree cover than for the overall assemblage. That is, they are lost from landscapes before the richness threshold is crossed. Other species may not show any threshold response. Setting management targets based on an assemblage-level response risks inadequate protection for the most sensitive species. Consequently, it will be important to explore the form and shape of the response to landscape pattern for a subset of species that represent different foraging types, migratory strategies and body size. It is not possible to manage the landscape for every individual species, however, and so we believe it is valuable to understand the response of faunal assemblages known to be of concern, such as woodland birds.

Landscape-level conservation of woodland birds

Achieving a balance between agricultural production and nature conservation in rural landscapes is difficult, but important. A primary goal is to maintain as large a proportion as possible of the biota indigenous to the region. This requires an understanding of the extent, configuration and quality of habitat required to maintain breeding populations that have a high likelihood of long-term persistence. These measures must be tempered by recognition that there is a time-lag in experiencing the full effects of the environmental change that has occurred (Bennett and Ford 1997). Rural landscapes in southern Australia have been stripped of their native vegetation within a period of <150 years (or usually much less), less than the lifespan of a single canopy tree.

Long-term persistence in rural landscapes also depends on maintaining breeding populations of woodland birds. In this context we caution that the analyses presented here are based on presence of species, not on occurrence of breeding populations. When summing species richness, a species is equally counted as 'present' for a sighting of a single bird as for a nesting pair or secure breeding population. A major limitation for conservation planning in rural environments is an almost complete lack of knowledge of the demographic status of woodland birds (and other fauna) in heterogeneous landscapes. We need to understand how the extent, configuration and quality of habitats at the landscape level are associated with parameters such as population age structure, breeding success and trends in population size.

References

Andrén, H. (1994) Effects of habitat fragmentation on birds and mammals in landscapes with different proportions of suitable habitat: a review. *Oikos* **70**: 355-366.
Barrett, G.W., Ford, H.A. & Recher, H.F. (1994) Conservation of woodland birds in a fragmented rural landscape. *Pacific Conservation Biology* **1**: 245-256.
Bellamy, P.E., Brown, N.J., Enoksson, B., Firbank, L.G., Fuller, R.J., Hinsley, S.A. & Schotman, A.G.M. (1998) The influences of habitat, landscape structure and climate on local distribution patterns of the nuthatch (*Sitta europaea* L.). *Oecologia* **115**: 127-136.

Bennett, A.F. & Ford, L.A. (1997) Land use, habitat change and the conservation of birds in fragmented rural environments: a landscape perspective from the Northern Plains, Victoria, Australia. *Pacific Conservation Biology* **3**: 244-261.

Blakers, M., Davies, S.J.J.F. & Reilly, P.N. (1984) *The Atlas of Australian Birds*, Royal Australasian Ornithologists Union and Melbourne University Press, Melbourne.

Fahrig, L. (1997) Relative effects of habitat loss and fragmentation on population extinction. *Journal of Wildlife Management* **61**: 603-610.

Ford, H.A., Barrett, G.W., Saunders, D.A. & Recher, H.F. (2001) Why have birds in the woodlands of Southern Australia declined? *Biological Conservation* **97**: 71-88.

Fuller, R.J., Trevelyan, R.J. & Hudson, R.W. (1997) Landscape composition models for breeding bird populations in lowland English farmland over a 20 year period. *Ecography* **20**: 295-307.

Hinsley, S.A., Bellamy, P.E., Newton, I. & Sparks, T.H. (1996) Influences of population size and woodland area on bird species distributions in small woods. *Oecologia* **105**: 100-06.

Hobbs, R.J. & Yates, C.J. (2000) *Temperate Eucalypt woodlands in Australia: biology, conservation, management and restoration*, Surrey Beatty & Sons, Chipping Norton.

Jansson, G. & Angelstam, P. (1999) Threshold levels of habitat composition for the presence of the long-tailed tit (*Aegithalos caudatus*) in a boreal landscape. *Landscape Ecology* **14**: 283-290.

Law, B. & Dickman, C.R. (1998) The use of habitat mosaics by terrestrial vertebrate fauna. *Biodiversity and Conservation* **7**: 323-333.

Mac Nally, R., Bennett, A.F. & Horrocks, G. (2000) Forecasting the impact of habitat fragmentation. Evaluation of species-specific predictions of the impact of habitat fragmentation on birds in the box-ironbark forests of central Victoria, Australia. *Biological Conservation* **95**: 7-29.

Mac Nally, R. & Horrocks, G. (2000) Landscape-scale conservation of an endangered migrant: the Swift Parrot (*Lathamus discolor*) in its winter range. *Biological Conservation* **92**: 335-343.

Radford, J.Q. & Bennett, A.F. (2004) Thresholds in landscape parameters: occurrence of the white-browed treecreeper *Climacteris affinis* in Victoria, Australia. *Biological Conservation* **117**: 375-391.

Radford, J.Q., Bennett, A.F. & Cheers, G. Landscape-level thresholds of habitat cover for woodland birds. *Biological Conservation* (submitted).

Robinson, D. & Traill, B.J. 1996. Conserving woodland birds in the wheat and sheep belts of southern Australia. RAOU Conservation Statement No. **10**.

Trzcinski, M.K., Fahrig, L. & Merriam, G. (1999) Independent effects of forest cover and fragmentation on the distribution of forest breeding birds. *Ecological Applications* **9**: 586-93.

Villard, M-A., Trzcinski, M.K. & Merriam, G. (1999) Fragmentation effects on forest birds: relative influence of woodland cover and configuration on landscape occupancy. *Conservation Biology* **13**: 774-783.

Wiens, J.A., Van Horne, B. & Noon, B.R. (2002) Integrating landscape structure and scale into natural resource management. Liu, J. & Taylor, W.W. (Eds.). *Integrating Landscape Ecology into Natural Resource Management*. Cambridge Univ. Press, Cambridge, UK 23-67.

With K.A. & Crist, T.O. (1995) Critical thresholds in species' responses to landscape structure. *Ecology* **76**: 2446-2459.

Thresholds of environmental sustainability: the case of woodlands

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Abstract

As a result of its commitment to sustainable development, society will have to face substantial costs for the protection and restoration of environmental systems. An understanding of the constraints or thresholds of sustainability is therefore required. To do this we must know the limits beyond which the integrity of ecological systems will be damaged in relation to the output of goods and services they provide for people. In this paper we examine the concept of sustainability thresholds in the context of the notion of 'ecosystem functions'. We show that a functional approach to landscape ecology is one which can help us define the concept of sustainability thresholds at the landscape scale more easily. An example is provided based on the analysis of patterns of recent woodland planting and the uptake of woodland management agreements in England. Such case studies illustrate that, for the future, landscape ecology needs to develop better methods of combining information about the biophysical constraints that apply in real landscapes, with the social and economic values held by different stakeholder groups. These types of analysis will enable us to both identify the limits associated with individual landscape functions and, more importantly, the trade-offs that needs to be considered in the management of multifunctional landscapes.

Introduction

If the landscape ecological community is to address current policy agendas, then a key issue that faces us is to explain how sustainability issues play themselves out at the landscape scale. Conventional wisdom suggests that if we are to achieve sustainable patterns of economic and social development then we must improve our understanding of the constraints imposed on our actions by the environment. To do this we must know the thresholds or limits beyond which the integrity of environmental systems will be damaged by people's activities - *but what are these thresholds and how do they apply when we look at whole landscapes?* The aim of this paper is to consider how such questions might be resolved by landscape ecologists, by using woodlands in England as an example.

In this paper we will argue that our understanding of sustainability thresholds is poorly developed because of the complexity of human-environment relationships, the limitations of current theories and the fragmentation of knowledge between different disciplines. One way forward, we recommend, is to consider landscape not in structural terms, but rather in relation to the output of ecosystem goods and services that they can provide. The focus on 'landscape functions' can, we suggest, potentially link different areas of understanding in ways that will broaden the impact and importance of 'thinking at the landscape scale'.

Landscape functions and the thresholds concept

The idea of ecosystem goods and services is now well established in the scientific literature (e.g., Daily, 1997, 2001; De Groot *et al.*, 2002 and in press). It involves the idea that the functioning of ecological systems can deliver a number of material and non-material outputs that have value for people. Thus we may think of woodlands, for example, as being 'multifunctional' ecosystems, delivering not only timber, but other wider benefits to society including biodiversity, carbon storage, flood and wind protection, scenic and historic value. Much of the debate surrounding the notion of ecosystem functions and the goods and services that they support, has focused on how to value these outputs in monetary and non-monetary terms, so that their significance can be recognised in wider planning debates (eg. Abel *et al.*, 2003; De Groot *et al.*, 2002; Millennium Assessment[1]). A key issue that remains to be addressed is how the functional approach or paradigm applies at the landscape scale (Haines-Young, 2000; Potschin and Haines-Young, in press).

Landscapes, like ecosystems, can be thought of as 'multifunctional' (Brandt & Vejre, 2003, 2004; Helming & Wiggering, 2003, Mander & Antrop 2003). However, when we come to deal with whole landscapes, the issues are even more complex than when we deal with single ecosystems, for landscapes typically consist of a mosaic of different ecosystems within a defined area. The mosaic elements are rarely isolated from each other. Rather it is usually the case that in most landscapes, changes in the structure and function of one ecosystem element can impact on another. Thus the 'health of woodlands' is often not only related to the management of the surrounding landscapes (cf. Bateman *et al.*, this volume) but may also have profound implications for the functioning of neighbouring ecosystems. The results of CS2000 (Haines-Young *et al.*, 2000, 2003) suggested, for example that part of the improvement of water quality between 1990 and 1998 in Great Britain could have been due to the successional change along watercourses. It is clear, therefore, that when we come to deal with whole landscapes we enter a new realm of complexity, in which we not only have to deal with multi-functionality at the individual ecosystem level, but must also seek to understand how these multiple functions combine across different ecosystems and how we may identify and attribute the outputs of environmental goods and services across large areas of the countryside.

The concept of ecosystem goods and services is valuable to landscape ecology, because it gives us one way of defining sustainability at the landscape scale. A sustainable landscape can, for example, be viewed as one in which the output of goods and services are maintained for current and future generations. As has been argued elsewhere (cf. Haines-Young, 2000), such a formulation is somewhat narrow, in that it is inherently 'anthropocentic'. Nevertheless, it has the advantage of linking the biophysical analysis of environmental systems to the needs and values of people. Ecosystem goods and services do not exist 'in the abstract'. They are things identified by people as being important or of value at a given time or place, and as society changes or knowledge grows, what we may seek to sustain or enhance may alter. The growing focus on re-establishing native woodlands in Britain illustrates just how values may change across generations.

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[1] "The Millennium Ecosystem Assessment", http://www.millenniumassessment.org/.

Figure 1 illustrates in a general way both how we might begin to represent and analyse landscape functions and, more importantly, how we might identify thresholds of sustainability. The diagram picks out two contrasting situations. Figure 1a describes a situation in which we may be dealing with a function which is essentially capable of being restored or recreated, while Figure 1b describes the situation where it cannot.

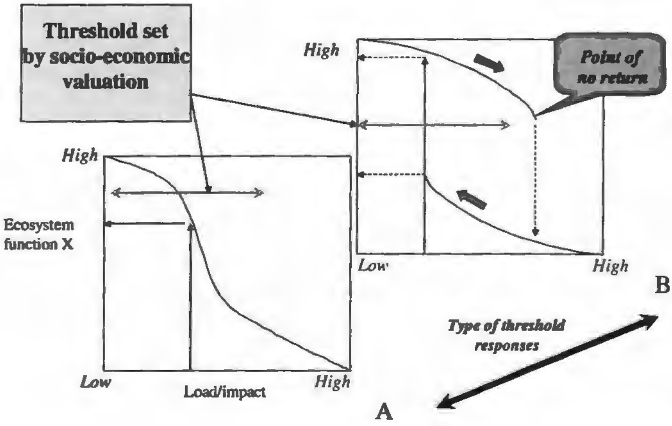

Figure 1. Sustainability thresholds and ecosystem function

Figure 1A illustrates that while the load or impact may cause the output of ecosystem goods and services to change, by regulating the pressure we may potentially achieve a level of output close to what might be desired. Hopefully, woodlands fall into this category of system. By creating and re-establishing woodlands in the landscape, we are seeking to enhance or restore the functionality of this element in the landscape – but what limits apply? How much new or re-established woodland do we need?

The model shown in Figure 1A suggests that there is, indeed, an upper limit, but we would argue that it is not one set entirely by biophysical considerations. It is also, we suggest, determined by economic and social factors relating to risks, costs, and the importance of particular benefits derived from the systems. Society, it can be argued, has eventually to 'take a view' on when 'enough is enough', and, while the answers may vary from place to place and from group to group, we cannot escape from the fact that sustainability is essentially a human construct, and if we choose to apply it, we have to specify what it is we seek to maintain.

The notion of a threshold involving biophysical, social and economic considerations is compatible with recent definitions suggested, for example, by the EU[2], which argued that environmental sustainability thresholds must be seen "as a physical value reflecting the extreme state of the environment ***and*** as an economic value reflecting the damage on monetary valuation or the restoration costs". Building in such ideas we would suggest that a major challenge for landscape ecology is to take their understanding of the biophysical constraints that apply in real landscapes and look at how these sit in relation to the goods and services that people require or desire from these same areas. In the case of woodlands, for example, we might ask what level of connectivity is necessary to enhance or sustain a given level of biodiversity? We might also ask what other thresholds would be transgressed if we moved towards this level of connectivity, say in terms of scenic values associated with particular landscapes. Would society be prepared to forego the benefits of environmental goods and services provided by other elements of the landscape given the level of importance it attaches to those associated with woodlands?

The model shown in Figure 1B presents a far more difficult situation. In this case there is no simple relationship between load and output, in that beyond a certain point the system collapses or transforms to some new configuration. Thus starting with the upper curve, as load increases the level of output of some function declines, until some point at which a 'catastrophic' change occurs and the structure and behaviour of the system changes fundamentally, so that the relationship between load and output is described by the lower curve. Although we may make efforts to reduce the load or impact to restore the functioning of the system, we may never be able to get back to the previous condition. Species extinction is a particularly dramatic example of a situation when the model in Figure 1B applies.

'Points of no return' are particularly important types of threshold that we should attempt to identify. Again, however, these thresholds are not solely determined by biophysical considerations alone, since society may judge it appropriate or acceptable to go beyond them on the basis of other considerations. Alternatively, being aware of them, society may seek to maintain the system in some state well away from these limits, so that the danger of collapse is reduced. Eventually, however, society has to 'take a view'.

Identifying thresholds: the case of woodlands

To what extent can sustainability thresholds be identified in the real world? Some of the outputs from a recently completed project for the Countryside Agency, called *Countryside Quality Counts*[3] (CQC), illustrate how some of these ideas might be applied in relation to woodlands.

The CQC project sought to develop a methodology to monitor changes in countryside quality. Although quality is a wide-ranging concept, it was suggested progress could be made by looking at the contribution that countryside character made to overall quality. This focus allowed the project team to focus on where the landscape elements that shaped countryside character, such as woodlands, were changing, and what significance these changes might have. The period selected for analysis was between 1990 and 1998.

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[2] European Commission Directorate General for Research (2002): Thresholds of environmental sustainability: the case of nutrients. EUR20170.

[3] http://www.countryside-quality-counts.org.uk/

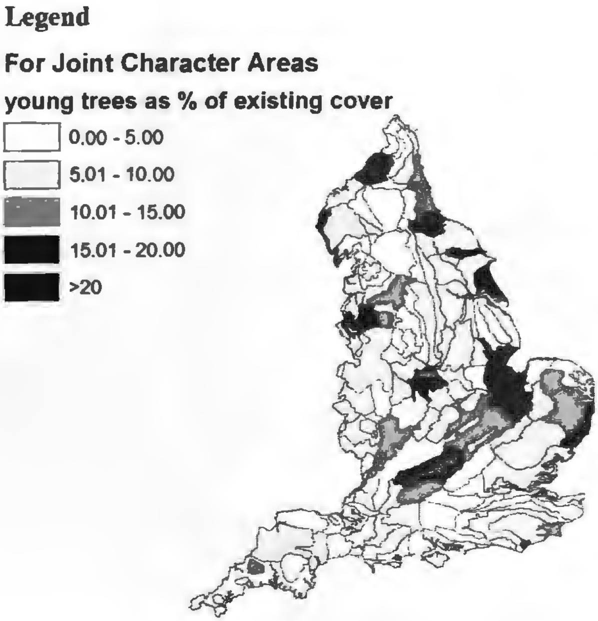

Figure 2. Area of 'young trees' as percentage of existing woodland cover by Countryside Character Area identified from the National Inventory of Woodlands and Trees. Source: Haines-Young *et al.* (2004)

The Forestry Commission's National Inventory of Woodlands and Trees (NIWT) provided an important source of information on recent woodland planting, in that one of the interpreted forest type categories on the woodland map was 'young trees'. These were trees judged by the photo interpreters to have been planted or replanted in the last ten years[4]. Using these data, the area and location of new planting in relation to the Joint Character Areas of England were determined (Figure 2). Information on the uptake of woodland grant scheme for the management of existing woodlands between 1990 and 1998 was also obtained. These were used to estimate, in an approximate way, the proportion of woodlands 'under appropriate management' in each area.

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[4] Some areas of young trees on recent updates of the NIWT also reflect the location of woodland grant schemes for new woodland.

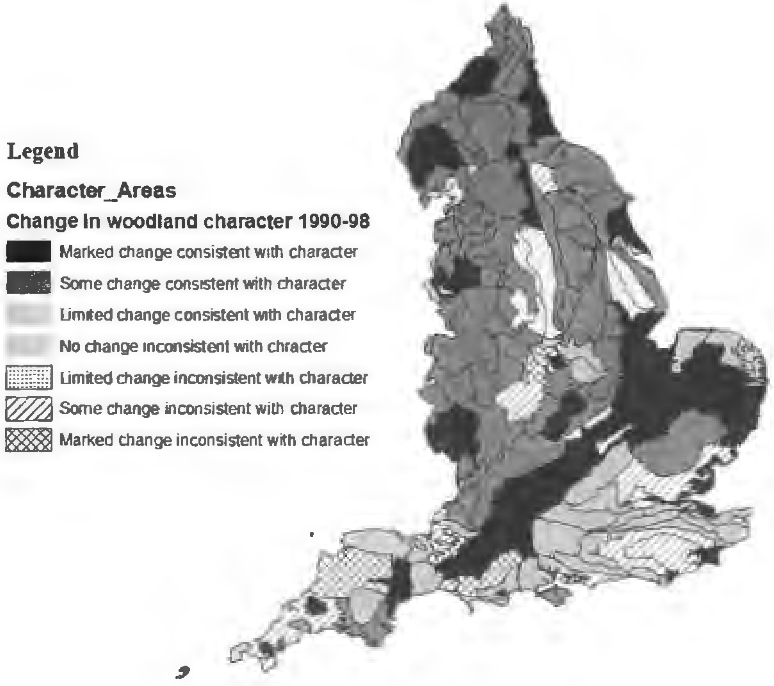

Figure 3. Preliminary analysis of change in woodland character 1990-98, based on analysis of change in extent of new planting and uptake of woodland management agreements by Character Area in England, using published landscape character descriptions to make the evaluation. Source: Haines-Young *et al.* (2004)

Both sets of data were considered in relation to the character descriptions for each Countryside Character Area. These documents were useful in that they contained statements about the direction and type of woodland change that would sustain or enhance landscape character, and the need for better management of the existing woodland stock. Figure 3 shows both the scale of new planting between 1990 and 1998, and the extent to which these changes and those related to the uptake of WGS were consistent or inconsistent with the existing character area descriptions.

The preliminary analysis (Figure 3) suggests that for woodlands, about one in six Character Areas showed either change that was inconsistent with existing woodland character, or only limited change that did little to restore the characteristic features that had been lost. By contrast, woodland character was stable in about 1 in 5 of the Character Areas. Elsewhere, the patterns of change were consistent with strengthening or restoring aspects of woodland character that were identified as important in the published Character Area descriptions.

Sustainability thresholds – linking people and the landscape

The analysis presented in Figure 3, is some way from a formal implementation of the kind of model suggested in Figure 1a,b. However, it does illustrate the importance of considering both where countryside change is occurring and its relationship to the values that people apply to those landscapes. The Character Area descriptions used for the analysis shown in Figure 3 are qualitative and generated by a process of consultation with 'informed' stakeholders. Nevertheless, they can be used to identify both the direction and magnitude of change that is 'appropriate', given prevailing values and priorities, for different landscapes. As such they can begin to serve as a ***proxy*** for the thresholds or limits that we need to identify in order to evaluate the significance of change in the wider countryside. Further research is required to specify these limits or thresholds more precisely in terms of quantifiable measures.

Work is therefore currently underway to develop the methodology used in the CQC project to refine the way in which the Character Area descriptions are designed so that they include more easily implemented notions of 'capacity' and 'landscape function' in relation to the various elements of landscape. This further work will also seek to understand the trade-offs that will have to be considered as the output of ecosystem services associated with one landscape element is affected by changes in another. For example, in the case of woodlands, what is the consequence of expansion in a particular landscape for the benefits or services delivered by the previous land uses? While studies such as *Space for nature* (Woodland Trust, 2002) have made some progress towards a better understanding of the targeting of the extent and context of future planting, the wider consequences of such changes for landscape character and the acceptability of such change to the people who live, work and visit those places is unknown.

Our understanding of sustainability thresholds is poorly developed because of limited ecological understanding of the biophysical constraints that apply in real landscapes, and the complexity of human-environment relationships. Notions of how to handle these different elements is also fragmented between different disciplines. One way forward, we suggest is to consider landscapes not only in structural terms, but rather in relation to the output of ecosystem goods and services that they can provide. A focus on 'landscape functions' can, we suggest, link more firmly the biophysical aspects of landscape with people. It not only helps us to overcome traditional disciplinary boundaries, but also potentially it will allow landscape ecology to broaden its impact and relevance to society and current policy debates. It is unlikely that landscape ecology can ever tell us what landscape configurations are 'optimal' in terms of sustainability. Through the notion of sustainability thresholds we may, however, be able to define the 'ecological choice space' within which different policy options and plans can be compared. In this way, through such work, landscape ecology may empower stakeholders with the knowledge they need to 'take a view' on what aspects of our natural capital we can and must maintain.

Acknowledgements

We acknowledge the use of the data produced by the CQC project, sponsored by the Countryside Agency and its partners, and the input of the many people who have contributed to the underlying methodologies.

References

Bateman, S. Field, R. & Smithers, R.J. (2004) Effects of fertiliser drift at ancient woodland edges. R.J. Smithers (Ed) *Landscape ecology of trees and forests*, IALE(UK).

Brandt, J. & Vejre, H. (2003) (Eds.*) Multifunctional landscapes. Vol. II Monitoring, Diversity and management*, WIT Press, Southampton, Boston.

Brandt, J. & Vejre, H. (2004) (Eds.*) Multifunctional landscapes. Vol. I Theory, values and history*, WIT Press, Southampton, Boston.

Daily, G.C. (1997) *Nature's services*, Island Press, Washington DC.

Daily, G.C. (2001) Ecological forecasts. *Nature* **411**: 245.

De Groot, R.S.; Wilson, M. & Boumans. R. (2002) A typology for the description, classification and valuation of ecosystem functions, goods and services" *Ecological Economics* **41(3)**: 393-408.

De Groot, R.S. (2004) Function analysis and valuation as a tool to assess land use conflicts in planning for sustainable, multi-functional landscapes. *Landscape and Urban Planning* (in press).

Haines-Young, R. (2000) Sustainable development and sustainable landscapes: defining a new paradigm for landscape ecology. *Fennia* **178(1)**: 7-14.

Haines-Young, R.H.; Barr, C.J.; Black, H.I.J.; Briggs, D.J.; Bunce, R.G.H.; Clarke, R.T.; Cooper, A.; Dawson, F.H.; Firbank, L.G.; Fuller, R.M.; Furse, M.T.; Gillespie, M.K.; Hill, R.; Hornung, M.; Howard, D.C.; McCann, T.; Morecroft, M.D.; Petit, S.; Sier, A.R.J.; Smart, S.M.; Smith, G.M.; Stott, A.P.; Stuart, R.C. & Watkins, J.W. (2000) *Accounting for nature: assessing habitats in the UK countryside.* London, DETR.

Haines-Young, R.H.; Barr, C.J.; Firbank, L.G.; Furse, M.T.; Howard, D.C.; McGowan, G.; Petit, S.; Smart, S.M.; & Watkins, J.W. (2003) Changing landscape, habitats and vegetation diversity across Great Britain. *Journal of Environmental Management* **67**: 267-281.

Haines-Young, R.H.; Swnwick, C. & Martin, J. (2004) *Countryside quality counts: final report.* Countryside Agency (in press)

Helming, K. & Wiggering, H. (Eds.) (2003) *Sustainable development of multifunctional landscapes.* Springer-Verlag Berlin, Heidelberg, New York.

Mander, Ü. & Antrop, M. (2003) (Eds.*) Multifunctional Landscapes. Vol. III Continuity and Change*, WIT Press, Southampton, Boston.

Potschin, M. & Haines-Young, R. (2004) 'Rio+10' and the landscape agenda. *Landscape and Urban Planning* (in press).

Woodland Trust (2002) *Space for nature: landscape-scale action for woodland biodiversity*, Woodland Trust, Grantham.

Exploring structural connectivity in Welsh woodlands using neutral landscape models

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Abstract

There is mounting interest in reversing habitat fragmentation through the creation of habitat networks, in order to improve the ecological viability of isolated populations. Many of the woodland habitat networks proposed to date have been primarily focused towards structural connectivity and use thresholds, linked to the amount of woodland cover, derived from the analysis of random landscape models. However, there are increasing concerns over the applicability and appropriateness of such structural approaches to assist the conservation of biodiversity within real landscapes.

This study explores the concept of structural connectivity and the derivation of potential thresholds. A selection of simple, random maps and more realistic fractal maps or neutral landscape models (NLMs), with varying degrees of woodland cover, were generated and analysed. The results of the various models were compared with a selection of Welsh landscapes with varying percentage of woodland cover to identify relationships and key thresholds. Results indicate considerable differences between these random models and Welsh woodlands, indicating the likely problems of applying simple thresholds. This reinforces the need to adopt a more functional approach utilising focal species in determining landscape connectivity to aid biodiversity conservation. It is clearly possible to have high functional connectivity in a structurally fragmented landscape, as long as the wider matrix supports the particular ecological process.

Introduction

In the UK, the conservation of biodiversity has been based on the protection of a series of small, isolated sites, but this approach is ineffective in preventing the continued declines in biodiversity. Conservation strategies within a highly fragmented landscape, such as some areas in Wales, need to include the surrounding 'wider countryside' in future plans. The impacts of habitat fragmentation are consistent with many of the reductions of biodiversity in the UK, and are widely considered as a major threat to biodiversity conservation, as identified in the woodland strategy for Wales, which aims to "create links between fragmented woodlands" (Forestry Commission, 2001).

Habitat networks

The reversal of habitat fragmentation, through the creation of habitat networks, is of increasing importance to improve the ecological viability of isolated populations (Peterken *et al.*, 1995, Ratcliffe *et al.*, 1998, Peterken, 2003, Ray *et al.*, 2003, Worrell *et al.*, 2003). Habitat networks are intended to reverse the deleterious effects of fragmentation by linking existing woodland to provide large connected areas of woodland habitat, which are capable of sustaining the life-cycle needs of critical species. Particular emphasis is placed on the development of strategic plans for the creation of large-scale woodland habitat networks, to provide a framework for habitat expansion and restoration. The woodland strategy for Wales claims that its success "will be shown if we can improve the quality of these woodlands, linking and expanding their habitat networks" (Forestry Commission, 2001).

Structural connectivity thresholds

Prior to discussion of levels of woodland connectivity, it is necessary to make the fundamental distinction between 'structural' and 'functional' connectivity. Structural connectivity is based on the degree of physical connection between elements of the same type, whilst functional connectivity is defined by an ecological process such as species movement and dispersal between habitat patches. Many woodland habitat networks proposed to date (Hampson and Peterken, 1998; Peterken, 2000) have been primarily focused towards 'structural' connectivity and use thresholds, linked to amount of woodland cover, derived from the analysis of random landscape models. These are different from 'functional' species-based approaches (Lambeck, 1997; Brooker, 2002).

A number of key structural thresholds have been identified at 30% and 60% woodland cover, by examining a number of simple landscape metrics applied to random landscape models. The thresholds are illustrated in Figure 1 and summarised below:

- The **number of patches** of woodland within a landscape decreases considerably between 20% and 60% cover as the patches start to coalesce and potential isolation is minimised.
- The **total edge** of woodland reaches a maximum at 50% woodland cover and reduces thereafter.
- The **largest patch**, as a proportion of the total woodland cover, increases considerably from 30% and contains the majority of the woodland by 50%.
- The **total core area** increases slowly depending on the patch size and the edge definition used; a 50m edge is used in this example.

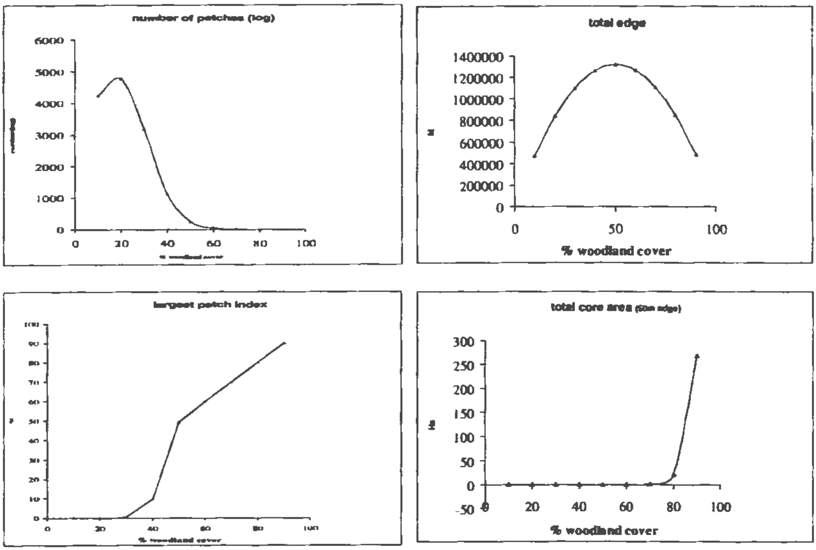

Figure 1. Simple landscape metrics derived from random landscape models

The 30% rule for woodland cover has become a widely accepted 'rule of thumb' when examining and designing habitat networks. The assumption being that "at 30% cover or more, woodlands become more ecologically resilient, i.e., that isolation is so small that woodland species can respond to changes in the pattern of woodland...the areas are functionally one forest" (Peterken, 2000). However, there are increasing concerns over the applicability and appropriateness of such thresholds to aid biodiversity conservation and their incorporation in planning and grant frameworks. In addition to woodland cover, woodland configuration and landscape context are likely to play a significant role in determining landscape connectivity.

Methods

This study explores the concept of structural connectivity, as an aid to the development of habitat networks and the reversal of fragmentation. This is achieved through the generation and analysis of random and fractal maps or neutral landscape models (NLMs), with varying degrees of woodland cover. It assesses whether the potential thresholds, particularly the 30% and 60% thresholds discussed earlier, occur within a selection of NLMs. These model outputs are then compared with a selection of Welsh landscapes with varying percentage of woodland cover to identify relationships with the NLMs and any additional thresholds.

Neutral landscape models

Neutral landscape models (NLMs) allow the production of a wide range of simple, random and more realistic spatially correlated fractal landscapes (Figure 2) with varying degrees of woodland cover (Gardner *et al.*, 1987; With and King, 1997; Turner *et al.*, 2001). Fractal maps are generated by an algorithm to produce spatially correlated patterns of land cover. Fractal maps are highly relevant to this study, as they are often used by investigators wishing to use random, but realistic, maps to simulate the movement and dispersal of organisms (With *et al.*, 1997). A key purpose of these artificial NLMs is to evaluate the effects of landscape structure on ecological process within a range of theoretical landscapes.

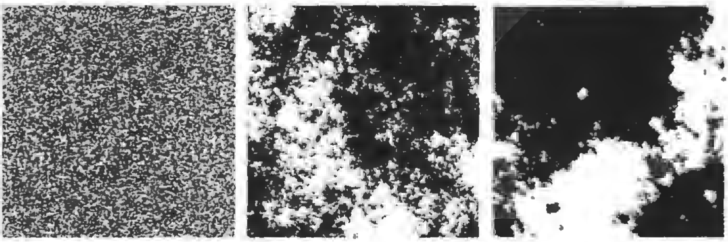

Figure 2. Random, low and high fractal landscapes all with 30% woodland cover

Each NLM is based on a grid 250 x 250 cells with a resolution of 20m, representing a 5km x 5km landscape. Random, low and high spatially correlated fractal NLMs were generated with between 10-90% cover, at 10% intervals. 10 replicates were generated for each in order to develop a representative sample, resulting in the analysis of 270 NLMs. The mean score for the various landscape metrics, as outlined in Figure 1, were calculated using Fragstats (McGarigal *et al.*, 2002).

Welsh landscapes

The whole of the Welsh land area was similarly divided into 5km by 5km squares with a resolution of 20m, thereby creating a 250 x 250 grid of similar size and resolution to the NLMs. A binary grid (i.e., habitat and non-habitat) was generated from woodland identified within the Countryside Council for Wales Phase 1 Survey; there were over 950 squares with woodland cover ranging from under 1 to over 85%.

Neighbourhood rules

Figure 3 illustrates three possible neighbourhood rules for defining habitat connectivity based on the surrounding 4, 8 or 12 cells. In order to explore structural connectivity in this study, the landscape analysis was based upon the 8 cell neighbourhood rule, which permits the formation of habitat clusters from contiguous cells that occur within the surrounding 8 cells, i.e., where there is a physical connection between them.

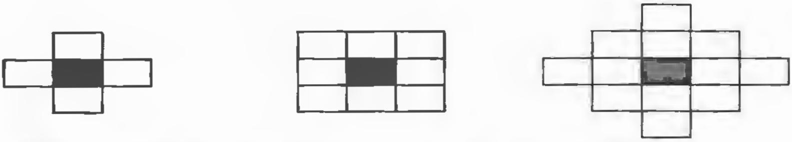

Figure 3. Illustration of the three major neighbourhood rules (4, 8, & 12 cell rules)

Binary landscapes

This analysis was conducted, and structural thresholds identified, using binary landscapes (i.e. habitat and no-habitat), which therefore takes no account of habitat quality or the impacts of the wider countryside.

Results

Overall, the Welsh wooded landscapes differed markedly from the random landscapes in the values of the selected landscape metrics (Figure 5). This is consistent with other studies that suggest there is great variation between random NLMs and real landscapes, particularly within the range of 10-50% woodland cover (Turner *et al.*, 2001). The Welsh wooded landscapes had greater similarity with fractal NLMs with high spatial correlation (Figure 5). Unfortunately, such fractal landscapes do not exhibit the same clear thresholds as identified for random landscapes. The wooded habitats in real landscapes are correlated with underlying landscape processes, such as topography and hydrology (Figure 4a) and anthropogenic land use, such as agriculture and plantation forestry (Figure 4b).

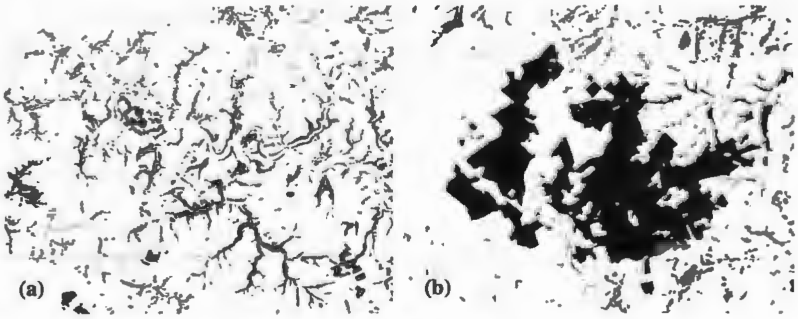

Figure 4. Distribution of woodland in riparian zone (a) and plantation forest (b)

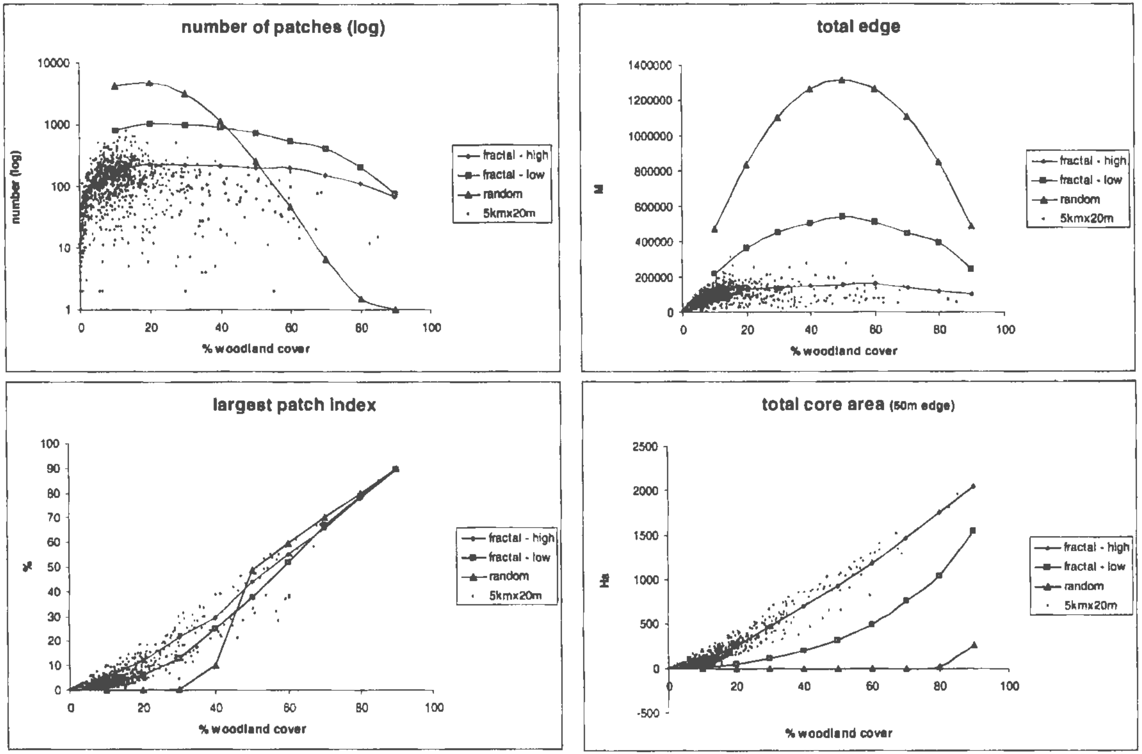

Figure 5 – Analysis of selected landscape metrics for Neutral Landscape Models and Welsh wooded landscapes (5km x 20m)

Discussion

Neutral Landscape Model theory

With and King (1997) suggest that the purpose of NLMs is "to provide null models of landscape structure as a baseline for comparison with real landscape patterns, or for evaluating the effects of landscape structure on ecological processes". Use of NLMs has now expanded beyond the domain of theoretical landscape ecology, for which they were originally developed, to applications in other areas of ecology. However, there is concern that such models may be used inappropriately, or that their function may be misunderstood or misinterpreted. Indeed, With and King (1997) emphasise "it would be a misuse of NLMs to assume that the results from simulations on neutral landscapes can be applied directly to real landscapes…it would also be naive to assume that real landscapes percolate *(are traversable using adjacent cells)* when at least 59.28% of the landscape is habitat".

Binary landscapes

Binary approaches do not account for habitat quality or the impacts of the wider countryside, raising doubt over the applicability of thresholds derived from random models in binary landscapes. The surrounding matrix will vary greatly in its degree of hostility or permeability to species movement, so real landscapes should be represented as 'shades of grey' in terms of habitat quality and landscape permeability rather than black and white.

Implications of scale

The scale of analysis (5km x 5km and a resolution of 20m, giving a 250 x 250 grid) can have an impact on the results, and further analysis of the implications of extent and resolution of scale is being undertaken. Alternative landscapes at 5 x 5km and 20km x 20km extent, with a resolution of 20m, 50m and 200m have been provisionally investigated. As expected, preliminary results revealed that the number of patches and the total edge decreases with decreasing resolution, whilst the total core area increases as complex shapes are simplified.

Conclusions

Maintaining landscape connectivity in a fragmented landscape, through the use of habitat networks, has now become a management imperative for many agencies. The concept of simple landscape thresholds, derived from random landscapes, is very appealing in the development of such plans and strategies. However, these results indicate there are considerable differences between these random models and Welsh landscapes, indicating the likely problems of practically applying simple thresholds derived from the former. This reinforces the need to adopt a more functional approach, utilising focal species, in determining landscape connectivity to aid biodiversity conservation. As With (2002) explains "landscape connectivity is far more complex than is implied by the notion of habitat corridors linking fragments (structural connectivity)". Clearly, it is possible to have high functional connectivity in a structurally fragmented landscape, as long as the wider matrix supports the particular ecological process. Therefore, a species-based perspective is the most relevant viewpoint if we are to develop meaningful and effective conservation strategies, and a functional approach to connectivity is likely to reveal very different thresholds.

Acknowledgements

This research was conducted under a contract from the Forestry Commission (Wales) and the Countryside Council for Wales (No. CT-W.H.N.S.W – 0303) to 'prepare a woodland habitat network strategy for Wales', and as part of the Forest Research Landscape Ecology Project with funding from the Forestry Commission. Duncan Ray and Chris Quine provided helpful comments on an earlier version of the paper.

References

Brooker, L. (2002) The application of focal species knowledge to landscape design in agricultural lands using the ecological neighbourhood as a template. *Landscape and Urban Planning* **60:** 185-210.

Forestry Commission (2001) *Woodland for Wales*, Forestry Commission, Edinburgh. (on behalf of the Welsh Assembly Government)

Gardner, R.H., Milne, B.T., Turner, M.G. & O'Neill, R.V. (1987) Neutral models for the analysis of broad-scale landscape pattern. *Landscape Ecology* **1:** 19-28.

Hampson, A.M. & Peterken, G.F. (1998) Enhancing the biodiversity of Scotland's forest resource through the development of a network of forest habitats. *Biodiversity and Conservation* **7:** 179-192.

Lambeck, R.J. (1997) Focal species: a multi-species umbrella for nature conservation. *Conservation Biology* **11:** 849-856.

McGarigal, K., Cushman, S.A., Neel, M.C. & Ene, E. (2002) *FRAGSTATS: spatial pattern analysis program for categorical maps*, University of Massachusetts, Amherst, www.umass.edu/landeco/research/fragstats/fragstats.html.

Peterken, G.F. (2000) Rebuilding networks of forest habitats in lowland England. *Landscape Research* **25:** 291-303.

Peterken, G.F. (2003) Developing forest habitat networks in Scotland. J. Humphrey *et al* (ed.). *The Restoration of Wooded Landscapes - Proceedings of a conference held at Heriot Watt University, Edinburgh, September 2000*, Forestry Commission, Edinburgh 85-91.

Peterken, G.F., Baldock, D. & Hampson, A. (1995) A forest habitat network for Scotland. 44, Scottish Natural Heritage, Edinburgh.

Ratcliffe, P.R., Peterken, G.F. & Hampson, A. (1998) A forest habitat network for the Cairngorms. 114, Scottish Natural Heritage, Edinburgh.

Ray, D., Watts, K., Griffiths, M., Brown, C. & Sing, L. (2003) Native woodland habitat networks in the Scottish Borders. Forest Research, Edinburgh.

Turner, M.G., Gardner, R.H. & O'Neill, R.V. (2001) *Landscape ecology in theory and practice: pattern and process*, Springer-Verlag, New York.

With, K.A. (2002) Using percolation theory to assess landscape connectivity and effects of habitat fragmentation. K.J. Gutzwiller (ed.). *Applying landscape ecology in biological conservation*, Springer-Verlag, New York 105-130.

With, K.A., Gardner, R.H. & Turner, M.G. (1997) Landscape connectivity and population distributions in heterogeneous environments. *Oikos* **78:** 151-169.

With, K.A. & King, A.W. (1997) The use and misuse of neutral landscape models in ecology. *Oikos* **79:** 219-229.

Worrell, R., Taylor, C.M.A. & Spittal, J.J. (2003) Highland Perthshire Forest Habitat Network. *Scottish Forestry* **57:** 151-157.

Session 4

Potential of trees to reduce impacts from farming

The role of trees in landscape planning to reduce the impacts of atmospheric ammonia deposition

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Abstract

The emission of atmospheric ammonia (NH_3) to the atmosphere and its deposition to ecosystems are highly spatially variable. While national (5 km resolution) maps form the basis of UK policy analysis, landscape level assessments show substantial local structure, with large gradients both adjacent to livestock farm buildings and fertilised/manured agricultural fields. The scale of expected ecological changes are such that it is not feasible to protect all ecosystems in the UK. To maximise the benefit/cost ratio of any NH_3 abatement strategy, it is therefore necessary to prioritise which ecosystems should be protected. Based on such a prioritisation, trees can play an important role in landscape planning to protect priority areas from NH_3 deposition. This is because: trees capture NH_3 at higher rates than other vegetation; they enhance the dispersal of emitted NH_3; and they can shelter sources, thereby reducing NH_3 emissions. Trees may therefore be planted 'sacrificially' around farms or conservation sites, as a strategic re-use of land (e.g., relevant to current agri-environment reforms). Finally, the assessment points to the need to prioritise the designation of large contiguous areas for nature conservation, allowing the inclusion of such buffer zones. By contrast, designating fragmented or linear conservation sites places them at greater risk due to the increased fraction that represents edges vulnerable to ammonia.

Introduction

Ammonia emissions are an increasingly recognised threat to the biodiversity of natural and semi-natural habitats. The extra nitrogen (N) provided by ammonia (NH_3) deposition leads to changes in species competitiveness resulting in altered species composition of particular relevance for the integrity of many statutory designated habitats (Sutton *et al.*, 1993; Fangmeier *et al.*, 1994; Bobbink *et al.*, 2003;). Emissions of NH_3 arise primarily from livestock farming and therefore occur in the rural environment, often close to areas of nature conservation interest. For this reason, there are very large local gradients in NH_3 concentrations and deposition, and the worst problems occur where a large intensive livestock farm is located close (e.g., < 2 km) to an area of nature conservation value (Dragosits *et al.*, 2002). This "near source" problem is additional to the "long range" air pollution problem, where NH_3 emissions are dispersed hundreds of miles and deposited mainly in precipitation. Technical methods are available to reduce ammonia emissions, but many of them are expensive to farmers and have only limited effectiveness (e.g., Cowell and ApSimon 1998). As a result there is a need to look at complementary measures that can help protect natural habitats. Trees can play a key role as additional means to help mitigate the effects of atmospheric ammonia (Theobald *et al.*, 2001). The rates of ammonia deposition to woodland are larger than to other vegetation surfaces, so the existence of tree belts around farms can help recapture ammonia emissions. The rough surface of trees also helps to generate turbulence to disperse the ammonia, reducing deposition to sites immediately beyond the trees. In addition, the sheltering effect of trees can reduce emissions, while keeping animals under trees is also expected to reduce emissions substantially (Theobald *et al.*, 2004).

This paper summarises current thinking on NH_3 recapture and investigates the possibility of novel landscape-level abatement measures. It is shown that tree belts need to be wide enough to capture significant ammonia, but that they can have significant benefits at costs comparable or better than many conventional abatement techniques. The approach is by no means a panacea, but it could make an important contribution, particularly in the context of future agri-environment reform and for the protection of nature conservation sites.

National assessment of ammonia sources, sinks and mitigation

Until now, most effort has been placed in quantifying ammonia emission fluxes and investigating abatement strategies at the national scale. The Gothenburg Protocol of the United Nations Economic Commission for Europe (UNECE) and the EU National Emissions Ceilings Directive commit the UK to a national target for 2010 of 297 kt NH_3 year^{-1}. This represents a reduction of 11% from the official estimate of emissions for 1990 of 333 kt NH_3 yr^{-1}, and of 7% from the 320 kt NH_3 yr^{-1} estimated for 2000 (Defra 2002). These changes will, however, be insufficient to protect most UK ecosystems from the effects of atmospheric ammonia. Overall, 59% of UK ecosystems (by area for 1999-2001) have been estimated to receive atmospheric N deposition that is in excess of the 'critical load' for nitrogen eutrophication, so that biological and ecological changes due to ammonia are expected for much of the country. The situation is even worse for woodland, with critical loads for N exceeded for between 92% and 98% of UK woodlands (depending on woodland type). Based on the Gothenburg/NECD emission ceilings for 2010, the figures are estimated to be 49% for all UK ecosystems and between 80% and 96% for woodlands (Hall *et al.*, 2004).

It is important to consider not just these average estimates, but also the spatial distribution of emissions, deposition and critical loads exceedance. Such estimates are made with a combination of models (see e.g., NEGTAP, 2001; Hall *et al.*, 2004). Figure 1 compares the national distribution of atmospheric N deposition, including both reduced and oxidised nitrogen, according to the NEGTAP approach with the critical loads exceedance for nutrient nitrogen effects on this habitat. As woodland/forest is aerodynamically rough and receives little mineral fertilisation, it is an efficient sink of ammonia, so that these rates of deposition are larger than for other vegetation types. The high spatial variability is striking, and reflects the different source/sink areas of ammonia, as well as high wet deposition in hill areas. The exceedances represent the largest values for any land cover type, and imply substantial effects of N deposition on woodlands across the UK. The clearest effects are likely to be changes in woodland ground-flora species composition (e.g., Pitcairn *et al.*, 2002; Bobbink *et al.*, 2003), while tree growth and carbon storage may also respond to the additional nitrogen.

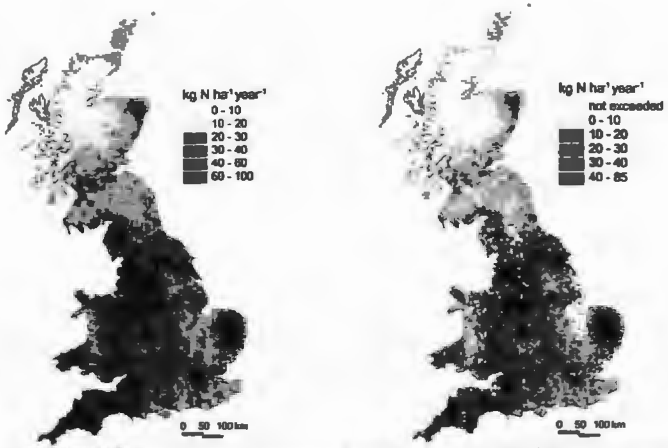

Figure 1. Estimated atmospheric nitrogen deposition and critical loads exceedance for nutrient nitrogen effects for woodlands/forest across Great Britain (estimates for 1995-1997).

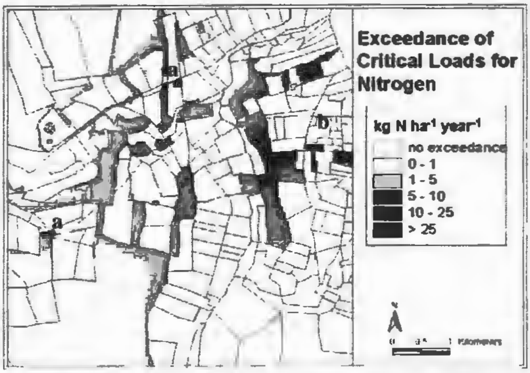

Figure 2. Exceedance of the critical load for nutrient N on woodlands and semi-natural vegetation for the modelled landscape of Ambridge in middle England (from Dragosits *et al.*, 2002). Farms are located at a. (small mixed beef farms) and b. (large poultry farm).

Landscape level variability in atmospheric N deposition and impacts

The national estimates of atmospheric N deposition are provided on a 5 km grid. These may be used directly for initial screening exercises, and are available through the UK Air Pollution Information System (APIS) (www.apis.ceh.ac.uk). It is, however, recognised that there is substantial sub-grid variability. This is an issue for NH_3 in agricultural areas, since sensitive ecosystems often occur in close spatial relationship to NH_3 sources (animal houses, manured and fertilised fields etc.) (Sutton *et al.*, 2003). Conducting a detailed assessment of the local variability of atmospheric N emissions and deposition is a significant task, but is very useful to investigate the spatial interactions between sources and sinks. Such an assessment has been made by Dragosits *et al.* (2002) for a rural landscape of 5 km x 5 km. For reasons of confidentiality, the location is referred to as "Ambridge", an unspecified location in middle England. Figure 2 shows the spatial pattern of critical loads exceedance for N deposition in the Ambridge study area. As critical loads do not apply for agricultural fields and farms, these areas are left white. The farm sources cause major gradients in critical loads exceedance for woodlands and other semi-natural vegetation, both adjacent to the farms and the fertilised agricultural fields. While Figure 1 may not show exceedance at 5 km resolution for this square, Figure 2 shows that localised exceedances can be extremely large.

Spatial prioritisation of ammonia abatement

The current national policies are limited for several reasons. Firstly, as indicated, the anticipated emission reductions will not be sufficient to avoid exceedance of the critical loads, according to the national assessment. Secondly, local scale variability in deposition indicates even more extreme "hot spots" of nitrogen deposition, for which a conceivable 10-20% reduction in emissions would be insufficient to avoid effects. Thirdly, the national emissions ceilings approach does not consider which ecosystems should be protected in which locations of the country. This last point is important, since, if it is accepted that not all ecosystems can be protected, then there is a need to prioritise which areas should be protected. For example, it may be agreed that it is of higher priority to protect designated nature areas (e.g., Sites of Special Scientific Interest, SSSIs; candidate Special Areas of Conservation, cSACs) than other ecosystems. If this is agreed, it paves the way to explore novel methods of spatial planning as a complementary approach to reducing the environmental impacts of ammonia. Existing woodlands or new plantings can play an important role in such strategies by acting as buffer zones to capture and disperse ammonia.

The multiple benefits of trees for ammonia mitigation

Trees have several benefits to help mitigate the environmental impacts of atmospheric ammonia. These can be summarised as four distinct roles:

1. Ammonia emissions from many sources are a function of wind speed. *Trees provide shelter from wind*, potentially to reducing emissions (e.g., naturally ventilated buildings).
2. Trees capture ammonia more efficiently than other vegetation. Therefore, planting trees around animal buildings or fields can *recapture a fraction of the ammonia on the farm*, reducing the amount that is deposited to more sensitive ecosystems further afield.
3. Trees represent a rough surface that encourages downwind dispersion. The *additional dilution of emissions leads to lower ammonia concentrations immediately downwind*, which can be of benefit for nearby nature areas.
4. Ammonia emissions per kg N excreted are smaller for animals kept outdoors than for housed animals. In addition, where stock is kept under trees, *overlaying vegetation is expected to recapture a substantial amount of any emitted ammonia.*

It is important to emphasise that trees are not a panacea for ammonia abatement: there are clear limitations. For example, the amount that is recaptured by trees down-wind of farm buildings may be <10%, while a similar additional 10-20% nearby may result from the dispersion effect. It is important to recognise that explicit design of farm woodlands to recapture and disperse ammonia is important to maximise the recapture. Effects of wind sheltering may be significant, e.g., 20-30% reduction in emissions (Theobald *et al.*, 2004), but these require further analysis in relation to interactions with ground surface temperature.

Finally, the keeping of stock under trees represents an ancient practice for the cheap provision of both food and shelter. Obviously, there are practical limitations regarding the number of stock and the protection of viable woodland, but the approach may be extremely attractive for niche markets, such as "woodland chicken" (www.faifarms.co.uk) or even "pannage pork".

As part of the AMBER project, Theobald *et al.* (2001; 2004) estimated the recapture of ammonia by a woodland downwind of a controlled field release of ammonia. Based on the woodland in question they also applied a new Lagrangian stochastic particle dispersion model (Loubet, 2000) to analyse the advective recapture within the woodland. They found that the woodland used for the experiment did not have the most optimal design, having a rather open under-story for its full width, which allowed ammonia to pass under the tree canopy. It was identified that an improved woodland design would be to ensure an open canopy under-story immediately adjacent to the source, but a dense canopy under-story at the down-wind edge, to ensure that the air is forced through the main tree canopy. While further work is clearly necessary to optimise such woodland design to maximise ammonia recapture, Figure 3 demonstrates the effect of width of the tree belt on ammonia recapture. This shows that a wider tree belt is estimated to capture substantially more ammonia. For a strip of woodland 15 m wide (with a 2 m high NH_3 source and 10 m high trees starting 5 m downwind of the source of the same canopy structure as in the experiment), 2.1% of the NH_3 was recaptured. By contrast, for a 60 m wide belt of trees the estimate was 7.1% recapture. This shows that a simple single row of trees will be insufficient to achieve substantial recapture, although the absolute values of the percentages might be increased by improved design of the tree belt.

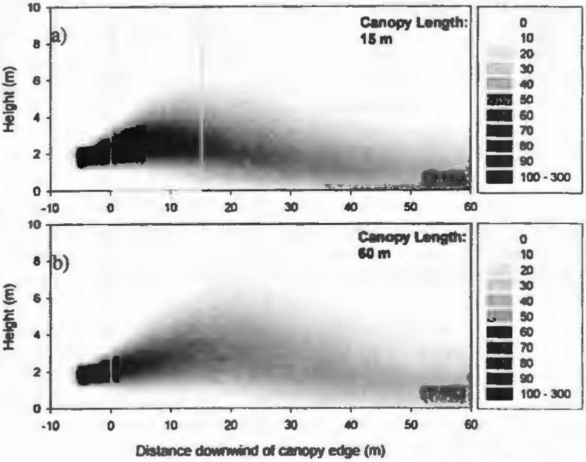

Figure 3. Results from the application of a Lagrangian Stochastic model (Theobald *et al.* 2004) to simulate the dispersion of ammonia in the canopy of an experimental woodland site. a. 15 m wide belt of trees; a. 60 m wide belt of trees. Units of shading: ug m^{-3} NH_3.

Landscape scenarios for ammonia abatement

The combined effects of recapture and increased atmospheric dispersion due to trees can be accounted for in the atmospheric dispersion modelling approach of Dragosits *et al.* (2002) for 'Ambridge'. This assessment was made using the LADD (Local Ammonia

Dispersion and Deposition) model, which includes the roughness effects and NH_3 affinities of different vegetation types in a simple way. By contrast, the effects of wind sheltering on NH_3 emissions or benefits of keeping stock under trees cannot currently be estimated in this model. The example of 'Ambridge' provides a starting point for the consideration of "what if" scenarios of landscape management for ammonia abatement. Two examples are considered here: a) the planting of woodland around sources of ammonia emissions (a large and a small livestock farm) and b) the planting of woodland around an area sensitive to ammonia deposition (a large and a small area for nature conservation). For the scenarios here, the intensive poultry farm in Figure 2 was 'moved' to the west ~2.5 km, while selected areas of woodland /semi-natural land were denoted as hypothetical SACs (hSAC). Figures 4a and 4b show the difference in deposition with and without a 50 m belt of trees round the farms and nature reserves, respectively for part of the study area.

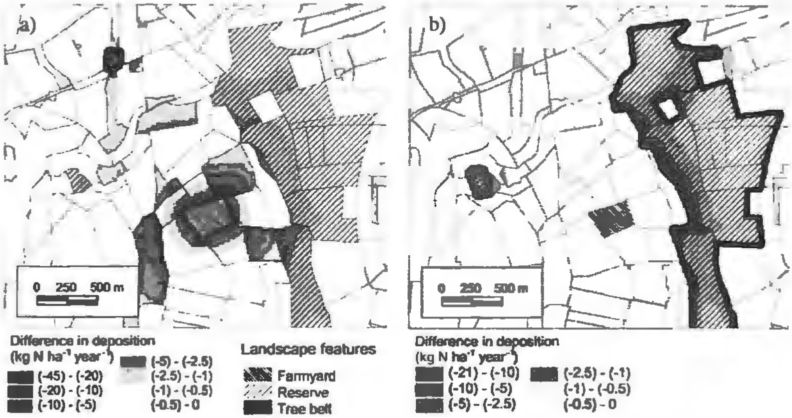

Figure 4. Effect of two scenarios of landscape planning using trees to reduce the impacts ammonia deposition on two hypothetical Special Areas of Conservation (hSAC). Reductions in deposition in neighbouring areas resulting from planting 50 m wide belts of trees a) around two farms, and b) around two hSACs.

The increased deposition to trees "planted" around the farm buildings (Figure 4a), and the increased dispersion leads to a reduction in deposition that is largest near the farms (beyond the planted trees) and extends into the hSACs. The estimated reduction in deposition in the hSACs is up to 5 kg N ha^{-1} yr^{-1}. If the critical load for the nature reserves were 10-15 kg N ha^{-1} yr^{-1}, this would represent a reduction up to 33-50% of the critical load. Substantial benefits are also seen in other nearby semi-natural and woodland areas, apart from the hSACs. The scenario of planting trees around the two hSACs produces benefits that are targeted almost exclusively on the hSACs themselves (Figure 4b). The benefits are largest at the edges of the hSACs nearest the largest sources of NH_3, with values up to 10 kg N ha^{-1} yr^{-1}, or 66-100% of the same critical load assumed above. This second scenario provides increased benefits for the small hSAC compared with the first scenario, since, by nature of its small size, all of the reserve is close to the trees. While these represent significant reductions in deposition in relation to the critical load, it is important to recognise that current deposition may be substantially larger above the critical load. Hence, while such measures will help, they may need to form part of a package of measures to protect the hSACs.

Discussion and conclusions

The UK analysis of critical loads exceedance for woodlands provides startling and worrying results: around 90% of all woodlands in the UK are estimated to receive deposition in excess of the critical load. This indicates that woodland flowers and other ground flora are being substantially affected by atmospheric nitrogen deposition, with the effects to a large extent being a result of ammonia emissions. Current UK policies, which aim at a ~10% reduction in ammonia emissions by 2010, are not sufficient to avoid this threat. This means, that we need to prioritise what should be protected (to maximise the benefits in proportion to the costs of any abatement strategy) and that imaginative approaches are needed to find complementary methods for mitigating the effects of atmospheric ammonia. Trees can play an important role in landscape planning to protect priority from N deposition. In this case, trees may be planted 'sacrificially' around farms or statutory conservation sites to do the job of capturing and dispersing ammonia. Effects of N deposition are therefore expected in such woodlands, in return for a reduction in effects on priority nature conservation areas.

It is important to note the constraints in this approach to ammonia abatement. Firstly, the undesirable effects on such woodlands need to be accepted (e.g., changes in woodland ground flora, increases in emissions of oxidised nitrogen). Other possible effects on protected sites also need to be considered, such as on water levels. By contrast, these woodlands may provide additional benefits, such as for visual screening, recreation, cover for wildlife and shelter for stock. The benefits of ammonia abatement could also be counted towards the scoring of woodland grant scheme proposals (Theobald *et al.*, 2004). Such benefits complement the use of woodlands in urban air quality planning, where they may be used to reduce particulate matter (PM_{10}) concentrations (Bealey *et al.*, 2004). The planting of trees is of particular interest in relation to current agri-environment reforms. Agricultural grants are becoming increasingly decoupled from production and linked more closely to environmental protection. Hence land around livestock farm buildings or nature reserves, which is currently grassland or arable land, might more easily be converted to farm woodland. The challenge is to ensure that the ammonia issues are considered as part of these agri-environment reforms.

Finally, the landscape analysis points to the benefits of designating statutory nature areas as large contiguous blocks, rather than isolated fragments or linear features. The habitats of small or linear conservation sites are at higher risk from ammonia deposition from adjacent agricultural activities, due to the increased fraction of the area that represents edges. Hence, by prioritising the protection of larger areas (over smaller fragments) it is more likely that there is space available for "buffer zones", while areas at the centre of the reserve are an increased distance away from the nearest ammonia sources. Such buffer zones could either be outside or inside the boundary of designated area (e.g., SSSI or SAC). The former might consist of avoiding use of manures on adjacent fields, or planting adjacent fields with woodland. Placing buffer zones *in* the designated areas would put less pressure on adjacent land managers, but would require new regulatory approaches, e.g., to SACs, where "significant adverse effects" would be considered as acceptable within the buffer zone. Such strategies point to the benefits of retaining existing woodland on the boundaries of conservation sites, even though a site may be designated for another habitat. For example, a site may be designated for its heathland, but retention of an existing belt of trees around the site would be useful to reduce the impacts of ammonia. In an agricultural region, this may be preferable to restoring the entire area of the site to heathland. Although restoration of woodland to heathland would increase the heathland extent, this would place the existing heath under increased threat from ammonia.

Acknowledgements

The authors gratefully acknowledge financial support from the NERC for the GANE 'LANAS' project and DEFRA for the 'AMBER' project (W0719).

References

Bealey W.J.; Nemitz E.; Fowler D. & McDonald A.G. (2004) An evironmental information system for planners to estimate the reduction of urban PM_{10} concentrations by trees *J. Environ. Planning and Man.* (under review).

Bobbink R.; Ashmore M.R.; Braun S.; Flückiger W. & van den Wyngaert I.J.J. (2003) Empirical critical loads for natureal and semi-natural ecosystems: 2002 update. In: *Empirical critical loads for Nitrogen* (Eds. B. Achermann and R. Bobbink) 43-170. SAEFL, Berne, CH.

Cowell D.A. & ApSimon H.M. (1998) Cost-effective strategies for the abatement of ammonia emissions from European agriculture. *Atmos. Environ.* **32** (3), 573-580.

Defra (2002) *Ammonia in the UK*, Defra, London. 89 pp.

Dragosits U.; Theobald M.R.; Place C.J.; Lord E.; Webb J.; Hill J.; ApSimon H.M. & Sutton M.A. (2002) Ammonia, emission, deposition and impact assessment at a field scale: a case study of sub-grid spatial variability. *Environ. Pollut.* **117** (1), 147-158.

Fangmeier A.; Hadwiger-Fangmeier A.; van der Eerden L. & Jaeger H.J. (1994) Effects of atmospheric ammonia on vegetation – a review. *Environ. Pollut.* **86**, 43-82.

Hall, J.; Ullyett, J.; Heywood, L.; Broughton, R. & Fawehinmi, J. (2004) Addendum: The Status of UK Critical Load Exceedances. April 2004. http://critloads.ceh.ac.uk

Loubet B. (2000) *Modélisation du dépôt se d'ammoniac atmosphérique à proximité des sources.* PhD thesis. Université Paul Sabatier, Toulouse, France (in French).

Pitcairn C.E.R.; Skiba U.M.; Sutton M.A.; Fowler D.; Munro R. & V.K. Kennedy (2002) Defining the spatial impacts of poultry farm ammonia emissions on species composition of adjacent woodland groundflora using Ellenberg indicators, nitrous oxide and nitric oxide and foliar nitrogen as marker variables. *Environ. Pollut.* **119**, 9-21.

Sutton M.A.; Asman W.A.H.; Ellerman T.; van Jaarsveld J.A.; Acker K.; Aneja V.; Duyzer J.H.; Horvath L.; Paramonov S.; Mitosinkova M.; Tang Y.S.; Achermann B.; Gauger T.; Bartnicki J.; Neftel A. & Erisman J.W. (2003). Establishing the link between ammonia emission control and measurements of reduced nitrogen concentrations and deposition. *Environ. Monit. Assess.* **82** (2) 149-185.**Sutton M.A.; Pitcairn C.E.R. & Fowler D. (1993)** The exchange of ammonia between the atmosphere and plant communities. *Adv. Ecol. Research.* **24**, 301-393.

Theobald M.R.; Milford C.; Hargreaves K.J.; Sheppard L.J.; Nemitz E.; Tang Y.S.; Phillips V.R.; Sneath R.; McCartney L.; Harvey F.J.; Leith I.D.; Cape J.N.; Fowler D. & Sutton M.A. (2001) Potential for ammonia recapture by farm woodlands: design and application of a new experimental facility. *TheScientificWorld* 1(S2), 791-801.

Theobald M.R.; Milford C.; Hargreaves K.J.; Sheppard L.J.; Nemitz E.; Tang Y.S.; Dragosits U., McDonald A.G., Harvey F.J., Leith I.D., Sneath I.D., Williams A.G., Hoxey R.P.; Quinn A.D.; McCartney L.; Sandars D.L.; Phillips V.R.; Blyth J.; Cape J.N.; Fowler D. & Sutton M.A. (2004) AMBER: Ammonia Mitigation By Enhanced Recapture. Impact of vegetation and/or other on-farm features on net ammonia emissions from livestock farms. Final report to Defra. CEH Edinburgh.

Investigating the impact of tree shelter belts on agricultural soils

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Abstract

There is growing concern that modern agricultural practices have reduced the infiltration capacity of the soil, thereby reducing the soil's ability to absorb rainwater. There are few quantitative data available, however, on the impact of land use on runoff and flood risk. A preliminary study was undertaken in the Nant Pontbren catchment, mid-Wales. This land is used extensively for grazing and experimental tree shelter belts were established in selected pastures. Infiltration rates were up to sixty times higher in areas planted with trees than in adjacent grazed pastures and significant differences were also observed for soil moisture and pH. Surprisingly, soil bulk density varied little between the two areas. The results indicate that more research is needed to gain a better understanding of the processes in operation. This study demonstrates that farm trees could represent a key landscape feature, reducing run-off even when only a small proportion of the land cover.

Introduction

The sustainable use of soil

There is increasing awareness of soil as a fundamental and non-renewable natural resource. The Royal Commission on Environmental Pollution's 19th Report 'Sustainable Use of Soil' (RCEP, 1996) introduced the notion that soils should be accorded the same priority in environmental protection as air or water. There is now an EU thematic strategy for soil protection and a framework directive is expected at the end of 2004. Soil protection is also influencing other policy areas, most notably land use policies such as the Common Agricultural Policy (CAP) (CEU, 2003). These policies have identified several issues of concern, one of which is soil structural degradation.

Soil structure

The structure of the soil can be degraded by a range of land use activities, such as housing and industrial development, the reclamation of mineral workings and spoil heaps, engineering works and forest management (Stevens *et al.*, 2002). The principal concern, however, has been with agricultural land management because of its dominant land use in England and Wales. There are different types of soil structural deterioration associated with agricultural practices. Effects such as surface capping, poaching, compaction of the soil surface and the development of a plough pan (compaction at depth) can all change the hydrology of the soil. One problem of particular interest is compaction of the soil surface (the topsoil). This can occur from vehicular movement, untimely operations and grazing animals. When soils are compacted, the pore space through which water moves into and through the soil becomes reduced (Lull, 1959). This leads to a reduction in the infiltration of rainfall and increases surface runoff (Holman *et al.*, 2002). If this occurs over large areas within a catchment it can alter the hydrology of whole rivers. There is anecdotal evidence that the response of rivers to rainfall events is becoming more intense because of agricultural soil management practices (Stevens *et al.*, 2002).

The effect of sheep on soil physical properties has been a subject of research for many years (e.g., Alderfer and Robinson, 1947; Tanner and Mamaril, 1959). It is of particular concern in Wales where 66% of the land is grassland, with grassland also accounting for 35% of England (Land Cover Map 2000). Research has shown that the pressure exerted by sheep hooves can compact the top 5 cm of the soil, reducing infiltration rate (Gifford and Hawkins, 1978) and increasing the soil bulk density (Greenwood *et al.*, 1997).

The impacts of trees

Several studies have examined hydrological processes in relation to large scale afforestation within single catchments in the UK (Robinson *et al.*, 1994; Neal, 1997) but there is a lack of quantitative data on the impact of lines of farm trees on the frequency and intensity of flooding. What evidence there is comes mainly from North America and tropical and subtropical agricultural systems (e.g., Shultz *et al.*, 1995; Chandler and Walter, 1998; McDonald *et al.* 2002). There are over a million narrow linear tree features in the UK countryside, comprising more than 60 million trees and extending to a length of over 90,000 km (Wright, 1998).

This paper discusses key findings from a preliminary study in the UK which examined the impact of tree shelter belts on the soil.

Methods

Study area

The Nant Pontbren is a headwater tributary of the River Severn in Montgomeryshire, Mid-Wales. The land is situated between 150 and 400 m above sea level with a mean annual rainfall of over 800 mm. The soils in this area consist of cambic stagnogleys with stagnogleyic brown earths on steeper slopes and cambic stagnohumic gleys at higher altitudes.

A large proportion of the catchment is farmed by the "Pontbren group"; a consortium of ten farming families. The group aims to provide a sustainable approach to farming, diversifying their farming practices and increasing their economic returns. Part of the group's management strategy has been to fence off small areas of land and plant trees to provide shelter for animals and wood products, notably coppice woodchip for animal bedding. This provides economical benefits through savings on the cost of bedding. The tree belts are also anticipated to have environmental benefits as buffer strips and wildlife habitat. This has provided an ideal opportunity to investigate the impacts of grazing and tree planting on the soil.

Sample design

Six grazed pastures were studied in November 2002. These pastures were used for grazing sheep with stocking rates around 2 to 3.5 Live Stock Units (LSU) per hectare (a sheep is equivalent to 0.15 LSU). Each pasture contained an enclosed area planted with trees. The trees were slot planted directly into the turf, with minimal soil disturbance, at a density of 2,500 stems ha^{-1}. The predominant species were Silver and Downy Birch (*Betula pendula* and *Betula pubescens*) and Alder (*Frangula alnus*), with small numbers of hedge species such as Blackthorn (*Prunus spinosa*), and Oak (*Quercus robur*) and Ash (*Fraxinus excelsior*). The sites were selected to represent a range of tree ages; 2, 6 and 7 years.

At each site two 10 m transects were located approximately 15 m from each other at right angles to the boundary fence. These transects were positioned between the planted trees. Sample points were located 1 m and 5 m from the boundary on both the tree and pasture side of the fence (Figure 1). A range of soil physical and chemical properties were measured at each sample point (Bird *et al.*, 2003). This paper presents the results for a selection of these properties: infiltration rate, soil bulk density, soil moisture, soil organic matter and soil pH.

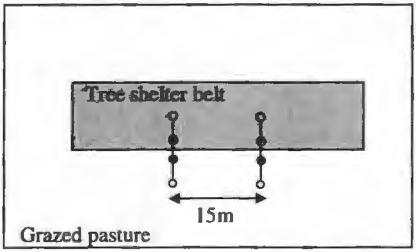


Figure 1. Sample design. Open circles are samples 5m from fence, closed circles are samples 1m from the fence.

Soil properties

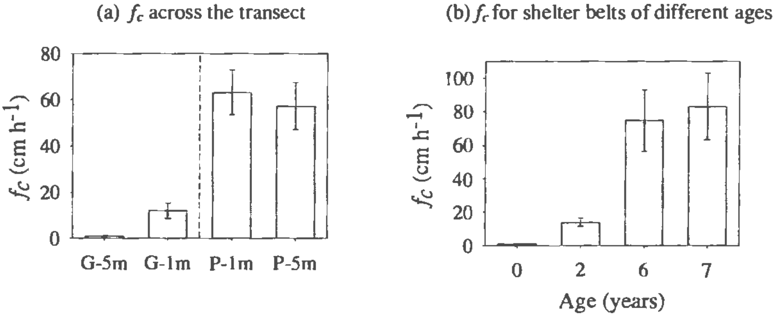
Infiltration rate was measured using a single ring infiltrometer. These rings were 12 cm in diameter, inserted to a depth of 10 cm and filled with water. Infiltration rates were calculated from the drop in water over fixed time intervals. This rate depends on (1) the moisture status of the soil, (2) soil texture, (3) soil structure and (4) vegetation cover. As the soil becomes saturated the rate will depend on 2 - 4 only as these remain unchanged and a constant infiltration rate (f_c) is reached. The average f_c was calculated for each sample point. Soil bulk density cores were taken using a slam-hammer soil bulk density sampler. Two samples were taken at each point: one at the soil surface and one at a depth of 20 cm. These cores were oven dried at 105°C to calculate soil moisture and bulk density. Soil samples were taken with a hand trowel at two depths: 0 – 5 cm and 20 – 25 cm. Organic matter was estimated by loss on ignition (LOI) in a muffle furnace at 375°C. The soil pH was measured with a standard pH meter for soil suspended in deionised water, using a ratio of 1:2.5.

Results and discussion

Infiltration rates

The mean constant infiltration rates (f_c) with associated standard error are shown in Figure 2a. The lowest f_c was found for points 5 m into the grazed area, with f_c increasing towards the boundary. It may be expected that f_c would be smallest close to the fence due to greater sheep trampling. This effect, however, was not observed. There was little difference between f_c 1 m and 5 m into the planted area. The results indicate that infiltration was up to sixty times greater in the planted area than in the adjacent grazed pastures. An analysis of variance showed that these differences were significant at the 99.99% level.

The mean f_c was also calculated for sites with trees of different ages (Figure 2b). The f_c for the 2 year old shelter belt was markedly different to that of the grazed areas (0 years, Figure 2b), suggesting that changes in the soil happen quickly. The f_c increased further from the 2 to 6 year old trees. There was little difference between areas where the trees were 6 and 7 years old. The sample size in this study was small and further research with more replicates of different aged tree belts is merited.

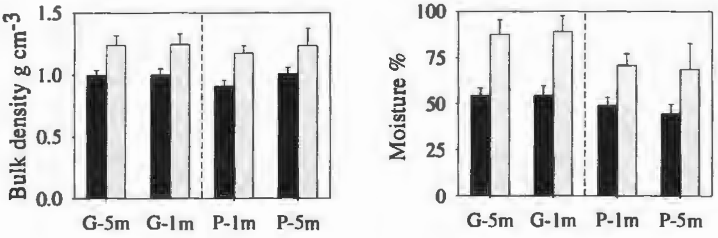

(a) f_c across the transect

(b) f_c for shelter belts of different ages

Figure 2. Mean infiltration rates (f_c) for (a) grazed, G, and planted, P, areas and (b) shelter belts of different ages.

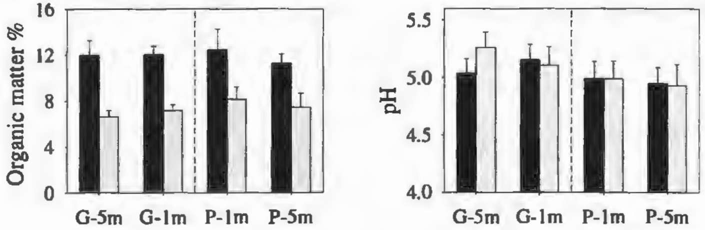
Soil bulk density

The average soil bulk density and associated standard error for the two depths is shown in Figure 3a and the values given in Table 1. It was expected that soil bulk density would be larger in the grazed pasture due to compaction by sheep. This effect, however, was not observed and there was little difference between the values in the two areas. The bulk density was larger for the subsoil, reflecting the heavier texture of the soil. The values for the two depths fall in the middle of the range for Welsh soils (Rudeforth *et al.*, 1984) suggesting that the soils are not heavily compacted.



(a) Bulk density

(b) Moisture



(c) Organic matter

(d) pH

Figure 3. Average values across the transect in the grazed, G, and planted, P, areas for (a) soil bulk density, (b) soil moisture, (c) organic matter and (d) soil pH. Dark bars represent 0 – 5 cm and grey bars 20 – 25 cm data.

Table 1. Average values from the 6 sites for each transect position

Property	Depth	Grazed 5m	Grazed 1m	Planted 1m	Planted 5m
Infiltration cm h^{-1}		12.08	1.00	63.17	57.25
Bulk density g cm^{-3}	0-5cm	1.00	0.99	0.91	1.01
	20-25cm	1.25	1.24	1.17	1.24
Moisture† %	0-5cm	89.12	87.58	70.86	68.83
	20-25cm	54.70	54.50	48.81	44.63
Organic matter %	0-5cm	12.05	12.01	12.46	11.24
	20-25cm	7.17	6.62	8.15	7.46
pH	0-5cm	5.16	5.04	4.99	4.95
	20-25cm	5.11	5.26	4.99	4.93

† g H_2O per 100g oven dry soil

The relationship between soil bulk density and f_c was examined through Pearson's product moment correlation coefficient (Table 2). Correlations were calculated for the grazed pasture and planted areas separately. The planted area showed a significant correlation (at the 90% level) between bulk density and f_c at both depths, although the grazed area showed no relationship. It appears, therefore, that although there is little difference between bulk density in the grazed and planted areas, the planted area shows a strong relationship to infiltration. This suggests that the ratio of micro to macro pores in the soil is important. When macro pores are present, such as those formed by tree roots, water can move through the soil easily. Bulk density provides a measure of the mass of soil per unit volume, it does not take into consideration the pore sizes.

Table 2. Product moment correlation coefficient for f_c and measured soil properties.

Property	Depth	f_c *planted*	f_c *grazed*
Bulk density g cm^{-3}	0-5cm	-0.70*	0.00
	20-25cm	-0.74*	-0.38
Moisture %	0-5cm	-0.32	-0.17
	20-25cm	0.32	0.17
Organic matter %	0-5cm	-0.12	0.20
	20-25cm	-0.06	-0.34
pH	0-5cm	-0.94*	-0.36
	20-25cm	-0.87*	-0.07

*** Significant at the 90% level**

Soil moisture

The moisture content for the soils was large in general, averaging around 80% for the surface and 50% at depth. This reflects the wet conditions during the sampling period. Overall, the average soil moisture content was larger for the grazed pastures than the planted areas (Figure 3b, Table 1). An analysis of variance showed significant differences between moisture in the grazed and planted areas for the soil surface (99% level) and at depth (95% level). It is possible that the smaller moisture contents were related to increased evaporation transpiration losses in the planted area, i.e., canopy interception, which could be high during windy periods. Transpiration losses, however, were probably low as sampling took place in autumn. The pattern in moisture content appears be related to the infiltration rate. When water moves more easily through the soil, typified by a large infiltration rate, it would be expected that the moisture content would be smaller.

Soil organic matter

The soil organic matter was highly variable both between transect points and between sites and little could be concluded from the data (Figure 3c, Table 1). The similarity between the grazed and planted areas is not surprising as it is probably too soon for leaf litter from the trees to have had a significant impact on the organic matter content. In addition, slurry and the excretal and manure returns from the grazing animals to the pastures may have increased the organic matter content in this area. Any differences present, however, may have occurred at the soil surface only, and have been lost by sampling to a depth of 5 cm. There was no relationship between soil organic matter and f_c (Table 2).

Soil pH

The soil pH was slightly more acidic under the tree planted areas than the grazed pastures (Figure 3c, Table 1). The differences between these areas were significant at the 90% and 95% level for the 0 – 5 cm and 20 – 25 cm depths, respectively. The pH for both depths showed a significant negative relationship to f_c in the two areas (Table 2). The soil pH also showed a negative relationship to soil moisture.

There are several possible processes that may give rise to the increased acidity beneath the trees. These include: i) leaching of organic acids from the developing humus layer; ii) capture of aerosols and particles by dry and wet deposition to exposed 'edges', this will be increased compared to short grass canopy; iii) lower moisture contents leading to relatively higher soil water concentrations (evaporative concentration effect), which will change the soil surface chemistry. This needs further work, however, before any assumptions can be made.

Conclusion

The results from this preliminary study indicate that tree shelter belts established on permanent pastures can greatly increase water infiltration into the soil. It also appears that changes in infiltration rate occur very rapidly (2 – 6 years). The soil moisture content is smaller under the planted areas than the pastures, suggesting better drainage in these areas. The soil pH suggests that in the planted areas the soil has also become more acidic.

The effect of tree shelter belts on the soil bulk density is unclear. There appears to be little difference between bulk density in the adjacent grazed and planted areas, although the planted areas show a relationship to f_c. It is possible that the ratio of micro to macro pores in the soil is more important than the bulk density. There was no difference in the organic matter content in the grazed and planted areas. The results from this study suggest that more detailed work is needed to gain a better understanding of the processes in operation in the two areas.

This preliminary study suggests that strategically placed, small scale planting of trees for shelter can be used to improve the infiltration capacity of extensive areas of grazed permanent pasture. This is likely to have positive impacts on run off rates, erosion and stream water quality at both farm and landscape scales.

Acknowledgments

This work was funded by the Welsh Assembly Government, Countryside Council for Wales and the Forestry Commission. Thanks to the Pontbren group for allowing access to their land and to David Jenkins, Coed Cymru for organising the work.

References

Alderfer, R.B. & Robinson, R.R. (1947) Runoff from pastures in relation to grazing intensity and soil compaction. *Journal of American Society of Agronomy* **39**: 948-958.

Bird, A.B; Emmett, B.A; Sinclair, F.L; Stevens, P.A; Reynolds, B; Nicholson, S. & Jones, T. (2003) *Pontbren: Effects of tree planting on agricultural soils and their functions.* Report to CCW, NAW and FC, March 2003.

CEU (2003) COUNCIL REGULATION (EC) No 1782/2003 (29 September 2003). 'Establishing common rules for direct support schemes under the common agricultural policy and establishing certain support schemes for farmers and amending Regulations. (EEC)No 2019/93, (EC)No 1452/2001,(EC)No 1453/2001,(EC)No 1454/2001,(EC)1868/94,(EC)No 1251/1999,(EC) No 1254/1999,(EC)No 1673/2000,(EEC)No 2358/71 and (EC)No 2529/2001'.

Chandler, D.G. & Walter, M.F. (1998) Runoff responses among common land uses in the uplands of Matalom, Leyte, Philippines. *Transactions of the ASAE* **41**(6): 1635-1641.

Gifford, G.F. & Hawkins, R.H. (1978) Hydrologic impact of grazing on infiltration: a critical review. *Water Resources Research* **14** (2): 305-313.

Greenwood, K.L; MacLeod, D.A. & Hutchinson, K.J. (1997) Long-term stocking rate effects on soil physical properties. *Australian Journal of Experimental Agriculture* **37**: 413-419.

Holman, I.P; Hollis, J.M. & Thompson, T.R.E. (2002) *Impact of agricultural soil conditions on floods – Autumn 2000.* R&D Technical Report W5B-026/TR. Environment Agency, Bristol.

Lull, H.W. (1959) *Soil compaction on forest and rangelands.* USDA, Forest Service Miscellaneous Publication No 768.

McDonald, M.A; Healey, J.R. & Stevens, P.A. (2002) The effects of secondary forest clearance and subsequent land use on erosion losses and soil properties in the Blue Mountains of Jamaica. *Agriculture, Ecosystems and Environment* **92**(2): 1-19.

Neal, C. (1997) (editor) Water quality of the Plynlimon catchments (UK). Special issue, *Hydrology and Earth System Sciences* **1**: 381-794.

RCEP. (1996) *Royal Commission on Environmental Pollution, 19th Report: Sustainable Use of Soil.* HSMO, London.

Robinson, M; Jones, T.K. & Blackie, J.R. (1994) *The Coalburn experiment – a twenty five year review.* National Rivers Authority, R&D Note 270.

Schultz, R.C; Colletti, J.P. Isenhart, T.M; Simpkins, W.W; Mitze, C.W. & Thompson, M.L. (1995) Design and placement of a multispecies riparian buffer strip system. *Agroforestry Systems* **29**(3): 201-226.

Stevens, P.A; Reynolds, B; Emmett, B.A; Thompson, T.R.E; Loveland, P.J; Jarvis, S.C; Haygarth, P; Thomas, H.R; Owen, D.H; Roberts, R. & Marsden, T. (2002) *Critical appraisal of state and pressure and controls on the sustainable use of soils in Wales.* Report to the Environment Agency and National Assembly for Wales, Contract 11406.

Tanner, C.B. & Mamaril, C.P. (1959) Pasture soil compaction by animal traffic. *Agronomy Journal* **51**: 329-331.

Wright, D. (1998) *National inventory of woodland and trees.* Forestry Commission Information Note 8. Forestry Commission, Edinburgh.

Session 5

Assessing the resource

Ecological change in British broadleaved woodlands between 1971 and 2000: a pilot re-survey of 14 of the 103 'Bunce survey' sites

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Abstract

We report a pilot analysis of ecological change in 14 out of 103 British broadleaved woodlands sampled in detail in 1971. Sites were visited again in 2000. At 16 fixed plot locations in each wood, soil, vegetation and management information was recorded. These data were used to detect and infer causes of change over the twenty-nine year interval. Three major changes were detected; a) average loss of six higher plants per plot occurred across the site series, b) tree and shrub populations saw a net cross-site increase in stems between 25 and 50 cm diameter at breast height (DBH) and net loss of stems in the youngest and oldest age classes, c) soil pH increased significantly. Species richness reductions in the ground layer were correlated with increased shading due to canopy growth, while soil pH change was hypothesised to reflect recovery from acidification. Linkage between change and potential causes will be explored further in an extension of the resurvey to all 103 sites.

Introduction

This paper presents the results of a pilot assessment of large-scale ecological change in 14 British broadleaved woodlands surveyed in 1971 and again in 2000. These sites were randomly selected from 103 woodlands that originated as a representative subset of 2,463 native broadleaved woodlands from which a UK-wide representative series of nature reserves were designated in the mid-seventies (Ratcliffe, 1977). Measurements of canopy population structure, canopy and ground layer species composition, management indicators and soil properties were recorded in both surveys from 16 plots randomly positioned in each site in 1971 and visited again in, as far as possible, the same locations. Hence, paired measurements were used to answer two questions: first, had change occurred in the intervening 29 years and, second, could observed changes in the species composition and richness of the ground layer be explained using explanatory variables relating to canopy growth, management and soil.

The analysis was designed as a pilot precursor to a full resurvey of all 103 sites. The purpose of the project was to address logistical issues and trial analytical approaches. Full details have already been published (Smart *et al.*, 2001). This paper aims to more widely publicise the key findings.

Methods

Field recording

Fourteen woodland sites were randomly selected from the original 103. Within each woodland in 1971, 16 relocatable 200m^2 plots were randomly placed. In each plot the herbaceous flora was recorded and soil samples were taken. In addition all tree and shrub stems over 1.3 m in height were counted and diameter measured. In 2000, all 16 plots on each site were located again and new records made of plant species composition and the age-class distributions of trees and shrubs. Soil sampling was also repeated. See Smart *et al.*, (2001) for full details.

Analytical methods

Mixed-model ANOVA (SAS Institute, 1997) was used to quantify change over time in the floristic, soil, management and DBH data. Floristic data were summarised as mean unweighted Ellenberg scores based on values recalibrated for the British situation (Hill *et al.*, 1999) and mean values for Grimes' three established strategies of plant growth; Competitors, Stress-tolerators and Ruderals (Grime *et al.*, 1995). Other variables analysed included: species richness of higher plants in the ground layer; soil pH; soil organic matter content; woody basal area; change in the shape of the DBH-class distribution curve for trees and shrubs; extent of open habitats; and change in the number of signs of recent grazing pressure.

Ordination techniques (CANOCO v4.0) were also used to extract the major gradients of floristic variation in the plot data recorded in 1971 and 2000 and to decompose variation into temporal and spatial components (e.g., Marrs & LeDuc, 2001).

Results

Patterns of change in age-class distribution of tree and shrub species

Between 1971 and 2000 there was a clear overall trend for reductions in the youngest and oldest age classes and a net increase in numbers of stems between 25 and 50 cm in diameter (Figure 1). The overall cross-site pattern indicated an ageing series of mixed tree species populations. Differences between sites and plots were not large enough to obscure this overall pattern.

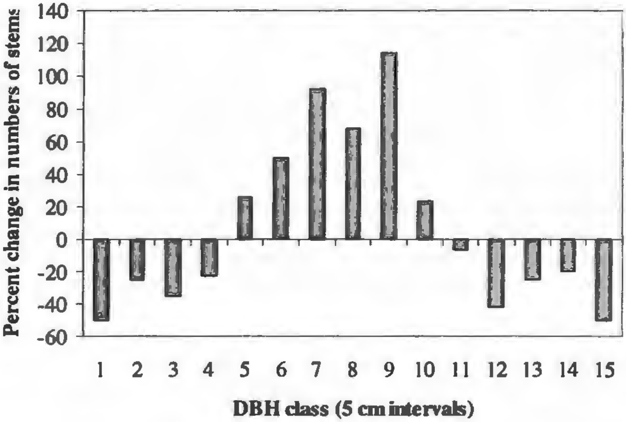

Figure 1. Net percentage change in counted stems across all sites and tree/shrub species between 1971 and 2000. DBH was measured at 1.3m. Classes start at 1 (0-5cm diameter).

Most canopy dominants saw ageing populations with little new recruitment. The only two species to see cross-site increases in younger stems were *Ilex aquifolium* and *Acer pseudoplatanus*.

Despite lack of recruitment to the youngest age-classes and ageing of tree populations, all sites had youthful population structures compared with North American or European old-growth forests.

Change in species richness

Across all sites and plots, ground layer species richness declined significantly (t=-2.65, df=13.6, p=0.0193) with a mean loss of six higher plant species per plot. The only two sites to show increases in species richness had been severely disturbed by clearfelling (Priestfield) or as a result of the effects of the October 1987 storm that swept the south east corner of Britain (Wellhanger) (Figure 2).

Change in soil pH

Soil pH increased between 1971 and 2000, especially in the more acidic soils. Increases were more a function of initial pH category rather than site. Hence, similar soil types saw similar changes overall.

Change in soil organic matter content

No significant overall change occurred between 1971 and 2000.

Changes in species composition

Simultaneous ordination of 1971 and 2000 ground flora plot records showed that only 1% of the total variation was explained by time. Site level effects explained 27% of the variation. Hence most variation was either residual or occurred between plots. When community patterns were constrained by potential correlates of ecological change, time was the tenth most important variable selected from a total of 33 variables and was significant by permutation test at p=0.005. Time correlated most strongly with ordination axis 2 which in turn was most highly correlated with a) the contribution of ruderal strategies to field layer species composition (tendency for higher ruderal contribution in 1971) and b) with soil pH (tendency for higher pH in 2000). Even though the variation accounted for by temporal change was small relative to the other sources of variation in the data, these explanatory variables suggested that shifts along gradients of shade and soil pH had occurred over time.

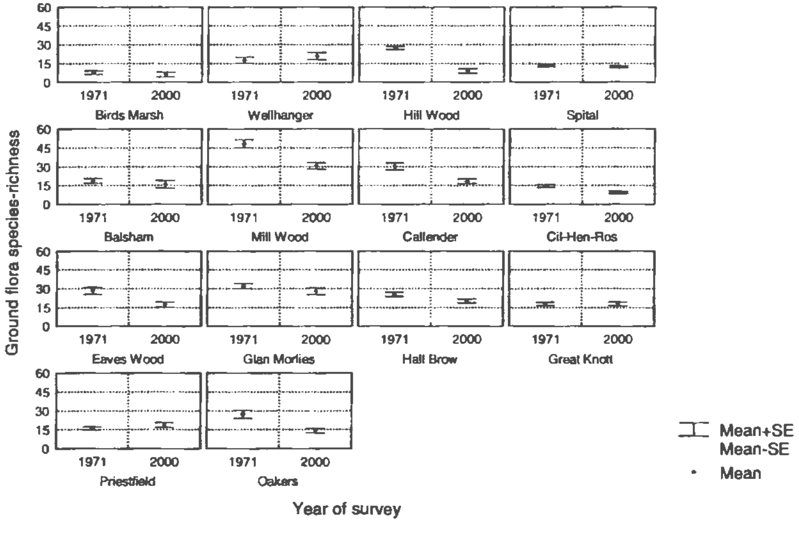

Figure 2. Changes in higher plant species richness in the ground layer in 14 British broadleaved woodlands between 1971 and 2000.

Which variables best predicted the change in species richness?

Changes in field layer species composition over time occurred alongside marked reduction in mean species richness within plots. Changes in species composition appeared to be linked to shifts in soil pH and an increase in shade. The relative importance of these condition changes was assessed by multiple regression of the explanatory variables used in the CCA analyses onto *change in* species richness (Tables 1 and 2).

Table 1. Multiple regression results for the dependent variable, change in species richness between 1971 and 2000 and derived correlates of species compositional change (r^2=0.52).

Effect	**df**	**t**	**p**	**Relationship between effect and response**
Change in proportion of ruderal traits	237	7.93	**<0.0001**	+
" stress-tolerant traits	236	2.12	**0.0347**	+
" competitive traits	236	1.65	0.1001	ns
" mean unweighted Ellenberg fertility score	230	-2.02	**0.0447**	-
" mean unweighted Ellenberg light score	229	2.99	**0.0031**	+

Table 2. Multiple regression results for the dependent variable, change in species richness between 1971 and 2000 and independent variables related to change in woodland canopy structure (r^2=0.31).

Effect	**df**	**t**	**p**	**Relationship between effect and response**
Change in skew of DBH curve	228	2.15	**0.0324**	+
Change in number of open habitats	226	3.80	**0.0002**	+
Change in woody basal area	222	-0.93	0.3510	ns

The results confirm the importance of change in shade/disturbance at ground level. Decreases in Ellenberg light score, decreases in the proportion of ruderal plants and increases in the proportion of competitive plants are all associated with decreased species richness. All the variables in Table 1 were not statistically independent of the actual change in richness but suggest the varying importance of change in plot conditions. Those in Table 2 were independent since woody species did not contribute to ground flora species counts in plots. The relationship between species richness change and the skewness of the DBH curve in 1971 is, however, not absolutely straightforward. Plots that had relatively lower counts of younger compared to older stems either lost or gained species. However, plots with the highest numbers of younger stems tended to mostly lose species richness.

A logical constraint on the size of species reductions is the number of species in each plot to start with. While there is not such a constraint on increasing richness, a plot cannot obviously lose more species than were originally present. Indeed the number of species lost per plot did appear to be proportional to the initial total. Differences in initial richness were actually correlated with northing, richer plots being more northern in 1971 (Smart *et al.*, 2001).

Discussion

What were the key changes seen across the 16 pilot sites between 1971 and 2000?

The most striking changes to have occurred were undoubtedly the cross-site decline in mean plant species richness and general increase in soil pH. There was also a widespread ageing of mixed tree populations across most sites characterised by a reduction in abundance of stems in the youngest DBH classes and a net shift in favour of stems between 25-50cm in diameter. This change was associated with a general increase in shade and reduction in field layer disturbance implied by variable changes in the proportion of competitive and stress-tolerant species but a clear, cross-site reduction in the contribution of more ruderal plants (Smart *et al.*, 2001).

What were the most likely causes of change?

The most obvious driver of soil pH change is reduction in acidification largely from reduced atmospheric deposition of SOx (NEGTAP, 2001). These results are consistent with those recently obtained from the Countryside Survey 2000 soil re-sampling that examined changes from 1978 to 1998/9 (see Haines-Young *et al.*, 2000), and also re-sampling data obtained by SSLRC from the National Soil Inventory and ADAS from the Regional Soil Sampling Scheme (both of which are dominated by agricultural soils) (Loveland and Chambers, pers. comm.). Hence, increases in soil pH appear to be a cross-ecosytem phenomenon, seen despite wide variation in management and floristic starting point.

The general reduction in mean species richness within plots was correlated with an apparent lack of disturbance on most sites as tree populations that comprised many more juveniles in 1971 aged over the twenty nine year period. This is perhaps not so surprising since the effect of canopy closure and disturbance on plant species richness in woodlands is well known. The interesting aspect from the pilot study is the prevalence of the change across plots and sites. This pattern maybe associated with recovery from much more open conditions that prevailed in the 1940s, as a result of war-time timber and underwood extraction from many sites across Britain.

The two sites that had experienced major canopy disturbance stood out in that species richness increased between 1971 and 2000. Indeed Priestfield, where canopy removal had affected the majority of plot locations, was the only site to have shown a significant increase in younger stems, an increase in the proportion of ruderals in the field layer and increasing mean plant species richness. Species richness also increased in Wellhanger Copse, large areas of which were blown down during the October 1987 storm. Whether such extensive disturbance as a proportion of these sites is or was desirable is another issue.

Although canopy removal is known to have a positive, medium-term effect on field layer species richness, soil changes could influence the quality of such gaps in the future. If one of the consequences of soil change is a gradual increase in nutrient availability then managed canopy disturbance intended to promote local increases in species richness within sites may have to contend with the vigorous response of field layer competitors whose consolidation and spread would be enhanced. This supposes that increasing soil fertility will result from increasing pH (e.g., Diekmann & Falkengren-Grerup, 2002).

A full resurvey and analysis of the 103 woodland sites is currently in progress. This will provide a geographically dispersed and more replicated dataset enabling us to more rigorously quantify change and infer causes of change in the wider population of native broadleaved woodlands. If, after having scaled-up, equivalent cross-site changes in species richness, canopy structure and soil pH are seen, we ought to have sufficient statistical power to explore potential drivers of soil pH change such as N and SOx deposition history, while also using site and plot level indicators of woodland management to investigate possible drivers of between and within-site succession. It should also be possible to more fully explore the roles of changing grazing pressure, lengthening growing season, intensity of surrounding land-use and site shape/area, as additional spatially variable modifiers of the rate of woodland change.

Acknowledgements

We extend our sincere thanks to all the landowners who granted access to their woods during the survey. We also thank regional staff at Scottish Natural Heritage, Countryside Council for Wales and English Nature for providing site ownership details. This work was jointly funded by English Nature, the Department of the Environment, Food and Rural Affairs, the Joint Nature Conservation Committee and the Forestry Commission.

References

Bunce, R.G.H. (1989) *A Field Key for Classifying British Woodland Vegetation: part 2.* HMSO, London.

Diekmann, M & Falkengren-Grerup, U. (2002) Prediction of species response to atmospheric nitrogen deposition by means of ecological measures and life history traits. *Journal of Ecology* **90**: 108-120.

Haines-Young, R.H; Barr, C.J; Black, H.I.J; Briggs, D.J; Bunce, R.G.H; Clarke, R.T; Cooper, A; Dawson, F.H; Firbank, L.G; Fuller, R.M; Furse, M.T; Gillespie, M.K; Hill, R; Hornung, M; Howard, D.C; McCann, T; Morecroft, M.D; Petit, S; Sier, A.R.J; Smart, S.M; Smith, G.M; Stott, A.P; Stuart, R.C. & Watkins, J.W. (2000) *Accounting for nature: assessing habitats in the UK countryside.* Countryside Survey 2000 main report. Department of the Environment, Transport and the Regions, London.

Grime, J.P; Hodgson, J.G; Hunt, R. & Thompson, K. (1995) *The Electronic Comparative Plant Ecology.* London. Chapman & Hall.

Hill, M.O; Bunce, R.G.H. & Shaw, M.W. (1975) Indicator species analysis: a divisive polythetic method of classification and its application to a survey of native pinewoods in Scotland. *Journal of Ecology* **63**: 597-613.

Hill, M.O; Mountford, J.O; Roy, D.B. & Bunce, R.G.H. (1999) *Ellenbergs' Indicator Values for British Plants.* ECOFACT Volume 2. Technical Annex. Department of the Environment, Transport and the Regions. CEH Monkswood, Huntingdon.

Marrs, R.H. & LeDuc, M.G. (2000) Factors controlling vegetation change in long-term experiments designed to restore heathland in Breckland, UK. *Applied Vegetation Science* **3**: 135-146.

Ratcliffe, D.A. (Ed.) (1977) *A Nature Conservation Review.* Cambridge University Press, Cambridge.

SAS Institute Inc. (1997) *SAS/STAT Software: Changes and enhancements through release 6.12.* Chapter 18. The MIXED procedure. Cary, NC.

Smart, S.M; Black, H.I.J; Bunce, R.G.H; Kirby, K; Isted, R; Ray, N; Singleton, D. & Bunce, F. (2001) *Measuring long-term ecological change in British woodlands (1971-2000): a pilot re-survey of 14 sites from the ITE/NCC 'Bunce 1971' woodland survey and two sites from the 1971 Native Pinewood Survey.* English Nature research reports. No.461. English Nature, Peterborough.

Veteran trees in the landscape: a methodology for assessing landscape features with special reference to two ancient landscapes

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Abstract

This paper sets out a field–tested methodology that can assess the relationship between selected environmental attributes of the landscape. Attributes have been measured from randomly selected sample squares across the Atlantic biogeographic region, as defined by Alterra's environmental land class system. The landscape form (dictated by hydrological factors), relative proportions of the broad habitats, and land uses within each square were also recorded. The methodology pays particular attention to biologically important veteran trees, defined based on a combination of tree size for each species and veteran characteristics. Special reference is made to two ancient landscapes, with significant populations of these veteran trees: the Caledonian Pinewoods of Glenmore in the Cairngorms of Scotland and the Bocage landscape of Brittany in western France. The paper investigates how the attributes measured by this methodology may be used as a measure of ecological functionality and landscape condition.

Introduction

Understanding the relationship between veteran trees and the landscapes in which they are found is important in trying to understand the bio-cultural heritage of where we live. Ancient landscapes are important in the context of past-cultural and present cultural landscapes, as well as future cultural landscapes we may wish to develop. These terms are used in the same way as developed by Peterken for natural woodland (Peterken, 1996).

Connectivity is in many ways the cornerstone of landscape ecology and the development of ecological networks, incorporating veteran trees and the ancient landscapes to which they belong, is vital. Ecological networks are being developed over much of Europe to provide eco-stabilisation, better connectivity between habitats and to support meta-populations of target or focal species. The development of ecological networks is the result of two different approaches with different philosophies behind them (Jongman *et al.*, *2003*). The eco-stabilisation approach is spatial, based on pattern and functional zoning to maintain a landscape where natural and productive zones form a functional network with an acknowledgement of the restorative powers of natural process. A more ecological, or process-orientated, approach is based on dispersal and the ability of species to form meta-populations across the landscape. Here the development of pattern is based on the ability of focal species to move between habitat patches or sites. Veteran trees have the potential to be both 'refugia' for natural processes and 'stepping stones' for species dispersal within ecological networks The development of future cultural landscapes will require habitat networks that takes a pragmatic approach that refers to all aspects of pattern and process.

A methodology that can help to assess the relationship between cultural, ecological and environmental attributes and also provide a measure of ecological functionality and landscape condition would go some way to help develop ecological networks across the range of bio-cultural landscapes.

Methodology

Veteran tree numbers and their distribution were investigated across 31 sites, 8 in the UK, 23 on the European continent. At each site, three 1-kilometre squares were searched and trees recorded that were either above minimum size diameters (75-150cm, depending on the species) or showed at least three 'veteran tree characteristics' (rot holes, hollowing, dead wood etc.). 409 veteran trees were surveyed in detail out of a total of 3,867 recorded that showed some 'veteran' characteristics.

The methodology required an objective definition of a veteran tree based on structural features. By using size classes (based on diameter at breast height: dbh) for different species combined with veteran tree characteristics, a methodology was developed that could be consistently applied to identify veteran trees that are potentially interesting and valuable in terms of nature conservation. (Read, 1999). This inclusive approach also enables us to look at the next generation of veteran trees. The location and surrounding land-use type were noted to give context and to allow for analysis of the relationship between veteran trees and selected environmental attributes of the landscape in which they are found. Full details can be found in Smith and Bunce (in prep).

Environmental attributes measured

Environmental zones/classes

The Atlantic biogeographic region, from within which the sample squares were allocated, was defined by Alterra's environmental land class system, which is a result of the development of scientific principles of environmental classification for strategic ecological survey across Europe.

Landscape form

The landscape form, dictated by hydrological factors, can be measured by the slope of the catchment, which in turn can be derived from the valley width and depth.

Habitats and land use

For the whole square the relative proportions of the broad habitats were estimated within ten percent bands, with the presence of categories occurring over one hectare being recorded as present (categories compatible with the Eunis Habitat Classification). The land uses within each square were also recorded.

Veteran trees

There is still debate about what constitutes a veteran tree, as there is no precise definition. It is a term, which can encompass a wide range of attributes, often associated with the ageing process. This methodology used a structural definition based on different tree sizes for different species and the presence of veteran characteristics as outlined below.

Tree size

The following size categories (dbh at 1.3 m) were used to determine veteran trees by species:

75 cm: field maple (*Acer campestre*), rowan (*Sorbus aucuparia*), yew (*Taxus baccata*), birch (*Betula pendula/pubescens*), holm oak (*Quercus ilex*), cork oak (*Quercus suber*) and Pyrenean oak (*Quercus pyrenaica*);
100 cm: other oaks (*Quercus* robur/petraea), ash (*Fraxinus excelsior*), Scot's pine (*Pinus sylvestris*), alder (*Alnus glutinosa*);
150 cm: sycamore (*Acer pseudoplatanus*), lime (*Tilia* spp), horse chestnut (Aesculus hippocastanum), elms (*Ulmus* spp.), poplars (*Populus* spp.), beech (*Fagus sylvatica*), willows (*Salix* spp.), other pines (*Pinus* spp) and exotics, such as *Eucalyptus*.

Diameters were estimated to the nearest 25 cm, being the realistic accuracy that can be achieved without measurement. No differentiation was made in recording of *Quercus petrea, Q. rober* or hybrids of the two. The category classes were referred to by the lower diameter limit (Table1).

Table 1. Tree size classes recorded

Size range	Category	Size range	Catagory
Under 50 cm	Under 50 cm	125 to 149cm	125cm
50 to 74 cm	50 cm	150 to 174 cm	150cm
75 to 99cm	75cm	175 to 199cm	175cm
100 to 124cm	100cm	Over 200 cm	Over 200cm

Tree attributes

These were based on the veteran tree recording methodology and its revieww(Fay and de Berker, 2003) and the attributes recorded were as follows:

Rot holes

The number of holes > 5cm in diameter (large enough for a bird to nest) were recorded, up to five. Over five holes were recorded as five. These holes can develop through limb loss and bark wounds and are expanded by micro-organisms and invertebrates. They may be occupied by birds and bats.

Rot sites

The number of major rot sites over $300cm^2$ were recorded. Five or more rot sites were recoded as five. Wood may be digested by the colonisation of rot holes (see above) by decay fungi. These sites can then become important for saproxylic species.

Dead wood

Attached and fallen deadwood were measured seperately as units of dead wood. A unit of deadwood was 1m in length and over 25cm in circumferance. Dead wood is often colonised by decay fungi.As fallen and attached dead wood support different suites of species, they were recorded separately.

Table 2. Showing deadwood unit scores

Deadwood units	Score
1-4 units	1
5-9 units	2
10-14 units	3
15-19 units	4
Over 20 units	5

Hollowing

Any hollowing in the trunk or major limbs was recorded as follows:

Table 3. Showing codes for trunk hollowing

Trunk	**Code**
Solid dead	1
Some hollowing	2
Hollow trunk	3
Partial trunk	4
Remnant trunk	5

Trees that showed over three of the above features were recorded as veterans for the purposes of this survey, whether or not they also exceeded the diameter limits set in 3.2. These trees show the habitat characteristics of veteran trees, which are thought to be important in terms of biodiversity. Their smaller size may reflect the influence of environmental factors (e.g., poor growing conditions) or tree management (lopping and topping, which reduces annual increments). Other features were recorded and these are detailed in the field methodology. Information on the landscape, habitats and major land uses of each square was recorded to give the context of the site.

Results

Distribution of veteran trees

The habitats or features associated with all the veteran trees recorded was noted in order to build a picture of where the veteran trees were found across the survey area. The distribution of the trees is summarised in Figure 1 below.

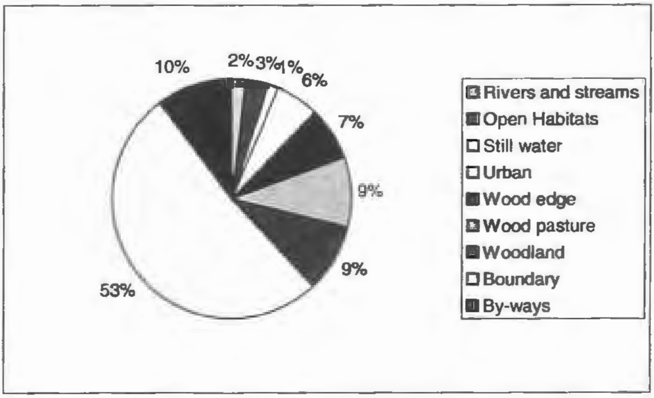

Figure 1. Distribution of veteran trees recorded

Boundary trees were by far the most common encountered across the sample area with over 50% of trees recorded from hedgerows, banks and ditches. Veteran trees associated with wooded habitats accounted for 25% of the recorded trees and a third of these were from wood-pastures. Although 10% were found associated with roads, tracks and avenues, these function in the same way as boundary trees. The 'open habitats' class is the amalgamation of these habitats (e.g., grassland types) and can be regarded as scattered veteran trees and made up only 3% of the trees recorded.

Ancient landscapes

The two examples of ancient landscapes represent the two main veteran tree distribution models: boundary trees within the bocage landscape of Brittany in northwest France and the wooded landscape of the Caledonian pinewoods of Glenmore in the Cairngorms of Scotland.

Caledonian pinewoods of Glenmore in the Cairngorms of Scotland

These forests formed part of the medieval earldom and diocese of Moray and were thought to have been carefully exploited as summer pasture, as part of a transhumance of livestock that occurred up until the1750's, and were in essence a pinewood–pasture. The entire landscape of Moray had been carefully divided into *dabhaichean*. This was done in such a way as to ensure the fair allocation of natural resources. The low-lying east coast of the province, however, lacked the high mountain grazing. The forests of Stratha'an and Glenmore were used for this grazing and the forest of Glenmore had 17 known shielings. The Royal Commission for Historic Monuments - Scotland's (RCHMS) historic land use map has identified six of these areas but the remainder have not been identified and are presumably within the area of existing woodland. Records show that between 1750 and 1753 a total of 3,098 animals were taken to the forest of Stratha'an for summer grazing and it is thought that this was low compared to earlier times where the pasturing of animals was a very profitable business (Alistair Ross pers comm.)

The Caledonian pine forests of Scotland support a wide range of well-recorded species often restricted to isolated populations in remote locations. This includes those associated with deadwood and mature timber habitats including records for the Glenmore awl fly (*Xylophagus junki*), found only once in 1913, which is one of the many that requires veteran Scot's pine (Alexander, 2002). The maintenance of the veteran tree populations that are remnants of this ancient landscape is essential for these saproxylic species as is the connectivity between them.

Bocage landscape of Brittany in Western France

The hedged field systems of the Bocage landscape are found throughout much of Europe. In Brittany this ancient landscape dates back to the 9th century and in many ways can be seen as a form of cultural old growth. The veteran trees on the boundary banks are cut every five to seven years and the foliage used as leaf hay.

Where these small intimate field systems have been maintained, the landscape is in effect acting as a wood pasture on a landscape scale, though divided into smaller management units. The boundary banks with their veteran trees show some of the characteristics associated with 'old growth' elsewhere (Alexander *et al* 2003). The pasture element, depending on past management, can have species-rich grasslands. The willow pollards associated with ditches and drains on flat valley floors can act in the same way. The traditional management of oak trees in Brittany (rascasse) by lopping and topping severely restricts the diameter growth of the veteran trees. Therefore, a tree with a diameter of only 50 cms could be over 200 years old. This is consistent with work by Slotte (2000), which shows that pollards, cut for leaf hay every 3-8 years in Sweden, had mean annual growth rings that were only 20%-35% of that of non-pollards.

The bocage landscape with its hedgerows and veteran trees acts as corridor for flora and fauna, including important carabids and diptera, but due to the limited movements of those species, it may take several generations to move along a corridor. A bird or a small mammal can make the same journey in minutes or days. (J. Boudray, pers comm.)

The veteran trees found in this landscape generally have fairly low amounts of standing dead wood compared to some of the very large structural veteran trees. They are, however, found in larger numbers with greater densities of standing deadwood (per unit area). This results in a high number niches suitable for species associated with veteran trees with good connectivity between them.

Discussion

The relationships between landscapes, habitats, land-use and veteran tree distribution can be further investigated by the use of multi-variate statistical analysis. The patches of woodland, lines of and individual veteran trees within sample squares could also be digitised from the 1:25,000 bases maps and field survey to create GIS polygons, lines and points. These can then be analysed using the 'Fragstats' program to measure selected attributes of connectivity (e.g., nearest neighbour distances between veteran trees). This will give a clearer picture of their distribution when combined with the inclusion of on going sampling.

Analysis of the veteran characteristics of trees sampled can also be undertaken to give 'habitat scores' compared across the different environmental classes to investigate whether the same habitat features might develop in a similar aged tree.

Connectivity

Restoring connectivity between veteran trees, habitats they are found in and other semi-natural habitats is important in terms of reversing habitat fragmentation. Populations of veteran trees that have potential for contributing to greater ecological integrity may be more suitable for restoration. Connectivity can be developed at three levels that relate to core, secondary and tertiary sites:

- Community connectivity to form ecological units with greater species viability and a higher biodiversity/conservation value. This effectively corresponds to consolidation of core sites
- Connectivity of ecological units to form larger areas of diverse grassland habitat of higher biodiversity/conservation value. This may particularly apply to the restoration of degraded sites adjacent to remnant communities of conservation value
- Connectivity of habitats to contribute to a landscape of higher biodiversity/conservation value. This may provide a useful criterion for the selection of sites for restoration or the creation of new semi-natural habitats.

Scattered trees

These trees may well be isolated in terms of dispersal of species, depending on the habitat in which they are found, though they could well add to the structural diversity of the landscape and are also likely to act as refugia for natural processes. It has been noted that isolated veteran trees are often found with woodland plant species and it may well be the case that other species are present relying on the long-established ecological process associated with the veteran trees. The restoration of semi-habitats adjacent to these trees, is likely to contribute to ecological networks

Lines of trees

Veteran trees found in lines may have fairly low amounts of standing dead wood compared to some of the very large structural veteran trees. They are, however, often found in larger numbers with greater densities of standing deadwood (per unit area). This results in a high number of niches suitable for species associated with veteran trees with good connectivity for dispersal between them. These lines of trees are important corridors in the development of ecological networks, not only for linking veteran tree habitats but also for other semi-natural habitats. This is especially the case where they are associated with hedgerows ditches and banks.

Wood pastures

Species' dispersal abilities within wood pastures will be dependent on the density of the trees and the open habitats between them. It is likely that the range of connectivity is wide from near isolated trees to wood pastures with good connectivity between the niches suitable for species associated with veteran trees. These are core areas for concentrations of veteran trees and are of particular importance where the pasture element is comprised of semi-natural open-ground habitats

Woodland

The woodland type and its structure around individual veteran trees will dictate the connectivity between these trees. Disparate veteran trees within a young conifer plantation are likely to be as isolated as those in an arable landscape. Again they may act as refugia for natural process with the potential to act as a locus for habitat restoration. Those in an ancient woodland setting on the other hand may well exist within and contribute to a functional habitat network.

Landscape condition

With further analysis of more sample squares across Alterra's Atlantic environmental classes it may be possible to identify attributes that could be used to measure ecological functionality and condition at a range of different scales. The condition of the veteran trees is one of the features that it is thought could give an indication of chemical, hydrological and physical disturbance within the landscape.

References

Alexander, K.N.A., 2002. The invertebrates of living and decaying timber in Britain and Ireland a provisional annotated checklist. *English Nature Research Reports*, No. 467, Peterborough.

Alexander, K.A.; Smith, M.; Stiven, R. & Sanderson, N.A., 2003. Defining old growth in the UK context. *English Nature Research Reports*, No. 494, Peterborough.

Fay, N. & de Berker, (2003) Evaluation of the Specialist Survey Method for Veteran Tree Recording. *English Nature Research Reports*, No 529, Peterborough.

Jongman, R.; Kulvik, M. & Kristiansen, I. (in Press) European ecological networks and greenways. *Landscape and Urban Planning*, Elsevier

Peterken, G.F (1996) *Natural woodland: ecology and conservation in northern temperate regions* Cambridge University Press, Cambridge

Read, H. (1999). *Veteran trees: a guide to good management.* Peterborough: English Nature.

Slotte, H. (2000). Lovtakt i Sverige och pa Aland. *Agraria* 236. Swedish University of Agricultural Sciences.

Smith, M. & Bunce, R.G.H. (in prep) A preliminary assessment of the distribution and abundance of veteran trees in North West Europe. *English Nature Research Reports*, No. 534, Peterborough.

Assessing the vulnerability of forests to environmental change

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Abstract

Many forests are currently subject to a range of threats, leading to widespread deforestation, habitat degradation and biodiversity loss. Information on the vulnerability of forests to such threats is of key importance to development of conservation strategies. Methods are required that enable the vulnerability of forest habitats to be analysed and mapped, in a way that can inform decision-making. This paper presents an analytical approach to vulnerability assessment, using probabilistic modelling of spatial data. Three dimensions of vulnerability are considered: exposure; severity; and response. These dimensions refer to the probability of a given threat occurring within a particular area, its intensity, and the potential impacts on biodiversity. Each of these dimensions may be analysed using appropriate spatial data together with probabilistic analysis, using techniques such as logistic regression and Bayesian inference with GIS. Case studies illustrate the application of these techniques to analysing the vulnerability of forests to environmental change at global, regional and local scales.

Introduction

It is widely appreciated that analysis of threats to biodiversity is fundamental to the development of effective conservation strategies (Salafsky *et al.*, 2002). At larger spatial scales, identification of those species or habitats that are at greatest risk is essential for developing appropriate priorities. At the scale of landscapes or individual sites, conservation practitioners require tools that enable diagnosis of the factors responsible for actual or potential biodiversity loss, to define which management interventions are appropriate. However, progress towards developing such tools has been limited.

Analysis of the threats to biodiversity faces a number of key challenges. Firstly, the precise causes of biodiversity loss may often be difficult to determine. Successful diagnosis of why a particular species has declined often requires detailed analysis of its natural history, involving an assessment of breeding success and survival at different stages of the life cycle in relation to patterns of environmental variation (Sutherland, 2000). Secondly, the different factors responsible for biodiversity loss typically vary both in space and time, and may act either cumulatively or interactively. Thirdly, and most importantly, the potential future impacts of a particular threat may be difficult to predict, as its relationship with its impact on the conservation status of a species and or habitat is often poorly defined. Analysis is often further hindered by lack of appropriate data describing the intensity or distribution of the threats involved. Methods are therefore needed that enable the risks of biodiversity loss to be estimated on the basis of uncertain information.

In this paper, we describe how progress might be made towards developing such methods. Firstly, consideration is given to the concept of vulnerability and how it may be defined and assessed, in relation to both species and habitats, by reference to current approaches. We then examine how vulnerability may be assessed using spatial analysis combined with methods that permit analysis of uncertain data, illustrated by recent research undertaken in forest ecosystems at both global and regional scales. We then consider how the concept of vulnerability might usefully be applied at the scale of individual sites or landscapes, where conservation management decisions are typically taken. In conclusion, we consider whether analysis of vulnerability might usefully be considered as an integral part of landscape ecology approaches that support conservation of forests and trees.

Current approaches to assessing vulnerability

In recent years, it has been increasingly recognised that the development of conservation priorities should incorporate some measure of the risk of biodiversity loss. Conservation assessments such as the ecoregion approach developed by WWF (Olson and Dinerstein, 1998) and the biodiversity 'hotspots' approach developed by Conservation International (Myers *et al.*, 2000) have attempted to define priorities on the basis of an analysis of current threats, as well as conservation value. However, such analyses have often used qualitative or subjective approaches to analysing threats and their potential impacts on biodiversity, and few have attempted to communicate uncertainty in their analyses (Wilson *et al.*, 2004a). While it is recognised that conservation scientists should place greater emphasis on analysis of threats (Balmford *et al.*, 1998), progress has been limited, largely because of lack of an appropriate analytical approach.

The term 'vulnerability' can usefully be used to refer to the likelihood or imminence of biodiversity loss as a result of current or impending threats (Wilson *et al.*, 2004a). The concept of 'vulnerability' has been interpreted variously by different authors working in a range of research fields, including pollution, land degradation, climate change and human development. Despite wide use of the concept, little attempt has been made to define it precisely or indicate how it may be analysed quantitatively. Wilson *et al.* (2004a) critically reviewed the concept of vulnerability and suggested that three dimensions of vulnerability could usefully be differentiated: (i) *exposure*, referring to the probability of a threat affecting an area over a specified period, (ii) *severity*, reflecting the intensity of a threat within an area, and (iii) *response*, indicating the effects of a threat on particular features of biodiversity. These elements are defined with respect to geographical areas affected. A key advantage of an area-based approach (*Wilson et al, 2004a*) is that it permits spatial analysis, and the production of map-based outputs to inform conservation decision-making. Separation of vulnerability into the three elements offers the possibility of analysing separately the responses of different biodiversity components to a particular threat.

Few other analytical frameworks are available that enable vulnerability to be analysed quantitatively. The most widely applied approach is that developed and implemented by the IUCN Red List process, for assessing the risk of extinction of individual species. In recent years, substantial progress has been made in developing a systematic approach to assessing extinction risk based on objective criteria (IUCN, 2003). Taxa are assigned to a particular category of extinction risk according to the population size and geographic range of the species, and assessments of their rate of decline. The development of the Red List criteria represents an important milestone in conservation science, as they provide the first rigorous, objective method for analysing the vulnerability of individual species. However, the approach does not attempt to detail the specific causes of biodiversity decline, and only relates to species, not habitats. During a Red List assessment, the assessment of threats, as well as the application of the criteria, is based largely on subjective expert judgement.

Wilson *et al.* (2004a) provide a comprehensive review of the different methods used to assess vulnerability, most of which focus on assessing exposure to threatening processes rather than the other two elements. Very few studies address response, highlighting the general lack of information about the specific impacts of different threats. Methods employed by previous researchers were grouped into six main categories, including expert consultation, the use of threatened species as proxies for threats, and previous occurrence of threats (Wilson *et al.*, 2004a). Following a comparative analysis of these different approaches, Wilson *et al.* (2004a) concluded that the use of spatially explicit, statistical or process-based models offer the most objective and reliable method for analysing vulnerability, at least for exposure. This approach can consider many threats concurrently, provide an assessment of their relative importance and allow interactions between threats to be identified. Opportunities also exist to explore uncertainty in the analysis. Despite the merits of such approaches, relatively few studies have employed these methods. Here we describe the results of some recent research attempting to apply such approaches to the analysis of vulnerability of forest ecosystems.

Analysing forest vulnerability at the global scale

A number of assessments have recently been undertaken at the global scale, with the aim of identifying priorities for conservation. These include the global ecoregion assessment undertaken by WWF-US (Olson and Dinerstein, 1998) and assessments of 'biodiversity hotspots' (Myers *et al.*, 2000). However, no current assessment has the explicit objective of assessing the conservation status of different forest ecosystems. As a result of widespread forest loss and degradation, a number of forest types have become the focus of conservation attention, including tropical rain forests, tropical dry forests and temperate rain forests. Yet no quantitative information is available to assess the relative conservation status of different forest types, or the relative importance of different threats affecting them.

For these reasons, a new assessment (entitled 'Forests at Risk') is currently being undertaken by the UNEP WCMC, UK, in association with collaborative partners. The approach involves compiling spatial data sets describing major actual or potential threats to forests, including fire, climate change, infrastructural development, conversion to agriculture, habitat fragmentation, and human population growth. Using a geographical information system (GIS), these data are overlaid on a map of global forest cover derived from remotely sensed data, and analysed to generate maps and statistics describing the forest area affected by each threat. The forest cover data used for this analysis is the MODIS Vegetation Continuous Fields (VCF) product (Miles *et al.*, 2004), which depicts percentage tree cover at a resolution of 500 m. These outputs can be used for comparative analysis of different regions and their relative exposure to threats.

An illustration of this form of analysis is provided here relating to tropical dry forests (from Miles *et al.*, 2004). The results indicate how dry forests in each tropical region are currently subject to multiple threats that affect differing relative areas between regions. For example, a higher percentage of forest area is exposed to three or more threats in Eurasia than in any other region (Figure 1). Results of this analysis indicated that approximately 97% of tropical dry forest is currently exposed to one or more of the threats considered, relative exposure differing between regions: while climate change is relatively significant in the Americas, habitat fragmentation and fire affect a higher proportion of African forests, whereas agricultural conversion is most influential in Eurasia (Miles *et al.*, 2004).

Such analyses can be of value for identifying priorities for conservation action. For example in the case of tropical dry forests, these analyses highlighted specific localised areas of both high conservation value and high vulnerability, which might be usefully targeted during the future establishment of protected areas. However, the approach is analytically simple, in that different threat data are combined with forest cover as an overlay within GIS; no attempt is made to analyse the data probabilistically or to assess interactions between threats. The analyses are also limited by data resolution. Although high-resolution data describing some threats, such as fire, are now becoming available from remote sensing technology, other threats (such as harvesting of forest products) remain difficult to quantify.

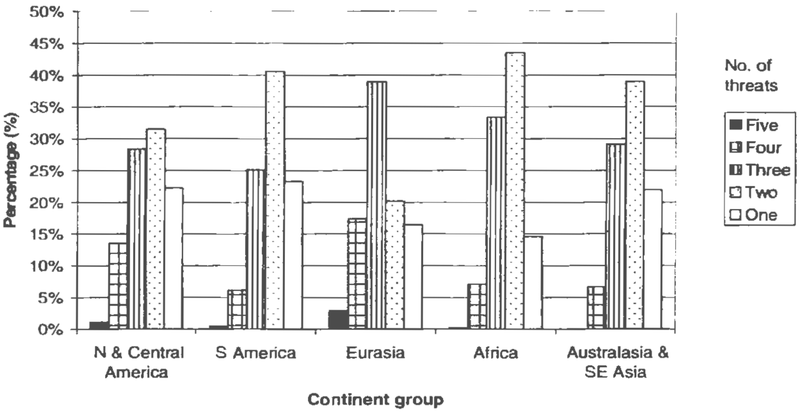
Analysing forest vulnerability at the regional scale

In order to develop and test methods of analysing forest vulnerability, we are currently undertaking a programme of research in the Valdivian ecoregion of Chile. Its temperate rain forests are now recognised to be of global conservation importance as a consequence of their high endemicity. The forests have been subjected to intense human pressure over the past 150 years, primarily due to timber extraction and conversion to agricultural land following European colonisation. These pressures have intensified in recent decades with development of the forest industry within Chile, which has led to establishment of extensive plantation forests, primarily of exotic pine and eucalypt species.

Our research aims to assess the relative vulnerability of remaining native forest areas in southern Chile to different threats, including conversion to plantations and habitat fragmentation. Firstly, we determined the extent and spatial distribution of native forest conversion to plantations, using the 'Catastro' digital data set of forest cover. This comprises 641 thematic 1:50,000 maps covering all of Chile derived from aerial photographs and satellite imagery dating from the late 1990's, and was undertaken by staff at the Universidad Austral de Chile, with CONAMA and CONAF. Mapping was supported by intensive field survey to ground-truth photo-interpretation and imagery classification. Approximately 61% of the native forest was described in the field, with a minimum mapping unit of 6.25 ha.

The vulnerability of native forest to plantation forestry was analysed through development of a multivariate, spatially explicit, statistical model using logistic regression (Wilson *et al.*, 2004b). The relationship modelled was between the binary dependent variable (1 = conversion has occurred; 0 = no conversion has occurred) and different environmental characteristics (annual precipitation, latitude, soil type, slope, altitude, distance to cities and distance to roads), using ArcInfo GIS and S-PLUS 2000 software. The best fit statistical model found probability of conversion to plantations to be positively correlated with gentle topography and low elevation, and negatively correlated with distance to cities, distance to roads, rainfall and latitude (Wilson et *al.*, 2004b). In addition, the probability was found to increase with particular soil types, namely red clay and mixed alluvial soils. The model of native forest conversion explained 43% of the deviance, with a Receiving Operating Characteristic Curve (ROC) statistic of 0.95.

Figure 1. Estimated percentage of 10-km cells containing tropical dry forest subject to multiple possible threats



The model was validated by applying it to a separate geographic area not included in the model parameterisation, and comparing predictions with the present distribution of plantations. The model was then used to make probabilistic predictions of where native forest conversion is likely to occur throughout the entire region (Figure 2). The results of this analysis indicate that native forests in northern and western parts of the region are at particularly high risk of conversion. This is an important finding, as although this region is of high conservation value few protected areas have been established. The results can therefore be used to prioritise sites for future protection. One of the key advantages of the approach employed is the use of logistic regression, which enables results to be expressed as probabilities. Decision-makers, who need to evaluate the relative risks of biodiversity loss caused by different threats, can easily understand this form of output.

Further analyses performed in the Valdivian region of Chile include an assessment of the degree of habitat fragmentation. The Catastro data were used in conjunction with a data set describing original forest cover, to examine changes over time in the extent of forest fragmentation resulting from human activity. Both data sets were analysed as grids, with pixel cell sizes of 150 m x 150 m. Forest fragmentation was analysed using a GIS-based approach, following Kapos *et al.* (2000), which provides an integrated measure of fragmentation based on separate measurements of: forest area or patch size; the degree of isolation from, or interconnection with, other patches; and the effect of the interface between forest and non-forest habitats (used as a proxy for edge effects). The results of this analysis (Figure 3) indicate that, for both forest types considered, native forests have become significantly more fragmented since European colonisation. Specifically, areas of relatively intact forest (i.e., with a spatial integrity index value of 10) have largely disappeared from this region. At least for the ***Siempreverde*** forest type, the forest area categorised as highly fragmented (low spatial integrity index) appears to have increased, due to the impact of clearance for agriculture, which occurred as a result of European colonisation.

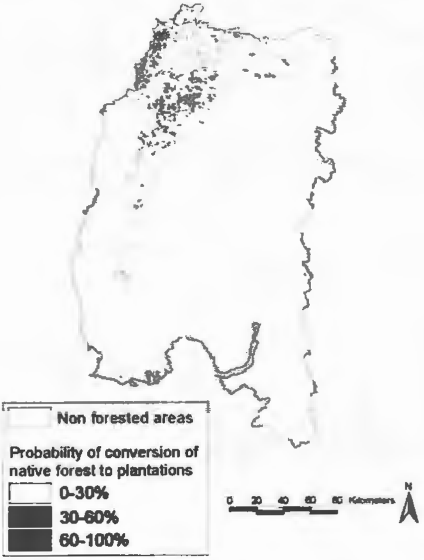


Figure 2. Probability of conversion of native forest to plantations in the Valdivian region of Chile, predicted from the results of a statistical model. From Wilson *et al.* (2004b).

Potentially, this form of analysis is also conducive to statistical modeling approaches; it should be possible to analyse various environmental and socio-economic factors responsible for forest fragmentation within the region, and identify those areas at high risk of future fragmentation. Current research is focusing on the analysis of time series data of forest cover derived from Landsat MSS, TM and ETM imagery within part of the Valdivian ecoregion, to determine both the rate and pattern of forest loss, and its impact on biodiversity. Preliminary analyses have confirmed the role of plantation forest establishment and clearance for agricultural land as major threats to native forests (Echeverría, 2003).

Analysing forest vulnerability at the local scale

Although the methods described are valuable for analysing forest vulnerability at large spatial scales, information needs at more local scales are very different. Rather than information for conservation priority setting, what site managers require are tools that enable diagnosis of the factors responsible for biodiversity loss, to define what management is appropriate. In the UK, the loss of native woodland is no longer a major threat; rather, the priority is to identify the principal threats to biodiversity within individual woodland landscapes, and assess their actual and potential impacts. However, the concept of vulnerability, as defined here, could again prove useful. Consideration of the three elements of vulnerability highlights how little information is available regarding the principal threats to woodland biodiversity in the UK.

To illustrate, Rackham (2003) notes pervasive deer browsing in many native woods. Exposure to this threat is high in many areas, yet information is scarce concerning factors influencing spatial variation in deer density, and the behaviour of deer populations at the landscape scale. While some information exists on the impacts of deer browsing, for example on woodland ground flora and coppice re-growth (Rackham, 2003), the potential long-term impacts of deer browsing are difficult to predict. We believe that a comprehensive analysis of such impacts requires some form of modeling, to enable the interactions between deer browsing and different components of biodiversity to be evaluated. Again, a probabilistic approach, such as Bayesian Belief Networks, could produce risk assessments of use to decision-makers.

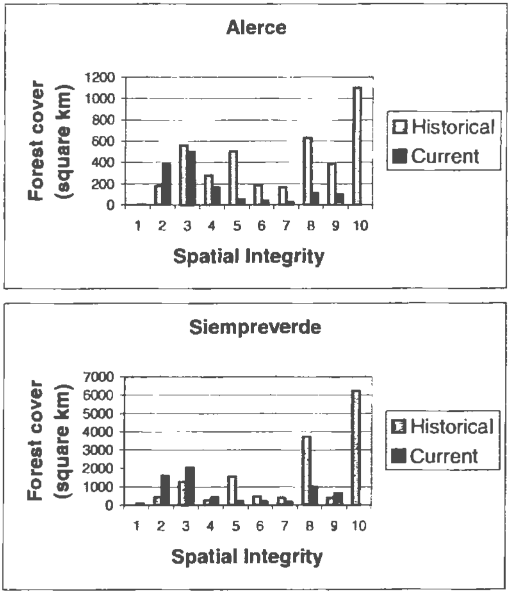


Figure 3. Assessment of forest fragmentation in two native forest types in Chile, using a combined index of spatial integrity (following Kapos *et al.*, 2000).

Conclusions: vulnerability and landscape ecology

In a recent review of priorities for conservation science, Balmford *et al.* (1998) highlighted the need for greater emphasis on assessing the processes responsible for biodiversity loss, to inform effective conservation. A landscape approach to assessing threats is essential, as many threats operate over large spatial scales, and may be explicitly spatial in origin (e.g. habitat fragmentation and edge effects). Strikingly, little emphasis is given to analysis of threats in recent texts reviewing the science of landscape ecology (e.g. Turner *et al.*, 2001), although the importance of a landscape approach to identifying threats and performing risk assessments is generally appreciated. We propose that the analysis of vulnerability, ideally employing spatial analysis and modeling approaches, should become a central component of both landscape ecology and conservation science in the future.

Acknowledgements

This research was supported by the European Commission as part of the BIOCORES project, INCO Contract ICA4-CT-2001-10095.

References

Balmford, A; Mace, G.M. & Ginsberg, J.R. (1998) The challenges to conservation in a changing world: putting processes on the map. In: *Conservation in a changing world*. ed. by Mace, G.M; Balmford, A. & Ginsberg, J.R. pp. 1-28. Cambridge University Press.
Echeverría, C. (2003) *Deforestation and forest fragmentation of temperate forests in Chile.* MPhil dissertation, University of Cambridge.
IUCN. (2003) *Guidelines for Application of IUCN Red List Criteria at Regional Levels: Version 3.0.* IUCN Species Survival Commission. IUCN, Switzerland and Cambridge, UK.
Kapos, V; Lysenko, I. & Lesslie, R.G. (2000) Assessing forest integrity and naturalness in relation to biodiversity. FAO/UNEP-WCMC, Cambridge, UK.
Miles, L; Newton, A.C; DeFries, R; Kapos, V; Blyth, S. & Gordon, J. (2004) A global overview of the conservation status of tropical dry forests. *Journal of Biogeography*. In press.
Myers, N; Mittermeier, R.A; Mittermeier, C.G; da Fonseca, G.A.B. & Kent, J. (2000) Biodiversity hotspots for conservation priorities. *Nature* **403**: 853-858.
Olson, D.M. & Dinerstein, E. (1998) The Global 200: a representation approach to conserving the Earth's most biologically valuable ecoregions. *Conservation Biology* **12(3)**: 502-515.
Rackham, O. (2003) Ancient woodland - Its history and uses in England. Castlepoint Press, Dalbeattie.
Salafsky, N; Margoluis, R; Redford, K.H. & Robinson, J.G. (2002) Improving the practice of conservation: a conceptual framework and research agenda for conservation science. *Conservation Biology* **16**: 1469-1479.
Sutherland, W.J. (2000) *The conservation handbook. Research, management and policy.* Blackwell Science, Oxford.
Turner, M.G; Gardner, R.H. & O'Neill. R.V. (2001) *Landscape ecology in theory and practice*. Springer Verlag, New York.
Wilson, K; Pressey, B; Newton, A; Burgman, M; Possingham H. & Weston, C. (2004a) Incorporating vulnerability into conservation prioritisation: a review of current approaches. *Environmental Management*. In press.
Wilson, K; Newton, A; Pressey, B; Burgman, M. & Weston, C. (2004b) A vulnerability analysis of south central Chilean forests. *Biological Conservation* In press.

Session 6

Spatial conservation strategies

Space for nature: developing landscape-scale action for woodland biodiversity

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Abstract

In a highly fragmented landscape, such as that of the UK, activities outside reserves for the conservation of biodiversity assume a far greater importance than those places with vast tracts of natural habitat that may be contained within individually large reserves. This is particularly relevant in the strongly cultural and densely inhabited landscapes in which we live. The Woodland Trust has developed an approach based on the identification of those site features that are major determinants of value to woodland biodiversity, which it has sought to encompass in the development of a series of simple yet meaningful surrogate measures (Woodland Trust, 2000). Its formulation highlighted the need for habitat creation to buffer and extend semi-natural habitats to increase their core area and thus their ecological resilience, rather than to simply link them. It also suggested that woodland biodiversity has greatest potential to be put on a more sustainable footing in areas where there is a high density of ancient woodland. Although the Woodland Trust wants to see improvements for woodland biodiversity across the whole of the UK, this paper sets out how targets have been determined specifically with regard to areas with a high density of ancient woodland. It is adapted from a report, *Space for nature* (Woodland Trust, 2002).

Background

The Woodland Trust's mission is to conserve, restore and re-establish woodland in the UK. We set out a clear vision for the future in our plan for action, *Keeping woodland alive* (Woodland Trust, 2003), including a wish to see the biodiversity of woods restored and improved.

Biodiversity is a concern of global importance and a measure of the quality of the environment in which not only plants and animals live but on which people also depend. The Trust believes that conservation should maintain and enhance biodiversity by enabling the widest range of habitats and species to survive and evolve (Woodland Trust, 1999). However, we appreciate that knowledge of the natural world will always be limited. As a result we have developed a series of simple yet meaningful measures to assess opportunities for future action relevant to:

- All habitats and species
- Existing habitats and habitat creation
- Landscapes and individual sites.

We believe that those site features that have greatest influence over woodland biodiversity are:

- Ancient woodland; areas wooded since at least 1600AD (Spencer and Kirby, 1992)
- Old-growth woodland; stands with more than 200 years' growth (Peterken, 1996) with a continuity of old trees reaching back into the past (Rose, 1992)
- Size
- Core area (Laurance and Yensen, 1991); area not affected significantly by edge effects from intensive land use (Figure 1)
- Woodland edge adjacent to other semi-natural habitats
- Density of semi-natural habitats
- Linkage of open-ground habitats.

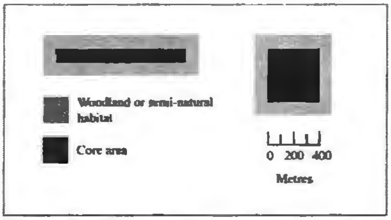

Figure 1. Two woods or semi-natural habitats of equal size affected by edge effects from intensive land use to markedly different degrees with core areas of 10 ha and 16 ha.

Our conclusion is that the contribution semi-natural habitats make to woodland biodiversity increases with age and size. In this context we have developed the following surrogate measures of woodland biodiversity:

- Density of ancient-woodland cover
- Percentage of ancient woodland which is semi-natural
- Cumulative core area of semi-natural habitats (area of semi-natural habitats as a whole not affected significantly by edge effects from intensive land use)
- Area of old-growth woodland.

The key features and resultant surrogate measures identified above are described in detail in *Expanding our horizons* (Woodland Trust, 2000).

Development of the biodiversity measures indicated that habitat creation should focus on buffering and extending semi-natural habitats to increase their core area and thus their ecological resilience from external impact, in preference to simple linkage between habitat patches (Figure 2). It also suggested that woodland biodiversity has greatest potential to be placed on a more sustainable footing in areas where there is a high density of ancient woodland. Both strategies not only address the fragmentation of semi-natural habitats by intensive land use but may also have much to offer in increasing the degree to which individual sites and whole landscapes welcome the establishment of species when they happen to arrive from a long distance. This is important because chance dispersal may be the only means by which many less mobile species will be able to survive rapid climate change (Wilkinson, 1999; Woodland Trust, 2001; Dawson *et al.*, 2004). The frequency with which species are able to move could be enhanced if in addition there is a general reduction in the intensity of land use over the UK as a whole.

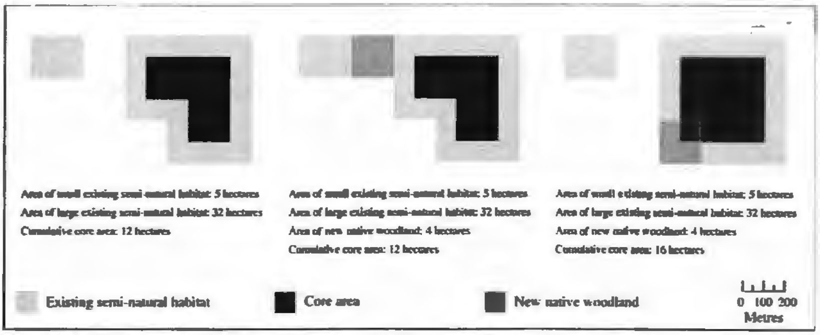

Figure 2. Habitat creation should focus on buffering and extending habitats

Actions for woodland biodiversity across the UK

In the context of our plan for action, *Keeping woodland alive*, and our biodiversity measures, the Woodland Trust will aim to restore and improve woodland biodiversity by:

- Preventing further loss of ancient woodland
- Seeking the conservation and extension of all areas of old growth
- Seeking restoration of ancient woodland planted with non-native conifers to semi-natural woodland (Pryor and Smith, 2002)
- Undertaking and promoting the buffering and extension of ancient woodland and existing semi-natural open-ground habitats in areas with a high density of ancient woodland through the creation of new native woodland and other semi-natural habitats
- Undertaking woodland-creation schemes either where the project contributes to a landscape-scale woodland initiative such as the National Forest, South West Forest or Community Forests, or where it contributes to a landscape-scale initiative led by other conservation interests whose focus may be semi-natural open-ground habitats, or where it is adjacent to an existing Trust property and benefits arise from economies of scale, or where it has arisen due to community-led demand
- Supporting the need to protect and maintain semi-natural open-ground habitats
- Supporting the removal of secondary woodland and plantations from important semi-natural open-ground habitats, where sufficient relict features survive to enable their successful restoration
- Seeking a general reduction in the intensity of land use, particularly adjacent to semi-natural habitats.

Areas with a high density of ancient woodland

Although the Trust wants to see improvements for woodland biodiversity across the whole of the UK, this paper sets out how targets have been determined specifically with regard to areas with a high density of ancient woodland. While major concentrations of ancient woodland are of paramount importance, local concentrations may also have a key role to play in supporting biodiversity and as stepping stones for the dispersal of species in the face of climate change.

Identification

The digitised ancient woodland inventories for England, Scotland and Wales (in draft) have been analysed spatially to identify major and local concentrations of ancient woodland (where ancient woodland in neighbouring 5-kilometre squares exceeds 5 per cent cover over 250 square kilometres and 100-250 square kilometres respectively): see Figure 3. Mapping has not been undertaken for Northern Ireland due to lack of an ancient woodland inventory.

The Land Cover Map of Great Britain (1990), produced from satellite images taken between 1988 and 1991, has been used to exclude all land cover classed as built development from the calculations of ancient-woodland cover. This ensures that the areas identified incorporate urban and urban-fringe areas, where appropriate.

Key features

The key features of each concentration of ancient woodland have been assessed through analysis of: the ancient woodland inventories; Land Cover Map of Great Britain (1990); National Inventory of Woodland and Trees; the Ancient Tree Forum's very provisional data of known concentrations of ancient trees; Forest Enterprise boundary data; preliminary data from English Nature on ancient woodland in public or conservation ownership in England; and the Woodland Trust's ownership boundary data.

Development of targets

It is not possible to calculate cumulative core area of semi-natural habitats from the Land Cover Map of Great Britain (1990), as information is only available summarised by 1-kilometre squares, so the Trust has based development of targets on a well-established mathematical model in which habitat patches are progressively and randomly added to a landscape. At 30 per cent cover a significant threshold is reached when it becomes difficult to add new patches that are isolated from existing ones, as, almost inevitably, they are next to them or in close proximity (Peterken, 2002). At between 50-60 per cent cover patches become contiguous for the first time, resulting in a significant increase in cumulative core area (Buckley and Fraser, 1998). Achievement of 30 per cent semi-natural woodland cover plus another 30 per cent semi-natural cover is therefore the ideal, as it is the optimum balance ensuring connectivity of woodland habitats, connectivity of open-ground habitats and fusion of semi-natural habitats at a landscape scale in a way that would reduce significantly edge effects from intensive land use.

Targets have been developed for each concentration of ancient woodland with reference to its key features and woodland ownership. Goals set to achieve the vision of 30 per cent semi-natural woodland cover take account of the existing area of semi-natural woodland, assume restoration of all ancient woodland planted with non-native conifers to semi-natural woodland, with woodland creation (including conversion of secondary plantations of non-native conifers) making up the balance, while goals for additional habitat creation have been set to achieve a total of 60 per cent semi-natural cover. Clearly it would be advantageous to biodiversity for the area of semi-natural habitat to exceed this threshold.

Existing built development has been excluded from the calculations as it would seem unrealistic to reverse this land use and it ensures targets are relevant to urban and urban-fringe

Figure 3. Concentrations of ancient woodland (derived using data from Countryside Council for Wales, English Nature, Natural Environment Research Council (supplied by Centre for Ecology & Hydrology) and Scottish Natural Heritage. © Crown copyright.

areas within concentrations of ancient woodland. Semi-natural cover is defined as excluding:

- Built development, arable and improved grassland (determined from the Land Cover Map of Great Britain, 1990)
- Mixed woodland and conifer plantation (determined from the National Inventory of Woods and Trees).

Targets

There are more than 100,000 hectares of ancient woods planted with non-native conifers requiring restoration in the concentrations of ancient woodland identified. Forest Enterprise manages 50 per cent of this area. The woodland-creation target within the areas identified is more than 400,000 hectares. Forestry Commission statistics show that 30,000 hectares of broadleaved trees were planted from 1998-2001 across Great Britain. Over a 50-year period, if this rate of planting were to be sustained, the target is achievable but would require a significant focusing of effort. The overall target for additional habitat creation to achieve 60 per cent semi-natural cover is only just over 20,000 hectares. This may seem surprisingly low but takes into account the extensive areas of low-intensity grassland shown on the Land Cover Map of Great Britain (1990). However, there is also a need to reduce significantly the intensity of land use across the other 40 per cent of land area within the concentrations of ancient woodland, as almost 750,000 hectares will comprise arable, improved grassland, conifer plantation or mixed woodland even once the other targets are achieved.

Delivering the targets

The targets need to be delivered as soon as possible, ideally within 50 years. The urgency is due to the pace of climate change and the time it will take both for new habitats to develop and for the pragmatic restoration of ancient woodland sites planted with conifers to semi-natural woodland. Semi-natural habitats are fragmented and exposed to edge effects from intensive land use, particularly in the lowlands, and although there is typically a delay following fragmentation before the onset of local extinctions, it seems that they are already occurring (Smart *et al*, 2004). With regard to ancient woods planted with non-native conifers, many sites have been planted for at least 30-40 years, so restoration needs to start as soon as possible if it is to be undertaken gradually to best effect (Curtis *et al.*, 2002).

It is important that woodland creation is targeted to arable and improved grassland wherever possible so that it also contributes to achieving 60 per cent semi-natural cover. It would be counterproductive to encourage woodland creation on existing semi-natural habitats in areas with a concentration of ancient woodland where there is less than 60 per cent semi-natural cover (e.g., in lowland England and Wales). However, in concentrations of ancient woodland where there is already more than 60 per cent semi-natural cover but less than 30 per cent cover of broadleaved or semi-natural conifer woodland, it may be acceptable to establish woodland on existing semi-natural habitats. Where this is the case, preference should be given to encouraging natural regeneration on semi-natural open-ground habitats through a reduction in grazing pressure. Such landscapes only occur in Scotland and it should be noted that the areas identified represent a relatively small proportion of the total area with extensive semi-natural open-ground habitats. Moreover, sensitive natural regeneration through reduced grazing pressure would maintain much of the open habitat while also creating a wooded landscape.

Although it may be regarded as ideal for semi-natural woodland and open-ground habitats each to occupy 30 per cent of the landscape, in reality, where the latter does not already exist, the scale of additional habitat creation required, landscape considerations, practicalities and willingness of organisations and individuals to undertake projects will determine in what proportions this comprises woodland or semi-natural open-ground habitats. With regard to the remaining 40 per cent of land cover in each concentration of ancient woodland, the relative degree to which there needs to be a reduction in the intensity of land use can be gauged from the proportion that is currently semi-natural open-ground habitats and low-intensity grassland.

Lack of suitable data prevents development of specific targets in relation to old-growth woodland but every effort should be made to conserve and extend old growth wherever it occurs. This is of particular importance in those areas where the significance of ancient trees is very high. As there is no quick way to develop old-growth woodland, existing old-growth stands and nearby woods should be managed to perpetuate old-growth characteristics. Additionally, in agricultural landscapes with significant concentrations of ancient trees regeneration to wooded conditions should be encouraged.

Areas of extensive semi-natural open-ground habitats

The Land Cover Map of Great Britain (1990) has also been spatially analysed to identify 5-kilometre squares, outside concentrations of ancient woodland, with more than 60 per cent cover of semi-natural open-ground habitats, not including low-intensity grassland (Figure 3). All land cover classed as built development has been excluded from the calculations. The 5-kilometre squares identified cover 25 per cent of Great Britain, although this rises to 60 per cent in Scotland, where the extent of moorland cover makes a major contribution, compared to 11 per cent in Wales and just 5 per cent in England. Within such areas, the Woodland Trust will assist wider conservation interests by:

- Supporting woodland creation through reductions in grazing pressure that allow woodland to become a dynamic and integral component of landscapes in which semi-natural open-ground habitats continue to predominate
- Supporting the need to protect and maintain semi-natural open-ground habitats at a landscape scale
- Supporting the removal of secondary woodland and plantations from important semi-natural habitats at a landscape scale, where sufficient relict features survive to enable their restoration.

Development of mapping

The location of concentrations of ancient woodland and extensive semi-natural open-ground habitats is not subject to change. As further data are tracked down or created, maps and targets can be refined with increasing accuracy. However, the Woodland Trust believes it is important that this does not delay setting programmes for action. Land Cover Map 2000 may enable the Trust to determine the cumulative core area of semi-natural habitats within each concentration of ancient woodland. It seems unlikely that woodland and habitat creation will occur at random within concentrations of ancient woodland, as the Trust and others are likely to target areas adjacent to ancient woodland and semi-natural open-ground habitats. We will seek to determine to what extent the existing landscape pattern and a targeted approach would increase the rate at which connectivity and fusion of semi-natural habitats, in terms of percentage cover, would be achieved. It seems likely that this will vary between different concentrations of ancient woodland. Mapping for Northern Ireland, analogous to that already done for Great Britain, will be carried out when the Woodland Trust completes production of an ancient woodland inventory for the Province in 2005-6, although interim analysis may be possible in 2004-5.

Acknowledgements

The Woodland Trust would like to thank English Nature, Scottish Natural Heritage, Countryside Council for Wales, Centre for Ecology & Hydrology, Forestry Commission, Forest Enterprise and the Ancient Tree Forum for supplying relevant data for analysis.

References

Buckley, G.P. & Fraser, S. (1998) *Locating new lowland woods.* English Nature Report No.283

Curtis, T., Pryor, S.N. & Peterken, G.F. (2001) *Restoring Plantations on Ancient Woodland Sites: conversion to native woodland or conservation of Ancient Woodland communities?* The Woodland Trust, Grantham.

Dawson, T.P.; Berry, P.M.; Harrison, P.A & Pearson, R.G. (2004) Modelling long-distance seed dispersal across fragmented landscapes. R.J. Smithers (Ed.) *Landscape Ecology of Trees and Forests.* IALE(UK)

Laurance, W.F. and Yensen, E. (1991) 'Predicting the impact of edge effects in fragmented habitats'. *Biological Conservation*, **55**: 77-92.

Peterken, G.F. (1996) *Natural woodland.* Cambridge University Press.

Peterken, G.F. (2002) *Reversing the habitat fragmentation of British woodlands.* WWF-UK.

Pryor, S.N. & Smith, S. (2001) *The area and composition of Plantations on Ancient Woodland Sites.* The Woodland Trust.

Rose F. (1992) 'Temperate forest management: its effects on bryophyte and lichen floras and habitats'. J.W. Bates and A.M. Farmer (Eds.) *Bryophytes and lichens in a changing environment.* Oxford University Press.

Smart, S.M.; Black, H.J.; Bunce, R.G.H.; Kirby, K.; Isted, R.; Corney, P.M.; Bunce F. & Singleton, D. (2004) Ecological change in British broadleaved woodlands between 1971 and 2000: a pilot resurvey of 14 of the 103 'Bunce survey' sites. R.J. Smithers (Ed.) *Landscape Ecology of Trees and Forests.* IALE(UK)

Spencer, J.W. and Kirby, K.J. (1992) 'An inventory of ancient woodland for England and Wales'. *Biological Conservation*, **62**: 77-93.

Scottish Natural Heritage (1997) Advisory Note 95.

Wilkinson, D.M. (1999) Plants on the move. *New Scientist* (Inside Science **119**).

Woodland Trust (1999) *Seeing the woods for the trees.* The Woodland Trust, Grantham. (www.woodland-trust.org.uk/publications/index.htm)

Woodland Trust (2000) *Woodland biodiversity: expanding our horizons.* The Woodland Trust, Grantham. (www.woodland-trust.org.uk/publications/index.htm)

Woodland Trust (2001) *A midsummer night's nightmare? The future of UK woodland in the face of climate change.* (www.woodland-trust.org.uk/publications/index.htm)

Woodland Trust (2002) *Space for nature: landscape-scale action for woodland biodiversity.* The Woodland Trust, Grantham. (www.woodland-trust.org.uk/publications/index.htm)

Woodland Trust (2003) *Keeping woodland alive.* The Woodland Trust, Grantham. (www.woodland-trust.org.uk/publications/index.htm)

Historic parks in the wider landscape

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Abstract

In a European context, the UK is of paramount importance for veteran trees in historic parks, pasture-woodlands and its wider farmed landscapes. Restoration and protection of the high historic designed landscape, archaeology and nature conservation interest in parks is ongoing and will ultimately lead to an increase in their quality. Open grown trees in the wider landscape provide the ecological and cultural context for parks but are threatened by farming practices, development and neglect. Conservation of these treescapes requires survey and strategic reviews to identify the most significant ones and allocate resources accordingly. This will require strong landowner support, together with judicious use of agri-environment schemes and the new Common Agricultural Policy reform measures.

Introduction

The UK holds a high proportion of the veteran trees in western Europe, many of them in historic parks and pasture-woodlands. Of these, historic parks and their surroundings present one of the greatest management challenges in the UK countryside. For some they are carefully contrived landscapes, the quintessence of a living art form perfected in the 18th C by designers such as Capability Brown. For others their significance lies in nature conservation interests such as the assemblages of invertebrates and fungi that rely on dead and dying timber in old trees (Kirby *et al*, 1995). Parks are important historic environments too; their long history of use as pasture can hold significant archaeology that has been lost due to modern agricultural practices and urban development outside the park boundary.

For a majority of parks the predominant land use remains grazing of livestock or deer, but the intensity of agricultural practice has increased so that this is often the most significant factor in terms of environmental impact. In this paper we explore the links between improvement of environmental quality within parks and the extent and quality of treescapes in the wider, surrounding landscape. We have used examples from parks owned by the National Trust with which we are most familiar but the general conclusions have wide applicability.

The National Trust (NT) is the single largest landowner of historic parks in the UK, with c. 150 documented (K.A. Fretwell, unpublished). Their special invertebrate fauna and lower plant interest has been subject to a preliminary evaluation (Alexander, 1995) and in many there have been more detailed saproxylic and lichen surveys. Typically parks surround a garden, which in turn surrounds a big house. Exterior to the park the estate usually held farmland, which, until intensification of management in the middle of the last century, would have held more field and hedgerow trees supporting some of the special wildlife of parklands and pasture woodlands. In addition, framing or defining vistas, and as eye catchers, they were often features of the "borrowed landscape" an important element in design of parks.

Tree surveys of open grown trees have been largely confined to parks and pasture-woodland and, at least around NT parks, the extent of this resource in the wider landscape is not well known. It seems likely that parks have been increasingly isolated from the surrounding land over centuries. Typically historic parks had well maintained boundaries to keep deer and livestock in and uninvited people out. Their cultural boundaries came to coincide with ecological and archaeological ones as the continuity of grazing of permanent pasture within them became increasingly left behind by agricultural intensification and development. Ecological isolation serves to reduce species' population sizes, and may restrict dispersal opportunities; both factors increasing extinction risk (Begon *et al*, 1990). Isolation also affects the aesthetic nature of parks, so many of which used borrowed landscape features well beyond the park boundary. There are, thus, aesthetic and ecological arguments for reconnecting parks with their surroundings to act as sources of propagules for recolonisation or reinforcement of species populations in trees in, for example, farmland. The first requirement is to maximise the quality of the parklands themselves.

In this paper we consider how improving the environmental quality of historic parks, so as to minimise population reductions of specialist species, should go hand in hand with conserving the treescapes surrounding parks. For this to be successful a long term strategic approach to conservation and management of open grown trees is needed to enable their special wildlife to move between parks and the wider landscape.

Approach

Three main sources of data were used: first, reports on NT historic parks from the last decade; second, specific requests for information on the condition of parklands in 2004; third, tree surveys in 2004 within and immediately outside two historic parks, Brockhampton (Herefordshire, SO682546) and Croome (Worcestershire, SO880443). Rowe *et al*, (2004) gives more details of these surveys.

Results

The quality of historic parks owned by the National Trust

Alexander (1995) and Fretwell (unpublished) summarised the special nature conservation interests and historic design features of NT historic parks respectively. Both noted that agricultural practices within parks damage nature conservation and historic environment/design features.

In a systematic survey Cox and Sanderson (2001) critically examined the impact of livestock production in 26 NT historic parks. A majority of parks (80.8 %) were classed as in an unfavourable condition (unfavourable no change; unfavourable declining; partially destroyed or destroyed). This was despite most having at least some grazing units in an agri-environment (AE) scheme such as Countryside Stewardship or Tir Gofal. Removal of dead or dying timber, and poor management of both parkland trees and grassland were largely responsible for the assessments being classed as unfavourable, and they are inter-related. Intensive livestock production requires high use of inputs (fertiliser, pesticides) and this in turn leads to mechanical and physiological damage to parkland trees, as well as removal of their timber to facilitate movement of farm machinery.

Trees are not only damaged by compaction from livestock and machinery but also by bark stripping and this too appears to be related to the intensity of agricultural practice. In a survey of tree damage by livestock in 47 grazing units in 18 parks in southern England, highest incidences were associated with a combination of high livestock density, low tree density and agriculturally improved swards; lowest incidences were associated with high tree densities and agriculturally unimproved swards (Jeffries, 1997).

In 2004 NT Nature Conservation Advisers made subjective assessments of the environmental state of 81 parks. These included most of the parks surveyed by Cox and Sanderson (2001), allowing an estimate of change to be made. Only one more park in their sample had gone into an AE scheme, but in six a shift to favourable condition was detected. Overall, 43.2% of at least one unit in each park in the NCA sample was in an AE scheme, many linked to planting plans. About two thirds (67.1%) were estimated to be in a favourable or unfavourable improving condition. In general, in NT parks, other factors that can threaten nature conservation interests (e.g., removal of dead timber for aesthetic reasons) were, and are, less important than the nature of the agricultural practice and often more easily resolved through negotiation. Thus NT parks do not seem to be declining in condition and many may be undergoing a long term, slow recovery.

The quality and extent of treescapes outside the NT parks

In farmland adjacent to two parks, Brockhampton and Croome, we compared the density, distribution, age/size structure, immediate land use, threats and health of open grown trees within and immediately outside historic designed parks. "Threats" were "potentially damaging operations" to which, we judged, a tree was or was likely to be exposed because of its location (e.g., root damage, fertiliser, tree surgery, removal of fallen dead timber etc.). Health was judged by canopy cover as "full" and "less than full", which included partial, residual, dead canopy, plus dead trees.

A majority of trees at both sites were oaks *(Quercus robur/Q. petraea)* (Rowe *et al* 2004) and only this species group is considered here. A majority of trees had a girth at breast height (gbh) of between 1.57 m and 3.13 m reflecting plantings between 100 and 200 years ago. Very old trees with gbh of more than 6 m were rare. Trees in arable habitats (including within arable fields, or in hedgerows bounded on both sides by arable/grass ley) were in poorer condition and subject to more threats than those growing in permanent pasture or hedgerows adjacent pasture/orchards. For the two commonest size classes these differences were statistically significant (Table 1).

Oak densities in and outside Croome Park were low and similar at c. 0.4 /ha. They were probably low in the park because it was largely under arable until recently and many trees had been removed. At Brockhampton, tree density was much higher within the park (c. 6 /ha) than in the surrounding farmland (c. 1/ha). Historical evidence (e.g., 1st edition, 6" to the mile, Ordnance Survey maps) reveals a higher tree density in the farmland until the early part of last century.

Table 1. Comparison of the threats and tree health of open grown oaks in 'arable' and 'permanent grass' sites in farmland surrounding Brockhampton and Croome Parks.

	Arable			**Permanent grass**		
gbh(m)	%		N	%		N
	1 + PDO	<F		1 + PDO	<F	
0- 1.56	76.5	3.9	51	80.0	0.0	15
1.57 – 3.13	95.0	14.6	240	58.2***	6.6*	153
3.14 – 5.99	97.3	31.7	183	67.4***	20.0*	95
6+	100.0	42.9	7	75.0	0.0	9

PDO = Potentially damaging operation (e.g., root, trunk, limb damage due to livestock/ vehicles, inputs). <F = less than full canopy (partial, residual, dead crown, dead tree). * and *** $p<0.05$ and $p<0.001$ using X^2, with Yates correction.

Discussion

This short study suggests that for the many parks owned and managed by the NT, declines in environmental quality due to agricultural practice were less evident in 2004 than in the previous ten years. There is recognition that intensive agricultural practice in parks damages many interests, including successful restoration of historic designed landscapes. In addition there is an increasing use of farm plans plus AE schemes to provide minimum environmental standards, all of which helps to protect parkland trees and their immediate habitats.

In England, some of the recently proposed Common Agricultural Policy (CAP) reform measures should increase support for extensive grazing in parks and in adjacent farm landscapes. These include the general principle of decoupling of aid from production and the need for cross compliance (meeting EU standards and measures that maintain the land in Good Agricultural and Environmental Condition) to be eligible for Single Payment farm subsidy. More specifically, cross compliance should lead to more grassland and buffer strips alongside current boundaries and, because of cost cutting, reduced use of inputs. In addition, protection of landscape features is likely to be reinforced.

Within parks there are additional measures that landowners such as the NT can take to prevent agricultural damage to this resource such as removing agricultural tenancies that are based on parks, and adding clauses to tenancy agreements that prohibit damaging practices such as ploughing. There is currently no statutory protection of designed landscapes in the UK and only in Wales are the borrowed landscape and views highlighted as features in need of protection. Tree Preservation Orders have a statutory basis, but they are clumsy, expensive to implement, and inappropriate for most parklands and treescapes. Scheduled Ancient

Monument designation is often very restricted in area. An increasing number of parks have been designated as Sites of Special Scientific Interest, or Areas of Special Scientific Interest, in recognition of their nationally important assemblages of old forest invertebrates and/or lichens. By default this designation is often the best statutory protection for other features in parks such as the designed landscape features and archaeology.

The problem remains that the treescapes outside parks are usually undervalued, unprotected and declining due to a combination of neglect, development and current agricultural practice. In order to prevent further decline in this resource more survey and a strategic reviews are urgently required. The reviews should evaluate the ecological, cultural and aesthetic significance of treescapes outside parks, together with the degree to which they are threatened. This could then be used to prioritise allocation of resources for maximum gain. As with all landscape scale projects a partnership approach will be necessary and the gains will be in the long term. Existing trees must be protected from damage due to agricultural practices (wide buffers, no arable, no inputs underneath the canopy etc.). In addition new trees should be planted that will become the open grown veterans of the future. Some of these trees will inevitably replace those lost from fields over the last 100 years. Prior to the recent CAP reform proposals this explicit reduction in agricultural productivity would have been unthinkable. It remains fraught with legislative difficulties and will need strong landowner, AE and CAP reform support. Without such measures parks, with their special wildlife and historic designed landscapes and archaeology, plus their borrowed landscapes will remain threatened and we will continue to preside over a resource that it is in decline.

Acknowledgements

We thank the NT staff at the properties of Brockhampton and Croome, the Conservation Directorate and the NT Nature Conservation Advisers for their support. We also thank the Agent and tenants of the Croome Estate for allowing access to their land.

References

Alexander, K.N.A. (1995) Historic parklands and pasture-woodlands; the National Trust resource and its conservation. D.J. Bullock & H.J. Harvey (Eds). The National Trust and nature conservation. *Biological Journal of the Linnean Society* **56 (suppl): 155-175**

Begon, M., Harper, J.L. & Townsend, C.R. (1990) *Ecology. Individuals, populations and communities.* Blackwell Scientific Publications, Oxford.

Cox, J. & Sanderson, N. (2001) *Livestock grazing in National Trust parklands.* National Trust, Cirencester.

Jeffries, R. (1997) *Some factors affecting bark stripping of veteran parkland trees by livestock and deer.* National Trust, Cirencester.

Kirby, K.J., Thomas, R.C., Key, R.S., McLean, I.F.G. & Hodgetts, N.G. (1995) Pasture-woodland and its conservation in Britain. D.J. Bullock & H.J. Harvey (Eds). The National Trust and Nature Conservation. *Biological Journal of the Linnean Society* **56 (suppl): 135-153.**

Rowe, S., Barker, S., Walmsley, N. & Bullock, D.J. (2004) Once and future veteran trees in farmland and around historic parks: two examples. R.J. Smithers (ed) *Landscape ecology of trees and forests*, IALE(UK).

The capercaillie as indicator of high species richness: potential and limitations of the umbrella species concept

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Abstract

Indicator species as surrogates for high species richness or a habitat type are widely used in conservation programs. The capercaillie (*Tetrao urogallus*), a large forest grouse, is often implicitly attributed an indicator function. To integrate habitat requirements for this endangered species in forest management practice is a major challenge for conservation in most central European countries. It has been predicted that capercaillie has an umbrella function for a rich mountain forest community. If this is true, conservation measures for capercaillie populations will favour overall diversity of naturally co-occurring species.

In this paper we present results of two studies that compare species richness and abundance of birds and beetles (*Cerambycidae, Buprestidae*) in forest stands preferred or avoided by capercaillie. We found that the capercaillie is a good umbrella for red-listed mountain forest bird species, but its potential as a surrogate measure for high species diversity of beetles is limited. We conclude that the applicability of the umbrella species concept can be (1) more effective among taxa than across taxa, (2) scale-dependent, and (3) limited by the habitat specific scale-sensibility of different taxonomic groups.

Introduction

In response to the "biodiversity crisis" and the limited resources in terms of time, funding and knowledge, conservation biologists have developed a growing interest in designing shortcuts for the protection of whole species communities. The concept of biodiversity-indicator taxa (e.g., Faith and Walker, 1996; Duelli and Obrist, 1998; Saetersdal *et al.*, 2004) and the umbrella species concept (USC; e.g., Wilcox, 1984; Noss, 1990; Simberloff, 1998; Suter *et al.*, 2002) are widespread in this context. The latter has received growing attention by conservationists aiming at using resources effectively. An umbrella species has specific habitat requirements combined with more extensive area demands than co-occurring species. Thus, conservation measures in favour of an umbrella species are expected to benefit a large number of naturally co-occurring species. Therefore umbrella species mostly are large-bodied animals with special habitat requirements in terms of size, resources and structural standards (reviewed in Roberge and Angelstam, 2004).

Two main criticisms are linked with the USC: (1) most of the papers only mention the theoretical significance of the concept without empirically evaluating its validity (Simberloff, 1999; Fleishman *et al.*, 2001; Roberge and Angelstam, 2004), (2) from the ecological point of view it is unlikely that the demands of one focal species would integrate those of many other taxa in the same habitat (e.g., Noss *et al.*, 1997; Basset *et al.*, 2001; Hess and King, 2002; Storch and Bissonette, 2003). However, the USC as a shortcut for the conservation of a whole biota is in use and will continue to be a conceptual tool within the conservation community. Thus, its applicability, potential and limits have to be evaluated.

The capercaillie (*Tetrao urogallus*) is a large forest grouse with specific habitat requirements and a large home range. In central Europe the populations are declining over much of its range (Storch, 2000). The species prefers structurally diverse, old-growth stands of coniferous forests with well-developed ground vegetation (Glutz von Blotzheim *et al.*, 1973; Klaus *et al.*, 1989; Sjöberg, 1996; Storch, 1993a, 1993b). In Switzerland, population size decreased from at least 1,100 to 450–500 males between 1968/71 and 2001 (Mollet *et al.*, 2003). Several authors refer to the umbrella function of the endangered bird for a rich forest community (Boag and Rolstad, 1991; Storch, 1994; Marti, 1995; Suchant, 1995, Mollet, 2001). If this holds true, conservation measures to increase the viability of capercaillie populations will favour overall diversity of naturally co-occurring species. Several studies evaluated this relationship empirically (Fischer and Storch 2001; Ménoni, 2001; Suter *et al.*, 2002; Pakkala *et al.*, 2003).

In order to identify conservation priorities for mountain forests in Switzerland and to use conservation resources efficiently, it is important to know to what extent and with which limitations the umbrella species concept applies to the capercaillie. In this paper we (i) present results of two studies by comparing species richness and abundance of birds (*Aves*) and beetles (Longhorn beetles *Cerambycidae*, Jewel beetles *Buprestidae*) in forest stands preferred or avoided by capercaillie, (ii) assess the structural habitat needs of the different taxa, and (iii) evaluate the relevance and limitations of the results for conservation.

Methods

The study was conducted in the Swiss Prealps of the canton of Schwyz. The study area comprises approximately 200 km^2, is dominated by coniferous forests, and mountain ranges extend from 800 to 1900 m above sea level (Suter *et al.*, 2002; Lienert, 1996).

Systematic inventories including indirect and direct evidence of the species' occurrence were used to spatially stratify the forest stands (see Suter *et al.*, 2002): (1) core area of capercaillie distribution; (2) areas with few or widely spaced records of capercaillie; (3) areas with no evidence of capercaillie presence. The categorisation of the study area was used as tool for a stratified sampling of bird and beetle taxa. We selected 30 study plots of 25 ha each, all of them located between 1,000 and 1,500 m: 8 plots in core areas (cat. 1), 9 in category 2, and 13 plots in forest stands not occupied by capercaillie (cat. 3, absence areas). Plots were balanced with respect to exposure, forest cover, elevation and species-specific bird detectability. Categories 1 and 3 were used as grouping variables when testing for differences in species richness and abundances. Category 2 is only used for habitat analysis of birds.

Bird census

An inventory of breeding birds was conducted in all plots from May to July 1998 applying the point-count or species-specific methods respectively (Suter *et al.*, 2002). Recorded bird species were assigned to one of the following groups: ubiquitous species occurring from lowlands to upper treeline; mountain species distributed mainly in the subalpine zone of Swiss mountain ranges; and species on the Swiss Red List in categories endangered or higher (Zbinden *et al.*, 1994). All mountain birds and three ubiquitous birds classified as red-list species.

Arthropod sampling

We chose six of the 30 study plots to collect for arthropod species richness. Three plots each were of category 1 and 3 respectively. In each plot two combi-traps (Duelli *et al.*, 1999) were placed with a minimal distance of 200 m to account for spatial independence. We checked the traps four times between June and August 2003. For the analysis, only ***Cerambycidae*** and ***Buprestidae*** were assessed. Because no difference in the seasonal phenology of the two groups was found, we pooled the samples for both taxa and each trap.

Habitat analysis

In both studies habitat structure was assessed. Besides geographical and structural variables, forest and vegetation-specific variables were quantified (Suter *et al.*, 2002; Debrunner, 2004). In the bird study, transects were distributed in each plot and comprised 10% of the total plot area. In the beetle study, we characterised the trap sites on a local scale (grid of 25m x 25m).

Data analysis

We assessed the relationship between capercaillie occurrence and species richness by using ANOVA. The number of landscape and habitat variables was reduced by principal component analysis (PVA). Spearman rank order correlations models were used to analyse relationships between habitat parameters and species richness. To explore the question of whether capercaillie and avian diversity responded in a similar way to habitat parameters we compared cluster trees. The software SPSS 11.0 was used for analyses.

Results

Species richness of birds an beetles

We registered 40 territorial bird species on the plots. Thirty were ubiquitous birds and 10 were mountain species. Bird species richness in the study plots ranged from 10 to 20. In total 17 species of *Cerambycidae* (15) and *Buprestidae* (2) were caught in the combi-traps, all of them being saproxylic beetles. Arthropod species richness per plot ranged from 4 to 10.

Mean species richness

Mean bird species richness grouped by capercaillie core and absence areas was 16.8 and 14.3 and differed significantly (Figure 1a; Mann-Whitney U=23.5, p=0.038). Also for mountain birds and red-list species significant differences emerged (U=6.0, p=0.001 and U=11.0, p=0.003, respectively). We could not demonstrate a difference for ubiquitous bird species (U=43.0, p=0.507) nor for overall beetle diversity (U=15.5, p=0.699). As only one of the encountered species of jewel and longhorn beetles is on the Red List of Germany (Geiser, 1998) and no such list is available for Switzerland, a comparable analysis could not be performed for these taxa.

Individual abundance

Bird abundance ranged from 32 to 46 individuals per plot and plot category; the equivalent figures for the arthropod traps were 14 and 135. For overall species richness, mean individual numbers did not differ for beetles by plot categories (Figure 1b). In contrast, capercaillie plots in core areas had more individual birds than plots without capercaillie (means 43.9 vs. 37.5, U=5.0, p=0.001). This effect was caused by differences in the abundance of mountain and red-list species between plot categories (U=1.0, p<0.001, and U=4.5, p=0.001). Such a difference did not exist for ubiquitous species. For the arthropod taxa considered, we could not document significant differences in the individual abundances of food specialists and generalists (U=30.5, p=0.180 and U=33.0, p=0.390 respectively). The two most abundant species (*Alosterna tabacicolor, Gaurotes virginea*) represented 67% of all individuals.

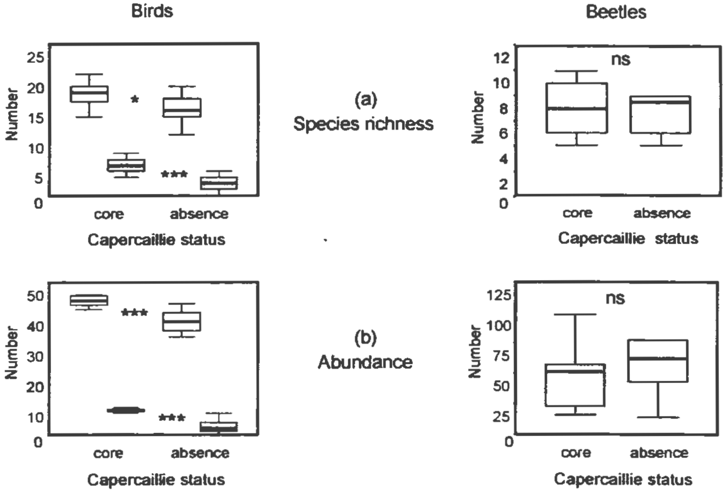

Figure 1. Species richness and abundance of birds vs longhorn and jewel beetles by study-plot category of capercaillie occurrence. Box-plot colours for birds: white – all species, grey – mountain bird species.

Species and vegetation

We analysed bird species richness in relation to habitat characteristics independently of the plot categories. In a PCA the first four factors explained 79.3% of the plots' total variation in vegetation. The factors consisted of 13 vegetation parameters. From these we selected five (multiple stories, ecotonal elements, canopy cover, ground vegetation, ericaceous shrubs) that had high factor loadings and are known from literature to be relevant habitat determinants of capercaillie. In a cluster analysis a dendrogram with three main clusters emerged. These coincided well with capercaillie categories 1 to 3 (see Suter *et al.*, 2002).

From the five vegetation parameters canopy cover was the only variable significantly correlated with beetle species richness in the habitat analysis of trap sites (Pearson correlation $r=-0.596$, $p<0.041$). Capercaillie itself prefers stands with canopy cover between 40 and 70 %. We registered a lower diversity of longhorn and jewel beetles at sites with canopy cover above 60 percent (mean±std: 7.9±1.6 vs. 4.8±1.0 species; $F=6.758$, $p<0.027$). Individual abundances were positively correlated to the amount of dead wood and rootplate ($r=0.651$, $p<0.022$ and $r=0.693$, $p<0.013$, respectively). At trap sites with more than five of these elements we registered more than twice as much beetles than at sites with less decaying wood (116.5±26.1 vs. 45.6±22.1 individuals; $F=16.502$, $p<0.002$).

Discussion and conclusions

In recent years the capercaillie populations in central Europe have increasingly received attention from wildlife agencies and forest managers (e.g., Klaus and Bergmann, 1994; Marti, 1995; Neet *et al.*, 1996; Zeiler, 2001; Schnidrig *et al.*, 2003). It would be desirable to have a single, charismatic species as an indicator of overall forest biodiversity at the regional scale for the conservation community (Noss, 1990; Pearson and Cassola, 1992). Our results do not advocate this shortcut. Rather they show a more differentiated relationship between large-bodied species with demanding habitat requirements and co-occurring taxa. We found that the capercaillie is a good umbrella for red-listed mountain forest bird species, but its potential as a surrogate measure for ubiquitous birds and for high diversity of *Cerambycidae* and *Buprestidae* is weak. The result that a large-bodied single species umbrella does not automatically protect invertebrate species richness is supported by other studies (e.g., Kerr, 1997; Poiani *et al.*, 2001; Rubinoff, 2001). Their common conclusion was that single species are unlikely to function as effective site-selection umbrellas for the conservation of other taxonomic groups, because some of the co-occurring taxa are limited by life history patterns or ecological factors that are not relevant to the focal species (e.g., see Berger, 1997; Storch and Bissonette, 2003). Contrary to these studies, Martikainen *et al.* (1998) demonstrated the white-backed woodpecker (*Dendrocopos leucotos*) to be an excellent umbrella for endangered saproxylic beetles, a result that at least partially is based on the trophic relationship between a predator and its prey species.

From the habitat parameters for classifying the three plot categories, canopy cover was of highest significance for arthropod diversity. Beetle species richness did not differ among capercaillie plot categories but was related to canopy cover. Thus, small stands with canopy cover below 60 % but encapsulated by dense forests not occupied by capercaillie have the same arthropod diversity as comparable plots embedded in large forest stands with capercaillie habitat standards. The beetle species of our study colonised open forest stands with a preferable microclimate independently of the surrounding matrix. Abundances were higher at trap sites with more elements of dead wood, a limiting resource for species richness of saproxylic beetles (Økland *et al.*, 1996; Martikainen *et al.*, 2000). Decaying wood is not an essential ecological factor for capercaillie but a frequent element of its habitat. Our results provide some evidence that *Cerambycidae* and *Buprestidae* rely on comparable structural and compositional habitat standards as capercaillie but on a much smaller scale. The lower abundance and more uneven distribution of these habitat factors in category 3 plots (larger scale, 250'000 m^2) did not result in a lower species richness of beetles at trap sites (small scale, 625 m^2) compared with sites in capercaillie core areas (category 1). Thus, efforts targeted towards capercaillie conservation are expected to be neutral for the diversity of saproxylic beetle species in our study area as long as small forest stands with low to medium canopy cover and a certain amount of decaying wood are still spread within capercaillie absence areas. In cases where the amount of such stands fall below a certain threshold, richness of beetle species is expected to be higher in capercaillie presence areas.

We conclude that the function of capercaillie as an umbrella species is (1) more effective among taxa than across taxa, (2) scale-dependent, and (3) limited by the habitat specific scale-sensibility of different taxonomic groups. With few exceptions (Caro, 2001, 2003) however, no study has assessed whether the application of conservation action targeted at umbrella species really enhances the viability of many co-occurring species (Roberge and Angelstam, 2004). This needs to be proven by researchers and conservationists and will be one of the major targets for future evaluation of the potential and limitations of the umbrella species concept.

References

Basset, Y.; Charles, E.; Hammond, D.S. & Brown, V.K. (2001) Short-term effects of canopy openness on insect herbivores in rain forest in Guyana. *Journal of Applied Ecology* **38**: 1045–1058.
Berger, J. (1997) Population constraints associated with the use of black rhinos as an umbrella species for desert herbivores. *Conservation Biology* **11**: 69–78.
Boag, D.A. & Rolstad, J. (1991) Aims and methods of managing forests for the conservation of tetraonids. *Ornis Scandinavica* **22**: 225–226.
Caro, T.M. (2001) Species richness and abundance of small mammals inside and outside an African national park. *Biological Conservation* **98**: 251–257.
Caro, T.M. (2003) Umbrella species: critique and lessons from East Africa. *Animal Conservation* **6**: 171–181.
Debrunner, R. (2004) *Das Auerhuhn als Schirmart für ausgewählte Käferarten. Eine Untersuchung zur Beziehung zwischen Auerhuhnvorkommen und Biodiversität*. Master thesis, Swiss Federal Research Institute WSL, Birmensdorf, and Swiss Federal Institute of Technology, Zürich.
Duelli, P. & Obrist, M.K. (1998) In search of the best correlates for local organismal biodiversity in cultivated areas. *Biodiversity and Conservation* **7**: 297–309.
Duelli, P.; Obrist, M.K. & Schmatz, D.R. (1999) Biodiversity evaluation in agricultural landscapes: above-ground insects. *Agriculture, Ecosystems and Environment* **74**: 33–64.
Faith, D.P. & Walker, P.A. (1996) Environmental diversity: on the best possible use of surrogate data for assessing the relative biodiversity of sets of areas. *Biodiversity and Conservation* **5**: 399–415.
Fischer, I. & Storch, I. (2001) Capercaillie and woodpeckers in Alpine forests: assessing the indicator species concept. The Wildlife Society (Ed.). *Proceedings of the 2nd International Wildlife Management Congress "Wildlife, land, and people: priorities for the 21st century"*, Gödöllo, Hungary, 376–379.
Fleishman, E.; Blair, R.B. & Murphy, D.D. (2001) Empirical validation of a method for umbrella species selection. *Ecological Applications* **11**: 1489–1501.
Geiser, R. (1998) Rote Listen der Käfer (*Coleoptera*). pp. 168-230. Bundesamt für Naturschutz. M. Binot.; R. Bless.; P. Boye; H. Gruttke & P. Pretscher (Eds.). *Rote Liste gefährdeter Tiere Deutschlands*, Schriftenreihe Landschaftspflege Naturschutz, Vol. 55. Bonn–Bad Godesberg. 434 + XVI pp.

Glutz von Blotzheim, U.N.; Bauer, K.M. & Bezzel, E. (1973) *Handbuch der Vögel Mitteleuropas*, Bd. 5, Galliformes-Gruiformes, Nachdruck 1981, Akademische Verlagsgesellschaft, Wiesbaden.

Hess, G.R. & King, T.J. (2002) Planning open spaces for wildlife. I. Selecting focal species using a Delphi survey approach. *Landscape and Urban Planning* **58**: 25–40.

Kerr, J.T. (1997) Species richness, endemism, and the choice of areas for conservation. *Conservation Biology* **11**: 1094–1100.

Klaus, S.; Andreev, A.V.; Bergmann, H.H.; Müller, F.; Porkert, J. & Wiesner, J. (1989) *Die Auerhühner*, Die Neue Brehm Bücherei, Band 86. Westarp Wissenschaften, Magdeburg, Germany.

Klaus, S. & Bergmann, H.H. (1994) Distribution, status and limiting factors of capercaillie in central Europe, particularly in Germany, including an evaluation of reintroductions. *Gibier Faune Sauvage* **11**: 57–80.

Lienert, S. (Ed.). (1996) *Interdisziplinäres Forschungsprojekt Ibergeregg: Nutzungsgeschichte und Waldvegetation – Geologie – Brutvögel – Fledermäuse – Insekten – Freilandvegetation – Epiphytische Flechtenflora – Pilzflora – Landschaftsschutz – Rechtsaspekte*, Berichte der Schwyzerischen Naturforschenden Gesellschaft, Heft 11.

Marti, C. (1995) Das Schweizerische Auerhuhn-Schutzprojekt. *Naturschutzreport* **10**: 47–56.

Martikainen, P.; Kaila, L. & Haila, Y. (1998) Threatened beetles in White-backed Woodpecker habitats. *Conservation Biology* **12**: 293–301.

Martikainen, P.; Siitonen, J.; Punttila, P., Kaila, L. & Rauh, J. (2000) Species richness of *Coleoptera* in mature managed and old-growth boreal forests in southern Finland. *Biological Conservation* **94**: 199–209.

Ménoni, E. ; Luigi, N. & Delfino, F. (2001) *Grand tétras et conservation de la biodiversité en forêt de montagne*, Rapport scientific, Office National de la Chasse et de la Faune sauvage, Paris, 56–61.

Mollet, P. (2001) *Auerhuhn und Waldbewirtschaftung*, Brochure, Swiss Agency for the Environment, Forests and Landscape (BUWAL), Bern, Switzerland.

Mollet, P.; Badilatti, B.; Bollmann, K.; Graf, R.F.; Hess, R.; Jenny, H.; Mulhauser, B.; Perrenoud, A.; Rudmann, F.; Sachot, S. & Studer, J. (2003) Verbreitung und Bestand des Auerhuhns *Tetrao urogallus* in der Schweiz 2001 und ihre Veränderungen im 19. und 20. Jahrhundert. *Ornithologischer Beobachter* **100**: 67–86.

Neet, C.; Leclercq, B.; Marti, C.; Dändliker, G. & Reymond, B. (1996) Quelle stratégie adopter pour la protection du Grand tetras? *Mémoires de la Société Vaudoise des Sciences Naturelles* **19.2**: 257–261.

Noss, R.F. (1990) Indicators for monitoring biodiversity: a hierarchical approach. *Conservation Biology* **4**: 355–364.

Noss, R.F.; O'Connell, M.A. & Murphy, D.D. (1997) *The science of conservation planning: habitat conservation under the Endangered Species Act*, Island Press, Washington, D.C.

Økland, B.; Bakke, A.; Hågvar, S. & Kvamme, T. (1996) What factors influence the diversity of saproxylic beetles? A multiscaled study from a spruce forest in southern Norway. *Biodiversity and Conservation* **5**: 75–100.

Pakkala, T.; Pellikka, J. & Linden, H. (2003) Capercaillie Tetrao urogallus - a good candidate for an umbrella species in taiga forests. *Wildlife Biology* **9**: 309–316.

Pearson, D.L. & Cassola, F. (1992) World-wide species richness patterns of tiger beetles (*Coleoptera: Cicindelidae*): indicator taxon for biodiversity and conservation studies. *Conservation Biology* **6**: 379–391.

Poiani, K.A; Merril, M.D. & Chapman, K.A. (2001) Identifying conservation-priority areas in a fragmented Minnesota landscape based on the umbrella species concept and selection of large patches of natural vegetation. *Conservation Biology* **15**: 513–522.

Roberge, J.M. & Angelstam, P. (2004) Usefulness of the umbrella species concept as a conservation tool. *Conservation Biology* **18**: 76–85.

Rubinoff, D. (2001) Evaluating the California Gnatcatcher as an umbrella species for conservation of southern California coastal sage scrub. *Conservation Biology* **15**: 1374–1383.

Saetersdal, M.; Gjerde, I.; Blom, H.H.; Ihlen, P.G.; Myrseth, E.W.; Pommeresche, R.; Skartveit, J.; Solhoy, T. & Aas, O. (2004) Vascular plants as a surrogate species group in complementary site selection for bryophytes, macrolichens, spiders, carabids, staphylinids, snails, and wood living polypore fungi in a northern forest. *Biological Conservation* **115**: 21–31.

Schnidrig, R.; Mollet, P. & Bollmann, K. (2003) Capercaillie conservation in Switzerland. *Grouse News* **25**: 12–13.

Simberloff, D. (1998) Flagships, umbrellas, and keystones: is single-species management passé in the landscape era? *Biological Conservation* **83**: 247–257.

Simberloff, D. (1999) The role of science in the preservation of forest biodiversity. *Forest Ecology and Management* **115**: 101–111.

Sjöberg, K. (1996) Modern forestry and the capercaillie. R.M. DeGraaf & R.I. Miller (Eds). *Conservation of faunal diversity in forested landscapes*, Chapman & Hall, London, 111–135.

Storch, I. (1993a) Habitat selection of capercaillie in summer and autumn: is bilberry important? *Oecologia* **95**: 257–265.

Storch, I. (1993b) Patterns and strategies of winter habitat selection in alpine capercaillie. *Ecography* **16**: 351–359.

Storch, I. (1994) *Auerhuhn-Schutz: aber wie? Ein Leitfaden*. Brochure, Munich Wildlife Society, Ettal, Germany.

Storch, I. (Compiler). (2000) *Grouse Status Survey and Conservation Action Plan 2000–2004*, WPA/BirdLife/SSC Grouse Specialist Group, IUCN, Gland, Switzerland and Cambridge, UK and the World Pheasant Association, Reading, UK.

Storch, I. & Bissonette, J.A. (2003) The problem with linking scales in the use of indicator species in conservation biology. J.A. Bissonette & I. Storch (Eds). *Landscape Ecology and Resource Management*, Island Press, Washington, USA, 73–92.

Suchant, R. (1995) Die Zukunft des Auerhuhns in einer mitteleuropäischen Kulturlandschaft. *Naturschutzreport* **10**: 73–90.

Suter, W.; Graf, R.F. & Hess, R. (2002) Capercaillie (*Tetrao urogallus*) and avian biodiversity: testing the umbrella-species concept. *Conservation Biology* **16**: 778–788.

Van Langevelde, F.; Schotman, A.; Claassen, F. & Sparenburg, G. (2000) Competing land use in the reserve site selection problem. *Landscape Ecology* **15**: 243–256

Wilcox, B.A. (1984) In situ conservation of genetic resources: determinants of minimum area requirements. J.A. McNeely & K.R. Miller (Eds). *National parks, conservation and development. The role of protected areas in sustaining society*, Smithsonian Institution Press, Washington, D.C., 639–647.

Zbinden, N.; Glutz von Blotzheim, U.N.; Schmid, H. & Schifferli L. (1994) Liste der Schweizer Brutvögel mit Gefährdungsgrad in den einzelnen Regionen. P. Duelli (Ed). *Rote Listen der gefährdeten Tierarten der Schweiz*, Swiss Agency for the Environment, Forests, and Landscape (BUWAL), Bern, Switzerland, 24–30 .

Zeiler, H. (2001) *Auerwild: Leben, Lebensraum, Jagd*, Österreichischer Jagd- und Fischerei-Verlag, Wien.

Applying landscape ecological methods to analyse and design ecological networks.

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Abstract

Ecological networks are a sound concept for creating sustainable spatial conditions in which species can survive. In recent decades at Alterra, much research has been done on ecological networks and the spatial conditions of species for sustainable networks. This has resulted in methods to assess the quality of ecological networks and to design sustainable ones.

In Cheshire (UK), among other habitats, woodland has been assessed and evaluated and a sound ecological network designed, involving stakeholders. We present this as a case study to illustrate that:

- ecological networks can be assessed and evaluated using the computer model, LARCH
- sustainable ecological networks can be designed and used in land use planning, based on simple guidelines and rules which have been developed
- the maps and guidelines enable stakeholders to input to the design process and final design.

Introduction

Planning sustainable landscapes includes ecological sustainability. Opdam *et al.* (in press) state that ecological networks are a sound basis for creating sustainable spatial conditions in which ecosystems and species can survive. Ecological networks have features in common with infrastructure networks: the need for spatial coherence and connectivity for proper functioning, and the hierarchical organisation of networks at different spatial scales. Ecological networks are the basis for functional ecologically sustainable landscapes, as infrastructure networks are the basis for economic and social mobility (Opdam *et al.*, in press).

In a fragmented landscape many species live in ecological networks, which consist of a number of populations (Figure 1). An ecological network may be defined as a spatially arranged set of habitat patches of one ecosystem type for a particular species or species group. The patches of a network are mutually connected by dispersal of individuals of a species. Ecological networks need to be of a certain size to support sustainable populations (defined as less than 5% chance of extinction in 100 years). The area required depends on the configuration of habitat: the more fragmented and smaller the individual habitat patches, the larger the total area required. The presence of a so-called "key area", a large habitat patch decreases the total area required considerably (Figure 2).

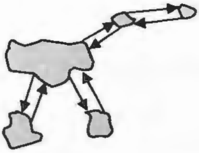

Figure 1. A metapopulation is a set of populations, connected by dispersal movements of individuals.

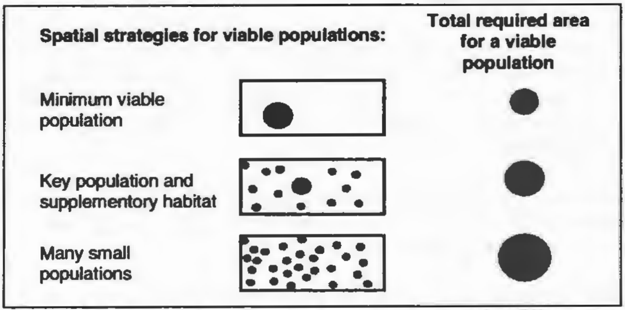

Figure 2. Spatial strategies for viable populations require different area of habitat.

Ecological networks are important in planning sustainable landscapes because: a) they provide an effective conservation strategy in a landscape under human pressure; b) they provide resilience in case of large scale disturbances (e.g., climate change); and c) they can provide other functions (e.g., recreation) (Opdam *et al.*, in press).

An important condition in planning sustainable landscapes is the involvement of local and regional actors in the planning process. This not only means involvement in developing regional targets and ambitions but also involvement in the actual design process. This may be difficult without knowledge about ecological processes. On the one hand regional stakeholders may lack sufficient ecological knowledge, on the other hand they may possess important regional knowledge critical in considering alternative ecological options for habitat locations. Ecological networks are important in stakeholder decision-making because they: simplify ecology in terms of area and shape for planning and design; provide sustainability on the proper spatial scale; are a flexible tool for non-ecologists to plan sustainable nature.

In this paper, we describe a case study in which we designed a long-term vision for an ecological network in Cheshire (UK). We limit ourselves to the results on woodlands. For the design we used a method that consists of applying a set of ecological guidelines for producing a set of alternative sustainable ecological networks in multifunctional landscapes (van Rooij *et al.*, 2003a and 2003b).

Method

Analysis of ecological networks with the LARCH-model

For analysis and evaluation of the sustainability of ecological networks, the LARCH-model has been developed (Landscape Analysis and Rules for the Configuration of Habitat). LARCH is a rule-based GIS-tool, where habitat networks of one or more species can be visualised and the sustainability of the network assessed based on: total network area; habitat quality; and on spatial cohesion of habitat patches. (Verboom *et al.*, 2001; Vos *et al.*, 2001; Opdam *et al.*, 2003).

The input for the LARCH model consists of a map representing the ecosystem of interest. The LARCH database contains the spatial characteristics of many species (e.g., dispersal distances, home range, densities and spatial requirements for sustainable populations). The data is based on extensive literature studies, field studies, and on elaborate population dynamic modelling. The LARCH-model itself uses a set of rules based on metapopulation theory. The output of LARCH consists of a map, outlining the ecological networks for the species of interest and visualising the sustainability of the networks and spatial coherence of the habitat patches.

LARCH is well-documented in reports and scientific papers. It has been used many times to analyse and evaluate ecological networks (e.g., Bolck *et al.*, 2004; van der Sluis *et al.*, 2001a and 2003) and scenario studies (van der Sluis *et al.*, 2001b).

In the Cheshire study, planners and ecologists in the County Council and local naturalists were involved in selecting target species and providing expert opinion on the parameters used and intermediate LARCH results.

Designing ecological networks using guidelines

Whilst LARCH is well-established, a design tool, based on the same knowledge and data, is just recently developed and has been applied in three case studies: in Cheshire, UK; Persiceto, Italy; and in the Gelderse Vallei, the Netherlands (van Rooij *et al.*, 2003a and 2003b). It consists of a procedure and guidelines for the design process, and an overview of the spatial characteristics of the species involved.

With the method, visualisations of possible sustainable spatial solutions are generated, which enable clear and functional communication with stakeholders, local experts and policymakers. Further, the method incorporates opportunities for regional stakeholder input, as required.

The procedure for the design process comprises four steps (Figure 3):

In the *first step* it has to be decided which ecosystems should be improved for biodiversity. It is important to consider the potential of the landscape for the required habitat; some ecosystems cannot be developed or have limited potential, e.g., peatland. The output of this step is selection of one or more ecosystems for which sustainable populations should be developed.

The *second step* is to decide on the level of ambition for each selected ecosystem, and to translate it to target species. As it is impossible to involve all species, we use so-called ecoprofiles. Each represents a number of species with similar spatial characteristics and habitat requirements. The selection of target ecoprofiles first requires analyses of which ecoprofiles are already offered good conditions for sustainable populations by the present landscape (e.g., ecoprofiles with small area requirements and large dispersal capacity) and which are not. Then it is necessary to consider for which ecoprofiles sustainable networks, or significant improvements of the networks, are feasible (in terms of money, hectares and time). In addition, both ecoprofiles of barrier sensitive species (e.g., most mammals) and ecoprofiles of non-sensitive species (e.g., birds) can be selected. In this step, involvement of decision-makers can be helpful; feedback can be provided immediately as to whether formal objectives are feasible in regard of available funds, hectares and/or time. Otherwise, the objectives or means can be adjusted. The output of this step is a selection of one or more target ecoprofiles per selected ecosystem, for which sustainable or well-defined improvements of the ecological networks should be provided in the final design.

In the *third step*, several spatial options to achieve sustainable networks for each selected ecoprofile are generated. To generate these options, a decision tree is drawn up. This is a tool to identify all possible areas where measures can improve the ecological network significantly. It also provides information on the ecological profit of each generated option. Therefore it is possible to ecologically rank the generated spatial options and priorities can be set from an ecological point of view. Besides the question of *where* the ecological network can be strengthened, there are also the different strategies of *how* this can be achieved. These strategies are to: enlarge habitat patches; increase the density of a habitat network; improve the connectivity of habitat patches; and improve habitat quality in one or more habitat patches.

The input of local experts and policy makers can significantly improve the sense of reality of the spatial options for consolidation and improvement of ecological networks. The output of this step consists of:

- Various spatial options where habitat can be consolidated and developed to obtain the desired level of sustainability. These spatial options are ecologically ranked (taking into account costs and benefits);
- Strategies that can best be applied to achieve the level of sustainability in an ecological network of interest.

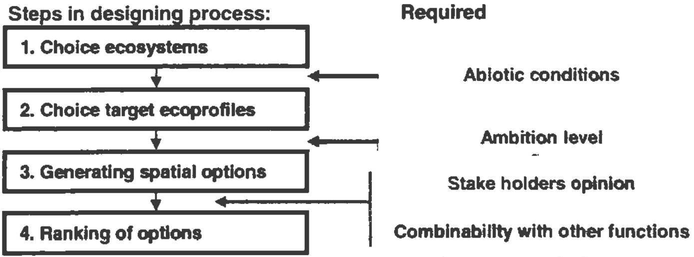

Figure 3. Steps in the method for designing sustainable ecological networks.

In the *last step* of the design procedure, the results of step 3 of the selected ecoprofiles of one or more ecosystems are integrated into one landscape design. Other information is involved in this integration step:

- maps of the present and future infrastructure in the study area;
- opportunities and limitations as a result of conflicting developments.

Involvement of stakeholders and policy makers in this step increases the feasibility of the final design, their support and commitment. The final result is a landscape design that meets the ecological objectives but might not be the ecologically optimal design. However, it is one that, taking account of other functions in a landscape, is feasible, has support and is ecologically sound.

Results

Analysis of ecological networks

In the Cheshire case study, the present landscape and its potential is assessed for viable populations in relation to five ecosystems (van der Sluis *et al.*, 2003). One of the ecosystems assessed is woodland using three species: purple hairstreak (*Querscusia quercus*) dormouse (*Muscardinus avellanarius*) and nuthatch (*Sitta europea*). In Table 1, some coarse indications of the spatial characteristics of the selected species are presented. The results of one of these species, dormouse, will be presented as an example to illustrate the method followed.

Table 1. Coarse spatial characteristics of selected species for woodland

Species	Sensitive for barriers?	Dispersal capacity	Ha requirement
Purple hairstreak	No	Small	Very small
Dormouse	Yes	Small	Small
Nuthatch	No	Medium	Small

From the results of the LARCH analyses it appeared that, if no barriers were considered, many woodland areas have potential for key populations or sustainable populations (data not shown). However, the connectivity of woodland habitat throughout the county is poor, due to fragmentation by the extensive road network. In Figure 4, the spatial cohesion of habitat is visualised. There is only a viable network in one area, in the north of the county (Table 2; spatial visualisations not shown), where there are few roads, some small old woodlands more or less adjoining, and a woodland hot spot close by in the neighbouring county.

Table 2. Results of the LARCH-analysis with the dormouse

Aspects of the ecological network	Result
Key Populations?	1
Viable networks?	One, restricted to a small part of the study area (?)
Connectivity of habitat?	Poor, locally some stronger patches

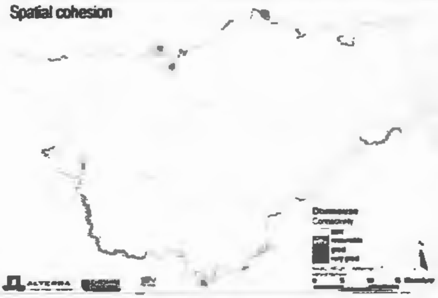

Figure 4. Results of LARCH-scan analysis of the present habitat of the dormouse in Cheshire County.

<u>Design of sustainable ecological networks</u>

One of the objectives set in step 2 (choice of target ecoprofiles) for the landscape design was to improve the woodland habitat in a way that gives rise to sustainable populations of species such as the dormouse throughout the county in the medium term.

In step 3 spatial options were generated to improve the ecological network for species such as the dormouse. Information used was:

- the present habitat in Cheshire and neighbouring counties
- quality of woodlands for species such as the dormouse (as quality improvement can improve the sustainability of ecological networks)
- information on infrastructure (as planning ecoducts can improve ecological networks)
- location of built up area
- Abiotic potential for creation of new suitable habitat throughout the county
- Parameters used for the spatial characteristics of species such as the dormouse (Table 3).

Table 3. Parameters used for spatial characteristics for the ecoprofile "dormouse"

Spatial characteristics	**Ecoprofile "dormouse"**
Key area	100 ha
Dispersal distance	1.5 km
Sensitive for barriers?	Yes
Area sustainable network (including key area)	150 ha
Area sustainable network (without key area)	200 ha

Using the decision tree for creation of sustainable ecological networks (not shown), we came up with a number of spatial solutions (Figure 5). The numbers in Figure 5 refer to the priority ranked in order of ecological gain related to the required effort.

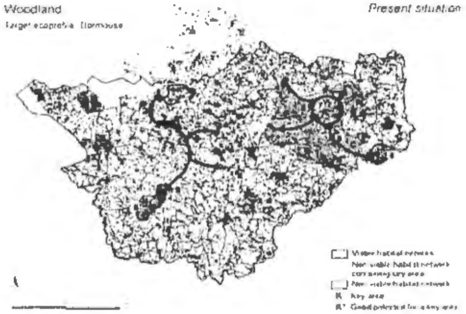

Figure 5. Result of generating spatial options for sustainable networks

For step 4 (ranking of options), we held a workshop with ecologists and spatial planners of the County Council. In preparation for this workshop, they discussed the results of step 3 with key stakeholders and asked their opinions about nature development in specific areas, listing opportunities (e.g., waste dumps) and threats (e.g., planned urban expansion).

Based on the results of step 3, the opinion of the stakeholders, future developments in infrastructure and the opportunities and threats for nature development, a final landscape design was drafted (Figure 6). This final design delineates an ecological network for Cheshire for five ecosystems that meets the objectives for biodiversity of the County Council and may receive maximal support among policy makers and stakeholders.

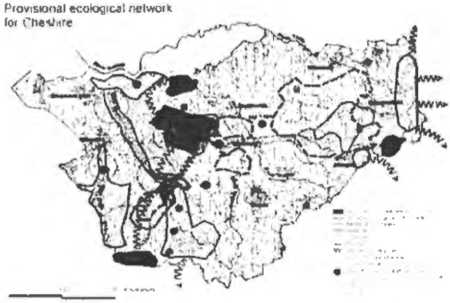

Figure 6. Final integrated landscape design, for 5 ecosystems in Cheshire County.

Discussion

The LARCH-model provided a very useful analysis of the present quality of the network of woodland habitat in Cheshire. The involvement of ecologists in the County Council and local naturalists increased the realism of the final results and acceptance of their diagnosis.

The design method and process for planning a sound ecological network for woodland also proved to be very useful. Reviewing the process, it appeared that:

- It was very time efficient
- It helped to set clear ambitions for nature and provided feedback on the feasibility of the ambitions within the available means (time, hectares)
- The scientific basis of the method provided a solid and neutral basis
- The method delivered unbiased options for nature development
- The input of local experts and stakeholders improved the final quality of the results and secured their support for the final design.

In conclusion, the method used successfully combined scientific rigour with local consultation. The method has only just been developed and is not yet fully operational for all ecosystems. Following the strategy "developing by doing", we are now processing available ecological data and knowledge, so that it can be used in the design process and are improving the process further.

References

Bolck, M., G. de Togni, T. van der Sluis & R. Jongman (2004) From models to reality: design and implementation process. R. Jongman & G. Pungetti (Eds.) Ecological networks, and greenways - concept, design, implementation. *Cambridge studies in Landscape Ecology* 128-150.

Opdam, P.F.M., E.G. Steingröver & S.A.M. van Rooij (*in press*). Ecological networks: a spatial concept for multi-actor planning of sustainable landscapes. *Landscape and Urban planning.*

Opdam, P., Verboom, J. & Pouwels R. (2003) Landscape cohesion: an index for the conservation potential of landscapes for biodiversity. *Landscape ecology* **18:** 113-126.

Rooij, S.A.M. van, E.G. Steingröver & P.F.M. Opdam (2003a) *Networks for LIFE. Scenario development of an ecological network in Cheshire County.* Alterra report 699. Wageningen, Alterra Green World Research.

Rooij, S.A.M. van, T. van der Sluis & E.G. Steingröver (2003b) Networks for LIFE. Development of an ecological network for Persiceto (Emilia-Romagna, Italy). Alterra report 729. Wageningen, Alterra Green World Research.

Sluis, T. van der, R.G.H. Bunce, H. Kuipers & J. Dirksen (2003) *Networks for life. Ecological network analysis for Cheshire County (UK).* Alterra report 698. Wageningen, Alterra Green World Research.

Sluis, T. van der, B. Pedroli & H. Kuipers (2001a) *Corridors for LIFE. Ecological Network Analysis Regione Emilia-Romagna – the plains of Provincia di Modena & Bologna.* Alterra report 365. Wageningen, Alterra Green World Research.

Sluis, T. van der, S.A.M. van Rooij & N. Geilen (2001b) *Meuse-Econet. Ecological networks in flood protection scenarios: a case study for the river Meuse.* Intermeuse report no. 4, Irma/Sponge. RIZA/Alterra. Wageningen, Alterra Green World Research.

Verboom, J., R. Foppen, P. Chardon, P.F.M. Opdam & P.C. Luttikhuizen (2001). Introducing the key patch approach for habitat networks with persistent populations: an example for marshland birds. *Biological Conservation* **100(1):** 89-101

Vos, C.C., Verboom, J., Opdam, P.F.M., ter Braak, C.J.F. (2001) Towards ecologically scaled landscape indices. *American Naturalist* **157:** 24-51.

Developing Forest Habitat Networks in southern Scotland

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Abstract

The Scottish Forestry Strategy identifies Forest Habitat Networks (FHNs) as the primary mechanism for expanding and restoring native woodlands in Scotland. The potential for FHN development in southern Scotland is assessed with reference to two contrasting case study areas. The Scottish Borders is a region in which native woodland has become significantly depleted. Native woodland cover in West Lothian is higher, but the resource is fragmented and threatened by commercial development. Problems associated with generic procedures for the identifying the native woodland framework for the FHN are identified, and examples given of the use of BEETLE (Biological and Environmental Evaluation Tools for Landscape Ecology) as a modelling tool for measuring the size, arrangement and function of wooded and open habitat networks using a generic focal species approach. Ways in which the approach might be used to target grant-aided native woodland expansion and management at a landscape-scale (in balance with other land-use objectives) are discussed.

Introduction

The concept of species conservation through habitat networks has developed rapidly in the decade since the signing of the Convention on Biological Diversity (Anon, 1992) prompting an international acceptance of the need to conserve biological diversity using an approach which includes the establishment of protected-area networks (UNEP, 2003). In Britain, plans to conserve biological diversity and ensure the future viability and integrity of semi-natural ecosystems are outlined in the UK Biodiversity Action Plan (UKBAP) (Anon, 1995). To date the focus has been on site protection strategies, but continuing declines in the biodiversity of designated sites indicates the ineffectiveness of this approach (Hawkins and Selman, 2002). In Britain, ancient semi-natural woodland is fragmented, particularly so across lowland Scotland (Badenoch, 1997). There is an increasing recognition and acceptance of the need to include the 'wider countryside' in rehabilitation and conservation strategies (Anon, 2000), adopting a more holistic view of biodiversity conservation at the landscape scale.

A practical method of addressing the conservation of forest biodiversity at the landscape scale, and one that has been widely accepted in Britain, is the development of Forest Habitat Networks (FHNs) (Peterken *et al.*, 1995, Fowler and Stiven, 2003, Worrell *et al.*, 2003). It is thought that woodland expansion targets within a FHN strategy will conserve forest biodiversity by reversing the consequences of woodland fragmentation and habitat loss. However this approach does not take a species-use view of habitat patches within the landscape mosaic, and proponents of FHNs have tended to accept a landscape structure approach in which landscape metrics, including the contiguity of patches are measured (Peterken *et al.*, 1995, Buckley and Fraser, 1998).

The Scottish Forestry Strategy (Anon, 1999) has identified that native woodland expansion should be targeted to develop FHNs, and the Scottish Forestry Grant Scheme (SFGS) provides some financial support to help meet these objectives. In this paper the generic focal species model BEETLE (Biological and Environmental Evaluation Tools for Landscape Ecology) (Watts, 2003) is used to prioritise where woodland expansion would be most cost effective to conserve forest biodiversity within FHNs.

Sites, data, models and methods

Study sites

Covering an area of 475,000 ha in south-eastern Scotland, the Scottish Borders supports intensive agriculture in the eastern lowlands, and grazing and commercial forestry in the western uplands. Although 19.5% of the land area is wooded, semi-natural woodland or woodland with a high semi-natural component (80-90%) covers less than 1% of the land area (Caledonian Partnership, 1999) In addition, a meagre 608 ha (0.13% of the land area) is classified as ancient semi-natural woodland (ASNW) (Kupiec, 1999), with 1,355 ha (0.28%) of plantations on ancient woodland sites. The existing framework woodland, from which a FHN could be developed, covers 6,792 ha, or about 1.4% of the land area and is distributed mainly in riparian corridors.

The county of West Lothian, to the south of the Forth Estuary in south eastern Scotland, covers about 94,500 ha, making it about a quarter of the size of the Scottish Borders. Total woodland cover is 6,750 ha (Smith and Gilbert, 2003), or about 14% of the land area. Other major land uses include intensive agriculture, urban areas and upland grazing. Approximately 20% of the woodland is broadleaved; 30% (4.5% of the land area) is either semi-natural or containing a high semi-natural component (80-90%), and 316 ha (0.33%) is classified as ancient semi-natural (Kupiec, 1999). The semi-natural woodland is so well distributed, that a 300 m buffer placed around all of it would cover almost 75% of the county. If this was accepted as framework for FHN development in West Lothian, the policy would amount to a continuation of unconstrained native woodland expansion, as occurred under Woodland Grant Schemes (WGS) I, II & III.

Data

Data availability varied between the Scottish Borders and West Lothian studies. For West Lothian a high quality, digitised Phase 1 Habitat Survey (P1) (Copyright, Scottish Natural Heritage) existed for all the semi-natural habitats in the county at a survey scale of 1:10,000. Only improved pasture and intensive arable fields had not been mapped for P1. P1 was not available for the Scottish Borders, instead a combination of the Land Cover Map 2000 (LCM) (Natural Environment Research Council, Centre for Ecology and Hydrology, Monks Wood) data and the Scottish Semi-natural woodland Inventory (Caledonian Partnership, 1999) woodland data were used.

Habitat Networks and the Biological and Environmental Evaluation Tools for Landscape Ecology (BEETLE) method

FHN objectives are often described from a forest management perspective, e.g., to "seek to link woodlands together into coherent areas which function better ecologically and are more rational to manage" (Worrell *et al.*, 2003). To measure network linkage from a perspective of ecological function, a distinction must be made between 'structural connectivity', the degree of contiguity between patches of the same type, and 'functional connectivity', an attribute defined by landscape processes such as species movement and dispersal (Gergel and Turner, 2002). It is possible to have high functional connectivity in a physically fragmented landscape as long as the wider matrix supports the particular ecological process (Farina, 1998). The effect of proposed woodland expansion on open networks and species also needs to be considered.

BEETLE measures the 'functional connectivity' of the landscape using a generic 'focal species' method, where the species profiles have been chosen to represent classes of dispersal ability and area requirements, and the patches scaled to represent matrix permeability for both woodland and open-habitat specialists and generalists. This idea changes the emphasis from a specific 'focal species' analysis to a more general classification of the suitability of a landscape to maintain a range of landscape ecological processes that indicate, in a broad sense, the functional range of biodiversity maintained within a landscape. Generic focal species have been classified within the BEETLE model according to Table 1, where the area requirement is defined as the minimum area to maintain a viable population, and the dispersal ability is defined as the maximum recorded dispersal distance. These definitions are intended to be general, and no distinction is made for example, between adult and natal dispersal, or dispersal differences between sexes. The classification thresholds are therefore quite arbitrary, and are intended only to identify points in the range of ecological processes operating at the landscape scale.

Model parameterisation

A weighted cost distance buffer (WCDB) surrounding each habitat patch was calculated up to the maximum dispersal distance within the habitat (Table 2). Weighting factors were represented by the permeability costs (Table 2) of different patches to dispersal (Adriaensen *et al.*, 2003). The WCDB was used to link habitat patches within the dispersal distance limit. At the threshold, the difficulty of dispersal was assumed to be too great, at which point a barrier was assumed to exist. All of the habitat patches, connected by virtue of falling within the WCDB, were classified within a single habitat network. This process was repeated for different generic focal species profiles.

In the Scottish Borders, concern has been expressed regarding the role of broadleaved woodland FHNs in providing corridors for grey squirrel (*Sciurus carolinensis*) dispersal into red squirrel (*Sciurus vulgaris*) core woodland areas. Also, there has been concern about the possible fragmentation of open-habitat as a result of FHN development, for example, affecting brown hare *(Lepus europaeus)* and the large heath butterfly *(Coenonympha tullia)*. The BEETLE generic focal species profiles (1 and 2, Table 2) attempt to address these issues. Since it is not possible to identify precisely where proposals for new woodland schemes will be offered, the approach in the Scottish Borders has been to buffer all the framework woodland by 500 m, and test the effect of this buffered expansion on woodland and open-habitat networks.

Table 1. Generic focal species classified by area requirements and dispersal ability (after Vos *et al.*, 2001).

	Short dispersal distance	***Medium dispersal distance***	***Large dispersal distance***
Large area requirement	High extinction - low colonisation Highest fragmentation sensitivity		High extinction – good colonisation Medium fragmentation sensitivity
Medium area requirement		Moderate extinction - medium colonisation Medium fragmentation sensitivity	
Small area requirement	Low extinction – low colonisation Medium fragmentation sensitivity		Low extinction - good colonisation Lowest fragmentation sensitivity

Within West Lothian, it has been accepted that a balance between woodland expansion, intensive agriculture and urban expansion is needed. The approach has been to define those areas with a high woodland biodiversity potential. Three, Core Woodland Biodiversity Areas (CWBA), were selected on the basis of the distribution of ASNW and the distribution of ancient woodland indicator plants, using the method suggested by Rose (1999) and Peterken (2000). Three woodland analyses (3, 4 & 5, Table 2) and one open-habitat analysis (6, Table 2) examine the current functional network linkages for a range of generic focal species. For both the Scottish Borders and West Lothian, the total area of habitat contained within each network was used to classify networks by habitat size class thresholds. Network suitability was judged only from habitat size criteria, whereas in reality, habitat quality criteria would also matter.

Results

In the Scottish Borders much of the semi-natural framework woodland is riparian. The effect of buffered expansion reduces the improved grassland of the valley floor (Figure 1), whereas the effect on heath, bracken, acid grassland and neutral grassland found on the mid and upper hill slopes was much less. Buffered woodland expansion reduces the number of networks for generalist woodland species (Table 3), but increases the total habitat contained, due to the increased functional linkage of woodland patches. Woodland expansion has a small effect on the open-habitat networks for heathland specialists with medium area requirements and dispersal ability. Following significant woodland expansion, 95% of the heathland network would remain connected within a single network (Table 3).

Table 2. BEETLE generic focal species profiles used for the Scottish Borders and West Lothian Forest Habitat Network studies. [Key: S – specialist species; G – generalist species; D Limit – Dispersal Limit; med – medium area requirements or dispersal ability], Note: Only a limited selection of permeability cost examples shown.

Beetle Analysis	**Habitat**	**S or G**	**Area (ha)**	**D Limit (km)**	**Permeability cost of habitat** **(a) 1; (b) 2**	**Permeability cost of matrix patches** **(a) 10; (b) 20**
1 Scottish Borders	Woodland	G	10 (large)	5 (med)	(a) Broad, mixed, conifer	(b) bracken, dwarf heath etc
2 Scottish Borders	Heathland	S	3 (med)	5 (med)	(a) Bog, dwarf heath, acid grass	(b) Calcareous-neutral & improved grass
3 West Lothian	Woodland	S	10 (edge impact 50 m)	2 (low)	(a) Broadleaved core woodland ; (b) mixed & conifer wood	(a) tall herb and fern, ruderal ; (b) Grassland, heathland, mire
4 West Lothian	Woodland	G	10 (med)	2 (low)	(a) All woodland	(a) tall herb and fern, ruderal ; (b) Grassland, heathland, mire
5 West Lothian	Woodland	G	10 (med)	10 (high)	(a) All woodland	(a) tall herb and fern, ruderal ; (b) Grassland, heathland, mire
6 West Lothian	Wet open-habitats	S	2 (small)	2 (low)	(a.) wet grassland, heathland & mire; (b) dry grassland	(a) tall herb & ruderal; (b) All woodland

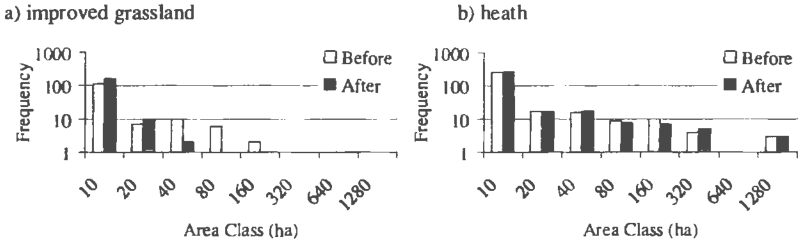

Figure 1. The effect of buffered expansion (500 m) of semi-natural native woodland on the patch size distribution of a) improved grassland; and b) heath, in the Yarrow catchment, Scottish Borders.

Table 3. The effect of buffered expansion (500 m) on the amount of habitat connected within networks in the Yarrow catchment, Scottish Borders for two generic focal species

	Woodland generalists (existing landscape)		**Woodland generalists (expanded woodland)**		**Open habitat specialists (existing landscape)**		**Open-habitat specialists (expanded woodland)**	
Habitat size class (ha)	Freq.	Habitat area (ha)	Freq.	Habitat area (ha)	Freq.	Habitat area (ha)	Freq.	Habitat area (ha)
less than 100	28	770	11	424	7	85	8	84
100-500	3	1058	6	1265	0	0	1	472
More than 500	2	1772	4	6281	2	14705	1	11585
Total	33	3600	21	7970	9	14790	10	12141

Table 4. Summary of habitat within networks in West Lothian for four generic focal species using a BEETLE analysis

Habitat Size Class (ha)	**Woodland Specialists (disp. limited)**		**Woodland Generalist (disp. limited)**		**Woodland Generalist (mobile)**		**Open-habitat Specialist (disp. Limited)**	
	Freq	Area (ha)	Freq.	Area (ha)	Freq.	Area (ha)	Freq.	Area (ha)
0-2	119	53	899	520	33	22	103	49
2-5	18	58	149	489	6	19	12	37
5-10	7	55	61	444	2	14	7	49
10-20	7	99	33	474	3	39	8	114
20-50	5	149	22	685	2	84	9	310
50-100	2	135	16	1167	1	66	3	195
100-200	0	0	5	694	1	163	3	403
200-500	0	0	2	702	1	317	0	0
500-1000	0	0	1	702	0	0	1	536
>1000	0	0	3	8338	1	13494	4	12101
Total	158	549	1191	14217	50	14217	150	13794

The degree of woodland habitat connectivity increases from left to right in Table 4. A total of 158 fragmented and localised networks (concentrated in each of the CWBAs) connected 549 ha of broadleaved core woodland in small patches, for dispersal-limited woodland specialists. Dispersal-limited woodland generalist habitat, although fragmented into 1,191 networks, was distributed evenly across the county. By comparison, 95% of woodland habitat connected into a single network for mobile woodland generalists. A large proportion of habitat suitable for dispersal-limited, open-habitat specialists, although slightly fragmented, was connected within 4 large networks, each containing more than 1,000 ha of habitat.

Discussion and conclusions

The fragmentation of native semi-natural woodland has been noted as a serious problem in lowland Scotland (Badenoch, 1997). The use of the BEETLE model illustrates both the extent and variation of fragmentation in terms of functional connectivity within the two study areas. The BEETLE method is versatile in terms of input data. LCM (broad habitat type) was used in the Scottish Borders, but other land-use data sets have also been successfully used such as Land Cover Scotland 1988 although it is recognised that considerable variation exists in the accuracy and precision of these data sets.

For models using a focal species approach, it has been recognised that an area of significant uncertainty occurs in the parameterisation of the model. For example, Lindenmayer *et al* (2002) suggested that a number of assumptions implicit in the focal species approach (Lambeck, 1997) may not be valid. While this may be true (as species have unique combinations of requirements of a landscape) the focal species method can provide a broad indication of the degree of habitat fragmentation within a landscape by classifying habitat area and measuring the effective distance between patches (Vos *et al.*, 2001). Opdam *et al* (2003) have argued that focal species profiles should be used as part of a tool kit to explore the functional connectivity of a landscape in a wider sense for a range of taxa, and not merely as direct targets. A general parameterisation of the model, with generic focal species profiles, allows a broad assessment of the functional connectivity of habitat patches in a landscape mosaic for specialists and generalists of woodland and open-habitats. The approach is intended to allow an assessment of the extent and size of habitat networks in a general sense, as a strategic tool to inform land-use change and land management options.

In Scotland, the SFGS rules are designed to guide the prioritisation of woodland expansion. Under SFGS new native woodland should be located within 300 m of existing native woodland, or within 300 m of plantation woodland on ancient woodland sites (PAWS) for which restoration is planned. However, the studies discussed in this paper have shown large regional differences in the amount and distribution of semi-natural native woodland described by the SSNWI dataset. These differences could influence the success of the SFGS in delivering the Scottish Forestry policy, for example, a riparian woodland network might develop in the Scottish Borders, while unconstrained native woodland expansion might result in West Lothian. For both regions, a more targeted approach to woodland expansion and restoration would be beneficial and the BEETLE method can help in this respect. To meet local conservation objectives, the spatially explicit habitat networks for a range of generic focal species can be used as a template for locating areas where restoration and expansion is required. This could be viewed as an adaptive approach, to increase the functional connectivity of habitats within agreed size thresholds so that, in the long term, landscapes may be managed to conserve biodiversity and, at the same time, accommodate multiple uses.

Acknowledgements

Part of this research was conducted under a contract from Forestry Commission Scotland, Scottish Natural Heritage, and Scottish Borders Council.

References

Adriaensen, F., Chardon, J.P., De Blust, G., Swinnen, E., Villalba, S., Gulinck, H. & Matthysen, E. (2003) The application of 'least-cost' modelling as a functional landscape model. *Landscape and Urban Planning* **64:** 233-247.

Anon (1992) Earth Summit. In: *United Nations conference of environment and development.* The Regency Press, London and Rio de Janeiro.

Anon (1995) *Biodiversity: The UK Steering Group Report*, HMSO, London.

Anon (1999) *Scottish Forestry Strategy*, Forestry Commission Scotland, Edinburgh.

Anon (2000) *Setting the agenda for biodiversity research: landscape ecology, habitat fragmentation, and land use change scenarios.* Report from a workshop held on 30 June 2000. UK Biodiversity Research Working Group, DETR.

Badenoch, C. (1997) Woodland origins and the loss of native woodland in the Tweed Valley. *Scottish Forestry* **51(1):** 15-20.

Buckley, G.P. & Fraser, S. (1998) Locating new lowland woodlands. Research Report 283, English Nature, Peterborough, UK.

Caledonian Partnership (1999) *Scottish Semi-Natural Woodland Inventory.* Highland Birchwoods.

Farina, A. (1998) *Principles and methods in landscape ecology*, Chapman and Hall.

Fowler, J. & Stiven, R. (2003) *Habitat networks for wildlife and people: the creation of sustainable forest habitats.* Joint Report ISBN 0-85538-299-6, Scottish Natural Heritage & Forestry Commission.

Gergel, S.E. & M.G.Turner. (2002) (eds) *Learning Landscape Ecology: A practical Guide to Concepts and Techniques*, Springer.

Hawkins, V. & Selman, P. (2002) Landscape scale planning: exploring alternative land use scenarios. *Landscape and Urban Planning* **60:** 211-224.

Kupiec, J. (1999) *Further development of the Inventory of Ancient and Long Established Woodland Sites and the Semi-Natural Woodland Inventory.* SNH Information and Advisory Note 104, Scottish Natural Heritage.

Lambeck, R.J. (1997) Focal species: a multi-species umbrella for nature conservation. *Conservation Biology* **11:** 849-856.

Peterken, G.F. (2000) Identifying ancient woodland using vascular plant indicators. *British Wildlife* **11:** 153-158.

Peterken, G.F., Baldock, D. & Hampson, A. (1995) *A Forest Habitat Network for Scotland.* Research, Survey and Monitoring Report 44, Scottish Natural Heritage, Edinburgh, Scotland.

Rose, F. (1999) Indicators of ancient woodland: the use of vascular plants in evaluating ancient woods for nature conservation. *British Wildlife* **10:** 241-251.

Smith, S. & Gilbert, J. (2003) *National Inventory of Woodlands and Trees.* Forestry Commission, Edinburgh.

UNEP (2003) Item 4.1 of 9th meeting of the subsidiary body on scientific, technical and technological advice, November 2003, Montreal, Canada. United Nations Environment Programme.

Vos, C.C., Verboom, J., Opdam, P.F.M. & Braak, C. (2001) Toward Ecologically Scaled Landscape Indices. *The American Naturalist* **157:** 24-41.

Watts, K. (2003) *Landscape Ecology Programme ROAME.* Forest Research.

Worrell, R., Taylor, C. & Spittal, J. (2003) Highland Perthshire Forest Habitat Network. *Scottish Forestry* **57:** 151-157.

Development of a forest habitat network for Wales: linking research with policy

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Abstract

Forest habitats networks are widely considered important for the conservation of biodiversity. They may offset the impacts of chronic habitat fragmentation, facilitating species migration and colonisation and defend against local extinctions, as well as providing contingency for the possible impacts of climate change. This importance has been recognised politically, and habitat networks are referred to within the UK Government's Biodiversity Action Plan, the EU Habitats Directive and the Welsh Assembly Government's strategy paper 'Woodlands for Wales'. In response, Countryside Council for Wales and Forestry Commission Wales are working with partners to develop a strategy for such a forest habitat network. The project has explored and refined the theoretical background, and used a prototype GIS-based model to identify functional habitat networks. Preliminary work has used a range of simulated species with variable area requirement and dispersal characteristics. This paper reviews progress and presents results from the underpinning research.

Introduction

In common with many countries, forest originally covered the majority of the land surface of Wales, and formed the matrix within which other habitats sat. Human influence has been profound, and millennia of clearance and intensive management have reduced forest cover to only a few percent of the land surface: forest fragments, or woods as they have become, are now islands within a matrix of non-woodland habitats. Forest species are adapted to a highly connected and extensive habitat, and fragmentation has inevitably had a major impact on them. Those with very large home-ranges will have become extinct rapidly, whilst the chronic interruption of dispersal, migration and metapopulation dynamics of many species will have caused a slow attrition of biodiversity. There are concerns that climate change will compound these effects, as species will not be able to track the movement of their climatic niches across landscapes and will become more susceptible to extinction (Harrison *et al.*, 2001).

Impacts of fragmentation have increasingly been recognised within the nature conservation community (UK Government, 1994), and consequently there is now much interest in developing forest habitat networks (Peterken *et al.*, 1995; Ratcliffe *et al.*, 1998;

Peterken, 2003; Ray *et al.*, 2003; Worrell *et al.*, 2003). Essentially networks are the strategic consolidation, expansion, and re-connection of woodland cover to reverse some of the impacts of fragmentation. The concepts have now moved into the political arena, and habitat networks are formally recognised through a number of nature conservation programmes and forestry strategies. Of particular relevance are the UK Biodiversity Action Plan (BAP) (UK Government, 1994), the EU Habitats Directive (European Commission, 1992), and the Wales Assembly Government's woodland strategy (Forestry Commission, 2001).

For example, the Upland Ashwood Habitat Action Plan (part of the BAP) (UK Biodiversity Group, 1998) includes the action: "Encourage the development of forestry/landscape strategies to provide a context for and to promote expansion and positive management of upland mixed ash woodland."

The need for habitat networks has been identified under article 10 of the Habitats Directive (European Commission, 1992). They are recognised as important for the long-term sustainability of the condition of Natura 2000 sites (SACs) and to deliver Favourable Conservation Status to endangered habitats and species. The Directive states:

"Member States shall endeavour, where they consider it necessary, in their land-use planning and development policies and, in particular, with a view to improving the ecological coherence of the Natura 2000 network, to encourage the management of features of the landscape which are of major importance for wild fauna and flora. Such features are those which, by virtue of their linear and continuous structure.....are essential for the migration, dispersal and genetic exchange of wild species."

The Welsh Assembly Government refers to woodland habitat networks in its strategy paper 'Woodlands for Wales' (Forestry Commission, 2001), which states: "The success of this strategy (in part) will be shown if we can improve the quality of these woodlands, linking and expanding their habitat networks....The prospect of global climate change demands that we consider how robust our woodland habitats are, so that we concentrate our conservation efforts on habitats that will be sustainable in the long term". It also includes the commitments:

1. "We will increase the quality of native woodlands for wildlife and implement the Biodiversity Action Plan targets for their restoration and extension, creating links between fragmented woodlands"; and
2. "We will increase the area of native woodlands, targeting extension and connection of existing woods and incorporating the concept of increasing the core area of native woodland habitats".

The Countryside Council for Wales (CCW) and Forestry Commission Wales are collaborating (with input from other partners) to develop and implement such a woodland habitat network in Wales. This paper reports on progress to date.

Project overview

The project was broken down into three major stages. The first was to refine the theoretical background. Habitat networks are potentially significant drivers of public funding and land-use change, and it was felt essential that the scientific basis for it was robust. The second stage was to explore existing woodland networks within Wales, to identify 'core areas', and produce indicative maps showing the most effective areas in which woodland could be encouraged to consolidate core areas and to increase connectivity between them. The third stage was to apply these results to forestry policy in Wales, so that the network could be realised.

Refining the theoretical background

This stage explored some of the fundamental concepts involved in habitat networks: the meaning of connectivity and the presence of woodland cover thresholds. It addressed these questions using neutral landscape models (NLMs) and is described in detail by Watts and Griffiths (2004). The results confirmed the presence of thresholds within random NLMs but revealed the difficulties in applying these to actual landscapes. Welsh landscapes exhibit a high degree of correlation with underlying landscape processes, such as topography, hydrology and land use. Networks were shown to be highly dependent on the species or ecological process under consideration, i.e., every woodland block in Wales may constitute a separate, fragmented network for a species with poor dispersal. There would appear to be no generic prescriptions for woodland cover that can be used to assist the generation and development of functional habitat networks.

Defining existing networks and identifying indicative linkages

Given that no universal network can be defined, the approach taken was to explore possible networks for different types of species. A prototype GIS-based model "BEETLE" (Biodiversity and Environmental Evaluation Tools for Landscape Ecology) being developed by Forest Research Agency was used (Watts, 2003; Ray *et al.*, 2004). Part of this model is a focal species (see Lambeck, 1997; Brooker, 2002) based tool that utilises habitat area requirements and dispersal characteristics to identify *functional habitat networks* for a given species. The model is innovative, in that it takes into account the probability of movement across the wider landscape matrix, and so does not require woodland to be physically continuous to be considered functionally connected. This explicitly makes the fundamental distinction between structural and functional connectivity (With, 2002). It is therefore likely to generate recommendations for relatively discrete new areas of woodland and extensification of other land uses, which may be more socially acceptable and compatible with the conservation of other habitats.

Due to limited autecological data a series of simulated *generic* species were used to explore the variation in potential functional habitat networks across Wales. These were defined to be representative of woodland specialists (virtually restricted to native broadleaved woodland), woodland generalists (able to live in all types of woodland including conifer plantation), and to have combinations of high and low habitat area requirements and dispersal abilities. Habitat data was available from CCW's Phase 1 Habitat Survey which records land cover across Wales down to a resolution of 0.01 ha.

The relative ecological permeability costs of intervening habitats were defined as shown in Table 1 and illustrated in Figure 1. Costs were expressed in terms of probability of movement through the surrounding landscape matrix (see Adriaensen *et al.*, 2003; Chardon *et al.*, 2003; Sutcliffe *et al.*, 2003), established in discussion with specialists for other habitats. The impact of altitude was also incorporated by using a digital elevation model. The dispersal element of the model was based around a 50 year timescale, to take account of long distance dispersal events. This is also a typical time horizon for the effects of climate change (Harrison *et al.*, 2001). The analysis was run for combinations of species types, dispersal and area limitation, and woodland type (all woodland, broadleaved, ancient etc.) and the functional networks that resulted were compared.

Table 1. The relative costs of non-woodland habitats used with the BEETLE model to determine functional woodland habitat networks for Wales.

	Habitat characteristic	**Examples**
Low ecological cost	Quasi-woodland habitats, with strong 3-D structure and known to readily accommodate woodland species.	Semi-natural scrub, bracken
Intermediate ecological costs	Unimproved semi-natural habitats with well developed structure	Heathlands, marshy grassland
	Unimproved semi-natural habitats with limited structure	Unimproved grasslands
	Semi-improved habitats	Semi-improved grassland, modified heathland
High ecological cost	Heavily modified and artificial habitats	Improved grassland, arable, urban areas

The results for generic species with extreme dispersal limitation were uninformative, as functional networks could only form where woodland blocks were separated by only a few metres: effectively each woodland block is its own functional network. These species are of course severely affected by fragmentation, but unfortunately are also those least likely to be aided by a strategic habitat network. Species with extreme dispersal limitation are unlikely to take advantage of any network, regardless of connectivity, during our time horizon of 50 years, except as a result of chance long distance dispersal, and are likely to be better protected by consolidation of individual woodland blocks and by management to improve habitat quality.

For these reasons we felt it appropriate to concentrate on more mobile species which realistically can move to take advantage of a habitat network. Our most mobile and least area restricted species generated a dramatic pattern of functional networks of broadleaf woodland across Wales (Figure 2).

Figure 1. Landscape permeability surface based on land cover and altitude

Figure 2. The largest provisional functional networks for broadleaf (low area, high dispersal) species across Wales – 'Core Areas'. Smaller networks have been excluded for clarity

Nested within these networks are smaller networks generated by species with lesser dispersal abilities and higher area requirements (Figure 3). Again, these are species that, although more limited, are typical of woodland species that can be expected to use a habitat network. Analysis using ancient woodland generated a similar pattern of networks, though obviously more restricted due to the more fragmented nature of the resource.

Figure 3. Sub-networks nested within larger networks

This work clearly identifies the largest functional woodland networks in Wales (Figure 2), which out of tradition and for convenience we have termed *core areas*. In addition, to the opportunity to target work to protect, improve, restore and expand the sub-networks within these core areas. Work is underway to identify where there is greatest potential to create ecologically functional links between these core areas. These will be presented as indicative linkage maps.

Application of the results

Results will be used to identify areas of Wales within which woodland protection, improvement, restoration and expansion, as well as changes to agricultural practice to enhance land matrix 'permeability', are likely to consolidate functional networks or link them and so contribute most to the conservation of biodiversity. This creates opportunities to develop policies and targeted incentives to encourage forest habitat networks via woodland management, expansion and alternative agricultural practices. This will increase the contribution to functional connectivity of the non-wooded habitats. The outcome based approach to the current re-development of the incentives for woodland expansion and woodland management in Wales ('Better Woodlands for Wales' - scheme yet to be finalised and launched), as well as the existing Tir Gofal scheme, are both well positioned to respond to the results of the ongoing modelling work.

Conclusion

The current focus on woodland habitat networks, to combat the consequences of habitat fragmentation, reflects their perceived importance for biodiversity conservation. The impacts of habitat fragmentation are consistent with many of the reductions of biodiversity and considered to be a major threat to its conservation.

The modelling and exploration of woodland connectivity and fragmentation through the BEETLE approach is built on a number of scientifically justifiable assumptions. A key strength of BEETLE is utilising a focal-species approach, which enhances our understanding and perception of habitat fragmentation by explicitly modelling functional connectivity. It is feasible to have a functionally connected network within an apparently structurally fragmented landscape. Landscape connectivity is more complex than the idea of habitat corridors linking fragments suggests (With, 2002).

A species-based perspective is the most relevant viewpoint if we are to develop meaningful and effective conservation strategies. The use of generic species, due to limited autecological data, is open to criticism. This approach has enabled the exploration of the range of potential functional networks. Identifying networks, which many woodland organisms may utilise, will assist future management and planning activities for biodiversity. The use of generic species has also proved useful in building consensus between multiple stakeholders.

Further development and testing is needed. It is necessary to explore the potential of using a limited number of real species, as the data allows, to further validate the present assumptions. Additional detailed analysis will be needed within the individual sub-networks (Figure 3). This will offer the opportunity to incorporate more detailed local information to guide local action.

Acknowledgements

The research was also supported by additional input from Steering Group members George Peterken, Richard Smithers (Woodland Trust), Hilary Miller and Patrick Green.

References

Adriaensen, F.,Chardon, J.P.,De Blust, G.,Swinnen, E.,Villalba, S.,Gulinck, H. & Matthysen, E. (2003) The application of 'least-cost' modelling as a functional landscape model. *Landscape and Urban Planning* **64:** 233-247.

Brooker, L. (2002) The application of focal species knowledge to landscape design in agricultural lands using the ecological neighbourhood as a template. *Landscape and Urban Planning* **60:** 185-210.

Chardon, J.P.,Adriaensen, F. & Matthysen, E. (2003) Incorporating landscape elements into a connectivity measure: a case study for the Speckled wood butterfly (Pararge aegeria L.). *Landscape ecology* **18:** 561-573.

European Commission (1992) *Council Directive 92/43/EEC on the conservation of natural habitats and of wild flora and fauna*, European Commission, Brussels.

Forestry Commission (2001) *Woodlands for Wales*, Forestry Commission (on behalf of the Welsh Assembly Government), Edinburgh.

Harrison, P.A.,Berry, P.M. & Dawson, T.P. (2001) Climate Change and Nature Conservation in Britain and Ireland: Modelling natural resource responses to climate change (the MONARCH project): Summary Report. UKCIP, Oxford.

Lambeck, R.J. (1997) Focal species: a multi-species umbrella for nature conservation. *Conservation Biology* **11:** 849-856.

Peterken, G.F. (2003) Developing forest habitat networks in Scotland. J. Humphrey, A. Newton, J. Latham, H. Gray, K.H. Kirby, E. Poulsom & C. Quine (eds). *The Restoration of Wooded Landscapes - Proceedings of a conference held at Heriot Watt University, Edinburgh, 14-15 September 2000*, Forestry Commission, Edinburgh 85-91.

Peterken, G.F.,Baldock, D. & Hampson, A. (1995) A forest habitat network for Scotland. 44, Scotish Natural Heritage, Edinburgh.

Ratcliffe, P.R.,Peterken, G.F. & Hampson, A. (1998) A forest habitat network for the Cairngorms. 114, Scotish Natural Heritage, Edinburgh.

Ray, D.,Watts, K.,Griffiths, M.,Brown, C. & Sing, L. (2003) Native woodland habitat networks in the Scottish Borders. Forest Research, Edinburgh.

Ray, D.,Watts, K.,Hope, J. & Humphrey, J. (2004) Developing Forest Habitat Networks in lowland Scotland. R.J. Smithers (ed.). *Landscape Ecology of Trees and Forests.* IALE(UK)

Sutcliffe, O.L.,Bakkestuen, V.,Fry, G. & Stabbetorp, O.E. (2003) Modelling the benefits of farmland restoration: methodology and application to butterfly movement. *Landscape and Urban Planning* **63:** 15-31.

UK Biodiversity Group (1998) *Tranche 2 Action Plans. Volume II - terrestrial and freshwater habitats*, English Nature, Peterborough.

UK Government (1994) *Biodiversity: The UK Action Plan (Cmnd 2428)*, HMSO, London.

Watts, K. (2003) *Landscape Ecology Project: Phase 4 Project Plan - Development of prototype landscape ecology evaluation tools to assist large-scale forest planning*, Forest Research, Woodland Ecology Branch, Alice Holt, Farnham.

With, K.A. (2002) Using percolation theory to assess landscape connectivity and effects of habitat fragmentation. K.J. Gutzwiller (ed.). *Applying landscape ecology in biological conservation*, Springer-Verlag, New York 105-130.

Worrell, R.,Taylor, C.M.A. & Spittal, J.J. (2003) Highland Perthshire Forest Habitat Network. *Scottish Forestry* **57:** 151-157.

AONBs in the South East: landscape-scale action for woodland fritillary butterflies

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Abstract

The South-East Areas of Outstanding Natural Beauty (AONBs) Woodlands Programme, funded by the Forestry Commission and Countryside Agency, seeks to increase the understanding of the environmental, landscape, and cultural importance of woodlands within these protected landscapes, identify key areas, and promote sustainable management.

A key aspect of the programme is Geographic Information System (GIS)-based, landscape-scale analysis to enable strategic targeting of resources to enhance woodland biodiversity. A particular focus is on key, threatened species, indicative of characteristic habitat conditions in southeast England. Using monitoring data supplied by Butterfly Conservation, analysis has been undertaken of the populations and range of the pearl-bordered fritillary (*Boloria euphrosyne*), and small pearl-bordered fritillary (*Boloria selene*) butterflies in the South East. Once more widespread in the coppice woodlands of the region, their populations have been in decline for many years and are now severely threatened.

Combining the mapping exercise with information about the ecology of these species and an understanding of their population dynamics, key woodland and semi-natural habitat areas have been identified where habitat management will help stabilise and build upon the remaining populations. Further work will help demonstrate how management effort can be co-ordinated at a landscape scale to allow further expansion of these butterflies across the region.

Introduction

The South-East Areas of Outstanding Natural Beauty (AONBs) Woodlands Programme has arisen out of a Joint Accord between the National Association for Areas of Outstanding Natural Beauty and the Forestry Commission. The AONBs covered by the programme are the High Weald, Kent Downs, Surrey Hills, Sussex Downs and East Hampshire AONBs. These are heavily-wooded protected landscapes, rich in ancient woodlands: the High Weald AONB has twice the amount of ancient woodland than any other AONB in England and Wales (based on GIS analysis carried out by the author). Overall, southeast England is calculated by the Forestry Commission as containing some 40% of the country's ancient woodlands. However, only some 17% of the ancient woodland resource in the Government Office for the South East (GOSE) region is given statutory designation as a Site of Special Scientific Interest (SSSI). Nevertheless, the SSSI series does contain many of the most important woodland sites, representing the wide range of different habitats and management regimes characteristic of the region. These include high forest (including beech and yew woodland), wood-pasture, bryophyte-rich gill woodlands, wet woodlands, and coppice.

Identifying priorities for targeting woodland management

Whilst considerable effort is being put into encouraging new markets for woodland products to support management, the need for ongoing management means that for the time being, conservation management is heavily dependent on grant support. With limited resources, it is clear that not all woodlands can be targeted effectively. However, identifying woodlands for strategic targeting across the AONBs in such a heavily-wooded area is not a simple process.

For many years, a site-based approach has often been taken as the basis for both protecting and managing wildlife. The SSSI series, in particular, has received increased protection and funding in recent years. But despite greater protection for both sites (and some species), many species of conservation concern continue to show population declines. In recent years, a greater understanding of the interdependence of populations in the wider countryside, and the role of population size, isolation, and habitat quality, have focused attention on the need to understand, and manage species metapopulations. Metapopulations have been defined as 'a collection of local populations (or colonies), connected by occasional dispersal, in which there are local extinctions and colonisations' (Oates 2004). Bourne *et al.* (2002) consider that changes in metapopulation structure 'are recognised as a major cause of population extinctions. This is because modern land-usage results in the survival or regeneration of too few, small or isolated ecosystem islands for persistence of the species.' This increased understanding has resulted in a shift in conservation thinking towards landscape-scale, rather than solely site-based, conservation management. In a recent report, English Nature (English Nature, 2004) has recognised that focusing solely on SSSIs will not address declines in key, threatened species:

> *'Although some species can survive within a single site, such a population is vulnerable to a local catastrophic event, such as severe drought on a wetland site, cutting of the vegetation at the wrong time of year, or fire. Understanding ecological processes and relationships on a scale greater than just the site enables consideration of the wider needs of species. This approach is especially important for species with populations in separated and isolated sites. It can also be used for those species that require several habitat patches within a given area, for example, the marsh fritillary butterfly. In a resilient environment sub-populations of this species will colonise and be lost from a suite of habitat patches in a cyclic fashion over time, as a response to parasite pressure. With increasing fragmentation of habitat patches, sub populations are less able to move between them and are lost. The eventual result is the disappearance of the species from the area.'*

For woodland conservation in southeast England, managing for key species can be seen as a means of targeting woodland management in general. Woodlands with a coppice structure (though often unmanaged) are one of the most widespread woodland types in the region. It is widely acknowledged that a decline in this management has led to a corresponding decline in species dependent on open woodland conditions. One approach would therefore be to identify key areas requiring the woodland structure provided by coppicing and wide ride management for a particular species or species group, and use these as a means for targeting efforts. However, outside of sites managed specifically for wildlife, the grant schemes that help deliver conservation management are often distributed on a 'reactive' basis, i.e., responding to contact from land owners and managers, rather than being directed proactively to deliver strategic management. This is understandable. Although greater use is being made of computer-based mapping (or GIS), there remains a paucity of widely-available data upon which to base decisions. Woodland is one of the few habitats for which there exists systematic national datasets (the provisional Ancient Woodland Inventory managed by English Nature, and the Forestry Commission's National Inventory of Woodland and Trees). This data can be used for a number of purposes, such as to identify and target woodlands according to size, connectivity, and management type, and for restoring plantations on ancient woodland. But whilst these can be valuable approaches, habitat data will not necessarily address the needs of species whose populations are continuing to decline.

GIS analysis of woodland fritillary butterfly populations

In common with many other habitats and species, many woodland species are dependent on the conditions created by continuous management. The less mobile species, restricted in their ability to cross unfavourable habitat to reach appropriately managed woodland, are those most under threat. Across the South East, a number of butterfly species associated with open woodland have shown severe declines in their range and populations. In particular, the pearl-bordered and small pearl-bordered fritillaries are rapidly declining species which once were far more widespread in the region. These species have shown, respectively, a 68% and 52% decline nationally in breeding areas since the recording period of 1970-1982. Both butterflies are now listed as high priority species in Butterfly Conservation's Regional Action Plan for southeast England, with the pearl-bordered fritillary a priority species in the UK BAP (see Table 1, below).

As well as understanding the population ecology of a particular declining species, it is also important to know its current and former extent, population sizes, isolation, habitat quality associated with remaining populations, and so on. Use of GIS greatly facilitates this work, through the digital mapping of species records, and the analysis of these in relation to other datasets such as habitat type, management, and the spatial relationship between remaining populations.

Table 1. Status of pearl-bordered and small pearl-bordered fritillaries (Butterfly Conservation, 2000).

Species	**BC priority rating for south-east region**	**BC national priority rating**	**UK BAP status**
Pearl-bordered fritillary	High	High	Priority species
Small pearl-bordered fritillary	High	Medium	Species of conservation concern

Ecology and management of the study species

It is beyond the scope of this short paper to provide a detailed account of the ecology of the two species examined in this study. More detailed information can be found in the national Species Action Plan for each species: pearl-bordered fritillary (Barnett and Warren, 1995b), and small pearl-bordered fritillary (Barnett and Warren, 1995a). In summary, for both species, the most frequently used larval foodplant is common dog-violet (*Viola riviniana)*. The pearl-bordered fritillary in the southeast England occurs in dry, open woodlands where the foodplant grows abundantly in sunny, sheltered conditions. The small pearl-bordered fritillary occurs in damp, open deciduous woodlands or damp grassland, with marsh violet (*Viola palustris)* also being used as a larval foodplant in wetter locations.

Research from coppiced woodland in southern England has shown that the small pearl-bordered and pearl-bordered fritillaries are largely sedentary species, forming closed colonies within discrete areas, with most individuals rarely moving outside the habitat patch where they emerge. The evidence indicates that if new areas of suitable habitat are to be utilised, they must occur close to existing populations, and within a few hundred metres of an existing colony to ensure rapid colonisation (Barnett and Warren, 1995a and 1995b).

For conservation management of these species, the aim is to target the remaining populations in the South East, identify the most appropriate management, and provide appropriate new habitat to enable population expansion and help re-establish species metapopulations. For both species, the best habitat conditions are likely to be provided by rotational coppicing and a network of wide, sunny rides with structural diversity in the ride-side vegetation. The pearl-bordered fritillary is one of the priority woodland species being targeted by the Forestry Commission in its contribution to the UK Species Action Plans. The Commission's aim is to target coppice management and restoration within 4km of existing woodland colonies of this species (Forestry Commission, 2000). The use of 4km buffer zones reflects the potential dispersal ability of these species. In a study of the pearl-bordered fritillary in Cirencester Park Woods in Gloucestershire, for instance, a marked female was found to have travelled 4.5km, across arable land and a wooded valley (Oates, 2004), suggesting 'that the butterfly is capable of ranging over a sizeable area of landscape' (Oates, 2003).

This 4km 'buffer' has been adopted for this GIS analysis for both fritillaries; targeting woodlands within 4km of the remaining populations will also provide management suitable for a range of other species which depend on similar management. As well as woodland management, the emphasis within the 4km zone also needs to be on restoring and creating area of semi-natural habitat, which will help facilitate species dispersal in the wider countryside.

GIS Methodology

The aim of the GIS analysis is to map the former and current extent of these species in the South East, and through this, target woodland management and habitat connectivity within 4km of the remaining populations. Butterfly records have been collected by Butterfly Conservation for many years, with two key recording periods: 1970 to 1982, and 1995 to 1999. Provisional records are currently available for 2000-2003. These records are held by Butterfly Conservation in spreadsheet format, including population size, breeding status, date and site information, and an Ordnance Survey grid reference. The grid reference can be used to geographically locate each species record in the GIS as point data, with the information for each record linked to its respective geographic locator point.

The use of Butterfly Conservation's records enables both the historic and remaining populations to be identified, together with the size of colonies, and their distribution through a woodland complex. For each remaining population, a 4km buffer is created using the central point ('centroid') from the woodland supporting the population. Within each buffer zone, the population data can then be compared with other datasets, such as ancient woodland status, woodland management type, grant scheme distribution, proximity of other species (particularly those benefiting from similar management), etc. To provide detailed statistics, each dataset (which may have national coverage) is first cut to the boundary of the buffer. The location and size of the buffers for all the remaining populations can then be defined, together with statistics for all other datasets within these areas.

Former populations are considered for this study to be those where a population was recorded between 1970 and 1982, but not in 1995-1999 or subsequently. Similarly, a population is also noted as not recorded if it does not appear in the 2000-2003 records, having appeared ion 1995-1999. However, it cannot be assumed that the absence of a species record from a site means that the butterfly no longer occurs there. Similarly, sites may have been missed in survey periods. Although the South East has strong butterfly recorder coverage, with the key sites for both species known and regularly monitored, the limitations on recording effort mean that small populations may have been overlooked. A national survey of the pearl-bordered fritillary is being undertaken in 2004 by Butterfly Conservation. It is hoped that this survey will confirm whether this species has been definitely lost from former sites, and highlight any populations missed from surveys in recent years.

Results

Lack of space prevents a detailed representation of the data which will be presented at the conference. For illustrative purposes, Figure 1 shows the distribution of pearl-bordered and small pearl-bordered fritillaries for the recording period 2000-03, for the area covered by the AONBs Woodlands Programme. When compared with the data for the previous recording periods, significant declines are shown in the range of both species. The map also shows the 4km management buffer which has been created around each of the remaining colonies. An example of the statistics from the combined buffer zones is shown in Table 2. Building on this, the mapping of the extent and distribution of all semi-natural habitat and existing grant schemes enables the identification of areas for targeted management and habitat restoration/ creation to provide habitat linkage between the key woodland areas supporting these butterfly populations. In addition, as well as identifying woodlands to target within the 4km buffer, the data also enables the identification of woodlands previously supporting colonies, which may provide suitable areas for species translocation.

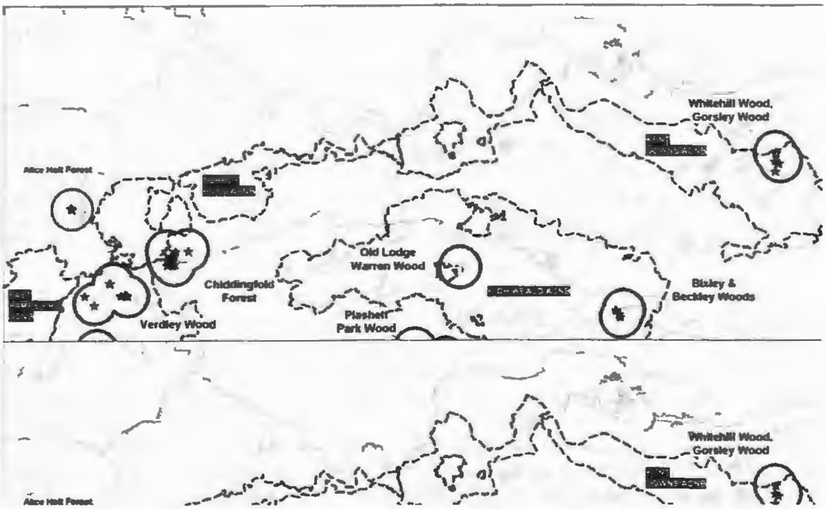

Figure 1. Records for pearl-bordered and small pearl-bordered fritillaries in the South East 2000-03 (provisional data), with 4km management buffer. Woodland names refer to main areas where these species were recorded.

Table 2. Woodland statistics from 4km management buffer for remaining colonies of pearl-bordered and small pearl-bordered fritillaries in the South East.

Woodland type	Area (ha)
Woodland recorded in the National Inventory of Woodland and Trees (NIWT) within 4km buffer (woods >2ha). National survey completed July 2000	17,666
Ancient woodland within 4km buffer (woods >2ha)	11,020 (62% of NIWT woodland)
Forest Enterprise-managed land in 4km buffer	3,447 (19.5% of NIWT woodland)
Woodland in Woodland Grant Scheme in 4km buffer (data as at November 2002)	6,177 (35% of NIWT woodland)
Areas managed as coppice in 4km buffer (from NIWT)	1,619 (9% of NIWT woodland)

Discussion

One of the primary aims of the GIS analysis of pearl-bordered and small pearl-bordered fritillaries is to demonstrate that addressing the decline of these species can only be achieved by looking at their populations at a landscape-scale, rather than only working at the site level.

For example, a small population of the pearl-bordered fritillary was last recorded in the Blean woods near Canterbury in 1997. The Blean is a large woodland complex well-managed for butterflies, but this species still declined and was lost from the area. There may be a number of reasons for this loss. The population may always have been small, and not continuously present at the Blean. Its presence there would have reflected changes in the wider population in the area. However, as the Blean population became more isolated with the loss of populations in the wider countryside, the lack of immigration of new individuals threatened the population's viability. This, together with pressures such as parasitism, predation, and poor weather conditions, would have led to the eventual extinction of this species from the Blean woods.

There is also evidence that rare butterflies such as the fritillaries were once more mobile than they are now. As habitat quality has declined across the countryside, suitable habitat may only be available several kilometres away from an isolated population. As a result, mobile individuals who leave isolated populations are unlikely to survive to leave progeny, leading to a remaining population of less-mobile individuals as a result of selection (Pollard and Yates, 1993). Whereas in the past the species in the Blean may have naturally died out, and then returned, this is now unlikely to happen without translocation of butterflies, given the lack of nearby populations, and the severance factors such as roads and urban areas that separate the Blean from the last remaining population of pearl-bordered fritillary in Kent, south of Canterbury.

As might be expected, the populations of both fritillaries shown in the Butterfly Conservation records of 2000-2003 tend to have been historically the strongest colonies. However, these population 'strongholds' are also in decline, and are unlikely to survive if suitable habitat, both in their existing sites, and in the wider countryside, is not provided and maintained. The conservation management needed, therefore, is a mosaic of habitat patches, at first within the buffers around the remaining populations, but then across the wider landscape. What is also an urgent requirement is the creation of suitable open space in woodlands (often through coppice and ride management) on a much larger scale than is currently being undertaken, and, above all, targeted in the right areas, at the landscape-scale. This will allow the colonisation of new areas, which ultimately is the key to survival of threatened species, rather than purely managing the sites where they still exist. Oates (2000), commenting on the decline of the Duke of Burgundy butterfly, stated that its decline 'should be measured not so much by the loss of colonies but by the paucity of new colonisations' (quoted in Roper, 2004).

Conclusion

The use of GIS maps provides an ability to analyse many factors relating to species populations, and enable targeted management founded on a clear evidence base. However, another important use of GIS is its simple, but important visual impact. Quite simply, no amount of words can compare with the mapped juxtaposition of records from a declining species' former and current extent. As the switch is made from the 'old' to the 'new' map, and the records disappear, the nature and extent of the decline becomes immediately apparent. This visual message, combined with our increasing knowledge of the ecology of these species, flags up the clear need for a change from both site-based, and reactive woodland conservation, to proactive, targeted efforts, based on the best information we have at our disposal.

Many of the remaining fritillary populations in the South East lie within and close to the AONBs covered by the Woodlands Programme. The results of the GIS analysis highlight the important role these protected landscapes have in conserving some of our most threatened woodland species, and providing a focus for landscape-scale woodland management. As a result, the Woodlands Programme is helping to influence the policies of agencies involved in habitat management across the region, with the work on threatened fritillaries included as a case study in the Regional Forestry Framework for the South East (Forestry Framework Steering Group, 2004). The work is also informing a joint Heritage Lottery Fund bid from Butterfly Conservation and the Forestry Commission that aims to address the declining fritillary populations across the South East.

References

Barnett, L.K. & Warren, M.S. (1995a) *Species action plan: small pearl-bordered fritillary.* Butterfly Conservation, Wareham.

Barnett, L.K. & Warren, M.S. (1995b) *Species action plan: pearl-bordered fritillary.* Butterfly Conservation, Wareham.

Bourn, N., Thomas, J., Stewart, K., & Clarke, R. (2002) Importance of habitat quality and isolation: implications for the management of butterflies in fragmented landscapes. *British Wildlife* 13 (6): 398-403.

Butterfly Conservation (2000) *Regional Action Plan South-East England.* Butterfly Conservation, Wareham.

English Nature (2004) State of nature: lowlands - future landscapes for wildlife. English Nature, Peterborough.
http://www.english-nature.org.uk/pubs/publication/PDF/SONlow.pdf

Forestry Commission (2000) UK Species Action Plans: priorities for the Forestry Commission in England. Forestry Commission England, Cambridge.

Forestry Framework Steering Group (2004) *Seeing the wood for the trees: a consultation on a Forestry Framework for the South East.* Government Office for the South East, Guildford.

Gilpin, M. & Hanski, I. (Eds.) (1991) Metapopulation dynamics: empirical and theoretical investigations. Academic Press, London.

Oates, M. (2000) Duke of Burgundy – conserving the intractable. *British Wildlife* 11 (4): 250-257.

Oates, M. (2003) The ecology & dynamics of the pearl-bordered fritillary *Boloria euphrosyne* in a Gloucestershire woodland. Butterfly Conservation Report No. S03-12. Butterfly Conservation, Wareham.

Oates, M. (2004) The ecology of the pearl-bordered fritillary in woodland. *British Wildlife* 15 (4): 229-236.

Pollard, E. & Yates, T.J. (1993) *Monitoring butterflies for ecology and conservation.* Chapman & Hall, London.

Roper, P. (2004) Are we managing habitats for woodland fritillaries in the right way? Unpublished paper. E-mail: patrick@prassociates.co.uk

Woodland Trust, The (2002) *Space for nature: landscape-scale action for woodland biodiversity.* Woodland Trust, Grantham.
http://www.woodland-trust.org.uk/policy/publications.htm

Forest habitat network development in the West Weald Woods

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Abstract

Woodland connectivity is examined at three scales; the wider landscape of 350km^2, a DPhil research site of 143km^2 and an individual nature reserve. Geographical Information Systems are used to describe current landscape elements and to model different woodland restoration scenarios. At a site scale, a near-natural, non-prescriptive management approach to creating pasture woodland through natural regeneration and grazing is being trialled.

Introduction

The Weald of northwest Sussex and southwest Surrey, England is a very important area for woodland nature conservation and could benefit from management at a landscape scale. 'Landscape scale' however, must be defined in terms of perceptual scale. Wiens, (1976) suggests that if a landscape, or habitat patches within it, is defined from a particular organism's perspective, landscape size will differ hugely in scale between organisms. In a UK context a kestrel's (*Falco tinnunculus*) 'landscape' will be different from that of one of its prey items, e.g., a bank vole (*Clethrionomys glareolus*). In the semi-rural landscape considered here, it is the human perspective that influences our actions in terms of land use, conservation and management.

This paper looks at the functioning of the landscape at three scales. First, at the wider landscape scale (350 km^2), we outline a developing project, the West Weald Woods Project, which aims to enhance the functional connectivity of forest habitat (including open habitats) in the area. Many of the woods are, or have been, pasture woodland. There is also much interest in open, grazed habitats. Future forest expansion may therefore take the form of natural regeneration of open pasture woodland, using grazing as a management tool, rather than through planting. Second, we look at a site of 143 km^2 in West Sussex, which is being studied for a DPhil research project. A Geographical Information System (GIS) is used to describe the landscape structure and configuration of woodland patches in a matrix of agricultural land and to model different management approaches to increase forest connectivity. Third, we look at an example of pasture woodland re-creation in a Sussex Wildlife Trust nature reserve. The proposal here is to develop a naturalistic grazing regime and allow habitat patterns to develop in a non-prescriptive way, arguably a landscape approach at a site scale.

The West Weald Woods Project.

The landscape unit under consideration has approximately 25% woodland cover, including ancient semi-natural woodland, shaws (narrow belts of woodland) and hedges, interspersed with open areas of pasture and arable land (Figure 1).

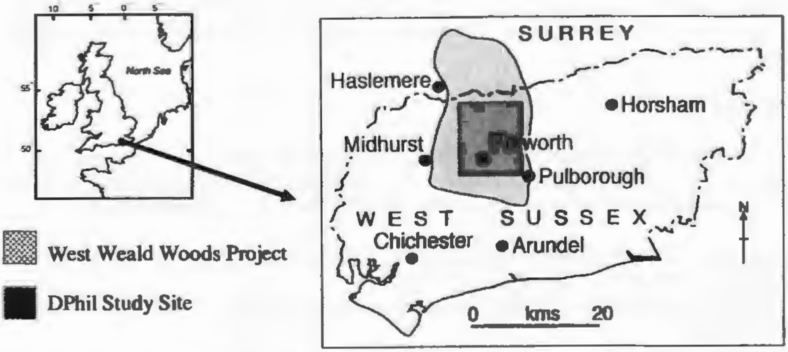

Figure 1. Location of the West Weald Woods Project area and the DPhil study site.

The clay vale of the Low Weald in West Sussex and Surrey is bordered to the west and north by a greensand ridge, while river valleys mark the eastern and southern edges. The unit's importance for nature conservation is recognised through European and national statutory legislation, plus local designations. There are two candidate Special Areas of Conservation (cSACs), one National Nature Reserve (NNR), three Sites of Special Scientific Interest (SSSI) and many Sites of Nature Conservation Importance.

Woodland type varies across the unit: Chiddingfold Forest (managed by Forest Enterprise) has a long history of coppice and timber production; Ebernoe Common is pasture woodland and The Mens is old growth high forest. Species found in these woodlands reflect past and present management, for example, rare butterflies, typical of forest glades and rides occur in Chiddingfold Forest. Ebernoe Common NNR is exceptional for its rich lichen flora, typical of large trees in open areas and for two species of bat, the barbastelle (*Barbastellus barbastella*) and the Bechstein's (*Myotis bechsteinii*). Both species are of European importance (as reflected by the site's cSAC status) and favour mature and veteran trees. The Mens is particularly important for fungi, containing three species of *Russula*, for which this is the only known site (English Nature, 1980). The challenge therefore, is to enhance connectivity across the landscape unit, taking into account the differing ecological requirements of a range of species at a variety of scales.

The guiding principles are to identify ways in which existing habitats can be enhanced at a range of scales. Within a large woodland complex such as Chiddingfold Forest, long-term management by Forest Enterprise aims to restore parts of the replanted forest to semi-natural condition, opening up small-scale clearings, to benefit rare butterfly populations. This will result in a network of interconnecting glades that will sustain and increase populations of the wood white butterfly (*Leptidea sinapsis*). Expansion of Core Forest Areas (CFAs) (Peterken, 2000) is also a key aim, which may be achieved through planting or natural regeneration. Maintenance of glades to ensure a forest cover/open habitat balance could be through low-intensity grazing, using rare breeds of livestock (see later in this paper).

The spatial relationship between woodland patches, plus the land use in the matrix can be critical in influencing connectivity in the landscape . The West Weald Woods Project is keen to utilise results from on-going DPhil research that includes Ebernoe Common and The Mens.

DPhil study site

This 143km^2 site in West Sussex (Figure 1) was chosen to investigate opportunities for linking two large CFAs of ancient semi-natural woodland, i.e., Ebernoe Common and The Mens, to create a forest habitat network using GIS. The objectives of the research are to:

- describe the current spatial arrangement of landscape elements, including baseline ecological and land cover data using ArcView 3.2
- quantify the degree of habitat fragmentation or connectivity, based on ecological principles
- test a series of different restoration scenarios to create a large scale, near-natural forest habitat network
- develop a layer of land valuation based on current agricultural land classification, development potential and ecological value
- evaluate the effectiveness of GIS as a tool for achieving the above.

This interdisciplinary research project will incorporate a range of data sets including land cover, land use, land value, land ownership and ecological value. GIS enables the complex interrogation of such data to explore a number of "what if…?" scenarios. It is clearly a very powerful tool, but how easy is it to use? How reliable are existing data sets? The aim is to evaluate the use of GIS as a tool to address real conservation issues in a real landscape.

Methods

A digital inventory of the landscape elements was created using ArcView 3.2 GIS. Existing data sets were reviewed and their quality and suitability for the research objectives was assessed. Data sources include: Ordnance Survey line and MasterMap data; protected areas under local, national and international legislation, Ancient Woodland Inventory (AWI) from English Nature; National Inventory of Woodland Type (NIWT) from the Forestry Commission; areas receiving grants under national agricultural environment schemes and orthorectified and georeferenced digital aerial photography, flown in 1999. Where existing data sets did not meet the research requirements, new data sets were created.

Ordnance Survey line data was customised using aerial photographs. Ground-truthing was undertaken where aerial photographic interpretation was difficulty. Ordnance Survey MasterMap data were referred to for clarification in some instances. This is an on-going process and once editing is complete, polygons will be created using ArcInfo and attribute data relating to land cover, land use, land ownership and land value will be added. Patches of high and low biodiversity interest will be identified, to aid in future modelling work. Landscape metrics will be generated to describe the spatial attributes of habitat patches, including area, perimeter, shape index, nearest-neighbour distance plus others, (still to be determined) relating to habitat fragmentation, isolation and connectivity.

Connectivity in a landscape is a function of the relationship between landscape structure and organism movement (Merriam, 1984). In order to model connectivity, relevant species must be identified and empirical data on their distribution and dispersal mechanisms obtained. The literature is full of terms such as 'indicator,' 'flagship,' 'keystone,' 'umbrella,' and more recently, 'focal' (Lambeck, 1997) species. All these terms have their pros and cons, as reviewed by Simberloff (1998) and Lindenmayer and Fischer (2002). In view of this, the term 'priority' species has been coined for the purposes of the current DPhil research. This reflects the importance of species in the context of the study site. The wood white butterfly, for example, is a local Biodiversity Action Plan species, the barbastelle and Bechstein's bat are of European importance, the marsh tit (*Parus palustris*) is causing concern nationally due to its declining numbers (Perrins, 2003). These four 'priority' species have been identified and data on their habitat requirements and dispersal mechanisms is being collected. Additional species may be identified, should sufficient data be available.

Data on bat movements have been obtained in the form of point data from radio tagged individuals from nursery roosts of the two bat species, (Greenaway *et al*, 2000). Breeding bird survey work started in spring 2004 and it is hoped to obtain point data on the wood white from the local branch of Butterfly Conservation.

Results

These are still very limited at present, but can be divided into three broad areas:

1. Preliminary review of existing digital data sets
2. Initial landscape metrics
3. Ecological data

Digital data sets

Review of existing data sets show it is rare that any given data will meet all the researcher's needs. A range of factors must be considered: what is the purpose of the data, how was it obtained, at what scale was it captured, in what format is it available? Comparison of two woodland data sets, the AWI and the NIWT are a case in point, both provide valuable information but with caveats. The former was created by interpreting areas of woodland from old maps, First Series Ordnance Survey and earlier. It was never envisaged that the data would be available in digital format. When overlaid on recent aerial photographs there are a number of discrepancies, both in area and woodland type and in some instances in actual land cover. Information is now sought from of it that was never originally intended. The NIWT was digitised from 1999 aerial photography and as such, represents more accurately what is on the ground. As with the AWI data, however, there are discrepancies where areas of particular woodland type are incorrect, or land that is classified as broadleaved woodland is actually pasture.

Landscape metrics

These are incomplete as editing of line data is still in progress. However shape indices have been calculated for a small sample of woodlands to get a feel for the type of information they provide (Table 1).

Table 1. Shape Indices for a sample of woodlands

Area (ha)	Perimeter (m)	Shape Index
1.76	662	1.40
2.52	645	1.14
3.30	1528	1.82
5.60	961	1.49
71.4041	7404	2.47
83.2792	8248	2.54

The two large woodlands are Ebernoe Common and The Mens, respectively. The remaining four small woodland patches lie in the agricultural matrix between these two CFAs. Further work is required to interpret the significance of the shape index, although the greater the perimeter/area ratio, the larger is the shape index.

Ecological data.

Point data from radio tagged barbastelles reveal a highly mobile species, with one female making a round trip of 18km in one night to a favoured foraging area. When the data are overlaid on aerial photographs, the spread of points show how the bats use all available woodland and hedgerow cover and watercourses during their foraging trips. Where cover ends, individuals will wait until complete darkness before flying rapidly across open and therefore potentially hostile areas of arable land. The Bechstein's bat in contrast, is very restricted in its movements from the nursery roosts, foraging in the immediate surroundings and only occasionally venturing outside the woodland boundaries. The implications in terms of landscape connectivity for these two species differ markedly. In simplistic terms, the barbastelle requires links of well-wooded vegetation between foraging areas, together with watercourses and open water bodies, whereas the Bechstein's requires expansion of existing mature woodland.

The breeding bird survey has only just started, however, the distribution of marsh tit is wider than previously thought, with records from four new woodland patches. This is probably a reflection of the level of previous survey effort rather than lack of suitable habitat. The survey will be completed by the end of May 2004 and results will be presented as an addendum to this paper at the conference.

Future work

- Complete editing of line data and generate attribute tables.
- Describe the current connectivity/fragmentation/isolation in the study site.

- Start modelling changes in the landscape to increase connectivity, e.g., expanding CFAs, creating/enhancing links between habitat patches, creating stepping-stones in the hostile arable matrix.
- Use land ownership data to target future land purchase/management agreements.
- Evaluate the use of GIS as a tool to achieve the research objectives

Butcherland/Ebernoe Common: site scale work trialing a landscape-scale approach.

Ebernoe Common is one of the best examples of prime pasture woodland habitat in the country. There are, however, ecological conflicts on the site, resulting from trying to maintain a diversity of wildlife on a relatively small site. Lichens require large old trees in open-canopy pasture woodland while bats such as the Bechstein's and barbastelle require dense old growth forest. This conflict indicates the need for more structural diversity at a larger scale, clearly the current site is too small to accommodate a range of habitat requirements.

In 2002, the opportunity arose for the Sussex Wildlife Trust to purchase 80 ha of farmland adjoining Ebernoe Common to the southeast. Butcherland consists of permanent grassland and arable fields with set-aside, separated by hedges and strips of woodland. The Trusts' objective is to create pasture woodland, so expanding the features found on Ebernoe and enhancing habitat connectivity. This will be achieved through a near-natural approach rather than traditional management planning. Normally, land is managed to achieve a definite set end point. However, setting such an end point, e.g., a particular habitat pattern, would be against the principle of a near-natural approach. Instead the Trust has a broad vision of pasture woodland and will encourage the natural processes that will lead to its establishment.

The lack of a clear end point, however, means there is no way of measuring progress towards it. A system is required whereby the Trust is not prescriptive about what habitats develop where, but which does allow the area to develop in the general direction of pasture woodland. This may be achieved by setting broad limits of acceptable change. Action will be taken if the site moves outside these limits, , guided by the principle of influencing the processes that deliver the required habitat. In the case of woodland cover, for example, the normal approach might be to determine areas of future woodland and write a planting plan or encourage natural regeneration in specific areas. The near-natural approach, however, would set broad limits for acceptable change such that woodland cover across the whole site is between 30% and 70%. The natural processes that could be influenced to maintain these limits are natural regeneration and grazing.

Management of Butcherland

The site is poorly wooded at present, so the aim is to encourage the expansion of woodland and scrub, whilst also encouraging the formation of future forest glades. Ten years before purchase most of Butcherland consisted of cultivated fields between woodland and hedges. There was, therefore, a sharp boundary between woodland edge and arable field. At about that time, however, a 6m strip of set-aside was established around most fields, which was cut annually. Scrub regeneration had established in the sward but could be promoted into the shrub layer.

Phase I

The objective after purchase was to establish a system allowing scrub regeneration but also encouraging the establishment of future glades (Figure 2). Stock grazing will be introduced and controlled to manage the interaction between grazing and regeneration. The set-aside zone has been fenced off, promoting scrub regeneration. Mown areas have been established towards the centre of the fields prior to the introduction of grazing, so encourage grass growth and tillering. Initially, commercial breeds that require rich fodder will be used. Cattle can wander throughout the fenced area but will tend to preferentially graze the mown areas. Sward height and structure will be carefully monitored and stock densities adjusted to maintain very low grazing intensity.

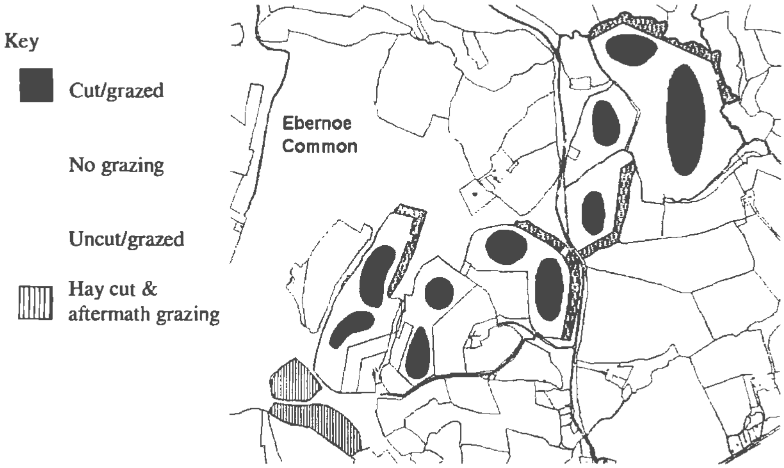

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Figure 2. Diagram of management zones across the Butcherland site.

Phase II

Approximately 10 years after purchase and about 20 years after the set-aside strips were first established, there should be a zone of tree and scrub regeneration around the preferential grazing areas. Elements of randomness may already be detectable, grazed areas may not be exactly the same shape as the original mown blocks and there may be grazed areas forming within the regeneration zone. At this point the fences around the set-aside strips will be removed and it may be possible to combine Butcherland and Ebernoe under one naturalistic grazing regime. Traditional breeds will now replace the commercial breeds of cattle, as they tend to prefer rough grazing or browse. A breed will be sought that is a reasonable analogue of wild cattle in that they might have a similar ecological effect as wild cattle.

An end point will not be set in terms of a defined habitat pattern, but a recognisable pasture woodland ecosystem should be forming after about 50 years. The ecotones between wooded areas and open habitat should be very wide, tree regeneration in patches of thorny scrub should be scattered throughout the area and preferential grazing areas should have turned into permanent forest glades. There will be a dynamic between wooded and open habitat, driven by regeneration on one hand and grazing pressure on the other.

It is hoped that the results from Butcherland will provide an illustration of what could be done in the wider West Weald Woods Project area. The objective is to develop a naturalistic grazing regime that allows the development of pasture woodland and through this, a complex network of open and wooded habitat. Applied over a large scale, this should deliver the habitat connectivity that is required by species of conservation concern in the area.

Summary

The West Weald Woods Project area is a landscape unit where increased connectivity, between and within woods, is required in order to conserve and enhance the habitats and species for which the area is important. GIS is a valuable tool for describing the current landscape resource and allows the modelling of a range of management scenarios to achieve particular conservation objectives. The Sussex Wildlife Trust's work at Butcherland is investigating how a near-natural approach could be developed that is applicable at a landscape scale. This is based on a non-prescriptive management approach, concerned with process rather than pattern, with the broad objective to create prime pasture woodland.

Acknowledgements

Mark Monk-Terry (Sussex Wildlife Trust), Frank Greenaway, English Nature, the Forestry Commission.

References

English Nature (1980) *The Mens SSSI citation.* English Nature, Peterborough.
Greenaway, F. & Hill, D. (2003). Bats of Ebernoe Common-Interim report. Sussex Wildlife Trust.
Lambeck, R.J. (2002) Focal species and restoration ecology: response to Lindenmayer *et al. Conservation Biology* **16:** 549-551.
Lindenmayer, D. B., & Fischer, J. (2003) Sound Science or a social hook- a response to Brooker's application of the focal species approach. *Landscape and Urban Planning* **62, 3:**149-158.
Merriam, G. (1984) Connectivity: a fundamental ecological characteristic of landscape pattern. *Landscape Research and Planning* **I:** 5-15. **Perrins, C. (2003)** The status of Marsh and Willow Tits in the UK. *British Birds* **96:** 418-426.
Peterken, G. (2002) Rebuilding networks of forest habitats. *Landscape Ecology* **25, 3**:291-303.
Simberloff, D. (1998) Flagships, umbrellas and keystones: is single-species management passé in the landscape era? *Biological Conservation* **83:** 247-257.
Wiens, J. A. (1976) Population responses to patchy environments. *Annual Review of Ecology and Systematics* **7:** 81-120.

Habitat re-creation and restoration at the landscape scale: woodlands in Wales

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Abstract

The aim of this three year project funded by the Countryside Council for Wales (CCW) is to develop techniques firstly, to refine and update existing targets for habitat restoration and re-creation at the landscape scale and secondly, to develop a GIS-based model for the implementation of those targets at the local scale. Landscape Character Assessment (LCA) is being used to map Landscape Types across the whole of Wales as the first stage towards setting strategic habitat targets. The GIS habitat model uses data from the digital Phase 1 Habitat Survey for Wales to determine the suitability of individual sites for restoration to specific habitat types, including broadleaf woodland. The long-term aim is to develop a system that strengthens the character of Welsh landscapes and provides real biodiversity benefits based upon realistic targets given limited resources for habitat restoration and re-creation.

Introduction

The aim of this three year project funded by the Countryside Council for Wales (CCW) is to develop a method to assist with habitat restoration (improvement in condition of habitats that have deteriorated) and re-creation (expansion of existing habitats) at the landscape scale in Wales. Firstly, at the strategic scale, the objective is to determine the potential of different landscape types to support priority habitats. Secondly, at the local scale, a GIS (Geographical Information System) habitat model based upon a set of ecological decision rules is being applied to identify suitable sites for implementing habitat targets on the ground. This paper describes the overall approach, concentrating in particular upon the challenge to 'scale-down' from strategic habitat restoration and re-creation targets to local implementation.

In the Welsh countryside loss of habitats, with many of the plant and animal species they support, has been considerable. Broad-leaved woodland, for instance, now covers barely four percent of the land area (86,000 ha) (Jones *et al.*, 2003). Much of what remains is intensively grazed by livestock, threatened by lack of management or invaded by aggressive plants such as rhododendron. Of the 188 rare and declining plant and animal species recorded in Wales, 55 are associated with broad-leaved woodland (CCW, 2003). Other habitats apart from woodland, have also suffered a decline in extent and quality to the point where only 15 percent of lowland Wales now supports semi-natural habitat.

In response to these continuing losses in extent and quality, there is renewed interest in the potential for habitat restoration and re-creation. The Tir Gofal agri-environment scheme is providing funds to manage and protect a number of priority habitats including heathland; bog and reedbeds; native broadleaved woodland and species-rich grassland. By the end of March 2003, 1,500 agreements had been signed, covering 2,030 holdings and 150,000 ha of land (CCW, 2003).

Habitat targets

The landscape character spatial framework

Landscape character is defined as a distinct, recognisable and consistent pattern of elements in the landscape. The fundamental building block of the landscape character spatial framework is the *Landscape Description Unit* (LDU). LDUs are distinct and relatively homogenous units of land, each defined by a series of physical and cultural attributes:

- physiography; geology (structure) & landform
- ground type; geology (rock type) and soil type
- land cover; tree cover and land use
- settlement; settlement pattern and farm type

Landscape Types have been mapped for the whole of Wales using a Landscape Character Assessment (LCA) approach (Countryside Commission, 1991; Swanwick and Land Use Consultants, 2002). The process of landscape character mapping using nationally available datasets enables broad patterns to be distinguished which assist with understanding the relationship between the many factors that contribute to landscape character. This systematic approach to landscape assessment which gives equal weight to the natural *and* cultural dimensions of the landscape provides, at a range of scales, a useful spatial framework within which to assess habitat potential and ecological quality.

Habitat potential is determined for each discrete Landscape Type based upon a combination of the physical (landform, geology, and soil type) and cultural (settlement pattern, tree cover, land cover, farm type and structure) attributes contained within the GIS landscape character database. Recently published work by CCW (Jones *et al*, 2003) summarises much of what is known about the environmental requirements of the 27 priority habitats in Wales and this information, with data from British Plant Communities (Rodwell,1991), is being used to determine the requirements of different habitats more precisely. Information of this type is also being supplemented with a GIS analysis of the type and extent of priority habitats, available as Phase 1 Habitat Survey data in digital form for the whole of Wales, within different Landscape Types. Thus, whilst it is known that 'upland oak woodland' occurs on base-poor to acidic soils under conditions of high rainfall (> 1,000mm per annum) (Jones *et al.*, 2003), the *range* of environmental conditions over which it is found at different locations in the country is less well known. An important aspect of the project therefore, is to use the opportunity provided by the availability of relatively recent Phase 1 Habitat Survey data to explore the link between each Landscape Type characterised by distinct cultural and physical attributes, and the presence, extent and quality of priority habitats.

This type of analysis is important to identify the constraints imposed upon the distribution of different habitats by cultural factors (Warnock *et al.*, 2001). The distribution and survival of semi-natural habitats in the Welsh countryside is closely related to the evolution of the landscape over millennia, the result of the complex interaction of people and environment. Cultural factors, for example the pattern of settlement, differences in woodland and tree cover, operate as constraints to biodiversity setting and habitat restoration and re-creation. For example, two of the National Vegetation Classification (NVC) community types that comprise upland oak woodland (W17 & W11) are the classic sessile oak woodlands, often 'hanging' on valley sides. This characteristic distribution pattern is a product of cultural factors as much as physical ones; hanging oak woodlands being 'left' between the lower in-bye or fridd and the open, unenclosed moorland above. Thus, whilst restoring and re-creating the links between fragments of remaining woodland along the contours of the slope may serve to connect isolated woodland patches and strengthen the historical character of the landscape, woodland re-creation in lower areas on less steep slopes that have been unwooded for centuries may detract from local character and diminish the diversity of landscape types across Wales.

Thus cultural patterns profoundly influence the capacity of a landscape to accommodate change of all types, including habitat expansion. An understanding of the constraints imposed by cultural patterns for the design of future landscapes is vitally important if habitat creation is to be sympathetic to historical continuity and variability in the landscape.

Habitat Action Plan targets

Both the restoration target and the expansion target for upland oakwood is 2,200ha, distributed across all 24 Local Biodiversity Action Plan (LBAP) areas (Jones *et al.*, 2003). Targets for upland mixed ashwood are 250 ha for restoration and 650 ha for expansion and for lowland beech and yew woodland, 150 ha for restoration and 300 ha for expansion. Similar targets have also been established for remaining priority habitats including; lowland mixed deciduous woodland, and lowland wood-pasture and parkland. These figures are national targets and the criteria used to develop the targets and, more importantly, to select suitable sites for habitat expansion remain ill-defined (Knightbridge, 2000) with implications for effective long-term policy.

An important aim of the project therefore, is to determine the habitat potential of each Landscape Type from knowledge of the environmental conditions under which it is typically found and to identify the cultural constraints that are characteristic of the Landscape Type. Thus, a Landscape Type with a high potential to support, for example, upland oak woodland but which only contains a few surviving and isolated fragments is deemed to be in poor condition but with the opportunity for significant habitat re-creation. Conversely, a Landscape Type with a low potential to support a specific habitat type, for example, in areas where a habitat is transitional or at the edge of its range, but high actual extent may be more suitable for restoration and protection of the remaining extant habitat. These are, of course, policy decisions but it is anticipated that this type of analysis will assist with formulating policies which protect those habitats that remain and identify suitable sites for expansion of habitats that have been lost.

Landscape Types are relatively large, covering in some cases 40 – 50 km^2 and incorporating a wide range of environmental conditions and cultural characteristics. At the next scale of analysis, the project is attempting to answer questions about 'which habitats, where and how much' within the context of updated habitat targets for each Landscape Type. This is being achieved using a GIS habitat model, thus providing the link between the regional strategy for habitat targets and the local implementation for restoration and re-creation at the scale of the field-parcel.

GIS habitat model

The local scale of analysis is necessary to translate regional biodiversity targets into change on the ground, for example, by targeting resources (e.g., under the Tir Gofal agri-environment scheme and its successor) on particular habitat types at specific locations. The GIS habitat model is based on a set of ecological decision rules to determine the suitability of each land parcel for re-creation to broadleaf woodland, or other habitat. The following describes briefly, the spatial functions that were applied to the Phase 1 Habitat Survey data in two 10km x 10km pilot test areas in South Wales.

'Existing' function

This function returns a suitability score for the Active Land Parcel (the current field parcel being considered by the program in the model) based upon the *existing* habitat of that parcel. This tests whether the existing habitat is suitable for conversion into the Target Habitat. A score of zero indicates that the exiting habitat is unsuitable for the creation of the Target Habitat; for example, if the target is broadleaved woodland and the parcel is already existing broadleaved woodland. Table 1 shows an example of parameters for the 'Existing' function that would favour the creation of broadleaved woodland in parcels of dense scrub over improved grassland and preclude the conversion of dry acid heath to woodland.

Table 1. Example parameters for the 'Existing' function for broadleaved woodland creation. The suitability code is ranked from 'a' (highly suitable) to 'e' (entirely unsuitable).

Existing Phase 1 Habitat	Suitability Code
Dense scrub (A.2.1)	a
Improved grassland (B.4)	c
Dry acid heath (D.1.1)	e

'Area' function

This function returns a suitability score for the Active Land Parcel, based upon the area of the parcel. This function allows targets to be set that will favour the creation of habitat patches that are greater than a specified threshold area.

Table 2 shows an example of parameters for the 'Area' function for creation of broadleaved woodland. Parcels with an area of greater than 25 hectares are considered most suitable for woodland creation because they would contain a significant 'core' area and are more likely to support viable populations of typical woodland plants and animals. Parcels with an area of less than 2 hectares are considered least suitable for woodland creation because they have a small 'core' and are strongly affected by woodland edge effects.

Table 2. Example parameters for the 'Area' function for broadleaved woodland creation.

Area of land parcel	**Suitability code**
>= 25 hectares	a
5 – 25 hectares	b
2 – 5 hectares	c
< 2 hectares	e

'Distance To' function

This function assigns a suitability code to the Active Land Parcel, based upon the distance from the boundary of the Active Land Parcel to a patch of habitat of the target habitat. This function allows targets to be set that will favour the creation of new habitat patches near to a potential source of species. Table 3 shows an example of parameters for the 'Distance To' function for creation of broadleaved woodland. Parcels within 20 metres of an existing patch of broadleaved woodland are considered most suitable for woodland creation because the nearby woods could provide a source of less mobile species such as plants or invertebrates. Parcels that are removed more than 250 metres from another patch of broadleaved woodland are considered least suitable for woodland creation because it is unlikely that less mobile species could disperse to the newly created woodland.

Table 3. Example parameters for the 'Area' function for broadleaved woodland creation.

Distance to broadleaved woodland (A.1.1.1)	**Suitability code**
<= 20 metres	a
<= 100 metres	b
<= 250 metres	c
> 250 metres	e

'Neighbourhood' Function

This function defines a fixed width buffer adjacent to the outside edge of the Active Land Parcel and examines the existing Phase 1 habitats within this buffer. On consultation with CCW ecologists it was decided that a simple application of this function would be to measure the proportion of semi-natural habitat in the area surrounding the Active Land Parcel. This permits targets to be set that favour the creation of new habitats in parcels that are surrounded by a high proportion of semi- natural habitats.

Parcels with greater then 50 percent cover of semi-natural habitat in the surrounding buffer were considered most suitable for the creation of new woodland because of the high 'naturalness' of its surroundings. Parcels with less than 1 percent cover of semi natural habitat in the surrounding buffer were considered least suitable because the parcel is likely to be isolated from other semi-natural habitats. The application of this function therefore, tends to reinforce the spatial concentration of semi-natural habitats.

Summary and discussion

The challenge is to modify the output from the model in the context of the strategic habitat targets derived for each Landscape Type based upon its physical and cultural attributes. This has been achieved so far, by weighting the derived suitability score for each Active Land Parcel according to the Landscape Type that the parcel intersects with. It therefore encourages habitat creation within Landscape Types that are particularly suited to the creation of that habitat. For example, heathland creation is particularly suited to landscapes characterised by humic, peaty soils but is not suited to alluvial landscapes with gleyed soils. The weights, which use a relative scale, were determined to reflect the suitability of each Landscape Type for re-creation of broadleaf woodland, lowland heath and unimproved grassland. Thus the 'valley topography with humic drift impoverished soils' was scored 'a' (highly suitable) for broadleaved woodland in one pilot area, in contrast to the 'sandstone plateau with shallow base poor soils' Landscape Type which was coded 'c' (moderately suitable).

However, it is recognised that this is a generalised and simplistic approach that; (i) only accounts for differences in physical attributes and ignores cultural attributes and, (ii) fails to recognise that more than one habitat may be equally suited to a particular set of conditions. Further work is required to determine how to modify the score allocated to a land parcel by the GIS habitat model with reference to the physical *and* cultural attributes of the Landscape Type. Equally important will be to analyse the spatial pattern of remaining woodland, especially within recently identified Core Forest Areas (unpublished) to ensure that the modelling takes into account the importance of linking isolated woodland fragments. In the long-term the aim is to develop a model that combines the important concept of 'habitat potential', largely determined by the physical factors of soil type, geology and landform at the broad scale, with decision-rule modelling based on principles of landscape ecology (Forman and Godron, 1986) at the field parcel scale.

Ultimately the model will need to be in a format that can be run by local ecologists and land managers in any part of Wales to assist with developing a vision for future landscapes based upon realistic habitat targets that strengthen local landscape character and provide biodiversity benefits. In the longer term it will be important to modify the model for selected priority species and to consider the future of Welsh landscapes for a range of climate change predictions.

References

Countryside Commission, 1991. *Assessment and conservation of landscape character: the Warwickshire Landscapes Project approach*. CCP 332.
Countryside Council for Wales, 2003. http://www.ccw.gov.uk/ (accessed April 2004).
Forman, R. T. T., and M. Godron,1986. *Landscape ecology.* John Wiley & Sons, New York. 619p.
Jones, P.S., Stevens, D.P., Blackstock, T.H., Burrows, C.R. & Howe, E.A., 2003. *Priority habitats of Wales – a technical guide.* Countryside Council for Wales, 140 p.
Knightbridge, R. 2000. *BAPS – who needs them*? ECOS, Vol, **21**, 2.
Rodwell J.S. (Ed.), 1991. *British plant communities: Volume Vol. 1 woodlands and scrub*, Cambridge University Press, 395p.
Swanwick, C. & Land Use Consultants, 2002. *Landscape Character Assessment: guidance for England & Scotland.* Prepared on behalf of the Countryside Agency and Scottish Natural Heritage, p.84.
Warnock, S., Griffiths, G.H., Porter J.R. & Simmons, E., 2001. The cultural component of landscape character assessment: data sources and typology. *Proceedings of IALE European Conference 2001, Development of European Landscapes*, pp.229 – 230.

Wildlife corridors and beyond: seeing the wood for the trees in conservation delivery

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Abstract

The relevance of wildlife corridors in mitigating the impacts of patch isolation has a controversial history in the ecological literature. In spite of this fact, the corridor concept has now become firmly embedded in conservation and land use management. Its intuitive appeal has created a powerful metaphor. The views of specialists and practitioners within English Nature were investigated through a seminar that considered the role of wildlife corridors and their alternatives. The content of a questionnaire and a summary of a series of presentations are reported.

Although the need to consider functional connectance across whole land mosaics was generally recognised and supported, there was a strong perception that corridors had become a useful 'social hook' on which to hang the delivery of a wider wildlife agenda that was not exclusively linked to the movement of species. The extent to which wildlife corridors mitigate the impacts of fragmentation remains uncertain. No direct benefits were considered to be present for the species and habitats that were examined. Alternatives are discussed.

Introduction

Several initiatives in the UK are attempting to define woodland networks at the current time that make explicit use of wildlife corridors. For example Scottish Natural Heritage (Fowler and Stiven 2003) have developed a joint strategy with the Forestry Commission that is hoped will lead to a "strengthening of core areas and the development of links and corridors". The role of corridors and the importance of connectivity is still clearly a current issue in conservation. Although the importance of connectivity between spatially divided populations has been generally accepted, considerable uncertainty about the importance of wildlife corridors continues to persist within the ecological literature. For example, Wiens (2002) states that the "evidence for the efficacy of corridors is nowhere near as compelling as the enthusiasm with which corridors have been embraced as a conservation and management tool". As he rightly points out, the emphasis that is placed on the identification and management of corridors tends to promote a simplistic view of landscapes where elements are classified according to their structure rather than according to their function.

The implicit assumption that corridors have some functional significance has been an area of considerable debate (e.g., Simberloff and Cox, 1987; Simberloff *et al.*, 1992; Beier and Noss, 1998). Much of this debate has been narrowly focussed on the importance of physical linkages associated with contiguous linear features (Bennett, 2003). Although a range of different ecological and social functions have also been associated with corridors, a recent review found that most studies still focussed on their role as "conduits" for species movement (Hess and Fischer, 2001). It is this particular function that has fuelled much of the debate. But as Bennett (2003) notes, it has largely "missed the point" because the connectivity of landscapes is about more than physical linkage between habitat patches. In spite of this fact, movement corridors continue to be an active area of research as figure 1 shows.

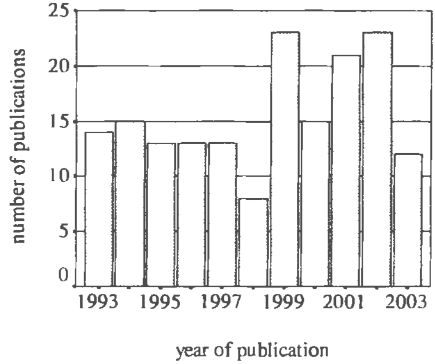

Figure 1: Publication rate for scientific journal papers containing the key words: wildlife corridor(s) and habitat corridor(s). Search was undertaken using relevent databases from Cambridge Abstracts and English Nature Wildlink database.

The importance of connectivity, as an explicit element of landscape structure, has been recognised for some time (Merriam, 1984). Connectivity is determined by the interaction of individual species with landscape structure and the mortality that they experience when they move across different landscape elements (Tischendorf and Fahrig, 2000). Purely structural definitions, such as the patch-corridor-matrix concept (Forman and Godron, 1981), are an oversimplification that assumes that movement is exclusively confined to linear features and that the matrix is uniformly hostile. This is clearly not the case. The matrix itself, although often ignored, not only helps to determine the viability of individual habitat patches but can also act as a critical reservoir for biodiversity (Franklin, 1993). The patch-corridor-matrix concept has also been viewed as diversionary as it obscures the need to think directly about connectivity (Wiens, 2002). Clearly new approaches are needed in conservation that explicitly consider connectivity and move beyond simplistic structural classifications that rely solely on physical linkage.

Species interact, either in an active or passive manner, with landscape structure across a very wide range of scales, which has led some authors to scale structure to individual species (Wiens and Milne, 1989). Even when structural factors, such as soil type, are measured directly, it is the species response that determines the heterogeneity of a landscape and not necessarily what human beings are able to measure (Kolasa and Rollo, 1991). The extent to which structural features can be used as surrogate measurements for connectivity is unclear at the present time and requires further research (Tischendorf and Fahrig, 2000). In spite of this fact, assumptions, about how the structural configuration of different landscape elements, their resistance to the movement of individuals, as well as the relative permeability of their boundaries, need to be made if more functional perspectives are to inform conservation practice. Information is needed on the behavioural response of focal species groups (e.g., Lambeck, 1997) to enable conservation to move beyond the rare and threatened. This is not an intractable problem, as some recent applications of "least cost path" analysis, "key patch" approaches and percolation theory have shown (Verboom *et al.*, 2001; Chardon *et al.*, 2003; Opdam *et al.*, 2003; With, 2002).

Conservation perspective

Even though thinking has changed in ecology, the same cannot be said for conservation. The promotion of corridors that physically link habitats is still seen as a crucial response to the problems associated with fragmentation and even issues such as climate change (Preston, 2004). Given the continuing debate within ecology on the importance of physical linkage and the general failure of conservationists to consider ecological research when taking management decisions (Pullin *et al.*, 2004), this is perhaps not surprising. In spite of the uncertainty that exists, the concept has led to the apparent exclusion of more balanced views of connectivity in land use planning, as Weins (2002) had in fact predicted. There are some clear exceptions to this situation which have adopted strategies that deserve wider consideration (e.g., Watts, 2004; Woodland Trust, 2000). Physical linkage has not only become entrenched in conservation but also more widely in land management policies and even legislation as the extracts in table 1 show. In the hands of policy makers it can clearly lead to what Bowman (2003) called "politically dangerous terrains".

The intuitive appeal of wildlife corridors and ease with which their role can be communicated have made it a powerful metaphor. There are clear parallels with the prevalence of surrogate species approaches in conservation, as well as similar problems (e.g., Lindenmayer and Fischer, 2003). While wildlife corridors may provide a useful "social hook" to stimulate landscape restoration, practitioners must be clear about the objectives and goals that might be reasonably delivered. Greenways or green corridors perhaps provide a better metaphor that emphasises the wider benefits of such structures without the need to make explicit claims about their function as wildlife conduits (Hawkins and Selman, 2002). Given the "lack of a clear and consistent terminology" (Hess and Fischer, 2001), this may be difficult. Provided terms that imply movement are avoided then such an approach would perhaps provide a more honest and inclusive basis for wider public engagement.

Table 1: Corridors in land use planning and management. Selected quotations.

Malvern Hills AONB Draft Management Plan (2003) - "Safeguard existing links or corridors that connect priority habitats and sites ... create and restore corridors to aid the movement of species".

Tamar Valley AONB Draft Management Plan (2003) - "Building a pattern of wildlife corridors across the landscape".

Surrey Hills AONB Draft Management Plan (2003) - "The maintenance and enhancement of existing designated sites and their extension through the creation of landscape and habitat corridors".

Blackdown Hills AONB Draft Management Plan (2003) - "It is equally important to retain and reinforce the corridors and stepping stones between these areas (*semi-natural habitats*), which enable wildlife to move between them".

English Nature Annual Report (2002) - "They (*SSSI's*) need buffer zones to protect many of them and to be linked by wildlife corridors, through which species can move".

Article 10, Habitats Directive (92/43/EEC) - Member states will develop "...policies encouraging the management of features" such as "...linear and continuous structure(s)" or "...stepping stones" that are "...essential for the migration, dispersal and genetic exchange of wild species."

Results

It was against the background of these considerations that an internal seminar was held for English Nature staff on the 30th March 2004 on "Wildlife corridors and functional mosaics". English Nature is a Government funded body whose purpose is to promote the conservation of England's wildlife and natural features. It has responsibility for ensuring the effective management of a protected area network that covers about 7.5% of the total land area of England. This involves working closely with about 25,000 owners and occupiers as well as a range of other public and non-governmental organisations. We employ over 900 permanent staff located in a series of 22 Area Teams and 8 National Teams. The aim of the seminar was to provide staff from both national and local teams with a practical, practitioner-based perspective on the circumstances in which wildlife corridors might be important and to evaluate the importance of functional connectance for various species and habitats. This paper will provide a brief summary of the views of a range of national specialists and the results of a questionnaire that was given to the participants. Full proceedings will be published in due course.

The seminar began with a series of presentations from national specialists who are responsible for advising staff and external organisations on the scientific issues relating to conservation. Presentations covered the ecology of landscapes, urban environments, invertebrates, woodlands, reptiles and amphibians, as well as marine and freshwater environments. Physical linkages in the form of linear features were generally viewed as unimportant in promoting movement with the exception of freshwater habitats (Clarke, 2004). The main ecological benefits were seen as being related to the role that linkages played in providing breeding and foraging habitat, especially for invertebrates and herpetofauna (Foster, 2004; Shepherd 2004). Although often thought to be vital within urban areas, green mosaics within close proximity to areas of high density settlement were seen as being more important for wildlife delivery than linear wildlife corridors. This was due to the fact that clearer benefits for wildlife have been demonstrated from early exposure to urban greenspace (Knight, 2004). Connectivity remains an issue for some species in urban areas, however, as the significantly lower genetic distances of toads (*Bufo bufo*) indicates (Foster, 2004). Genetic distance is a standard metric that is used in population genetics to measures the amount of variation shared between populations. While woodland habitat networks were seen as important, no standardised corridor structure was thought to be suitable or desirable (Kirby, 2004). The consolidation of small woodland clusters and alteration of shape characteristics were generally viewed as more effective (Kirby, 2004). Unlike many other habitats, broadleaved woodland has not experienced fragmentation in recent years and cover has actually increased but the functional isolation of these areas continues to be an issue (Kirby, 2004). Connectivity was generally emphasised as a key issue, as was the need to move beyond simplistic views of landscapes (Catchpole, 2004).

The next aspect of the meeting that I want to consider relates to the answers that were given to two questions: 1.) What do you think about the role of corridors in conservation? 2.) What is your most important information need? The results for question one are shown in figure 2.

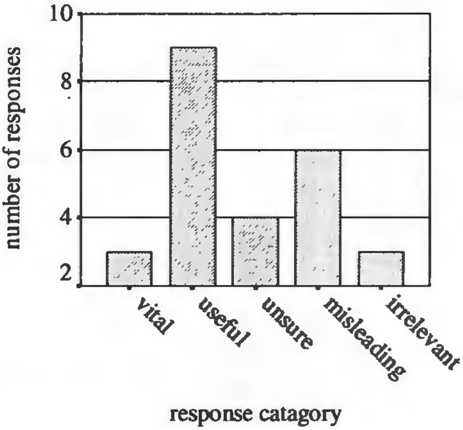

Figure 2: Role of corridors in conservation. Summary of practitioner responses.

In spite of the emphasis that was placed, by the speakers, on the need to take actions and develop strategies that directly considered connectivity, most of the delegates still felt that wildlife corridors had a useful role to play. The main reason for this appeared to be that they felt that it had become an effective approach around which strategies of public engagement have successfully been built; in other words a "social hook". Only four delegates believed their role was primarily related to the movement of species. Eleven delegates felt that they had other uses which were mainly centred round public engagement and habitat provision. A sizable proportion did, however, find corridors to be either misleading or irrelevant. This appeared to reflect the underlying ecological uncertainties about the importance of physical linkage as well as the need to think directly about functional issues. The results for the information needs were harder to summarise because of the highly varied responses that are shown in figure 3.

Many of the categories of information needs only corresponded to single individuals. The most common centred round clearer practitioner orientated guidance and information on the identification and management of functional mosaics. The next most common requirements were for better information on species autecology and case studies where more integrated, functional approaches had been successfully implemented. The demonstration of successful outcomes is a critical issue for conservation at larger scales. In a recent review of corridor implementation, Vos *et al.*, (2002) noted that not only had there been no evaluation of their effectiveness but also no cases were found that addressed fragmentation in a coherent manner. When large amounts of public money are being spent, this is a serious issue. The need for evidence on the effectiveness of corridors and alternative focal species approaches were also felt to be important by a number of delegates. All the other categories are self explanatory with the exception of 'time series data' which one delegate felt might be useful in determining the effectiveness of landscape-based conservation delivery.

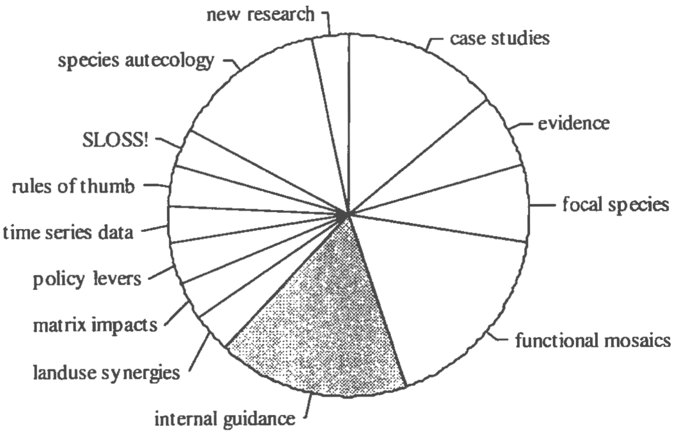

Figure 3: Information needs. Summary of practitioner responses. Hatched areas indicate categories where N = 5. N being number of respondents?

Summary

In conclusion, the need to move beyond simplistic structural definitions of landscapes when implementing land use policy was recognised as was the value of corridors as a 'social hook' on which wider conservation objectives could be hung. Careful use of different terms and clear objectives need to be associated with wildlife corridors. These need to be communicated to wider audiences, as well as more functional perspectives. The need for wider participation and engagement was seen as crucial in delivering effective outcomes. Greater knowledge of species autecology in relation to habitat preferences and dispersal behaviour were also important, as were more integrated and accessible ways of conceptualising and managing land/sea mosaics.

Acknowledgements

I would like to thank Jean Tither for running the literature search. I would also like to thank Richard Smithers for comments on an earlier draft of this manuscript and Steve Preston for supplying the AONB quotations.

References

Bennett, A.F. (2003). Linkages in the landscape. The role of corridors and connectivity in wildlife conservation. *Conserving forests and Ecosystems Series,* 1. IUCN

Beier, P. & Noss, R.F. (1998). Do habitat corridors provide connectivity? *Conservation Biology*, **12:** 1241-1252.

Bowman, D. (2000). Frontier landscape ecology and the conquest of *Terra Australis*. Plenary Address. Crossing frontiers: landscape ecology down under. IALE 2003 World Congress, Darwin, Australia.

Catchpole, R.D.J. (2004). Oral presentation to wildlife corridors and functional mosaics seminar. English Nature Science Series Seminar, 30th March 2004.

Chardon, J.P., Adriaensen, F. & Matthysen, E. (2003). Incorporating landscape elements into a connectivity measure: a case study for the speckled wood butterfly (*Pararge aegeria* L.). *Landscape Ecology*, **18:** 561-573.

Clarke, S. (2004). Oral presentation to wildlife corridors and functional mosaics seminar. English Nature Science Series Seminar, 30th March 2004.

Forman, R.T.T. & Godron, M. (1981). Patches and structural components for a landscape ecology. *Bioscience*, **31:** 733- 740.

Foster, J. (2004). Oral presentation to wildlife corridors and functional mosaics seminar. English Nature Science Series Seminar, 30th March 2004.

Franklin, J.F. (1993). Preserving biodiversity: species, ecosystems, or landscapes? *Ecological Applications*, **3:** 202-205.

Fowler, J. & Stiven, R. (2003). *Habitat networks for forests and people: the creation of sustainable forest habitats*. Forestry Commission Scotland & Scottish Natural Heritage.

Hawkins, V. & Selman, P. (2002). Landscape scale planning: exploring alternative scenarios. *Landscape & Urban Planning*, **60:** 211-224.

Hess, G.R. & Fischer, R.A. (2001). Communicating clearly about conservation corridors. *Landscape & Urban Planning*, **55:** 195-208.

Kirby, K. (2004). Oral presentation to wildlife corridors and functional mosaics seminar. English Nature Science Series Seminar, 30th March 2004.

Knight, D. (2004). Oral presentation to wildlife corridors and functional mosaics seminar. English Nature Science Series Seminar, 30th March 2004.

Lambeck, R.J. (1997). Focal species: a multi-species umbrella for nature conservation. *Conservation Biology*, **11:** 849-856.

Lindenmayer, D.B. & Fischer, J. (2003). Sound science or social hook – a response to Brooker's application of the focal species approach. *Landscape & Urban Planning*, **62:** 119-180.

Merriam, G. (1984). Connectivity: a fundamental ecological characteristic of landscape pattern. J. Brandt & P. Agger (Eds), *Proceedings of the First International Seminar on Methodology in Landscape Ecological Research and Planning*, Roskilde University, 5-15.

Opdam, P., Verboom, J. & Pouwels, R. (2003). Landscape cohesion: an index for the conservation potential of landscapes for biodiversity. *Landscape Ecology*, **18:** 113-126.

Preston, S. (2004). Personnel communication.

Pullin, A. S., Knight, T.M., Stone, D.A. & Charman, K. (2004). Do conservation managers use scientific evidence to support their decision making? *Biological Conservation*, in press.

Sheppard, D. (2004). Oral presentation to wildlife corridors and functional mosaics seminar. English Nature Science Series Seminar, 30th March 2004.

Simberloff, D. & Cox, J. (1987). Consequences and costs of conservation corridors. *Conservation Biology*, **1:** 63-71.

Simberloff, D., Farr, J.A., Cox,J. & Mehlman, D.W. (1992). Movement corridors: conservation bargains or poor investments? *Conservation Biology*, **6:** 493-504.

Tischendorf, L. & Fahrig, L. (2000). On the usage and measurement of landscape connectivity. *Oikos*, **90:** 7-19.

Verboom, J., Foppen, R., Chardon, P., Opdam, P. & Luttikhuizen, P. (2001). Introducing the key patch approach for habitat networks with persistent populations: an example for marshland birds. *Biological Conservation*, **100:** 89-101.

Warming, E. (1909). *Ecology of plants: an introduction to the study of plant communities.* Clarendon Press.

Watts, K. (2004). Personnel communication.

Wiens, J.A. (2002). Central concepts and issues in landscape ecology. K.J. Gutzwiller (Ed), *Applying landscape ecology in biological conservation*, Springer-Verlag, 3-21.

Wiens, J.A. & Milne, B.T. (1989). Scaling of 'landscapes' in landscape ecology, or, landscape ecology from a beetle's perspective. *Landscape Ecology*, **3:** 87-96.

With, K. A. (2002). Using percolation theory to assess landscape connectivity and the effects of habitat fragmentation. K.J. Gutzwiller (Ed), *Applying Landscape Ecology in Biological Conservation*, Springer-Verlag, 105-130.

Woodland Trust (2000). Woodland biodiversity: expanding our horizons. Woodland Trust, Grantham. www.woodland-trust.org.uk/publications/publications.htm

Vos, C., Baveco, H. & Grashof-Bokdam, C.J. (2002). Corridors and species dispersal. K.J. Gutzwiller (Ed), *Applying landscape ecology in biological conservation*, Springer-Verlag, 84-104.

Balancing site-based protection versus landscape-scale measures in English woodland

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Abstract

Woodland conservation in England is based on a mixture of site-protection and broader landscape-scale measures. The balance between these two has varied over time, but there are currently efforts to work more at the broader scale. The principles of landscape ecology, including species-area relationships, core-edge comparisons, the benefits of building on existing habitats, suggest a framework for how landscape-scale conservation might operate. However the heterogeneous nature of real sites and processes limit their use in predicting the outcome of conservation actions. The agencies need to be aware of developments in landscape ecology, but beware of applying generalised solutions to particular circumstances.

Introduction

Woodland conservation in England since 1949 has been a mixture of special-site protection (largely based on the system of Sites of Special Scientific Interest (SSSIs)) and wider landscape protection measures, delivered via land-use policies and incentives. Initially protected sites took priority because the Nature Conservancy, subsequently Nature Conservancy Council (NCC), had little influence over agricultural, forestry, or development policies that were leading to major habitat and species losses in the countryside (NCC, 1984; Kirby, 2003). In addition the agencies have limited resources; if there is pressure to increase the size of the protected site series, or to increase the protection such sites receive, for example through changes in the legislation, then less effort can be put into landscape-scale work. This is particularly relevant to the conservation of ancient woodland, where only a relatively small percentage (c.15% by area) is within the SSSI series. However, since the late 1980s, and particularly since the introduction of the UK Biodiversity Action Plan, the potential for trying to operate more at the landscape scale has increased.

Interest in landscape-scale conservation has long featured in both NCC's and English Nature's research programmes, including studies on species-area relationships, habitat fragmentation and the value or not of corridors for local movement or to allow for species migration in response to climate change (Dawson, 1994; Kirby, 1995; Peterken & Game, 1984). Following from these studies a number of landscape-level projects have been undertaken to look at the practicalities of restoring habitats in a targeted fashion.

While there is a reasonably well-developed rationale behind the programmes for selecting and subsequently protecting special sites (NCC, 1989) there is far less agreement as to the best ways to promote woodland conservation at a landscape scale; see for example the debates about corridors (Dawson, 1994). Ideas from landscape ecology have influenced our activities, but have not always proved as useful in practice as might have been hoped.

Three examples illustrate some of the potential complexities of translating what may be sound theories into action on the ground:

a) the use of species-area relationships in developing conservation priorities;
b) ideas on applying the core-edge concept to British woodland;
c) a case for adding new woodland to small rather than to large existing woods.

Species-area relationships

An early proposal was that one large site was better as a reserve than several small ones of the same total extent. Game and Peterken (1984) showed that this was not necessarily the case for woodland vascular plants: sites are not uniform in their characteristics apart from size and isolation; large sites do not necessarily contain all the species contained in smaller woods (Rackham, 2003). More recently Peterken and Francis (1999) suggest that, for species associated with open space, there may be step changes in the relationship between numbers of plant species and woodland area. These step changes might occur where woods first become big enough to contain internal glades (c.3 ha) and when they are big enough that they are likely to be managed to create structural diversity (c.20-30 ha).

The data from another survey organised by George Peterken (Goodfellow & Peterken, 1981) shows a highly significant log-log relationship between the number of native woodland species (shade-bearers, shade-casters, wood-edge species), adjusted to allow for possible under-surveying of some woods, and woodland area for seventy Norfolk woods ($p<0.001$; r-squared = 60%). However if only the 37 woods between 5 and 30 ha are considered the log-log relationship, though still significant, shows a much greater scatter; r-squared drops to just 13%. The species-area relationship confirms that large woods (in a Norfolk context) are species-rich, very small woods poor. However it is of limited use in discriminating between sites in the intermediate size categories where it would be most useful to have such a guide (Figure 1).

Core-edge comparisons

Large sites have a lower proportion of 'edge', which may be important where there are interior species that either do not occur, or do less well close to edges (Forman, 1995). Habitat patches are often illustrated with an edge zone distinct from a central 'core' and it is assumed that the greatest conservation benefits will arise from increasing the size of this core (e.g., Woodland Trust, 2002). However the assumptions behind this conclusion may not always apply, including:

- That there are distinct core species, that are more valuable than edge species;
- That edge effects are negative.

A further common assumption is that making sites more compact is the most effective way of increasing the core area, yet all of these assumptions can be questioned.

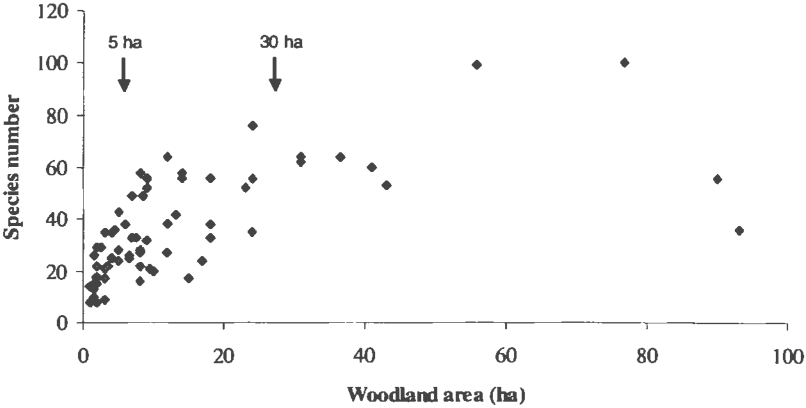


Figure 1. Woodland plant species number (adjusted for under-recording) versus woodland area for seventy Norfolk woods (Goodfellow & Peterken, 1981). When transformed to log scales a highly significant linear relationship is produced: $\text{Log}_{10}\text{Species} = 1.12 + 0.406\ \text{Log}_{10}\text{Area}$.

Do core species exist in British woods?

Core species are assumed to be those that require a large area of continuous habitat, although what counts as a 'large area' for a beetle may be very different from what would be needed for a woodland bird or large mammal. However are 'core woodland species' likely to be a significant part of the current British fauna and flora? Most ancient woods are less than 20 ha, and have been managed for centuries leading to a high degree of regular internal disturbance. If species that required large areas of continuous woodland were present in the past they would now be extremely rare, or, more likely, extinct. If Vera's model of a mosaic of groves, grassland and scrub (Vera 2000) is taken as the paradigm for what our forests should be like, then should conservation priority be given to edge species, rather than to core species anyway?

Edges - good or bad?

Whether a particular edge type is considered better or worse than another depends on the species/species-group and the edge processes that are being considered (Table 1). There may be edge effects (halos around the wood) from the woodland on to the adjacent habitat: e.g. increased predation risk from tree-nesting corvids on breeding birds of wet grassland, or the increased tree invasion of heathland from adjacent woods. Positive effects, for example trees providing nesting sites for some farmland birds, or increased shelter for invertebrates using grassland next to the woodland edge also occur.

Table 1. The influence of different types of habitats on processes at a woodland edge.

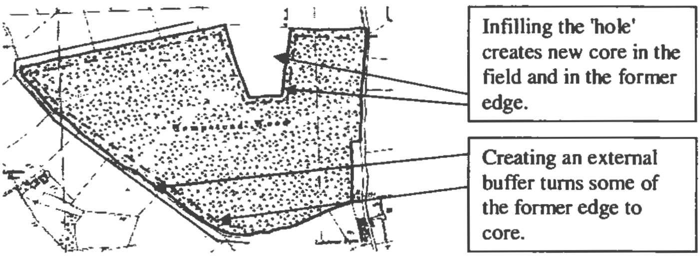
Possible adverse effect arising from different habitats immediately next to an ancient (ungrazed) oak woodland edge, indicated as negative (-) or very negative (--). The adjacent habitats are: 1 intensive arable; 2 'organically farmed' arable; 3 western hemlock plantation; 4 plantation of oak of Hungarian origin; 5 semi-natural pasture; 6 suburban garden.

Process potentially affecting edge	**1**	**2**	**3**	**4**	**5**	**6**
Spray drift (fertilizer, pesticide)	--	-				-?
Blocking of light to edge shrubs and flowers			--	-		-
Drying out of edge strip through air movement	--	--			--	-
Risk of stock damaging the wood-edge by grazing					--	
Cross-fertilisation with non-local oak pollen				--		
Increased predation on edge-nesting birds by cats						--
Invasion of oak woodland by self-seeding conifers			--			

At any junction between two habitats both positive and negative edge effects may be operating on the woodland, while simultaneously different edge effects operate on the non-woodland component. One of these, for example spray drift, may be so strong compared to the others that it rightly determines the necessary action. However the influences may be more evenly balanced: for example, in Table 1, how important is it to reduce cross-fertilisation by non-local oak pollen compared to the increased drying out of the wood-edge that will result?

Increasing the area not affected by edge-effects - the example of Hempstead Wood, Essex

Hempstead Wood, Essex is a large ancient wood in an intensive agricultural setting (Figure 2). If a fixed area of new woodland is to be added, what are the benefits of filling in the hole in the northern edge, versus creating a narrow buffer strip of woodland round the outside?



Infilling the 'hole' creates new core in the field and in the former edge.

Creating an external buffer turns some of the former edge to core.

Figure 2. What is the best way to increase the area not affected by edge effects? (Copyright Ordnance Survey).

Table 2. Area of new 'core' woodland created for different assumptions about the width of edge effect. (All figures rounded to nearest 0.1 ha)

		Width of edge effect		
Total woodland area 69.6 ha		5 m	12.5 m	25 m
Existing core woodland (ha)		67.6	64.3	59.6
Filling in hole: new core created	From former edge	0.3	0.8	1.6
	On former field	5.2	5.0	4.7
12.5 m buffer strip: new core created	From former edge	2.1	5.1	5.1
	On former field	3.0	0	0

Filling in the hole (5.3 ha) creates a small new area of 'core' woodland on former field, plus some areas of former edge around the hole that are no longer edge. Almost exactly the same area (5.1 ha) would allow the creation of a 12.5 metre-wide strip of woodland round the edge of the wood (perimeter 4.1 km); this then results in an equivalent-width strip within the wood no longer being edge-affected. Table 2 compares the outcomes for different assumptions about the width of the edge effect.

The option of filling in the hole is likely to be the least costly. However for these widths of assumed edge effect, it is ecologically less effective than putting the new planting round the edge of the wood as a buffer. With the buffer strip approach a much higher proportion of the potential new core is former edge, i.e. existing woodland rather than former field, and hence is more likely to have soils suitable for woodland species to colonise. Reducing edge effects by new buffer planting will often be more effective than trying to make woods more compact.

All the potential new core is within 25 m of existing 'core' woodland, depending on the width of the edge effect. If estimates of short-distance spread of ancient woodland species are taken to be about 0.3 and 1.0 m per year (Brunet and Oheimb, 1998; Rackham, 2003) potentially all the new core woodland could be colonised within 75 years. By contrast, if the infill approach is taken, the centre of the former field is 100 m from the edge of the existing woodland: it could therefore take up to 375 years for ancient woodland plants to spread throughout the new core.

Caution must however be applied to using such colonisation rates for prediction, both here and in other studies, since they are very much less than the rates of spread that must have applied in post-glacial times: a rate of 1m a year represents a maximum spread of only10 km in 10,000 years! Either ancient woodland plants have evolved to be more sedentary, or a major dispersal agent has been missing for the last 500 years at least, or, and this seems most likely, short-term, and necessarily short distance, studies of spread miss rare long-distance colonisation of new sites.

New small woods are not a waste of time!

The main new planting of broadleaved woodland in recent years has been as small patches (<2 ha) on farmland. Isolated small blocks have been criticised as not being an effective way of meeting any of the government objectives for forestry, although in time they may come to substitute for hedges and clumps of trees lost during recent periods of agricultural intensification (Kirby, Buckley & Good,1999). There are likely to be higher conservation benefits if new woodland is created next to existing sites, but should this be by building on already large woods, or by making small woods bigger?

The relationship developed for Figure 1 is used in Table 3 to explore the predicted consequence of adding new woodland to existing sites in terms of the number of species they might eventually support (ignoring colonisation issues). The predicted number of species for the seventy woods of different sizes was used to give baseline values (to eliminate the variation due to other factors such as soil type). A ten percent increase in total woodland area (100 ha) was then simulated. In scenario 1, ten hectares was added to each of the ten biggest woods; in scenario 2, two hectares was added to the fifteen smallest woods (2 ha or less) and five hectares to the next fourteen smallest (5 ha or less). The new species-richness values were calculated for each wood. Under scenario 1, the mean increase for the ten biggest woods was 5.4 species (range 4-7); under scenario 2, the mean increase for the twenty-nine woods that benefited was 7.8 species (range 6-11).

These scenarios assume that perfect colonisation of the new woodland is possible and do not take account of which species might be expected to benefit from new area added to large woods rather than to small ones. Nevertheless the results suggest that the option of adding new woodland to existing small woods may be worth further consideration. Buckley and Fraser (1998) used a rule of adding new woodland to existing small sites for one of their scenarios for targeting new woodland creation. It led to the emergence of a potential series of woodland 'stepping stone' patches in their Rockingham Forest study area. More generally Drechsler and Watzold (2001) show that whether it is more efficient to add to areas where a habitat is already abundant or to where it is scarce depends on the relative shapes of the cost-benefit functions.

Table 3. Is species-richness best served by increasing the size of large or small woods?

No of woods	Mean area (ha)	Range (ha)	Area added per wood	Initial species no: Mean	Initial species no: Range	Increased species no: Mean	Increased species no: Range
1. Add 100 ha to largest woods							
10	52.3	24-93	10	64	48-83	5.4	4-7
2. Add 100 ha to smallest woods							
15	1.7	1-2	2	16	13-17	6.3	6-8
14	3.8	2.5-5	5	22	19-25	9.5	9-11

Conclusions

The changing climate, whether physical, political or institutional, provides opportunities for the conservation agencies to make more of an impact on conservation at a landscape-scale than in the past. As such opportunities allow, English Nature will strive to promote conservation outside of the SSSIs. However, within a fixed resource base, this increase will only be achievable if the pressures to work on the protected site series do not also increase.

Landscape ecology has produced valuable ideas that have helped considerably with the explanation of why species and assemblages occur where they do. However the degree of variation that exists across sites has made some of the theoretical assumptions appear too simplistic to be useful yet in a quantitatively predictive, and hence prescriptive, way. The relative costs and benefits of different options need to be part of analyses, including the potential value of 'owner goodwill'. If 80% of benefits can be achieved for 20% of costs, because this is acceptable and supported by the owner (whereas going further would lose that good-will); then there are funds available for use on other sites. The agencies need to be aware of developments in landscape ecology, but beware of applying generalised solutions to particular circumstances.

References

Buckley, G.P. & Fraser, S. (1998) *Locating new lowland woods.* English Nature, Peterborough (Research Report 283).

Dawson, D. (1994) *Are habitat corridors conduits for animals and plants in a fragmented landscape?* English Nature, Peterborough (Research Report 94).

Drechsler, M. & Watzold, F. (2001) The importance of economic costs in the development of spatial conservation management. *Biological Conservation* **97**: 51-59.

Forman, R.T.T. (1995) *Land mosaics.* Cambridge University Press, Cambridge.

Game, M. & Peterken, G.F. (1984) Nature reserve selection strategies in the woodlands of central Lincolnshire, England. *Biological Conservation* **29**: 157-181.

Goodfellow, S. & Peterken, G.F. (1981) A method for survey and assessment of woodlands for nature conservation using maps and species lists: the example of Norfolk woodlands. *Biological Conservation* **21**: 177-195.

Kirby, K.J. (1995) *Rebuilding the English countryside.* English Nature, Peterborough (English Nature Science Report 10).

Kirby, K.J. (2003) Woodland conservation in privately-owned cultural landscapes: the English experience. *Environmental Science and Policy* **6**: 253-259.

Kirby, K.J., Buckley, G.P. & Good, J.E.G. (1999) Maximising the value of new farm woodland biodiversity at the landscape scale. P.J.Burgess, E.D.R.Brierley, J.Morris & J.Evans (Eds). *Farm woodlands for the future.* Bios Publishers, Silsoe, 45-55.
NCC (1984) *Nature conservation in Great Britain.* Nature Conservancy Council, Peterborough.
NCC (1989) *Guidelines for the selection of biological sites of special scientific interest.* Nature Conservancy Council, Peterborough.
Peterken, G.F. & Francis, J. (1999) Open spaces as habitats for vascular ground flora species in the woods of central Lincolnshire. *Biological Conservation* 91: 55-72.
Peterken, G.F. & Game, M. (1984) Historical factors affecting the number and distribution of vascular plant species in the woodlands of central Lincolnshire. *Journal of Ecology* 72:155-182.
Rackham, O. (2003) *Ancient woodland (revised edition).* Castlepoint Press, Dalbeattie.
Woodland Trust (2002) *Space for nature: landscape-scale action for woodland biodiversity.* Woodland Trust, Grantham.
Vera, F.W.M. (2000) *Grazing ecology and forest history.* CABI, Wallingford.

Session 7

Landscape-scale action

Achieving landscape-scale change: the National Forest, a case study

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Abstract

The National Forest is creating a vast, new woodland landscape for the nation covering some 200 square miles of the English Midlands.

This paper describes the ambitious scale of landscape and ecological transformation that is planned and already underway. It also describes the approaches being adopted by the National Forest Company to manage this major undertaking.

Two case studies from the environmentally blighted Leicestershire and South Derbyshire coalfield and the rural farmland landscapes of the Mease and Sence lowlands, demonstrate the extensive environmental and socio-economic transformation that the Forest has achieved, in a little over 10 years.

The paper concludes by highlighting the important quality of life and sustainable development benefits that are being met through the Forest's landscape-scale ecological change.

What is the National Forest?

The National Forest aims to create a vast new forested landscape for the nation covering some 200 square miles of the English Midlands. Spanning parts of Staffordshire, Derbyshire and Leicestershire, the Forest area is rich in historic, ecological and cultural interest and reflected in a varied landscape. The new Forest aims to link the remnant ancient forests of Needwood and Charnwood and includes a substantial area of the former Leicestershire and South Derbyshire coalfield, the industrialised Trent Valley corridor and a number of towns and other settlements.

It will be a genuinely multi-purpose Forest that will transform the landscape, the environment and economy of the area. The Forest will create a new resource for recreation and tourism, create rich new wildlife habitats, restore damaged landscapes and offer alternative, productive uses of farmland. Woodland will also be grown to produce high quality timber and be used as a spur to broader economic regeneration of the area.

What are the Forest's origins?

The National Forest was conceived by the then Countryside Commission in 1987. The inspiring idea caught the public imagination and in 1991 the Countryside Commission established a development team to draw up a strategy for its creation. This strategy was endorsed by Government and in 1995 a new, independent public company, the National Forest Company (NFC), was set up to spearhead the Forest's creation. The NFC is sponsored by the Department of Environment, Food and Rural Affairs and works in partnership with the public, private and voluntary sectors, landowners and local communities to coordinate the Forest's creation.

Achieving landscape-scale change

From its original 6% (3,010 ha) starting point the aim is that trees and woodlands will ultimately cover around a third of the area (16,560 ha). The Forest is already becoming increasingly evident in the landscape, as wooded cover has more than doubled to 15% (7,640 ha) in its first 10 years.

However, the Forest is about far more than trees and is also creating a mosaic of other wildlife habitats that reflect the landscape and ecological character of the area. Already, over 400 ha of non-wooded habitats have been created and over the life of the Forest around 1,000 ha will be added to the existing ecological resource. This will include both large-scale habitat creation, as well as the incorporation of smaller non-wooded habitats within new woodland schemes.

This knitting together over time of some 19,000 ha of new and existing woodlands, hedgerows, meadows, heathlands and wetlands will create landscape-scale biodiversity change across the area. It will also achieve Forest-wide habitat connectivity by linking areas of existing high quality habitat with fragmented wildlife sites through woodland and other habitat creation.

This combined activity will significantly reduce the overall fragmentation of habitats, increase the viability of isolated wildlife sites and enable wildlife species to thrive and disperse across the Forest's landscape. Over time the Forest will attract and support thousands of more common wildlife species as well as many declining and some rare species. In so doing it will also significantly help to redress the loss of habitats that the area has faced in the past.

Planning landscape-scale change

This scale of landscape and ecological change is being guided by several important mechanisms:

- the National Forest Strategy, 2004-14.
- the National Forest landscape character assessment.
- an Indicative Planting Strategy.
- the National Forest Biodiversity Action Plan.
- the National Forest broadleaved planting and local provenance policies.

The National Forest Strategy

The Forest Strategy sets out the overall, long term, vision for the Forest and how all its various land uses will fit together. It is key to ensuring that the Forest's landscape and ecological objectives are well balanced, integrated and achievable, alongside the equally important socio-economic objectives for the Forest. The strategy sets out the overall targets for the Forest in terms of woodland and habitat creation and how these will be achieved.

Landscape Character Assessment.

Fundamental to creating the Forest is understanding the area's landscape evolution, its physical, historic, ecological and current land use influences and the character of its landscapes. This was achieved through undertaking a landscape assessment in 1994. This assessment was updated in 2004 in response to the scale of woodland and other habitat creation, which has already dramatically changed parts of the Forest landscape.

The landscape character assessment identifies six landscape character areas, which are individually distinctive and form parts of wider landscapes that extend beyond the Forest boundary; and six landscape types, which are defined to a greater or lesser extent by woodland, agricultural, urban and industrial influences.

Taken together the landscape character areas and types describe the subtle differences in landscape that occur across the Forest area. The landscape types guide the type, scale and design of woodland planting suited to different parts of the Forest. This is essential to ensure that new planting matches the scale of different landscapes and fits with their character. The NFC promotes planting design guidance for each landscape type as a guide for landowners when they draw up new woodland schemes.

The landscape character areas and types are also important in helping to guide other types of habitat creation suited to different parts of the Forest.

Indicative Planting Areas

The NFC has developed a strategic approach to woodland creation by identifying Indicative Planting Areas (IPA). These have been developed from a strategic assessment of the area's planting potential and how new planting should best relate to landscape character.

An IPA map has been produced that classifies the Forest into nine types of area that are suited to different types and scales of planting. These include:

- woodland expansion areas; which have the best potential for larger-scale planting, to match their large-scale landscape character
- woodland linkage areas; for planting to connect existing woodlands
- well-wooded landscapes; with scope for only small-scale, strategic planting that links existing woodlands
- floodplain landscapes; for small scale tree planting and wet woodland creation
- historic enclosures; for small-scale planting and hedgerow trees
- urban /urban fringe; a focus for community woods and urban forestry
- transport corridors and gateways; for roadside and visual impact planting
- ancient woodland priority areas; for extending ancient woodlands
- parklands; to target restoration and reinstatement of parkland.

This provides a strategic framework to relate new planting to landscape potential, with the aim of achieving a spread of woodland cover across the area and avoiding too many trees being planted in certain areas.

Biodiversity Action Plan

A Biodiversity Action Plan (BAP) for the Forest area was produced by the NFC and conservation partners in 1998 and has been updated in 2004. This focuses biodiversity activity on 13 of the 38 UK BAP's nationally important habitats, 7 habitats of local importance and several key wildlife species. Table 1 highlights the Forest's targets for these habitats and species and shows that substantial progress is being made. Especially notable are gains that have been made against nationally important habitats (e.g., hay meadows, heathland and hedgerows) and the fact that otters have returned to rivers in the area.

The Forest BAP is currently being updated and will include a 'BAP map' to provide a more geographically focused approach towards habitat management, creation and species enhancement works. This will complement the Forest's Indicative Planting Areas and provide a strategic, Forest-wide, framework for integrating woodland creation, other habitat creation and the management of existing habitats. It will also help to maximise landscape and ecological character by concentrating activity on habitats that are characteristic of certain areas (e.g., heathlands in Charnwood, or wetlands in the Trent Valley) and by giving priority to extending existing habitats and linking fragmented ones.

Broadleaved planting and local provenance

Two other important aspects of NFC policy are to achieve an essentially broadleaved forest and to encourage the use of local provenance planting stock. Of the 5.5 million trees so far planted the current planting split is 86% broadleaves to 14% conifer. Since 2002 the NFC has also adopted an annually monitored local provenance policy, linked to its grant schemes. A local provenance project has also been established with Charnwood Borough Council to make effective links between seed collectors, tree nurseries and landowners planting trees. Since 2002, over 97% of the trees planted in the Forest have been from Midlands seed zones 402 and 403, or from UK sources. This emphasis on broadleaved planting and using local provenance stock is helping to maximise the ecological benefits of the Forest's landscape-scale change.

Case studies in landscape-scale change

So, how is the National Forest's approach manifesting itself 'on the ground'? Well ten years into the making and the Forest jigsaw is well and truly coming together. A new woodland landscape is unifying the Forest area and meeting the original aim of linking the remnant ancient Forest landscapes of Needwood and Charnwood.

This change is taking place through a variety of means, notably the National Forest Tender Scheme; land acquisition (by the NFC, Forestry Commission, the Woodland Trust, local authorities and Wildlife Trusts); restoration of mineral worked and derelict land; agri-environment schemes; development-related planting; and urban forestry.

Landscape change has been most extensive in the Leicestershire and South Derbyshire Coalfield and the farmland landscapes of the Mease and Sence Lowlands. The following two case studies are a microcosm of the widespread change that is planned and underway across the breadth of the Forest area. They give a feel for the scale, pace and type of landscape transformation that has been achieved in a little over 10 years.

Coalfield dereliction to environmental showcase

The Ashby Woulds, covering around 6 square miles, is at the heart of the National Forest area. In the early 1990s it had the single most intensive concentration of coal and clay workings and derelict land in the Leicestershire and South Derbyshire coalfield , some 250 ha.

In the space of a decade the Ashby Woulds has been transformed from being a blighted place with black dereliction into a green environment that is now the main focal point of the Forest as a visitor destination. This dramatic change has been achieved with:

- the creation of CONKERS; a £16m new woodland Discovery Centre, attracting over 250,000 visitors a year.
- the reclamation and planting of over 300 ha of land to new woodland sites, with associated wetland and grassland habitats.
- bringing fragmented wildlife sites back into management and creating strategic links with new woodlands.
- the re-watering of 2km of the derelict Ashby Canal.
- the creation of widespread new public access, with networks of forest trails and cycleways linking key sites, plus all-abilities routes.
- the renovation of the historic Moira Furnace as a visitor attraction.
- and economic regeneration through new housing and business development alongside tourism and leisure development.

The Ashby Woulds is now an area with renewed community spirit and pride where people want to live and visitors want to come to experience the new Forest environment and its related attractions. This transformation was formally recognised in 2002 when the area won the national Royal Town Planning Institute Silver Jubilee Cup for Planning Achievement.

Farmland landscapes of the Mease and Sence lowlands

The open, mixed farmland landscapes of the Mease and Sence Lowlands in South Derbyshire were badly hit in the 1970s by Dutch Elm disease (caused by *Ceratocystis ulmi*). Biodiversity interest was limited and confined to hedgerows, scattered hedgerow trees, remnant but unmanaged field ponds, several small woods and one large ancient woodland (80 ha).

Over the last 10 years there has been a major conversion of farmland to forestry and other habitats as farmers and organisations such as the Woodland Trust have taken advantage of the National Forest Tender Scheme. Both the Woodland Trust and Forestry Commission have also purchased land for woodland creation.

Over a significant part of this area (around 9 square miles) there has been a particular concentration of Forest creation activity. Over 360 ha of new woodland has been added to the landscape, including 140 ha strategically linked to the remnant ancient woodland at Grangewood. In addition a whole host of other habitat creation and enhancement works are underway including:

- 6 ha of new wildflower meadows and over 60ha of rough grassland have been created.
- 12 field ponds have been brought back into management; plus 7 new water bodies and seasonal wetlands have been created.
- extensive hedgerow planting and management.
- around 10 ha of existing woodland has been brought into management.
- special new woodland biodiversity projects including, natural regeneration adjoining ancient woodland; creation of new wet woodland; plus new areas planted as hazel coppice with standards.
- BAP species enhancement works including black poplar (*Populus nigra*) planting, sowing of bluebells in new woodlands, inclusion of bat boxes, plus wetland works for ruddy darter dragonfly (*Sympetrum sanguineum*).
- development of an extensive trails network linking key sites, all of which offer new open public access.
- development of new farm-based leisure pursuits including fishing, cross country and carriage driving courses, plus new tea rooms.

This is now an environmentally diverse and thriving farmed landscape. It demonstrates the benefits of an environmentally focused approach to agricultural diversification over a wide area for; recreation, tourism, wildlife and landscape enhancement.

A Forest for the nation and national exemplar

The National Forest's creation is transforming the environment, the landscape and economy of a great swathe of the English Midlands, and is significantly improving the quality of life of local residents and visitors to the area. It is sensitively creating a vast new landscape for this and future generations to enjoy and is re-connecting people with nature by creating new, accessible, green space close to where they live, work and spend their leisure time.

An ecologically richer landscape is contributing to physical and spiritual well-being. It is also important for tourism, as visitor surveys have highlighted nature and wildlife as one of the main reasons why people are coming to visit the National Forest. Increasingly the Forest is also becoming known as a place to experience and learn about the natural world – with Forest visitor centres already attracting over 36,000 educational visits per year.

The National Forest is recognised by Government as a national exemplar of sustainable development. It is contributing to some 33 sustainable development indicators, related to national programmes which are addressing and monitoring landscape and biodiversity change, economic development and people's quality of life. The National Forest model of achieving landscape-scale ecological change offers many lessons from which others can learn, both in this country and worldwide.

Table 1. National Forest BAP targets and progress

Key habitats in the National Forest	National Forest BAP restoration/creation targets	Summary of achievements 1998 – 2004		
Wet woodland	Create 50 ha by 2010	12.23 ha	24%	
Lowland wood pasture and parkland	Restore 100 ha by 2010	93 ha	93%	
Lowland hay meadows	Create 50 ha by 2010	230.38 ha created	460.76%	*TARGET EXCEEDED*
Lowland calcareous grassland	Create 10ha by 2005	nil		
Lowland dry acid grassland	Create 50 ha by 2010	21.8 ha	43.6%	
Lowland heathland	Restore / recreate 100 ha by 2010	42.24 ha created/managed	42.24%	
Floodplain grazing marsh/wet grassland	Re-create 50ha by 2010	24.55 ha created and grazing introduced to 42 ha=68.34 ha	136.68%	*TARGET EXCEEDED*
Reedbeds	Create 50 ha by 2010 (in 2 large blocks)	3 ha created. Long term plans for over 20 ha in Trent Valley as part of mineral restoration	6%	
Mesotrophic lakes	Maintain rare plant and animal communities	Management ongoing	100%	
Eutrophic open waters (including field ponds)	Restore 2 water bodies each year until 2010 (24 over 12 years) + create one new body of 1ha each year to 2010 (12 over 12 years) + create 5 new ponds each year until 2010 (60 over 12 years)	78 restored 6 over 1 ha created 70 new ponds and c12 ha scrapes	325% 50% 117%	*TARGET EXCEEDED* *TARGET EXCEEDED*
Ancient or species-rich hedgerows	Restore 50km pa + plant 3 km pa to 2010 (over 12 yrs = 600 km restored; 36 km planted)	59.11 km restored 54.73 km planted	9.85% 152%	 *TARGET EXCEEDED*
Field margins	Establish 2-6m margins on 10% farms by 2010 (i.e., 66 farms) Encourage 6m margins on all watercourses	11. 57 km established on 9 farms 0.47 km of buffer strip created Wet grassland around ponds in many schemes	13.6%	
Ancient semi-natural woodland	Extend 5 sites by 2010	11 sites extended	220%	*TARGET EXCEEDED*
Plantation/secondary woodland	Plant 400 ha of trees per annum to 2010	2174.93 ha	45%	
Post industrial sites, parks, gardens, buildings	Initiate 2 urban wildlife projects per annum i.e. 24 over 12 years	6 initiated Many schools projects and BTCV work	25%	
Roadside verges	Encourage designation of roadside nature reserves – 5 per annum until 2010 i.e., 60 over 12 years	17 designated	28.3%	

National Forest BAP species	Species targets	Summary of achievements 1998-20 04
Noctule bat *(Nyactalus noctula)*	Maintain existing populations and increase distribution. Establish a Forest wide bat box campaign by 2002	Woodland management ongoing; bat boxes installed Forest wide
Otter *(Lutra lutra)*	Maintain and expand the otter population along major watercourses by 2010. Improve water quality of the Rivers: Trent, Tame, Sence and Mease and their tributaries. Provide suitable otter habitat along 80% of the watercourses by 2010	10 new holts; streamside management
Redstart *(Phoenicurus phoenicurus)*	Increase the population to 30 breeding pairs by 2010	Boxes installed across Forest; 500 ha of woodland management
Adder *(Vipera berus)*	Determine the range of the species and centres of population by 2002 Increase the number of sites and populations. Promote the adder to the public to raise awareness of the adder's needs	4 new hibernacula; habitat management; new sightings
Ruddy darter dragonfly *(Sympetrum sanguineum)*	Increase the number, size and range of populations	Pond and scrape creation/ management; ditch and streamside management
Bluebell *(Hyacinthoides non-scriptus)*	Maintain and extend the distribution of the species and protect key sites	61 new sites; 2 extended
Black poplar *(Populus nigra)*	Halt the loss of and damage to existing trees. Increase the population particularly in areas where the species is native e.g., Trent Valley. Source stock of appropriate provenance for distribution and planting	43 new sites; cuttings taken; database of all sites; Derbyshire Wildlife Trust survey across Derbyshire

Putting theory into practice

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Abstract

As Britain's largest land manager, the Forestry Commission is a leader in restoration and management at a landscape-scale. The paper gives practical examples over the last 25 years including: the restructuring of Kielder Forest; heathland re-creation in Dorset; landscape-scale planning and restoration in the New Forest; conservation management in the western Cairngorms; and restoration of ancient woodland in England. It draws from Dutch and Polish models to look into the future, identifying socially driven low-intensity land management as the biggest opportunity for new landscape-scale conservation development. It identifies, as the key drivers to practical action: vision; scale; natural processes; people and partnerships.

Restructuring Kielder

Twenty-five years ago in England's largest forest the Forestry Commission developed restructuring and set in train events leading to some of Britain's most spectacular landscape-scale conservation. The first rotation, the results of the re-afforestation that dominated the Forestry Commissions first half-century, was coming to an end as large scale felling started. The Forestry Commission was faced with a huge, even-aged forest dominated by sitka spruce (*Picea sitchensis*) plantation. Simply felling as trees matured would 'roll up the carpet', clearing huge areas and re-creating the same problems. After toying with detailed landscape plans for the most prominent landscapes, managers realised that a more radical approach was needed and had the vision to see that having created the landscape, they could change it.

Instead of felling all at once, restructuring broke up the ages of the future forest by bringing forward some felling and delaying others. Endemic windthrow, where stands could be expected to blow over once they passed a certain top height, complicated the process. Roads, watercourses and other breaks in the forest were used in the search for a stable edge. Watercourses were also used as a large-scale network for introducing new diversity, broadleaved woodland and open ground, into the forest.

Today Kielder is over half way through this transformation. It is a patchwork of different aged stands and over half the watercourses, eventually totalling 5,000 ha, equivalent to a whole year's new planting of broadleaved trees for England, are in place. These watercourses are no grudging 20 metres back from the stream edge; they are in scale with this big landscape, often 400 metres across. However, this was not a one off vision, the thinking behind the transformation of this huge forest from single purpose plantation to a multi-purpose landscape goes on apace, a major EU LIFE project has funded tree removal and drain blocking to restore extensive areas of the Kielder Mires SAC. Continuous cover forest management is under development in the least windy parts of the Kielder valley. The latest design plans will develop 2,000 ha of irregular broadleaved edge for black grouse (*Lyrurus tetrix*).

Although a completely artificial forest, Kielder has developed a unique and fascinating landscape (Petty & Avery, 1990; Jardine, 1988). It has one of the few predator-on-predator ecosystems in England, with goshawk (*Accipiter gentilis*) at the top of the tree and tawny owl (*Strix aluco*)further down significantly impacting species like sparrowhawk (*Accipiter nisus*), kestrel (*Falco tinnunculus*) and short-eared owl (*Asio flammeus*). It is valued for what it is, a huge conifer forest, and is increasingly important as the best refuge for the declining and endangered red squirrel (*Sciurus vulgaris*) in England.

We live in an age when, despite our prosperity, it is sometimes hard to see beyond the self-imposed barriers we have created. Foresters by their nature look ahead. Twenty-five years on, the restructuring of Kielder shows that it is possible to have big, new visions for the future and to make them happen.

The Forestry Commission

As Britain's largest land manager the Forestry Commission is in a unique position to develop landscape-scale thinking for sustainable management. It manages large areas of non-plantation habitat, including the largest areas of lowland heath, pasture woodland and Caledonian pine in Britain. Through the examples in this paper I want to show how theory can be put into practise. All the examples involve the Forestry Commission – and they are only a fraction of what we are currently doing but I recognise, and applaud, the many others involved in visionary landscape-scale projects, from the Woodland Trust's Glen Finglas in the Highlands of Scotland, through Tomorrow's Heathland Heritage to the Great Fen project.

The Dorset Forests and Heathland Project

Launched in 1990, the Dorset Forests and Heathland Project (Forestry Commission, 1991) was a milestone for the Forestry Commission because of its landscape-scale conception and because it involved the removal of trees to re-create heathland. Forestry, with agriculture and urbanisation, is a major cause of heathland loss in Dorset. Forests fragment the remaining heathland and a range of heathland species, particularly sand lizard (*Lacerta agilis*), were threatened and conserved in small, intensively managed reserves. Although small by today's standards, at 150 ha, at the time this was a radical, culture-challenging change.

The project was planned along landscape ecology lines, looking in particular to create landscape linkages at two levels. Firstly, links between existing areas of heathland, in particular, fragmented SSSIs were linked by very large (300 metresplus) corridors. Secondly, by re-linking the sequence of bog, moist heath and dry heath. Almost all the dry heath had been planted because it was best for trees.

Finally, as the project developed it became clear that grazing was the key to future management. English Nature led in bringing together several ownerships and funded the fencing of a 400 ha grazing enclosure.

New Forest – New Future

What started in Dorset laid the seeds for larger scale action in the New Forest. The landscape-scale action now underway started with a £100,000 sponsorship from ESSO's Trees of Time & Place project, leading to the appointment of Jonathan Spencer to develop the conservation of the Ancient & Ornamental (pasture) woodlands of the New Forest (Goriup, 1999). The thinking around this project led to a much wider view of the future of the New Forest. There was, as there tends to be in both conservation and agriculture, a 'little and often' approach to much Forestry Commission conservation. The New Forest was big and had lots of top habitat, whereas the thinking was that improvements should be focused on the big upland forests, introducing broadleaves and taking conifers back from watercourses.

However, we recognised that the New Forest was the only place we could increase the scale of the biggest heaths and pasture woodlands in northwest Europe. The theoretical benefits of scale to biodiversity have been thoroughly covered in this conference but the effect of scale is something one can feel empirically in the New Forest. There just seems to be more wildlife than in equivalent smaller areas of habitat in the lowlands. So, New Forest – New Future set out to make the best better. The physical work of the project was only one component. The interest and diversity of stakeholders in the New Forest is intense and it is not solely voluntary, as rights are shared between the state and many hundreds of Commoners, administered by the Court of Verderers. Stakeholder buy-in was vital but so too was the deep knowledge and advice on the ecology and culture of the forest of the wide range of people who formed the core forum for the development of a new plan for the New Forest.

Put baldly, this will restore or develop new habitat on over 2,000 ha of the timber 'enclosures', about a quarter of the total area. But this involves the improvement of much larger areas of the 27,000 ha SAC, the development of innovative techniques and novel habitat restoration. The plan also led to what may be the most fundamental change of all, the revision of the Minister's Mandate under which the Forestry Commission manages the New Forest to give priority to management for 'conservation of the natural & cultural heritage'.

In practise, wide ranging and exciting habitat restoration is underway on a grand scale, supported by nearly £5m of EU LIFE funding. Heathland is being restored from conifer plantation. The most costly projects are the restoration of the forest's 90 valley mires, the largest surviving series in Europe. Hydrological studies showed that simply blocking drains would disrupt the 'trickle through' movement of water and a range of techniques, especially drain blocking using heather bales are being implemented. Erosion on the heathland is being repaired. Heavy thinning of nineteenth century oak plantations to move them towards the more open structure of the pasture woodland is underway.

For the planning process, Forestry Commission landscape architect, Roger Worthington developed new methods for visual presentation of landscape change. Starting with hand drawn illustrations from ground level and oblique aerial views, somewhere between a map and a photo, he moved onto a sophisticated photo-montage approach which, by using photos of existing habitat, gives a near-virtual impression of what the landscape will look like over time.

This year more mires are under restoration and, in partnership with the Environment Agency, work has moved downstream to re-align rivers straightened in the 1960s to speed drainage from the forest, with significant implications not just for the natural flood plain but also downstream flood management.

One of the key things that make the New Forest so special is a function of its scale and the 'fenceless' management of the 'open' forest, which includes almost all the heathland and pasture woodland. Natural processes are at work. As grazing patterns shift, scrub moves out onto the heathland in one place, woodland becomes more and more open in others. In lowland England we live in an environment of sharp, hard habitat edges. Most managers never have the experience of this sort of natural change and for some, and perhaps the way we approach conservation management, it is scary and counter-cultural, after all that is European designated heathland that scrub is spreading out onto!

The Cairngorms

The last 20 years have seen the quiet development of a landscape revolution around the western fringes of the Cairngorms. In 1980 the RSPB owned the famous Loch Garten reserve and the Forestry Commission had all but converted Glen More forest to non-native conifers. RSPB first bought Ben Macdui, a superb area of the Cairngorm plateau, then made its biggest ever purchase of the Abernethy estate. To the south the Forestry Commission's spectacular programme to restore its native pinewoods has transformed Glen More forest, with the removal of large areas of non-natives and to the south the young, largely pine, Inshriach Forest will develop as a further huge extension of what is now tens of thousands of hectares of continuous natural landscape. Already a LIFE project is funding the restoration of wet woodland in an area of forest bogs so similar to Scandinavia that their forest waders must surely follow!

Most important of all, however, is that at both Abernethy and Glen More young pine are moving out from the forest edge. This is the result of concerted cross-boundary action by RSPB and the Forestry Commission to tackle the biggest threat to the future of the pinewoods; excessive deer numbers. It demonstrates another key theme for practical landscape action common to every one of these examples; working together in partnership.

Ancient woodland restoration

Forestry Commission action for native pinewoods is a key component of a much wider programme to restore ancient woodlands. Since the work of Peterken (1981) and Rackham (1980) the irreplaceable value of ancient woodland has increasingly been recognised. Crabtree (2003), in a major review of the economics of forestry in England, recognises the case for ancient woodland to be considered strongly sustainable and a key component of natural capital. The drive for timber production resulted in the conversion of many ancient woodlands in England to non-native conifers. In contrast to woods grubbed out for agriculture, Forestry Commission restoration work stretching back to small scale, tentative projects in the early 1980s has demonstrated that these woods can show a remarkable level of recovery.

In contrast to the other examples, ancient woodland restoration involves hundreds of sites, but the total involved is impressive, 15,000 ha by 2020. The Forestry Commission plans to carry out extensive restoration in all its ancient woodlands, with the majority restored to native woodland. In contrast to so many conservation programmes, we have good baseline information, thanks to a major survey of our ancient woodland and a simple classification of its state of 'nativeness' (Spencer 2002). Additionally and crucially, the survey also predicts likely National Vegetation Classification type for the restored native woodland.

The Ancient Woodland Project in the Northants Forest District (covering 5 counties of the East Midlands) is the biggest single project, restoring over 5,000 ha. With most woods in this intensely arable area protected historically by heavy clay soils, 50 years struggle planting conifers climaxed in 1990 when whole stands of middle-aged Norway spruce started dying of drought stress. The impacts and appropriateness of the 'conventional' replanting of felled sites were challenged by UK Woodland Assurance Scheme auditors and led to a complete re-appraisal and change of direction towards restoration by the most natural means possible, with the key objective the establishment of native woodland cover. This change was accompanied by a major communications campaign which has resulted in a huge increase in interest in these woods and in the value local people put on them.

Faced with the ferocious weed growth characteristic of the clay soils and high deer pressure, we went into the project expecting to be left with many areas where no trees regenerated. The last three years have been an eye opener, as young trees force their way through the weed growth, delicate primroses appear amongst the rank clumps of *Calamagrostis epigejos* (wood small-reed) and bare sites are transformed into young native woodland. A major archaeological survey has opened a new dimension not just to these woods but across our ancient woodlands as a whole as it has become clear how well the woods have conserved heritage features lost from the intensively farmed landscape.

And, to top it all, the District Office at Fineshade has become home to the red kite (*Milvus milvus*) project, with CCTV coverage of a nest of the red kites recently re-introduced to the area.

Learning from others

Many of us have been inspired by the Dutch plan for nature and the superb achievement of vision and application that the Oostvaardersplassen reserve represents. Grazing and Frans Vera's theories can hardly fail to be one of the talking points of the conference, but I'd like to draw out some other lessons I took away from Holland. The first, obviously, was the vision. It is a vision we can share.

The Dutch are taking head on the big issue of traditional primary production as against social and environmental values in determining land use. Frans explained how grazing for biodiversity had to be vigorously protected against intense 'mission creep' back towards conventional agriculture and away from conservation management outcomes.

The second is scale. At one level Oostvaardersplassen is huge at 5,000 ha. This one site contains more reedbed than the whole of England. It also appears to have become the northwest European population driver for at least two bird species; spoonbill (*Platalea leucorodia*) and bearded tit (*Panurus biarmicus*). It is less than 0.05% of the area of England and roughly the same size as the new woodland currently planted in England annually. It is quite a conservation outcome for such a small area of land.

The other example is a natural one, but not woodland either, the Biebrza Marshes in Poland (D'Arcy Shillcock, 1993). Going back to natural processes, the thing that impressed me so much about this area was the effect of the intimate mix of different habitats, from open water and reedbed through meadowsweet (*Filipendula ulmaria*) rich grazing to wet woodland, and all full of birds including lekking black grouse, a cacophony of corncrakes (*Crex crex*), thrush nightingales (*Luscinia luscinia*) and great snipe (*Gallinago media*).

I was amazed, reading Mike Shrubb's (Shrubb, 2003) new book on the history of agriculture and birds, by the species lost to the enclosure of wastes and commons and the drainage of the fens. Dating back to the late 1700s and early 1800s, well before *The historical atlas* (Holloway 1996), he shows that species considered largely upland today including black grouse, hen harrier (*Circus cyaneus*)and corncrake were East Anglian birds, whilst bittern (*Botaurus stellaris*) were common and tasty enough to be a star feature at feasts. Time and again Shrubb describes how species have lost out through simplification rather than just loss of habitat with the species that need the Biebrza or New Forest type intimate diversity produced by natural processes losing out first.

Into the future

The New Forest is a wonderful ecosystem. It is also one of the most popular countryside recreation resources in England. Looking into the future, I want to come back close to towns where the Forestry Commission has developed 3,500 ha of new community woodland and is working with the new Land Regeneration Trust towards the restoration of a further 10,000 ha of damaged land.

These are small woods, aimed at people and don't at first site look like a big opportunity for landscape-scale biodiversity. Many are in the community forests and the National Forest and on damaged land, especially the coal mining legacy. Whilst the Forestry Commission is developing many of the larger sites, we are not alone, with the Woodland Trust also playing a major role in developing small woods. However, whilst that may be nice for the people who live there it is not very relevant to landscape ecology surely? And is not the National Forest the forest without trees? No and no, it takes time and vision but drive along the M42 or through the National Forest and suddenly a decade's work is starting to show and there are trees everywhere.

And those little Woodland Trust woods? Actually they are not so small after all and in the National Forest alone together extend to over 1,000 ha. They are a wonderful improvement for the people who live next to them and now there is another wood beside them and the landfill across the road is nearly finished and on the way to woodland. Gradually, but in the life of trees very fast, a new network of green space is starting to emerge and the vision of 'green grids' through our cities is something more and more people are starting to understand. These woods are for people first: the Government's Capital Modernisation Funding to the Forestry Commission built 75 km of new all-ability access close to where tens of thousands of people live.

But these sites are equally important for biodiversity. First, they have significant wildlife value of their own. In Thames Chase community forest, sites buffer London's biggest reedbed and downstream the river leads to RSPB's new Rainham Marsh reserve. In Nottinghamshire, wet grassland, pools and gravel at Bevercotes is a superb reserve for waders, including redshank (*Tringa totanus*), lapwing (*Vanellus vanellus*) and little ringed plover (*Charadrius dubius*), that are increasingly rare in the wider countryside.

They are also the launch point for wider, landscape-scale habitat restoration. They show what green space can do for people, and I believe this is the route to new landscapes, particularly in the lowlands. Large-scale low-intensity public access land could play a key role in the Government's plans for the development of new housing. It is the social benefits, which will drive thinking; quality of life, health, accessibility. But sheer scale, as we see today in the New Forest, can make real multi-purpose land management feasible, with great biodiversity gains linked to social benefits (Garforth & Dudley 2003).

Conclusion

What we are already achieving proves we can put theory into practise. Perhaps the greatest barriers are in our own perception. In the Forestry Commission it has taken deep cultural change to, for example, cut down timber-producing trees for heathland. Similar but different barriers are there in conservation, agriculture and planning. To see the future we need to think:

- Vision
- Scale
- Natural processes
- People & partnership

References

Crabtree, R. (2003) *Economic analysis of forestry policy in England*, DEFRA.
D'Arcy Shillcock, R. (1993) *Portrait of a living marsh*, Inmerc BV, Netherlands.
Holloway, S. (1996) *The historical atlas of breeding birds in Britain & Ireland 1875-1900*, Poyser.
Forestry Commission (1991) *Dorset Forests and Heathlands Project* (available from author's address).
Garforth, M. & Dudley, N. (2003) *Forest Renaissance*, Forestry Commission, Edinburgh .
Goriup, P. (Ed) (1999) *The New Forest woodlands*, Pisces Publications
Jardine D.C. (Ed) (1988) *Wildlife management in forests*. Institute of Chartered Foresters, Edinburgh.
Peterken, G.F. (1981) *Woodland conservation and management*, Chapman & Hall.
Petty S.J. & Avery M.I. (1990) *Forestry and Birds*. Forestry Commission.
Rackham, O. (1980) *Ancient woodland*, Edward Arnold.
Shrubb, M. (2003) Birds, scythes and combines CUP
Spencer, J.W. (2002) *Ancient woodland on the Forestry Commission estate in England*, Forest Enterprise, Bristol (available from authors address)
Wigbels, V. (undated) *Oostvaardersplassen – New nature below sea level*, Staatsbosbeheer, Netherlands.

Policy implications of landscape-scale action for trees and forests in the UK

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Abstract

Delivering landscape scale action for trees and forests presents a major challenge to existing policy frameworks. Governments and their agencies aspire to 'joined up thinking' in many areas of public policy but achieving major shifts in land use and land management as required by landscape scale action is a task which will challenge the most avid proponents of integrated thinking.

Yet there are some emergent areas of policy where the potential to achieve a less sectoral and more coherent approach to land management is evident: the Water Frameworks Directive focuses on planning at the level of whole catchments and river basins; both the Common Agricultural Policy Mid Term Review proposals for reform of Pillar 1 payments and the Pillar 2 agri-environment scheme review will have a major impact on land management practices.

This paper briefly assesses the potential of these and other areas of public policy development and their associated institutions both to advance and to hinder the development of landscape scale action.

Introduction: chasing the rainbow?

The imperative of landscape scale action for the conservation of trees and forests and indeed for all semi-natural habitats has only been recognised fully in the past two or three years. At present many current land use policies relevant to forests and other semi-natural habitats have been derived from accepted thinking and conservation paradigms, which are a decade or more old. There is an inevitable lag effect between the development of aspirational ideas, the widespread acceptance of such ideas in primary legislation and mainstream policy documents (such as country forestry and biodiversity strategies, the Habitats Directive and the Water Framework Directive) and the development of new policy instruments to deliver them. Major revisions to policy instruments, such as forestry and agri-environment grants or planning guidance for example, are unlikely to happen more than every three to five years and key policy documents may be revised even less frequently.

A particular example of such a time lag is in relation to current biodiversity policy. This is currently based on the paradigm of site-based measures to prevent loss and damage to an elite suite of designated sites, and to enhance management of these sites. The two main instruments to achieve these objectives are wildlife legislation including the Countryside and Rights of Way Act (2000), the Nature Conservation Bill (Scotland) due to become law this summer, plus their antecedents the Wildlife and Countryside Acts (1981 and 1992), and the UK Biodiversity Action Plan (BAP) (Anon 1994). The BAP does encompass targets and actions beyond designated sites in the wider countryside and contains actions, which encourage restoration, buffering and expansion of habitats, but in their present form, neither of these tools will be able to deliver sufficient action to achieve landscape scale conservation. Nonetheless the revision of the UK Biodiversity Action Plan by early 2006 is a major opportunity to restructure targets and priority habitats and species lists to achieve a more cohesive landscape scale approach to the conservation of all semi-natural habitats.

Paradoxically, other policy areas relevant to land use decisions are changing faster than biodiversity policy instruments. The Common Agricultural Policy's Mid Term Review of 2003 has seen the introduction of reforms that include breaking the link between subsidy and production. These reforms are likely to have a number of environmentally beneficial outcomes, although much still depends on implementation of certain aspects, such as Member State's definitions of Good Agricultural and Environmental Condition. In addition all UK countries have developed or are developing an Entry Level Scheme (or equivalent), a good illustration of a landscape scale policy instrument, as it will be available to all farmers and will see increased environmental outcomes across all farmland. However, the potential for stimulating and rewarding environmentally benign land management in the farmed landscape is still yet to be fully achieved particularly as the Rural Development Regulation expenditure (Pillar 2), which includes funds for agri-environment schemes, accounts for only 10% of the EU total CAP budget of about 40 billion Euro (£24 billion), 90% of which still supports Pillar 1 production subsidies.

Water policy is starting to become more sensitive to operating at a whole catchment scale through the transposition of the EU Water Framework Directive (WFD) into UK law. The WFD is principally concerned with quality and quantity of water, so agriculture and land-use generally are crucial to its implementation. It is potentially a powerful catalyst to creating joined up policy delivery through linkages with diffuse pollution, biodiversity conservation (through floodplain wetland and woodland habitats for example) and flooding.

In some cases the legislative framework is already in place to deliver landscape scale action. However there seems to be a primary time lag from this legislation to its interpretation into delivery. For example, Article 10 of the Habitats Directive talks about stepping stone habitats, which is a promising platform for delivering landscape scale action, but this has not yet been transposed into UK and national legislation. In England and Wales, the Countryside and Rights of Way Act (2000) gives English Nature and Countryside Council for Wales the power to buffer and extend SSSIs, but this has yet to be translated into policy instruments or actual delivery.

Such promising signs may be compromised by the additional complication that land management policy developments may well be pulling in contrary directions simultaneously. For example, the Common Agricultural Policy Mid Term Review implementation in the UK may end up creating some unforeseen consequences for woodland management and woodland creation on farms (Land Use Policy Group 2004) such as the current concern about grubbing out of traditional orchards in England as a result of the proposed eligibility rules for Single Farm Payments.

There are also tensions created by policy developments in other sectors such as housing and transport policy which are running counter to the need for landscape scale action for wildlife. The imperative for 'joined up thinking' within Governments has been partly a direct result of two or more departments paying out public money for activities that are mutually contradictory. Perverse agricultural subsidies have been a phenomenon for many years (an example has been the existence of grants to grub out hedgerows and grants to plant hedgerows being simultaneously available) but perverse policies persist to the present time. For example, plans for major housing expansion in the South East of England and airport expansion are being promoted by the Office of the Deputy Prime Minister and the Department of Transport in direct contravention of the Department for Environment, Food and Rural Affairs' (DEFRA) England Biodiversity Strategy, the Government's UK Sustainable Development Strategy and other commitments to environmental protection.

The consequences of such tensions are that the basic building blocks of landscape scale action, existing semi-natural habitats, are threatened. Ancient woods are still subject to damage and loss through development and other causes (Allison, 2003). If semi-natural habitats, such as ancient woodland, are continuing to be lost, what hope is there of getting to grips with landscape scale action? Joined-up thinking within and across Government departments, throughout the UK, is key to achieving landscape scale action because of the need to work across traditional disciplines. In practice, this represents a huge and complex task in the face of political realities, interdepartmental tensions and the timeframes necessary to translate new ideas into practicable policy instruments.

Tools for the job?

In trying to assess whether the policy instruments at our disposal are fit for the task of trying to achieve landscape scale change, it is necessary to define what is meant by landscape scale action (creating ecologically functional landscapes with a far higher proportion of semi-natural habitats than exist at present in a more sympathetic matrix of land use) in terms of identifiable changes in land management practices, and then to evaluate the current instruments against these changes. This will then allow an assessment of whether existing instruments will deliver our objectives or whether we need to seek out or create new kinds of tools for the job.

Changes in land management are needed:

- to prevent any further loss and fragmentation of ancient and semi-natural woods and forests in the UK to protect the essential seed corn of woodland biodiversity which exists now and from which future dispersal can take place, assuming conditions are favourable. The same is needed for other semi-natural habitats too
- to restore degraded habitats, such as conifer planted ancient woodland sites (PAWS), which nonetheless retain important relict features. Ancient woodland is irreplaceable so restoring PAWS is the only way to increase the amount of woodland with ancient semi-natural characteristics in the landscape
- to ensure that targeted woodland and habitat creation is carried out to increase the cumulative core area of semi-natural habitats (Woodland Trust, 2000) associated with ancient woodland, including wood pasture and other areas supporting concentrations of ancient trees
- to achieve a reduction of land use intensity in relation to forestry, agriculture and urban development. This breaks down into specific requirements, such as reductions in diffuse pollution, improved soil management, fewer herbicide and pesticide inputs, reductions in grazing pressure to deliver more sympathetic management of existing

habitats and re-establish more transitional habitats (ecotones), and more natural green space within urban environments.

By doing a crude tabulation (Table 1) it can be seen that some tools have or will have the potential to produce some of the desired outputs but by no means all of them.

The policy instruments shown here are not broken down by country variations in order to keep the evaluation as simple as possible but there is already ample evidence that the differences in implementation of the Common Agricultural Mid Term Review, particularly the method of payment calculation for the Single Farm Payment in the four countries within the UK, are likely to lead to different prognoses for the delivery of more environmentally sensitive land management.

Nonetheless this top line evaluation shows that few existing policy instruments, even when considered in total, have specific provision to deliver changes at present. There remains much negotiation over the finer points of many of these new instruments as they are being developed. For example, the definition and implementation of Good Agricultural and Environmental Condition, which will form part of the cross-compliance package associated with the new Single Farm Payment Scheme, may yet prove to be a crucial tool in reducing herbicide and pesticide inputs and creating ecotones. Equally the implementation of the Water Framework Directive may lead to new perspectives on the role of more benign land management options in flood alleviation and the new spatial element to regional planning in England and Wales may help to prioritise those elements of biodiversity conservation which can be accommodated within the planning system. However these remain at best possibilities with the potential to deliver rather than being tried and tested, and their ability to work in a joined up way is uncertain.

Table 1. Evaluation matrix of policy instruments against desired land management outcomes

	Woodland Grant Schemes	Regional Spatial Strategies/ Wales Spatial Plan	CAP Mid Term Review reforms	Agri-environment Entry Level Scheme	Water Framework Directive	UK BAP	Article 10 Habitats Directive/	Planning Policy Guidance/ Statements	CROW Act
Protection of ancient woodland	⇨⇨	⇨	⇨	⇨	⇨	⇨⇨	⇨⇨	⇨⇨	⇨
Targeted woodland creation	⇨⇨	⇨	X	⇨	⇨	⇨	⇨	⇨	⇨
Restoration of degraded habitats	⇨⇨	XX	X	⇨	X	⇨⇨	⇨	X	⇨
Reduction in diffuse pollution	X	XX	⇨	⇨	⇨	⇨⇨	⇨	XX	⇨
Fewer chemical inputs	⇨	XX	⇨	⇨	⇨	⇨	⇨	XX	⇨
Reduction in grazing	X	XX	⇨	⇨	XX	⇨	⇨	X	⇨
Creation of ecotones	X	X	⇨	⇨	⇨	X	⇨	X	⇨
Green space in urban areas	⇨⇨	⇨	XX	XX	⇨	⇨	X	⇨⇨	X
Soil management	X	X	⇨	⇨	⇨	⇨	⇨	X	X

⇨⇨ = provisions already within policy instrument to secure this output

X = no evidence that this instrument is securing/will secure this output

⇨ = potential beneficial impact

XX = no likelihood it will secure this output

Institutions for the job

Possibly more worrying than the efficacy of tools is the lack of suitable institutional structures to develop and implement them effectively. There are no truly integrated land management institutions that can take a holistic view of natural resources protection at a landscape scale.

Within each country of the UK there are at least two or three agencies dedicated to different aspects of resource protection and land management. In England, the Environment Agency deals with water quality and flood control issues and various elements of enforcement of environmental standards such as emissions, while English Nature is responsible for the protection of biodiversity and the Rural Development Service of DEFRA administers the agri-environment scheme budget. The current Modernising Rural Delivery Review being undertaken by DEFRA is intended to strengthen rural delivery and integrate various land management initiatives but the Review is as much about political posturing and cost control than it is about genuine integration of land management and adopting a holistic approach to managing natural resources. In any event the outcome of this Review is likely to mean that there will still be more than one agency responsible for various aspects of rural delivery.

In Northern Ireland, the lack of any parallel institution to the Environmental Agency or the Scottish Environment Protection Agency means that there is an institutional deficiency of a serious nature (Macrory, 2004).

Achieving landscape scale action for forests and trees

The existing instruments examined above are still for the most part too new to deliver the land management outputs needed to achieve landscape scale action except the Habitats Directive, which, as already shown, has yet to deliver it potential. Many are still in development and the chance must be seized to influence the process of building and testing them. Advocacy and campaigning by voluntary organisations and agencies within Government is crucial in this regard and political awareness and intelligence gathering is necessary to identify suitable openings. Indeed there are many windows of opportunity at this very moment, such as the UK Sustainable Development Strategy Review and the reviews of the role of the public forest estate in Scotland and England, which within a matter of months will open and close.

As well as attempting to strengthen the instruments that will shortly be coming on stream, it is also crucial to identify new opportunities and potential instruments. Strategic Environmental Assessment is one such tool which on first inspection has only limited relevance to the delivery of landscape scale action but which needs further examination to see whether its provisions could open up new avenues.

Nonetheless, there are also considerable threats to the implementation of broad land management policies that will embrace the need for landscape scale action. Many of these are linked to the institutional issues identified above, such as the political necessity of the Modernising Rural Delivery Review driving out the chance for real change and resulting in the loss of any likelihood of creating a truly integrated land management agency in England. There are also many uncertainties about the long term structure and future of the Forestry

Commission which, as a result of its unique combination of regulatory, advisory and funding role combined with a massive publicly owned land holding, is probably best placed of all institutions to catalyse landscape scale action for forests and trees. Added to this, there is no real history amongst private landowning interests in the UK of co-operative working towards a common goal, which may also confound our best efforts to achieve landscape scale action, neither have they been engaged in any formal way in the development of landscape scale thinking and the actions which arise from it. This could prove to be a serious barrier to achieving change. All of these threats may in turn be compounded by the reality of political timescales, which are measured in intervals between elections rather than sustained long term programmes to achieve change of the order required.

Conclusion

Some progress has been made; some existing legislation and many policies and strategies are couched in sufficiently broad terms to enable a landscape-scale approach to be taken if the political will to develop policy instruments appropriate to the task were present. Do governments have the stomach to see this through with genuine joined-up cross-departmental co-operation operating over timescales much longer than the normal political calendar allows? It seems that there are some genuine opportunities to make landscape scale action more likely over the next one or two decades as new and developing policy instruments come on stream. It will require political skill to maximise these opportunities but the uncertainties associated with institutional structures which will be leading their development and delivery are immense. The future is difficult to predict but perhaps the imperative for landscape scale action on human society, as climate change takes hold, will ultimately prove to be the trigger for action that benefits the natural environment as well.

References

Allison, H.M. (2003) *From wild wood to concrete jungle? – a contemporary attitude to ancient woodland and its protection.* Paper given at South Yorkshire Biodiversity Research Group and Network and the Landscape Conservation Forum May 2003.
Anon (1994) *The UK Biodiversity Action Plan*, HMSO.
Land Use Policy Group (2004) *Common Agricultural Policy and Forestry seminar.* March 2004.
Macrory, R. (2004) *Transparency and Trust – reshaping environmental governance in Northern Ireland.* Centre for Law and the Environment, UCL.
Woodland Trust (2000) Woodland biodiversity: expanding our horizons www.woodland-trust.org/publications/index.htm

Poster papers

Effects of fertiliser drift at ancient woodland edges

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Abstract

Ancient woodland in the UK is highly fragmented, exposing sites to impacts from the external landscape matrix. This study investigates the impact of fertiliser drift on soil and ground vegetation at the edges of ancient woodland set within arable matrices. Data from soil analysis and vegetation surveys were analysed to determine whether an anthropogenic soil nutrient and vegetation gradient exists within ancient woodland edges adjacent to arable fields. Woods next to semi-natural habitat were used as control sites. No significant gradient was found for phosphorus, though significant gradients for soil pH (measured and Ellenberg reaction), Ellenberg nitrogen, magnesium and potassium were detected, with edge effects extending 20-30m into the woodland for soil pH and nitrogen, and 10-20m for potassium and magnesium. Woodland edges bordering arable fields showed a higher mean abundance of some nutrient-demanding species than edges next to semi-natural habitat. In particular, *Galium aparine* showed such an edge effect that extended 30-40m. Buffer zones may offer some protection to ancient woods.

Introduction

In the UK, fragmentation of ancient woodland, defined as woods where there has been continuous cover since at least 1600 AD (Spencer and Kirby, 1992), has resulted in 48% of ancient woods in the Ancient Woodland Inventory (AWI) in Britain being less than five hectares (46% in England, 55% in Scotland, 50% in Wales) (Woodland Trust, 2002). The isolation of remaining ancient woodland fragments has been aggravated by the intensification of land use in the surrounding landscape (Peterken, 1991). In agricultural landscapes such as those typical of the UK, one type of environmental variation has been little studied in relation to woodland fragmentation, that of anthropogenic soil nutrient gradients.

Anthropogenic soil nutrient gradients may be expected across woodland edges in intensive agricultural landscapes due to a direct input of nutrients, mainly nitrogen, phosphorus and potassium, by fertiliser drift and run-off from adjacent agricultural land (Kleijn and Snoeijing, 1997). It is known that artificial nutrient addition can have major consequences for plant species diversity, originating from changes in competitive relations between species (Lee, 1998). Honnay *et al* (2002) investigated the invasion of ancient forest edges by weedy plant species, adjacent to intensive arable fields in Belgium. They found that plant community composition of forest edge zones differed significantly from that of the forest interior, with a relatively high abundance of competitive species at forest edges. Longer-term studies support findings of such short-term investigations. Kleijn and Snoeijing (1997) undertook a three-year experiment into the effects of fertiliser misplacement on the botanical diversity of arable field boundary vegetation and found that fertiliser application resulted in a decline in species richness through a loss of species of low stature. A nine-year experiment into the effects of NPK fertilisation on boreal forest vegetation in Canada (Turkington *et al*, 1998) revealed that overall vegetation composition changed as more competitive species increased in abundance. These studies suggest that woodland edges adjacent to land that has been subject to artificial nutrient input through the use of fertiliser, may be vulnerable to invasion of more competitive, nutrient-demanding species to the detriment of native woodland flora. This is of concern in relation to ancient semi-natural woodland, as it is regarded as the most important category of woodland for nature conservation in the UK (Peterken, 1996). It generally has a richer ground flora than more recent woodland (Rieley & Page, 1990) and tends to contain a high proportion of rare and vulnerable woodland species, some of which have poor dispersal and colonising abilities (Peterken, 1996).

This study investigates the effect of proximity to arable land on soil nutrient levels and invasion by weedy, nutrient-demanding species at the edges of ancient semi-natural woodland. Control sites are used (woods adjacent to semi-natural habitat), against which data from woods adjacent to arable land are compared. This controls for the possibility that any soil nutrient gradient found at woodland edges may be due in part to atmospheric ion deposition. The study addresses the following questions:

1. Does an anthropogenic soil nutrient gradient exist across ancient semi-natural woodland edges adjacent to arable land as compared to edges adjacent to semi-natural habitats?
2. Does plant species composition change across ancient semi-natural woodland edges adjacent to arable land as compared to edges adjacent to semi-natural habitats?
3. What is the effect of orientation on plant community composition and edaphic factors?

Methodology

Twelve ancient semi-natural woods from the AWI were selected within Lincolnshire (nine woods) and Derbyshire (three woods). Transects were established at randomly selected points along the woodland edges, extending perpendicularly into the woods. In some larger woods, two transects were established at least 100m apart. All transects were located at least 100m from any woodland edge other than the one of interest. Overall 16 transects within the 12 woods were surveyed (11 arable and 5 semi-natural transects). Parallel replicate transects were established at randomly chosen distances, between 10m and 20m from the original transects. This resulted in 32 transects being surveyed in total. Quadrats were located at 0, 2, 5, 10, 20, 30, 40, 50, 75 and 100m from the woodland edge in both original and replicate transects, and were 1m x 1m in size (subdivided into 100 subunits). A vegetation survey, which recorded percentage cover of all vegetation under 2m in height, was undertaken in each. Soil samples were taken, at a depth of 20cm using a Dutch auger, from each quadrat in the original transects and from 0, 50 and 100m quadrats in the replicate transects. Samples were analysed for available phosphate, magnesium, potassium and soil pH. Available nitrogen was not measured because of financial constraints. From vegetation data, values for nitrogen and reaction (soil pH) were calculated based on Ellenberg indicator figures (Hill *et al*, 1999) via weighted averages for each quadrat (Diekmann & Falkengren-Grerup, 1998).

Spearman's rank correlation was used to identify relationships between distance from the woodland edge and the dependent variables: phosphorus; soil pH; potassium; magnesium; Ellenberg values for nitrogen (EVN); Ellenberg values for reaction (EVR); and cover of selected nutrient-demanding species with Ellenberg values of six and above. This was performed separately for transects adjacent to arable land and those adjacent to semi-natural habitat to enable comparisons between the two land uses to be made. Spearman's rank correlation was used because the normality assumption of Pearson's correlation was violated in some cases. To further investigate the effect of adjacent habitat, edge orientation and distance on the dependent variables, analysis of covariance (ANCOVA) was used. In order to quantify the penetration distances of edge effects, pairwise comparisons were made of dependent variable values on either side of all possible cut-points along the transects, using Mann-Whitney U–tests. For example, quadrats 0m from the woodland edge vs all the others, quadrats 0m and 2m from the woodland edge vs all the others, etc; increased likelihood of Type I errors was accounted for by setting the significance level at $\alpha=0.001$ for this analysis.

Results

Soil analysis

For woodland adjacent to arable fields (hereafter 'arable woodland edges'), there were negative correlations with distance from edge for soil pH ($r_s=-0.357$, $p<0.001$), potassium ($r_s=-0.227$, $p=0.008$) and magnesium ($r_s=-0.174$, $p=0.045$), indicating weak gradients, but not for phosphorus ($r_s=-0.026$, $p=0.77$). For woodland next to semi-natural habitat ('semi-natural woodland edges'), distance from edge did not correlate with phosphorus, magnesium or pH ($p>0.2$ in each case) but did with potassium ($r_s=-0.447$, $p<0.001$). There was no significant difference in potassium levels between the arable and semi-natural woodland edges, but the penetration distance of the edge effect was greater for arable edges (c.10-20m) than for semi-natural edges (c.2m). All relationships and differences involving phosphorus were non-significant. In ANCOVA for magnesium, a weak reduction with distance from edge was found ($F=5.93$, $p=0.016$), the slope of which did not differ significantly between arable and semi-natural edges. However, significantly more magnesium ($F=11.1$, $p<0.001$) was present in soils of arable woodland edges (controlling for the orientation effect – see below). Penetration into arable woodland edges was c.10-20m (but $p=0.012$, so this was not significant at $\alpha=0.001$). Soil pH declined with distance from edge (ANCOVA: $F=18.0$, $p<0.001$), with no significant difference in slope between arable and semi-natural edges. The pH was significantly higher ($F=14.4$, $p<0.001$) in soils of arable woodland edges, for which penetration distance was c.20-30m. Orientation only affected magnesium levels, with significantly more magnesium found in soils of north-facing than south-facing woodland edges ($F=17.6$, $p<0.001$).

Vegetation analysis

The results for EVR and EVN were very similar. Both correlated negatively with distance from the woodland edge for arable edges (EVR: $r_s=-0.300$, EVN: $r_s=-0.236$; $p<0.001$ for both) and more weakly for semi-natural woodland edges (EVR: $r_s=-0.276$, $p=0.0074$; EVN: $r_s=-0.165$, $p=0.114$). In ANCOVA in both cases there was a significant decline with distance from woodland edge (EVR: $F=9.83$, $p=0.0019$; EVN: $F=8.01$, $p=0.0050$); in neither case was there a significant difference in distance-decay slopes between arable and semi-natural edges, nor was edge orientation significant. In both cases, significantly higher Ellenberg values were found in arable than semi-natural edges (EVR: $F=55.7$, EVN: $F=56.6$; $p<0.001$ for both). In both cases the proportion of variation explained was low (adjusted $R^2=0.172$ for EVN and 0.170 for EVR) but were slightly higher than in the soil analyses. Reassuringly, the results for EVR were also similar to those for measured soil pH (see above), though the correlation between EVR and pH values was only moderate (overall: $r_s=0.608$; arable edges: $r_s=0.553$; semi-natural edges: $r_s=0.547$; $p<0.001$ for all). Penetration distances were c.20-30m for both EVR and EVN in arable woodland edges.

Of the nutrient-demanding species, *Rubus fruticosus, Galium aparine* and *Urtica dioica* were all significantly more abundant in arable than semi-natural edges (Mann-Whitney U: $Z>3.5$, $p<0.001$ in each case); there was no significant difference for *Stachys sylvatica* and *Geum urbanum*. Although all these were more abundant on south-facing edges, this was only significant for *Urtica dioica* ($Z=3.08$, $p=0.002$), though *Stachys sylvatica* and *Geum urbanum* were marginal ($p\approx0.05$). The penetration distance analysis only produced clear results for *Galium aparine*, for which the penetration distance was estimated as 30-40m in arable woodland edges. The other nutrient-demanding species tended to be found at low densities throughout the transects in arable edges while rarely being found more than 5m from the woodland boundary in semi-natural edges, except for *Rubus fruticosus,* which was common throughout the transect in both types of woodland.

Conclusion

As ancient woods are valuable conservation assets in the landscape, it is important that measures are taken which afford some protection to ancient woodland plant species. Buffer strips at the edges of cultivated fields that border ancient woods may be sufficient for the protection of native plants at woodland edges. Our results suggest that a buffer zone of 30-40m around arable woods may be sufficient to protect woodland interiors from effects of fertiliser drift. One area for future research concerns whether the vegetation composition of buffer zones influences their effectiveness.

References

Diekmann, M. & Falkengren-Grerup, U. (1998), A new species index for forest vascular plants: development of functional indices based on mineralization rates of various forms of soil nitrogen, *Journal of Ecology* **86**: 269-283

Hill, M.O.; Mountford, J.O.; Roy, D.B. & Bunce, R.G.H. (1999), *Ellenberg indicator values for British plants,* ECOFACT Volume 2 Technical Annex, DETRA

Honnay, O.; Verheyen, K. & Hermy, M. (2002) Permeability of ancient forest edges for weedy plant species invasion. *Forest Ecology and Management* **161**: 109-122.

Kleijn, D. & Snoeijing, G.I.J. (1997) Field boundary vegetation and the effects of agrochemical drift: botanical change caused by low levels of herbicide and fertiliser. *Journal of Applied Ecology* **34**: 1413-1425.

Lee, J.A. (1998) Unintentional experiments with terrestrial ecosystems: ecological effects of sulphur and nitrogen pollutants, *Journal of Ecology,* **86**: 1-12

Peterken, G.F. (1996) *Natural woodland: ecology and conservation in northern temperate regions,* Cambridge University Press.

Rieley, J.O. & Page, S. (1990) *Ecology of plant communities: a phytosociological account of the British vegetation,* Longman Scientific, Harlow.

Spencer, J.W. & Kirby, K.J. (1992) An inventory of ancient woodland for England and Wales, *Biological Conservation* **62**: 77-93.

Turkington, R.; John, E., Krebs, C.J.; Dale, M.R.T.; Nams, V.O.; Boonstra, R.; Boutin, S.; Martin, K.; Sinclair, A.R.E. & Smith, J.N.M. (1998) The effects of NPK fertiliser for nine years on boreal forest vegetation in northwestern Canada. *Journal of Vegetation Science* **9**: 333-346.

Woodland Trust (2002) *Space for nature: landscape-scale action for woodland biodiversity,* Woodland Trust, Grantham.

Bats in the United Kingdom, a landscape scale perspective: considering the importance of habitat connectivity and the threats posed by fragmentation

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Abstract

Bats are a diverse group of nocturnal, insectivorous, flying mammals, occupying a variety of habitats although showing strong preferences to certain landscape types. They make unique use of connective landscape features for navigation, using echo-location to find their way in the dark and hunt their insect prey. Species' populations are spread across a wide area of countryside using favoured roost sites and foraging areas, the knowledge of which is passed down through generations. Swarming sites, used for mating, are utilised only once a year, but could be essential for the genetic sustainability of large populations or entire species. Hibernation sites are equally important and it is thought that some have been used for thousands of years. Summer roost locations may be more disparate and their locations often transitory, but a range of suitable sites close to good foraging areas will increase their value to bats.

Introduction

There are more native species of bat in the UK than any other group of mammals, with 16 known to be breading, and several vagrants. However, we know relatively little about their ecology or interactions with our environment. Recent use of high technology detection and tracking devices has given an insight into their short-term use of the landscape (Greenaway, 2001).

This is a brief review of our understanding of 'bat landscapes' making reference to published literature. We have concentrated on the interaction between bats, their habitats, wider landscape use, the importance of connectivity and threats posed by fragmentation.

The link between the dramatic changes in landscape character in the UK, especially during the 20th century, and plummeting bat populations may appear evident, but is difficult to verify because of few records prior to the 1970's. What is clear though is that alterations in habitats result in changes to the composition of bat communities (Stebbings, 1988) and we need to consider further the potentially far-reaching impacts of landscape management decisions.

We can only guess at the number of bats that could have been sustained by the rich and abundant habitat of the past, but we can be sure that it was many more than in our present comparatively degraded countryside. Anecdotal though much of this evidence may be, it is nevertheless strong (Altringham, 2003).

There has been increasing research into British bat species and their population dynamics. As a result our understanding is growing, but so too is the appreciation of our knowledge gaps. As small, highly mobile animals capable of flying long distances in the dark, bats present many problems for those wishing to study them (Stebbings, 1988).

It has been demonstrated that bats moving any distance have a preference to follow linear landscape features; hedges, woodland edges and rivers (Racey and Swift, 1985). On a dark night most bats will have less hesitation in crossing big open fields but such areas are normally avoided in early evening or on moonlit nights. A well-connected landscape therefore aids these movements by providing cover (Greenaway, 2001).

Greenaway (2001) made observations on the apparent importance of landscape features for bats along the North Downs in Surrey. This well-connected landscape provides a linkage to more distant habitats essential for the viability of bat populations. He suggests that over the last fifty years changes in the landscape beyond the Downs appear to have eliminated three of the fifteen species recorded in the Box Hill area. Landscape fragmentation appears to have damaged commuting routes for these species, and reduced their choice of hibernation and roosting sites. Limpens and Kapteyn (1991) also noted that linear landscape elements are important to bats, and their loss may have contributed to the decline of some species. Important research carried out by Altringham (2003) found that good hibernation sites appear to be rare and some attract large numbers of bats; loss of single sites can therefore have drastic consequences.

The impact of land management on roost site selection

Different bats derive different benefits in different seasons from their roosts, and the enormous diversity in ecology, habitat and climate has led to an equal diversity in roost site and roosting ecology (Altringham, 2003). The availability of both natural and artificial features, which offer conditions suitable for use by bats as roost sites, will influence their ability to utilise food resources, even when these occur in abundance. An understanding of their roosting ecology is therefore fundamental to an understanding of bats (Altringham, 1996).

Most bat species are highly mobile and occupy a wide variety of roost locations across the countryside according to their species, sex, seasonality, weather and food availability. Some species, such as the noctule (*Nyctalus noctula*), will commute many kilometres from their roosts each night to forage and are aware of the best places to visit for each season and set of conditions. Others, such as the brown long-eared bat (*Plecotus auritus*), have a more localised home range and rarely move more than 2-3 km from their preferred roosts (Greenaway, 2001).

Individual bats may use roosts separated by distances of a few metres or hundreds of kilometres. However, movements between roosts follow seasonal patterns and are not haphazard wanderings (Stebbings, 1988). From even a simple understanding of bat biology it is known that those features of the environment essential to bats' survival are being fragmented, degraded and destroyed (Altringham, 2003). A collaborative approach to landscape management is therefore as important as the maintenance of a well-connected countryside.

Migrations to swarming and hibernation sites

An important consideration for conservation land management is that there are sites that have been recorded with concentrated use by bats in only one season of the year. However their utilisation of the particular landscape feature may be fundamental to the sustainability of a local population or even an entire species.

The possibility that bats from widely dispersed colonies are faithful to one swarming site raises important issues about individual site protection. Bats from a large area might be less likely to find mates or hibernation sites should swarming sites be altered or destroyed. Protection of mating sites for swarming bats should be a conservation priority and habitat management should encompass a wide area around such sites (Parsons and Jones, 2003). It is therefore important that well-connected commuting corridors are maintained between these seasonal sites, some of which may be separated by many kilometres, is ensured to sustain their successful use by bats.

Landscape use during foraging

Bats are dependent on insects as a food source. Flight characteristics and bat physiology, as well as environmental factors, such as weather and available light, influence how different bats use feeding habitats. Hutson (1989) identified that there is little point in protecting roosts if feeding areas are destroyed. This aspect of bat ecology must, therefore, be understood in order to fully protect them. However, because bats range over large areas in search of insects, management needs to be considered over many square kilometres, even for small colonies (Stebbings, 1988).

Ancient woodland provides a vast diversity of structure, food and roosts together with a large bat community. Younger or plantation woodland has a simpler structure, a less continuous food supply, fewer roost sites and a less diverse bat community (Greenaway, 2001). It is therefore anticipated that alterations in habitats result in changes to the composition of bat communities (Stebbings, 1988). However, when these changes affect key sites such as hibernacula or places where bats gather to swarm, the impact could be devastating.

The importance of landscape connectivity

Bats can live more than twenty-five years. Once a young bat has established a successful pattern of movement and roost use, it can be repeated annually, so reliable environmental conditions benefit it greatly (Greenaway, 2001). This includes use of historic flyways or commuting routes (Limpens and Kapteyn, 1991). As bats are known to teach their young how to find roost sites and good foraging areas, some established commuting routes could have been in use for decades or even centuries by successive generations of bats.

Trees and woodlands influence local climate. Tree felling operations impact on the density of insect populations, which thrive in the shelter of leafy canopies. Surface wind speeds are increased following the removal of woodland and hedgerow shelter, making it more difficult for insects to fly on windy nights and therefore reducing the amount of food available for bats. The increased exposure of remaining trees also increases the wind-chill factor in winter, potentially reducing suitability of remaining hollow trees for use as hibernacula (Stebbings, 1988).

Conclusion

The UK has numerous protected sites, designated landscapes and nature reserves. However, good conservation practice relies on teamwork, since no individual can be an expert in all of the essential fields. In the past, despite their prominent place in Britain's vertebrate fauna, bats have often been forgotten or considered peripheral, but they now have a growing presence in the minds of the public and professionals (Altringham, 2003).

Landscape modification is often considered the principal cause of population decline in many bat species. Thus schemes for bat conservation rely heavily on knowledge about species: landscape relationships. However, few studies have quantified the possible influence of landscape structure on large-scale spatial patterns in bat communities (Jaberg and Guisan, 2001). Effective conservation of British bat species requires a range of conservation measures at local, regional, and national levels, covering all of the seasonal and ecological niches utilised by bats. Maintaining and enhancing landscape connectivity is vital.

References

Altringham J.D. (1996) *Bats: biology and behaviour*, Oxford University Press, Oxford.
Altringham J.D. (2003) The new naturalist, S.A. Corber, S.M. Walters, R. West, D. Streeter & D.A. Ratcliffe (Eds.) *British Bats*, Harper Collins Publishers, London.
Greenaway F. (2001) *The Box Hill Book of Bats*, The Friends of Box Hill, Surrey
Hutson, A.M. (1989) Where do bats feed and how do they get there? A review of survey techniques. *Batchat*,**12.** Nature Conservancy Council.
Jaberg, C. & Guisan, A. (2001) Modelling the distribution of bats in relation to landscape structure in a temperate mountain environment. *Journal of Applied Ecology*, **38**, 1,169-1,181
Limpens, H. & Kapteyn, K. (1991) Bats, their behaviour and linear landscape elements. *Myotis* vol. 29 (*Proceedings of the fifth European bat research symposium*, Denmark, 1990).
Parsons K.N. & Jones G. (2003) Dispersion and habitat use by *Myotis daubentonii* and *Myotis nattereri* during the swarming season: implications for conservation. *Animal Conservation*, The Zoological Society of London.
Racey P.A. & Swift S.M. (1985) Feeding ecology of *Pipistrellus Pipistrellus* (Chirpotera: Vespertilionidae) during pregnancy and lactation. 1. Foraging behaviour. *Journal of Applied Ecology* (1985), **54**: 205-215.
Stebbings RE. (1988) *Conservation of European Bats, IUCN/SSC Chiroptera Specialist Group*, Christopher Helm, London.

The UK Biodiversity Research Advisory Group (BRAG)

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What is BRAG and what does it do?

The UK Biodiversity Research Advisory Group (BRAG) was set up as a recommendation of *Science in action for biodiversity* (UK Biodiversity Research Working Group, 2001), and is organised through a Service Level Agreement between the Department for Environment, Food and Rural Affairs (Deffa) and Joint Nature Conservation Committee (JNCC).

BRAG comprises the key researchers, practitioners and policymakers concerned with delivering the UK Biodiversity Action Plan (BAP), and operates through a full-time Secretariat located within JNCC. BRAG exists to:

- promote and facilitate research that seeks to support delivery of the UK Biodiversity Action Plan (UK BAP)
- provide a forum for sharing ideas and information about biodiversity research
- co-ordinate UK engagement with related EU and international initiatives.

What key themes are being addressed?

At the core of its work, BRAG is identifying the key research issues associated with the following six cross-cutting themes:

1. Conservation of genetic and native species diversity
2. The impacts of introduced species
3. The roles of biodiversity in ecosystem function
4. Monitoring of biodiversity and evaluation of actions
5. Management of habitats and ecosystems
6. Developing tools to optimise policies to favour biodiversity.

Additionally, BRAG is considering the socio-economic constraints and opportunities associated with the delivery of the UK BAP, and seeks to respond to new developments and needs in biodiversity research.

1. Conservation of genetic and native species diversity

This programme is focused on native species, to provide better knowledge of genetic variation, functional ecology, and taxonomy of lesser-known groups. It addresses a wide range of issues, including:

Genetic conservation
- variation in native species

- factors eroding genetic diversity
- strategies and policies.

Functional Ecology
- critical functional marine species
- species at landscape scale.

Taxonomy
- lesser-known groups, especially marine.

2. Impacts of introduced species

This programme aims to develop the scientific basis to address the most significant, negative impacts of non-native, translocated and purposely-bred and released species. Issues under consideration include:

- protection of biodiversity from introduced species
- review nature and scale of environmental damage
- identify species, habitats and ecosystems most at risk; priorities for action
- how to manage and reverse damage; policy and practical responses.

3. The roles of biodiversity in ecosystem function

This programme aims to understand mechanisms and processes underpinning ecosystem function, using a whole-systems approach. Some of the issues being considered include:

- how to maintain large-scale ecosystem function
- distinguishing natural and anthropogenic ecosystem change
- role of marine microbial communities
- ecology of soils at landscape scale
- mechanisms and processes of change
- what is a healthy ecosystem?

4. Monitoring of biodiversity and evaluation of actions

This programme aims to develop the best approaches to measuring biodiversity at a range of spatial scales, e.g., regional, catchment and site scales for terrestrial habitats. Issues being addressed include:

- what are the best methods and indicators for measuring status and change?
- how do we identify causality and significance of change?
- how should we evaluate the success of our actions?

5. Management of habitats and ecosystems

This programme aims to develop the knowledge, best practice and tools to enable management for biodiversity at a range of spatial scales, e.g., regional, catchment and site scales for terrestrial habitats. A wide range of issues are being considered, including:

- implications for biodiversity of current and alternative management systems
- socio-economics of management options
- how good are current landscape management measures at delivering optimum landscapes for biodiversity?
- decision support tools.

6. Developing tools to optimise policies to favour biodiversity

This programme aims to develop tools to enable society to make decisions and put in place actions and policies based on good science, to optimise biodiversity in the context of sustainable development objectives. It aims to:

- improve the evidence base and techniques for objective and transparent decisions, and
- help explore mechanisms for more effective delivery of biodiversity objectives.

Socio-economic issues

Socio-economic constraints and opportunities for progress in biodiversity conservation require a full analysis, and so BRAG has worked closely with a wide range of experts, to:

- identify current and potential future research requirements to meet the needs of the UK BAP and developing country strategies
- evaluate the socio-economic implications and current status of research undertaken on the six themes
- identify links and potential synergies between UK BRAG socio-economic research and other biodiversity research programmes, and
- recommend a cost-effective process for delivering the research agenda.

Taking BRAG further

In addition to its work setting research priorities on these issues, BRAG has numerous roles, both nationally and internationally. The following are examples:

- building links with other national biodiversity research programmes via the European Platform on Biodiversity Research Strategy (EPBRS)
- offering advice on mechanisms for coordination and collaboration
- incorporating horizon scanning into analysis of research needs
- shaping research programmes and influence funding bodies.

How does BRAG communicate?

A key role for BRAG is to communicate ideas and engage the research, policy and practitioner community. It does this in a variety of ways:

- presence on the UKBAP website www.ukbap.org.uk/Groups/BRAG.htm
- links to related websites
- variety of publications and workshops
- full-time Secretariat (within JNCC).

The BRAG webpages offer users access to the latest meeting and research reports produced by the group, up-to-date information on research-related symposia and workshops, a variety of consultative exercises, and sources of biodiversity research information (via a developing research catalogue).

Reference

UK Biodiversity Research Working Group (2001) *Science in action for biodiversity: a report of the Biodiversity Research Working Group 1998-2001.* DEFRA. www.ukbap.org.uk/Library/BRWG06.PDF

Forest modelling based on landscape ecology concepts

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Abstract

There is an increasing need in landscape modelling to study landscape dynamics and to test related ecological hypotheses. While neutral models create fixed and artificial mosaics, explicit landscape models simulate realistic landscape scenarios through identified processes. There is a need for models to emerge at their interface, as it should be possible to simulate realistic landscape dynamics by means of statistical processes. This study proposes such a model based on an adaptation of the Gibbs process already used in forestry to define tree positions within a woodlot. We construct a potential function with pure attributive or pure geometrical operations in the landscape units (the woodlots) to reproduce their possible temporal modifications. The result is a real forested landscape progressively modified to fit the previously chosen property values. Realism and convergence with the adapted Gibbs process is much better than a basic selection of random operations on the same landscape units.

Introduction

Today, there is an increasing need for landscape modelling, as a response to various objectives, such as understanding changes in the environment or making decisions on landscape management. Models can offer spatial support for biophysical fluxes and movements of organisms, they can help to study landscape spatial patterns and landscape dynamics, and to test hypotheses related to ecological objectives (Fall and Fall, 2001). Spatial structures of landscapes that can be characterised by different parameters (e.g., heterogeneity, connectivity or fragmentation) influence many ecological phenomena, like plant and animal population dispersal, abundance, diversity, forest fire spread, etc. Many modelling studies have explored the possible relationships between landscape patterns and ecological processes by the use of artificially or more realistically generated landscapes. Neutral landscape models for instance produce an expected pattern in the absence of specific landscape processes (and with the help of random functions). Yet, such models are generally partial, non realistic and without dynamics (Turner *et al.*, 2001). At the opposite extreme, explicit landscape models reproduce processes explicitly described in the model, which are explanatory but often need detailed measurements to be validated and quite heavy work to be implemented. The aim of this study is to propose an intermediate model, focusing on landscape dynamics to reproduce artificial (or even real) landscape patterns on the basis of explicit statistical processes. We expect the model to be explanatory, although no ecological mechanism has been coded. After justification of our methodology, we detail how the Gibbs process has been adapted to categorical landscapes. An illustration of this model is given here on a forested landscape. The landscape has been partitioned into woodlot units that are modified and can (geometrically) interact to make the landscape evolve. Landscape properties are quantified with the evenness (heterogeneity) index.

Method

Neutral landscape models are generally used to generate patterns in the absence of the specific processes being studied. They are often grid-based models fixed and "non realistic", but with similar statistical properties to real landscapes. Potential (gravity) models are also used in spatial analysis to create virtual landscapes. We hypothesise that modelling the landscape as a patchy mosaic and modifying these patches to simulate their possible dynamics will lead to more realistic patterns than those of virtual models existing today. Patches are adjacent uniform polygons with an attribute value (the landscape class) and geometrical properties such as shape and gravity centre. All of them are susceptible to change according to landscape rules included by the model, progressively leading to a landscape evolution scenario. The main difficulty comes from the patchy structure of the landscape and manipulating polygons instead of elementary cells or pixels. In this context, geometrical "actions" applied to landscape units are only six fold; they can appear or disappear, dilate or erode, merge or divide, while it is possible to reduce them to gravity centre movements. Similarly, actions on attribute units can only change their value. With these definitions, we identify at least three ways to make a patchy mosaic evolve (without identified ecological processes):

i) To apply random actions (either geometrical or attributive) on random landscape units to modify the landscape and keep only its spatial configuration. We tested this option which has the disadvantages of not being explanatory and showing slow (or impossible) convergence;

ii) To apply precise actions on specific landscape units guided by spatial constraints, such as direction and/or intensity of unit modifications (e.g., Kernel density function). We did not explore this option which requires very detailed knowledge on the simulated landscape;

iii) To statistically apply actions on landscape units on the basis of already validated methods such as the Gibbs process (Stoyan and Penttinen, 2000). We implemented this option in various ways, either with a) attributive changes, or b) pure geometrical actions.

Gibbs process

In forestry, the marked Gibbs process has shown a powerful ability to plot trees within woodlots, with realistic 2D positions or attributive distributions. The process is based on an attraction/repulsion potential function, indicating how to place trees depending on their relative distances. This function represents non-parametric measures of explanatory statistics. Strong hypotheses are made with such a methodology: the woodlot is supposed to be at thermodynamic equilibrium and the approach supposes that it is possible to use a local rule to control the global evolution of woodlots. The present paper shows how marked Gibbs processes can be used for parametric description of the development of forest woodlands and their dynamics. A global cost function (the landscape "energy") is used to estimate the distance to the final expected configuration and to quantify the error made. The main difficulty of adapting the Gibbs process to patchy landscapes is the choice of the landscape unit's potential function. In a first step, we linked in a very simple manner landscape operations to the heterogeneity parameters (modified Shannon indices; Li and Reynolds, 1993). We used simple energy values with the assumption that a positive energy enhances local heterogeneity and a negative energy reduces it. Hence, each modification reduces the total landscape energy (i.e., summed over all the units) and corresponds to a quantified gain or loss of landscape heterogeneity (the controlled property). We identified two easy ways to test forms of potential functions:

Attributive landscape modifications

By ignoring the geometrical shape and properties of a landscape unit, it is possible to only modify the unit attribute (i.e., to change landscape class) in a way that reduces the cost function. Landscape units, to be modified, are randomly chosen and their neighbourhood energy (over all the adjacent units or "clique") is estimated. Here, energy is set to 1 if two adjacent units are of the same class and to -1 elsewhere. There exist a limited number of landscape class permutations for which an energy value can be estimated and summed. This adaptation of the Gibbs process seems relatively basic, but has the advantage of being applicable to every kind of categorical landscape.

Geometrical landscape modifications

It is much more difficult to adapt the Gibbs process to geometrical actions, because of the number of degrees of freedom imposed by the polygonal nature of landscape units (many pixels have to be moved together, with possible conflicts). One way to simplify this question is to reduce each unit to its gravity centre (with the hypotheses of convex and compact units) and to define its boundary with a simple rule, such as a nearest neighbours interpolation. By this means, units are reduced to points and their dynamics are assimilated to point movements similar to tree position optimisation in a Gibbs process. In addition to the simulation advantages of this approach, this adaptation of the Gibbs process ensures convergence and efficiency of the pattern's evolution. In this case, the energy potential is fixed to a high value if units are closed, and even higher if they have different classes, and low negative energy for higher inter-distances. All gravity centre pairs are examined here. This method also gives clues to a clear and even analytical interpretation of landscape dynamics (not shown).

Data and results

Results on a real French forested landscape show different dynamics according to the pure attributive and pure geometrical unit modifications, above. The site is located north of Ille-et-Vilaine (Brittany, western France) and is at ~31.9 m altitude. The landscape comprises approximately 769 ha of categorical mosaic, with 5 classes (the 5 dominant species), plus the background. Landscape dynamics are synthesised by the evolution of the energy and heterogeneity curves (Figure 1). In all tests, the forested landscape dynamics is simulated over 1,000 iterations and progressively evolves to the final state characterised by higher or lower evenness values. It is possible to quantify the convergence rates of each approach and to compare the initial and two final landscape modifications (Figure 2). All the algorithms involved here have been developed with Matlab® software and a typical run on this illustration has a ~23' duration on a Pentium III PC computer (for the second method).

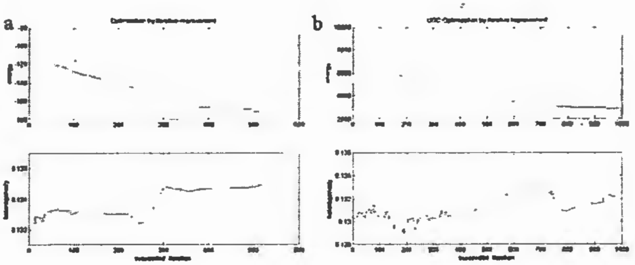

Figure 1. Evolution of energy (cost function) and heterogeneity arising from the attributive (a) and gravity centre (b) methods.

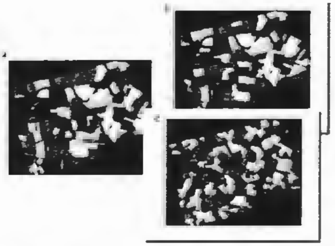

Figure 2. Evolution of Brittany landscapes resulting from the two methods. Landscape units are visualised with their identification number for the initial date (a) and their landscape attributes for the two final dates (b and c for first and second method respectively).

Discussion

Results show realistic colonisation of landscape units, though no ecological processes have been explicitly described in the model. Only the statistical Gibbs rule has been applied to woodlot shape or class. Landscape properties are fully controlled during the scenarios and these simulations offer interesting alternatives to functional, as well as neutral, landscape modelling. They allow testing of ecological hypotheses and focus on specific landscape properties, without neglecting the temporal dimension. We are currently developing further the geometrical adaptation of the Gibbs process to landscape evolution, based on the operations mentioned.

References

Fall, A. & Fall, J. (2001) A domain-specific language for models of landscape dynamics. *Ecological modelling* **141**: 1 – 18.

Li, H. & Reynolds, J.F. (1993) A new contagion index to quantify spatial patterns of landscapes. *Landscape ecology* **8**: 155 – 162.

Stoyan, D. & Penttinen, A. (2000) Recent applications of point process methods in forestry statistics. *Statistical science* **15 (1)**: 61 – 78.

Turner, M.G., Gardner, R.H. & O'Neill, R. (2001) Landscape ecology in theory and practice. Springer, New-York, pp. 401, new edition.

Modelling natural restoration potential of marginal olive groves in south Spain

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Abstract

Olive (*Olea europaea*) groves are the most frequent component of the landscape in the region of Andalucia, in the south of Spain, comprising more than 1,400,000 ha. The present landscape is the result of past interactions between different socio-economic factors that have led to enlargement of the area of olive groves. But history shows us that there have also been periods when the area has been reduced at a local or regional scale. In this sense, there is a clear link between historic processes and present landscape character.

This paper focuses on inappropriate olive groves in Andalucia. Although marginality takes many forms (it can be physical, ecological, economic, social or cultural, or a combination of any or all of these), a physical approach has been chosen, focusing on olive groves whose environmental attributes are not suited to sustainable management. Physical marginality has been defined on topographical factors (degree of slope) and edaphic factors. A GIS has been developed in order to distinguish between marginal and non-marginal olive groves. As a result, 16.4% of the surface of olive trees in Andalucia has been identified as marginal.

Marginal lands are likely to be abandoned and subsequently recolonised by natural vegetation. Olive groves could play a key ecological role in this process. On this basis, it is proposed that a simple restoration model should be used as a decision-support tool for policy-makers and managers. Distance to source of propagules (patches of trees and shrubs) has been considered fundamental to the process of succession. Accordingly, we estimate that 4% of all olive groves in Andalucia (25% of marginal olive groves) could be afforested solely by natural processes.

Introduction

Olive groves are of important ecological and environmental value. They often serve to protect against soil erosion and provide nest sites and food for birds. Olive groves dominate the landscape in the region of Andalucia, in the south of Spain, with 1,480,162 ha (Consejería de Agricultura y Pesca, 2003). The present landscape is the result of past interactions between different socio-economic factors that have led to enlargement of the area of olive groves. But history shows us that there have also been periods when the area has reduced at a local or regional scale. In this sense, there is a clear link between historic processes and present landscape character.

Some olive groves are situated on land unsuitable for agricultural purposes. Cultivation of this marginal land leads to environmental degradation, including erosion and is not sustainable.

The definition of marginality is based on physical attributes, such as, degree of slope, soil characteristics, climatological parameters and other factors. It is assumed that physical and economic marginality are inextricably linked and that the probability of these olive groves being abandoned is great. Some old olive groves are already neglected, although this is not generally the case, as yet.

After abandonment, natural processes led to restoration of natural vegetation cover. As a consequence of the high heterogeneity of mediterranean ecosystems, it is difficult to predict the future trajectory of succession.. Nevertheless, decison-support tools are needed to aid management of these areas.

Objectives

A decision-support tool has been developed, based on land evaluation and a plant cover restoration model. Identifying olive groves with a high probability of abandonment due to the low capability of the land has been the first step to determine a vegetation restoration model.

Methods

The land evaluation model for olive plantations has been based on land evaluation assessment (De la Rosa and Moreira, 1987; USDA, 1961; FAO, 1976), recognising as limiting factors; the degree of slope, and a number of soil properties (depth, texture, fertility, $CaCO_3$ content). Digital map information on degree of slope (from a Digital Elevation Model with topography at 20 m resolution) and soils (1:400,000 scale Soil Type Map of Andalucia, CSIC – IARA, 1989) have been used. The distribution of olive groves in the region has been determined from the Crop Map of Andalucia (Consejeria de Agricultura y Pesca, 2001), which was derived from 1:60,000 scale aerial photographs.

Five land suitability classes for olive cultivation were specified for each 20 x 20 m polygon using Arc View 3.2.; 1 (highly suitable), 2 (suitable), 3 (moderately suitable), 4 (marginal) and 5 (very marginal). The last two classes were considered to be at high risk of abandonment.

Marginal lands are inclined to be abandoned and, consequently, to be recolonised by natural vegetation. On this basis a simple restoration model is proposed as a decision-support tool for policy-makers and managers. Distance to source of propagules (patches of trees and shrubs) has been considered as the key factor for the restoration model. The GIS layer of actual vegetation cover (Mapa de Vegetación y Recursos Forestales de Andalucía) has been a 1:100,000 scale map based on the Forest Map of Spain (Ruiz de la Torre, 1990). An index of colonisation potential was defined for each natural vegetation type according to the capacity for establishing wood or shrubland communities in adjacent marginal olive polygons. A distance of 500 m was considered to be the maximum effective dispersal distance, considering the variability of the reproductive strategies of mediterranean plant species and the scale of work (Debussché *et al.*, 1985; Debussché and Lepart, 1992; Herrera, 1995; Guitián and Sánchez, 1992).

Through the intersect module of ArcView, marginal olives polygons situated less than 500 m from natural vegetation, with the potential to be colonised by forest or shrub communities, were distinguished from marginal olive groves with no such opportunities.

Results

According to the land evaluation and using the Digital Crop Map of Andalucia, 16.4% (218,780 ha,) of olive plantations in Andalucia grow under marginal conditions. Roughly 25% of this marginal area has no potential for restoration of forest and shrub vegetation (**Figure 1**), equivalent to 4% of the area of olive groves in Andalucia.

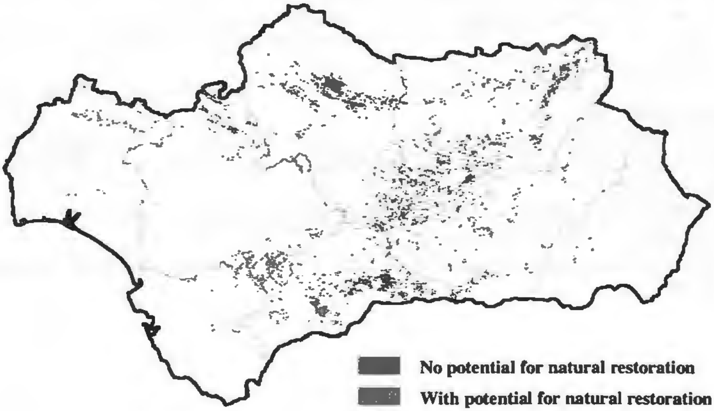

Figure 1. Map of potential for natural restoration of marginal olive groves to forest and shrub communities in Andalucia, south of Spain.

Discussion

This is certainly a prudent model. The polygons identified as having no potential for restoration are those clearly isolated and with a high degree of marginality in terms of physical factors. In these areas it would be necessary to develop specific plans and interventions to restore vegetation, to avoid the risks involved in abandonment and to redefine the landscape. Whereas, the marginal olive groves with opportunities for restoration by natural means require more detailed research to determine the short to medium term consequences of abandonment.

References

Consejería de Agricultura y Pesca, (2001) *Mapa Digital de Cultivos y Aprovechamientos,* Junta de Andalucía, Sevilla.

Consejería de Agricultura y Pesca, (2003) *El olivar andaluz,* Junta de Andalucía, Sevilla.

CSIC - IARA (1989) *Mapa de suelos de Andalucía,* Sevilla.

Debussché, M.; Lepart , J. & Molina, J. (1985) La dissémination des plantes á fruits charnus par les oiseaux: rôle de la structure de la vegétation et impact sur la succession en région mediterranéenne. *Acta Oecologica. Oecologia Generale* **6**: 65-80.

Debussché, M. & Lepart, J. (1992) Establishment of woody plants in Mediterranean old fields: opportunity in space and time. *Landscape Ecology* **6**: 133-145.

De la Rosa, D. & Moreira, J.M. (1987) *Evaluación Ecológica de Recursos Naturales de Andalucía. Aproximación al conocimiento necesario para planificar el uso y protección de las tierras,* Agencia de Medio Ambiente, Junta de Andalucía, Sevilla.

FAO (1976) *A framework for land evaluation.* Soils Bullettin, 32, Roma (http://www.fao.org/docrep/X5310E/X5310E00.htm).

Guitián, J. & Sánchez, J.M. (1992) Seed dispersal spectra of plant communities in the Iberian Peninsula. *Vegetatio* **92**: 157-164.

Herrera, C. M. (1995) Plant-vertebrate seed dispersal systems in the Mediterranean: ecological, evolutionary and historical determinants. *Annual Review of Ecology and Systematics* **26**: 705-727.

Ruiz de la Torre, J. (1990) *Memoria del Mapa Forestal Español,* ICONA - Ministerio de Agricultura, Pesca y Alimentación, Madrid.

USDA (1961) *Land capability classification,* U.S.D.A. Soil Conservation Service, Handbook nº 210, Washington.

New machine learning tools for predictive vegetation mapping after climate change: Bagging and Random Forest perform better than Regression Tree Analysis

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Abstract

More and better machine learning tools are becoming available for landscape ecologists to aid in understanding species-environment relationships and to map probable species occurrence now and potentially into the future. To that end, we evaluated three statistical models: Regression Tree Analysis (RTA), Bagging Trees (BT) and Random Forest (RF) for their utility in predicting the distributions of four tree species under current and future climate. RTA's single tree was the easiest to interpret but is less accurate compared to BT and RF which use multiple regression trees with resampling and resampling-randomisation respectively. Future estimates of suitable habitat following climate change were also improved with BT and RF, with a slight edge to RF because it better smoothes the outputs in a logical gradient fashion. We recommend widespread use of these tools for GIS-based vegetation mapping.

Introduction

The world's climate has always been undergoing change but now there are new reports almost every month linking recent changes in the climate to some biological trend. It has been estimated that the composition of one-third of the earth's forests could change markedly due to climate changes associated with a doubling of atmospheric CO_2 (e.g., Melillo, 1999). Plant species are expected to shift in range and relative abundance as the climate changes, and this has been the thrust of our research for the eastern half of the United States (e.g., Iverson & Prasad, 1998; 2001). For this paper, we tested three statistical prediction tools for modelling current and potential future suitable habitat under climate change, for four common tree species from eastern North America.

Statistical Models

Regression Tree Analysis (RTA)

RTA differs from classical statistical methods in that it constructs a set of decision rules via recursively partitioning on the predictor variables (Breiman *et al.*, 1984; Iverson and Prasad 1998). These rules allow for the possibility of interactions and non-linearities among variables (Moore *et al.*, 1991), and enable mapping of the predictors with the greatest influence on distributions geographically, which can provide insights into the spatial influence of the predictors (Iverson & Prasad, 1998). RTA predictions can, however, be unstable in that small changes in data can produce largely different models.

Bagging Trees (BT)

Bagging uses a regression tree technique as well but uses many (30-80) training data sets by resampling the data with replacement (on average 67% of cases appear in bootstrap sample), then averages the outputs, so that the variance component of the generalization error is reduced (Breiman 1996). The portion of the data drawn into the sample in a replication is known as the "in-bag" data while the portion not drawn is the "out-of-bag" data. The "out-of-bag" data is not used to build or prune any tree but used to give better estimates of node-error and other generalisation errors for bagged predictors (Breiman, 1996). The main disadvantage of bagging is that the large number of models makes it difficult to interpret the results, especially for species that have relatively unstable models. In contrast, for species with stable models, the interpretation of the original RTA tree may be suitable.

Random Forest (RF)

Random Forest is relatively new, but has been shown to produce very accurate predictions without overfitting models to the data (Breiman 2001). RF is essentially very similar to BT in that bootstrap samples are drawn to construct multiple trees. The differences, however, are that the each tree is grown with a randomised subset of predictors, i.e., the number of predictors (initially fixed) used to find the best split at each node is a randomly chosen subset of the total number of predictors, and that a very large number (500-2000) of trees are grown (hence 'forest' of trees). Like BT, the trees are grown to maximum size without pruning and aggregation takes place by averaging the trees. A main advantage of RF is that the output depends mainly on only one user-selected parameter, i.e., the number of predictors to be chosen randomly at each node, and even this parameter is not highly sensitive. Obviously, it is not possible to interpret each of the trees in RF, but the procedure does provide tables on relative importance among predictor variables.

To our knowledge, we are the first to use BT and RF in the field of ecology (Prasad *et al.,* submitted). Furthermore, it appears that only one biologically based study has used RF (Furlanello *et al.,* 2003).

Methods

We used the three statistical modelling techniques on a data set consisting of: (1) tree importance values, based on tree density and basal area from over 100,000 inventory plots in the eastern U.S. (Iverson and Prasad 1998), for four tree species (red spruce (*Picea rubens*), jack pine (*Pinus banksiana*), white ash (*Fraxinus americana*) and chestnut oak (*Quercus montana*)); (2) 36 environmental (predictor) variables describing climate, soil, land-use, landscape, and topography of each grid cell; and (3) potential future climate based on the Canadian Climate Centre (CCC) global circulation model (Boer *et al.,* 2000). Each cell was 20 x 20 km, for a total of 9,782 cells in the study area. We first ran the models using the dataset with current climate variables and then re-ran the models using CCC variables to get the, CCC-derived, future predictions of suitable habitat. We used the statistical software R (R Development Core Team, 2003), based on the S language (e.g., Chambers & Hastie, 1993), to run RTA ("rpart", Therneau and Atkinson, 1997), BT ("ipred", Peters *et al.*, 2002), and RF ("randomForest", Liaw & Wiener, 2002).

We used three map similarity measures to conduct a pixel-by-pixel comparison (actual vs. predicted) among the three models and four species; correlation, Kappa, and fuzzy Kappa (Hagen 2003). With the Kappa statistic, the level of agreement between maps is based on a contingency table, which details how the distribution of categories in map A differs from map B. Fuzzy Kappa recognizes that categories are often not crisp, i.e., there are grades of similarity between pairs of cells in two maps. The fuzziness of location is set with a function that defines the level to which the neighbouring cells influence the target cell (Hagen 2003).

Results and discussion

Although space does not allow displaying of maps for this paper, the RTA, BT, and RF maps all replicated current distributions well. However, the correlation, Kappa, and fuzzy Kappa tests verified that the BT and RF models were clearly superior to RTA in predicting current distributions of the four species studied (Table 1). The "multiple-perturbed" trees in BT and "multiple-perturbed-randomised" trees in RF allow for better prediction capabilities. While the outputs of BT and RF are fairly similar, we prefer the output of RF for two reasons: 1) RF slightly outperformed BT in most statistics; and 2) RF smoothes the response more than BT. We believe this smoothing provides an advantage because the IVs grade smoothly from lower to higher and there are no abrupt changes or skips in IV classes. These abrupt changes happened more in BT.

Table 1. Correlation and Kappa scores for RTA, BT, and RF among four tree species.

	Correlation			Kappa			Fuzzy Kappa		
	RTA	BT	RF	RTA	BT	RF	RTA	BT	RF
Picea rubens	0.864	0.945	0.953	0.576	0.586	0.589	0.660	0.659	0.660
Pinus banksiana	0.734	0.896	0.919	0.430	0.447	0.477	0.497	0.517	0.539
Fraxinus americana	0.693	0.907	0.923	0.357	0.417	0.441	0.375	0.443	0.455
Quercus montana	0.795	0.940	0.947	0.506	0.513	0.532	0.567	0.579	0.590

Predictions of potential future suitable habitat were biogeographically reasonable and logical, especially for BT and RF. The RF models were slightly more biogeographically realistic because of the smoothed output, though we realise that this kind of reasoning about potential future habitat is subjective and fraught with uncertainty because many factors, including many not considered here, will influence the final distribution. Additional support for the superiority of the RF model compared to BT has also been shown by others (Hawkins and Musser 1999; Meyer *et al.*, 2003; Svetnik *et al.*, 2003).

We propose to use RTA, BT, and RF as a toolbox for species modelling. The superior prediction capability of RF is best used to map future scenarios, while RTA and to some extent BT can be used for their interpretive abilities. If the individual trees (among BT) are similar, a single RTA tree can be used to map what predictors are driving the distribution of the species spatially; a very unique aspect of RTA that offers additional insights into the species distribution (Iverson and Prasad, 1998; Iverson *et al.*, 1999). We are currently using this procedure to model the future climate distributions of 135 eastern US tree species. Our website (http://www.fs.fed.us/ne/delaware/4153/4153.html) has an online atlas of an earlier version of this work (RTA only) which will be updated using these new tools. We advocate this package of tools for widespread use in predictive biological mapping.

References

Boer, G.J.; Flato, G.M. & Ramsden, D. (2000). A transient climate change simulation with historical and projected greenhouse gas and aerosol forcing: projected climate for the 21st century. *Climate Dynamics* **16**, 427-451.

Breiman, L. (1996). Bagging predictors. *Machine Learning* **24**, 123-140.

Breiman, L. (2001). Random forests. *Machine Learning* **45**, 5-32.

Breiman, L.; Freidman, J.; Olshen, R.; & Stone, C. (1984). Classification and regression trees. Wadsworth, Belmont, CA. 358 pp.

Furlanello, C.; Neteler, M.; Merler, S.; Menegon, S.; Fontanari, S.; Donini, A.; Rizzoli, A. & Chemini, C. (2003). GIS and the random forest predictor: integration in R for tick-borne disease risk assessment. K. Hornik, F. Leisch, & A. Zeileis, (Eds.), *Proceedings of the 3rd International Workshop on Distributed Statistical Computing*, 1-11, Vienna.

Hagen, A. (2003). Fuzzy set approach to assessing similarity of categorical maps. *Geographic Information Science* **17**, 235-249.

Hawkins, D.M. & Musser, B.J. (1999). One tree or a forest? Alternative dendrographic models. *Computing Science and Statistics* **30**, 534-542.

Iverson, L.R. & Prasad, A.M. (1998). Predicting abundance of 80 tree species following climate change in the eastern United States. *Ecological Monographs* **68**, 465-485.

Iverson, L.R. & Prasad, A.M. (2001). Potential changes in tree species richness and forest community types following climate change. *Ecosystems* **4**, 186-199.

Iverson, L.R.; Prasad, A.M.; Hale, B.J. & Sutherland, E.K. (1999). *An atlas of current and potential future distributions of common trees of the eastern United States.* General Technical Report NE-265. Northeastern Research Station, USDA Forest Service.

Liaw, A. & Wiener, M. (2002). Classification and regression by Random Forest. *R News* **2/3**, 18-22, URL: http://CRAN.R-project.org/doc/Rnews/.

Melillo, J.M. (1999). Climate change: Warm, warm on the range. *Science* **283**, 183-184.

Meyer, D.; Leisch, F. & Hornik, K. (2003). The support vector machine under test. *Neurocomputing* **55**, 169-186.

Moore, D.E.; Lees, B.G. & Davey, S.M. (1991). A new method for predicting vegetation distributions using decision tree analysis in a geographic information system. *Journal of Environmental Management 15*, 59-71.

Peters, A.; Hothorn, T. & Lausen, B. (2002). Ipred: improved predictions. *R News* **2**, 33-36, URL :http://CRAN.R-project.org/doc/Rnews/.

Prasad, A.; Iverson, L.; Liaw, A. (submitted). A comparison of four computer intensive modelling techniques: RTA, MARS, Bagging Trees and Random Forest, for predicting current and future distribution of trees. *Ecosystems.*

Svetnik, V.; Liaw, A.; Tong, C.; Culberson, J.C.; Sheridan, R.P. & Feuston, B.P. (2003). Random forest: a classification and regression tool for compound classification and QSAR modeling. *Journal of Chemical Information and Computer Science* **43**, 1947-1958.

Therneau, T.M. & Atkinson, E.J. (1997). *An introduction to recursive partitioning using the RPART routines.* Technical Report #61. Mayo Clinic. Rochester, MN.

Integrating indicators of biodiversity into a tool to enhance participatory planning of landscape change: a component of the VisuLands project

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Abstract

There is an increasing call for indicators to guide both policy and practice in land management, and an associated proliferation of recommendations and suggested schemes at a variety of spatial scales. Many of the proposed indicators are being developed to monitor the consequences of decisions and, if necessary, trigger corrective action. However, the use of proactive indicators, allowing predictions of the impacts of potential actions, would appear to be inherently more appealing. The potential for incorporating such indicators into a participatory planning tool is outlined here. VisuLands is a complex EU-5th Framework project that seeks to develop a toolkit to enhance public involvement in the management of landscape change. A core premise is that it is possible to create realistic visualisations of the consequences of land use decisions, particularly those involved in discrete allocation of use. The technical barriers to creating such visualisations have declined recently due to developments in software, computing power, and digital data availability. However, there remains a significant challenge of representing future values of other landscape functions, such as amenity, productivity, biodiversity in the same convincing manner. The scope and use of indicators of biodiversity required to fulfil this role are discussed here.

Indicators of biodiversity

There has been a huge proliferation in the derivation and recommendation of the use of indicators (Büchs, 2003). Schemes have been proposed encompassing scales from the global to the local. For example, the Organisation for Economic Co-operation and Development (OECD) recommend a set of key environmental indicators (OECD, 2001), a number of UK indicators of sustainable forestry have been proposed (Forestry Commission, 2002), and indicators at the scale of the forest stand have been suggested (Larsson, 2001). All too often, the context and subsequent usefulness of the indicators is loosely developed. Another feature is that many of the proposed indicators are concerned with monitoring the state of the environment. This is clearly captured in the Pressure-State-Response (PSR) model of decision-making as proposed by the OECD, and the expanded variant of Driving Force–Pressure-State-Impact-Response (DPSIR) model of the European Environment Agency (EEA) (EEA, 1999, Weber and Hall, 2001). The EEA also clarify the purpose of indicators and distinguish descriptive (a measure of what is happening), from performance (a measure against a set target), and efficiency indicators. However, there is an inherent danger in reliance on schemes of performance indicators. The impact will have already occurred before the response is triggered, and there is an inherent assumption that corrective action is possible. However, many commentators (and indeed many landscape ecology case studies) have shown that such problem-solving is rarely easy, particularly when the dimensions of the problem encompass ecological processes at a range of temporal and spatial scales. It would be better to avoid the impacts in the first place. This is clearly the focus of much targeted conservation effort and legal protection for special sites, but the incorporation of such values in the wider planning context and in decisions that do not concern the designated sites is more challenging.

Participatory planning tools

The recent focus on sustainability in land management has placed a greater emphasis on taking cognisance of future values. One response has been to broaden the base for decision-making so that more views are sampled during the planning process. This trend towards participatory planning has led to a demand for methods of assessing future options, and a requirement for new tools. The production of visualisations of future landscapes (and even virtual landscapes) is now technically feasible, and allows some assessment of the visual impact of land use changes (Sheppard and Harshaw, 2001). However, is it possible to visualise (in the broadest sense) the consequences for other land use values?

The scope and use of indicators within the VisuLands toolkit

The EU Quality of Life Shared Cost project VisuLands (QLK5-CT-2002-01017) seeks to develop a set of tools, at least in a generic sense, to aid participatory planning. The concept includes the visual representation of future land use choices, and a supporting assessment of the implications on other landscape values, including cultural, environmental and economic. The methods are being developed and applied in case study areas in the UK, Sweden, Norway, France, Switzerland, France and Portugal.

Developments in visualisation techniques, including virtual reality, have made it possible to provide persuasive images of new landscapes. However, providing a means by which users can assess the non-visual qualities of such landscapes is more problematic. It is proposed that this can be tackled within the VisuLands project by applying a sub-set of indicators. In particular, indicators that are descriptive and provide decision criteria (Failing and Gregory, 2003) are needed rather than performance and monitoring indicators. There is also a requirement for indicators that can be utilised proactively (testing scenarios before they are implemented), rather than reactively (testing after implementation) (Lawson *et al.*, 2003). Finally, the requirement is for indicators of change in 'state' and 'impact' (using EEA terminology); the driving forces and pressures will be identified in constructing the scenario, whilst the response will be the decision informed by the visualisation and supporting analysis.

A review of potential indicators has been conducted. There is an obvious tension between the indicators that are easy to calculate – and those that have the desired theoretical under-pinning and relevant scope. Three broad categories of indicators for biodiversity were identified and are summarised in Figure 1. We distinguished indicators of landscape composition, indicators of landscape configuration, and a catch-all group of indicators that reflect species or habitat suitability; the latter place particular demands on supporting data and models but permit the most specific predictions.

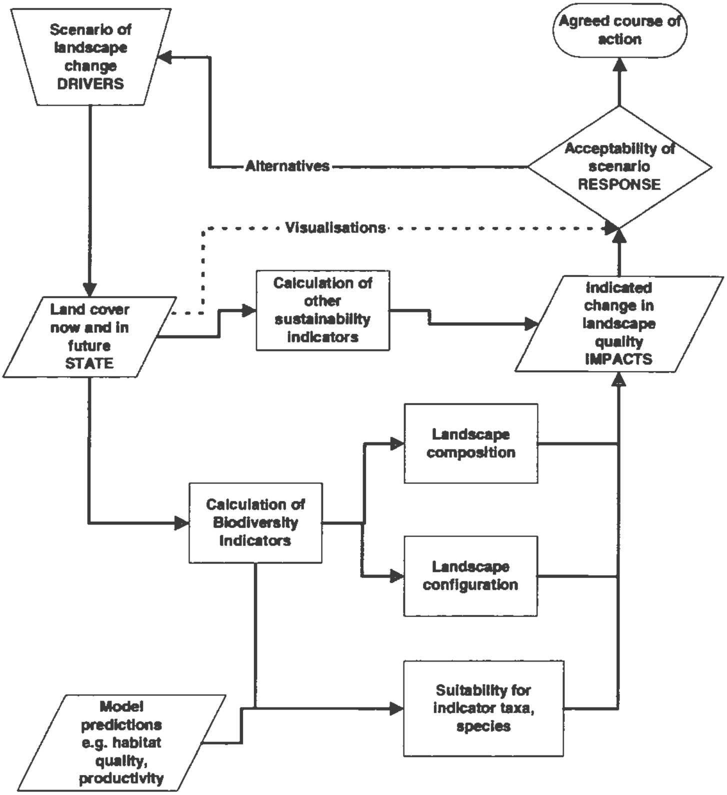

Figure 1. Scheme for use of indicators of biodiversity within the VisuLands project.

Next steps

Further selection of suitable indicators will be made on the basis of ease of calculation and visual representation, simplicity of understanding, and transferability of results. There are trade-offs to be established between simplicity (e.g., direct measures applied to land cover maps), and specificity. Although species-based approaches are desirable, they place heavy demands on data and ecological knowledge and these will not always be available for the type of application intended by the project. A hybrid approach, combining indicators based both on metrics and on generic focal species may be most suitable, incorporating clear statements of assumptions and uncertainty. The final selection requires input from end users to establish priorities for indicators, the level of sophistication required, and the scope of the scenarios to be modelled.

Acknowledgements

Thanks to colleagues at Forest Research, and within the VisuLands project, for stimulating discussions.

References

Büchs, W. (ed) (2003) Biotic indicators for biodiversity and sustainable agriculture. Elsevier, Amsterdam, 550pp.

EEA (1999) Information for improving Europe's environment. An introduction to the European Environment agency. EEA, Copenhagen.

Failing, L. & Gregory, R. (2003) Ten common mistakes in designing biodiversity indicators for forest policy. *Journal of Environmental Management* **68**: 121-132.

Forestry Commission (2002) UK Indicators of Sustainable Forestry. Forestry Commission and Forest Service (Northern Ireland), Edinburgh.

Larsson, T.-B., et al (2001) Biodiversity evaluation tools for European forests. *Ecological Bulletins* **50**: 1-237.

Lawson, S.R., Manning, R.E., Valliere, W.A. & Wang, B. (2003) Proactive monitoring and adaptive management of social carrying capacity in Arches National Park: an application of computer simulation modeling. *Journal of Environmental Management* **68**: 305-313.

OECD (2001) Key environmental indicators. OECD Environment Directorate Report Organisation for Economic Cooperation and Development.

Sheppard, S.R.J. & Harshaw, H.W. (eds) (2001) Forests and landscapes: linking ecology, sustainability and aesthetics. IUFRO Research Series 6. CABI and IUFRO, Wallingford xxiv + 294.

Weber, J.-L. & Hall, M. (2001) Towards spatial and territorial indicators using land cover data. EEA Technical Report 59, European Environment Agency, Copenhagen.

Priority woodland in the landscape for two bat BAP species: the importance of ancient trees and woodlands

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Abstract

All 16 species of bat in the UK and Eire are on Annex IV of the EU Habitats Directive. Five are also on Annex II and are Priority Species under the UK's Biodiversity Action Plan (BAP). Three of these have high dependence on ancient trees and woodlands for roosting and foraging requirements; *Rhinolophus hipposideros* (lesser horseshoe bat), *Myotis bechsteinii* (Bechstein's bat), and *Barbastella barbastellus* (barbastelle bat). This paper looks at the requirements of the Bechstein's bat and the barbastelle bat and makes recommendations for tree and woodland management to help conserve them.

Bechstein's bat

The Bechstein's bat is a tree-dwelling bat distributed in Europe from the Iberian Peninsula east to the Ukraine and Moldova, north to southern Sweden. Localised populations are scattered in southern England and Wales (Mitchell-Jones *et al.*, 1999). It is considered one of the rarest bats in western Europe, and one of the UK's rarest mammals with a 1995 British population estimated at 1,500 individuals (Harris *et al*, 1995). Population decrease has been reported in most of its range, the significant decline in forest cover said to be a major cause. Archaeological excavations show a high proportion of Bechstein's bones in comparison with other bat species, indicating that 3,000 years ago it was much more common than today (Stebbings, 1988). It is on the IUCN Red List of Threatened Species as vulnerable (Hutson *et al.*, 2001).

The Species Action Plan (UK Biodiversity Group, 1998) identifies objectives and targets:

- Maintain the known range and populations
- Increase the national population size of this species by improving woodland age structure to enhance roosting and foraging opportunities.

It goes on to say the retention of old trees and woodland around roost sites is essential to the survival of the Bechstein's bat. Detailed population assessment and monitoring are currently difficult because little is known about the bat, so more research into its ecology and behaviour is needed. It will be important to find and protect further key (particularly maternity and hibernation) roosts. Factors causing loss or decline include further loss and fragmentation of open ancient deciduous woodland habitat, and loss, destruction and disturbance of roosts or potential roosts (particularly in old trees) (UK Biodiversity Group, 1998).

Although the Bechstein's bat is known to use mainly broadleaf woodland, little is known about roosting and foraging requirements and this makes it difficult to advise on habitat requirements and makes them vulnerable to habitat management methods. A study by Vincent Wildlife Trust of Bechstein's bats in Dorset found that during the summer females use large mature broadleaf trees with rot/woodpecker holes and dead branches; they will move roost using different trees until they produce their young. Their foraging distance from the roost was relatively small (0.96km) compared with other bats. Because of this it may be that Bechstein's bats from the same colony will compete with each other for food, but this is balanced by the reduced commuting time; more time can be spent eating, which is especially important for pregnant and lactating females. However, this does mean that further reduction of already small woodlands that host Bechstein's bats can have devastating effects on the viability of its colony since the bats' limited range limits their choice of alternative foraging areas in these instances (Schofield & Morris, 2003).

Both the Dorset and a Sussex study found Bechstein's to prefer closed canopy areas with well-developed understorey, especially near water. The understorey moderates temperatures and wind speeds, and maintains higher humidity. Although tree lines and hedgerows were found in the Sussex study to be of marginal importance to Bechstein's as they prefer to forage in woodland, they are probably of more importance in fragmented landscapes.

Barbastelle bat

The barbastelle occurs throughout Europe except Iceland, Northern Ireland, Scotland, most of Scandanavia, Estonia and much of southern Europe (Mitchell-Jones *et al.*, 1999). Although distributed across Wales and southern England, it is thought to have been significantly under-recorded. Records for Britain up to 1995 had been too low and scattered to represent a viable breeding population, leading to the hypothesis that a population of perhaps 5,000 individuals had to exist but remained undiscovered (Harris *et al.*, 1995). Since then, however, five colonies have been discovered, probably due to improved detecting technology (Greenaway, 2001). Population decrease has been reported in most of Europe (JNCC, 2004) and, it is on the IUCN Red List of Threatened Species as vulnerable (Hutson *et al.*, 2001).

The Species Action Plan says that threats to this species are poorly understood, but its low population density and slow population growth make it particularly vulnerable to factors such as:

- Further loss and fragmentation of ancient deciduous woodland habitat
- Loss/destruction/disturbance of roosts/potential roosts in buildings, trees and underground
- A reduction in numbers of insect prey due to habitat simplification acting through factors such as fertiliser use and intensive grazing.

Although barbastelle ecology is not well known, it is believed to be mainly an upland and forest species, with a liking for wooded river valleys and woodland containing water bodies. Its diet is highly specialised, consisting mainly of small lepidopteran species. It forages in mixed habitats and seems to prefer cracks and crevices in wood for breeding, such as those in old or damaged trees. Hibernation sites include trees, caves and underground structures.

Studies of a nursery colony in West Sussex found that the adult females formed continually shifting, dividing and regrouping roosts within the woodland between April and September, with the largest groupings occurring in August. Radio-tracking identified 56 barbastelle roosts in this area, of which 52 were in trees (pre-1995 records identified a preference for man-made and underground structures, probably because bats are difficult to locate in trees) (Greenaway, 2001). Roosts within large hollow trees surrounded by dense holly understorey were favoured in cold winters, although they will use buildings or underground sites if the cold spell is lengthy. During spring and autumn, even if air temperatures get close to freezing, roosting barbastelles can be found relatively exposed in trees cracks caused by storms, narrow splits and behind loose bark of old trees. The study also found that dead tree stumps projecting above the holly into sunshine are often chosen as roosts in warmer conditions. In late spring pregnant females tended to form new roosts (communally) in splits and cracks, whilst still returning to their original foraging areas at night, and just before birth gather together in a more protected space within a tree. The holly understorey is thought to moderate temperatures and wind speeds, and maintain higher humidity; all thought to benefit barbastelles in different ways depending upon lifecycle stage. Both the west of England and the Welsh colonies are in wooded valleys with relatively high humidity (Greenaway, 2001).

Radio tracking studies by the Vincent Wildlife Trust have shown individuals to forage up to 18km away from the day roost. In this way they probably avoid competing with other members of the colony for food (VWT 2002). However, foraging areas up to 3km away are important for nursery colonies, and juveniles forage up to 7km away about six weeks after birth. Flightlines to foraging areas are therefore important both near to and some distance from the roost. Partly because of differing habitats occupied by smaller moths as the seasons change, so foraging habitats for barbastelles change; meadows and hedgerows in summer, quite dense woodland in winter, and riverine in spring. (Greenaway, 2001). Connectivity to a variety of habitats from the roost area is therefore important.

Studies indicate that a complex ancient/semi-natural woodland structure is needed to provide the range of roost sites breeding barbastelles prefer. Old trees with loose bark, dead hollow trees, hollow branches, windthrow and storm damage are important. Also important are new roost sites to replace those lost by natural processes. Although this type of habitat would have been common in years gone by, modern management techniques make it a rarity today.

Recommendations with respect to tree and woodland management

Bechstein's bat

- Identify and protect existing foraging areas, particularly those within 2km of maternity roosts (more than recorded foraging area to act as a buffer)
- Where maternity roosts are to be designated SSSI/SAC, include their foraging area
- Manage woodland around roosts to promote closed canopy with understorey
- Safeguard potential habitat; mature broadleaved woodland and copses, hedgerows and tree lines
- Avoid felling or surgery to large mature broadleaved trees within 1km of known roosts, maintain these trees wherever possible in other areas

Barbastelle bat

- Identify and protect existing foraging areas
- Where maternity roosts are to be designated SSSI/SAC, include their foraging area
- Maintain unmanaged ancient and semi-natural woodland to keep a range of tree ages and conditions and understorey
- Avoid felling or surgery to large mature broadleaved trees, especially those with splits, cracks and holes; where necessary to make safe, consider removing branches and leaving trunks
- Maintain and gap up existing and potential hedge lines leading from roosting woodlands to river corridors and other foraging areas

Woodland managers

There is a training need for all involved with tree related occupations so they can identify and locate signs of bats and bat roosts and take appropriate action. However, a precautionary approach must also be taken as bats do not always conform to current thinking on habitat use; practitioners must be aware that new research changes current thinking. Habitat enhancement is a complex issue, and survey to record existing species should be undertaken preferably before enhancement takes place; non-specific habitat enhancement for bats may result in non-target species being attracted at the expense of vulnerable species. However, diverse ancient tree and woodland habitats are important for these vulnerable species, and woodland size and connectivity in relation to the roosts' locations is important. Careful management of these is crucial to the maintenance (and hopefully enhancement) of populations.

References

Greenaway, F. (2001) The Barbastelle in Britain. *British Wildlife*: **12**: 326-334.

Harris, S., Morris, P., Wray, S. & Yalden, D. (1995) *A review of British mammals: population estimates and conservation status of British mammals other than cetaceans.* Joint Nature Conservation Committee, Peterborough.

Hutson, A. M., Mickleburgh, S. P. & Racey, P. A. (comp.) (2001) *Microchiropteran bats: global status survey and conservation action plan.* IUCN/SSC Chiroptera Specialist Group. IUCN, Gland, Switzerland and Cambridge, UK.

Joint Nature Conservation Committee (2001) *Habitat management for bats.* JNCC, Peterborough.

Macdonald, D. W. & Tattersall, F. (2001) *Britain's Mammals; the challenge for conservation.* People's Trust for Endangered Species, London.

Mitchell-Jones, A. J., Amori, G., Bogdanowicz, W., Krystufek, B., Reijnders, P. J. H., Spitzenberger, F., Stubbe, M., Thissen, J. B. M., Vohralik, V., & Zima, J. (1999) *The Atlas of European Mammals.* T. & A.D. Poyser, London.

Richardson, P. (2000) Distribution Atlas of Bats in Britain and Ireland. BCT, London.

Schofield, H., W., & Morris, C. (2004) *Ranging behaviour and habitat preferences of female Bechstein's bat, Myotis Bechsteinii (Kuhl, 1818), in summer.* Vincent Wildlife Trust, Ledbury.

Stebbings, R.E. (1988) Conservation of European Bats. Christopher Helm, London.

Vincent Wildlife Trust (2002) Barbastelle Bat. Vincent Wildlife Trust, Ledbury.

English Nature UK (1998) Biodiversity Group Tranche 2 Action Plans. Volume 1. English Nature, Peterborough.

Once and future veterans in farmland around historic parks: two examples

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Abstract

The UK has a very high proportion of the veteran trees in western Europe, thousands of which are in historic parks and other pasture-woodlands. Open grown old trees also occur in farmland, where there have been very few surveys or evaluations of this resource. In order to better assess the quality of these treescapes, we surveyed the trees in and outside two historic parks; Brockhampton (Herefordshire) and Croome (Worcestershire). Oak (*Quercus spp.*) accounted for at least 40% of the trees. Densities in surrounding land were lower (Brockhampton) or similar (Croome) to those within the parks, and constituted a substantial resource in the wider landscape the conservation of which deserves more attention.

Introduction

The UK has a very high proportion of the veteran trees in western Europe, many of which exist in historic parks and pasture-woodlands. However an unknown but potentially large number of open grown trees occur in the wider landscapes around historic parks. In order to gauge the true nature of the tree resource, and so its wildlife and landscape value, much more information is needed about the trees around parks. This study describes tree surveys within and immediately outside two historic parks in England. The objective of the study was to compare the distribution, abundance, age and health of open grown tree species in and outside historic parks. Greater knowledge of the tree resource at the landscape scale allows a strategic approach to be adopted, and a more efficient allocation of resources.

Method

Two historic parks were selected: Brockhampton (NGR: SO682546) and Croome (NGR: SO880443) both owned by the National Trust (NT) and surrounded by farmland, in the former case also owned by the NT. Within the areas available for survey, the following data were collected for individual trees and clumps: species; location; girth at breast height (gbh); canopy cover (full, partial, residual, dead crown); dead/alive/stump; immediate land use (e.g., arable , permanent grass etc.); potentially damaging operations (e.g., potential for root damage through compaction by livestock or vehicle); trunk damage (bark stripping by livestock or through machinery); limb damage; tree surgery; fertilisers and other inputs; and other details (e.g., tree form). Photographs and location data will allow us to track the fate of individual trees in the future. Only the preliminary analyses are presented in this paper.

Results

A total of 1,794 individual trees of 33 species were recorded. Inside and outside both parks, oaks (*Quercus spp.*) constituted at least a third of all trees. In Brockhampton Park there were significant numbers of beech (*Fagus sylvatica*), ash (*Fraxinus excelsior*), lime (*Tilia spp.*) and hawthorn (*Crataegus spp.*). In Croome Park there were more open grown hawthorns (48% of sample) than oaks. This probably reflects the death and removal of oaks when the park was under arable (Table 1).

Table 1. Population size and structure for native oak (*Quercus robus/petraea*) and all trees in and outside Brockhampton and Croome Parks

	Brockhampton			Croome		
	Park	Outside	Overall	Park	Outside	Overall
% oak	38.8	42.0	40.7	34.2	57.4	49.4
% oak in 1.57–3.13 m gbh	22.9	55.2	42.7	48.9	64.3	57.9
Density of trees (no/ha)	5.9	1.1	1.6	1.0	0.6	0.8
Density of oak (no/ha)	2.3	0.5	0.7	0.4	0.4	0.4

At Croome, and outside the park at Brockhampton the size structure of the oaks was strongly skewed in favour of the 1.57-3.13 m gbh size class and so aged between 100-200 years, revealing deep "age class" crises. In Brockhampton Park, this skew was much weaker reflecting a planting programme over the last century (Table 1). Very old oaks, with a gbh of more than 6 m and estimated to be at least 500 years old, were rare. There were twelve and four at Brockhampton and Croome respectively.

Overall densities of oaks were low (<0.5 / ha) except in Brockhampton Park (c. 2 / ha) where tree density generally was much higher than elsewhere (c. 6 / ha) (Table 1).

We assumed that the density of open grown trees would decrease from the big house surrounded by first a garden and then a parkland, out into the surrounding land. To test this the density of open grown oaks was tracked in 250 m wide concentric bands from the big house to the furthest extent of the tree survey (4,000 m). For both Brockhampton and Croome, the predicted "density gradient" was only evident within the first 1,000 m from the house. Thereafter density increased, albeit erratically. Thus the local densities of open grown oaks were often higher *outside* the two historic parks in our study.

Discussion

The tree surveys described here were quick and easy to conduct and required a minimum of specialist knowledge. They revealed large populations of open grown trees in farmland adjacent to parkland. In many areas of the surrounding farmland, the density of open grown field and hedgerow oaks was higher than within the adjacent parks.

Open grown trees in the wider landscape have an obvious aesthetic value. They may also hold populations of the species that rely on the dead and dying timber of old trees that are usually associated with parks and pasture-woodlands. Early in this study we assumed that the ecological "sources" of, for example, saproxylic invertebrates would be the parks and that dispersal from these ecological islands would be fruitless, as there were no suitable habitats in the wider countryside "sinks".

The tree surveys described here suggest a need for surveys of saproxylic invertebrates and other specialist assemblages associated with parks and pasture woodlands in treescapes in farmland. It is possible that source and sink have been misidentified. In a sister paper which uses other data from the tree surveys (Bullock *et al,* 2004) we show how oaks growing in intensive arable farming habitats are in poorer health than those growing in permanent pasture. , The landscape value of trees in farmland is sufficient to demand that they are given more protection from the excesses of modern agricultural practices, irrespective of their biological worth, which may be underestimated.

Acknowledgements

We thank the NT staff in the Brockhampton and Croome Parks for facilitating the tree surveys and NT Conservation Directorate staff for their help in the field. We thank Katie Fretwell (NT) for alerting us to tree survey reports for the parks. We are grateful to the Agent for The Society of Merchant Venturers land at the Pirton Estate and the tenants for granting us access.

Reference

Bullock, D.J., Fretwell, K.A. & Wainwright, A. (2004) Historic parks in the wider landscape. R.J. Smithers (Ed) *Landscape ecology of trees and forests*, IALE(UK).

Landscape ecology of woody tree and scrub species in hedgerows and woodland edges. Assumptions on relations between landscape composition and species occurrence, based on a field-study

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Abstract

The paper summarises the main conclusions and theoretical foundations from a field-study in southern Sweden on relationships between vegetation structure, landscape structure and woody species occurrence in field boundaries and woodland edges. Landscape ecological concepts such as connectivity, connectedness, and the principle of a positive feedback loop are discussed in relation to the dispersal mode of individual plant species. In the field study, landscape connectivity and connectedness were related to dispersal mode and to species preferences for forest versus open habitat. Connectedness seemed to be most important for animal dispersed species with heavy propagules and may relate to higher activities of dispersing birds or mammals in nodes of a higher density hedgerow network. A positive relationship was found between animal dispersed species and woodland edges with highly developed structure at the site level. A higher number of animal dispersed species was found in hedgerows compared to woodland edges. This relates to the assumption that activities of dispersing animals are higher in hedgerows than in woodland edges. The importance of interpreting landscape scale studies taking into account the biological significance of the results is emphasised.

Introduction

With an increasing intensification of agricultural and forested land use, linear wooded habitats such as wooded field boundaries, hedgerows, river corridors and woodland edges are getting increasingly important in managed landscapes as potential refuges for a diverse tree and shrub flora. Few of the woody plant species found in these remnant habitats are considered as endangered, but trees and shrubs can contribute to a wide range of biological resources in the landscape. They are the base of complex food webs and key-species for insects and bird species (Harper and Hawskworth, 1995) as well as providing: food sources for several mammal species (Herrera, 1989); habitats and corridors for numerous plants and animals; filters against pollution and windbreaks. Wooded linear boundaries also influence the landscape visually, and contribute to landscape character as well as providing resources for amenity and recreation (Sarlöv Herlin, 1999). Edge habitats with a high diversity of plant-species and a continuity of food resources during the season are particularly beneficial to wildlife (Sarlöv Herlin, 1999; 2001). In order to enhance structurally diverse and species rich edge habitats through management, an understanding of relationships between landscape structure and woody species occurrence is needed.

A field study was carried out place in the province of Scania in Southern Sweden. The study area consisted of 2,800 ha of mixed agricultural and forested land on a major estate, characterised by species-rich forest edges and hedgerows with a homogeneous and well documented land management. The occurrence of woody species was recorded in 262 forest edges and 315 hedgerows that varied in site structure and landscape structure, and the results were interpreted with consideration to dispersal characteristics for different woody species. The study focused a diversity of woody plants and the 21 most frequent species were included in the analysis. As the results also have been published elsewhere (Sarlöv Herlin, 1999; Sarlöv Herlin and Fry, 1999; Sarlöv, 2001), the aim of this paper is to summarise the main conclusions and theoretical foundations of the field study and particularly to discuss the landscape ecological concepts of connectivity, connectedness and the principle of a positive feedback loop in relation to the biological significance of the results.

The landscape ecology of woody plants in edges

Woody plants are easy to record, as always being present in the landscape, but not always easy to study. Their long life and large size can complicate studies of their population dynamics, and the clonal growth of many woody species can make the measurement of age difficult. In natural conditions, the composition of vegetation is a result of soil type, climate, dispersal and regeneration of plant species. In managed landscapes, potential sites for colonisation and further dispersal are affected by land use practices that can vary strongly due to human factors.

One of the core themes in landscape ecology is the relationship between landscape structure and species composition. Edges or boundaries have a particular role as regulators of the flow of organisms and material in the landscape (Forman, 1995). The landscape ecology of woody plants is strongly related to the landscape ecology of their dispersal vectors. A majority of tree and shrub species that occur in edge habitat are bird dispersed, although dispersal by hoarding animals or carnivores, wind and water also take place (Herrera, 1989; Kollman and Schill, 1996). Trees and shrubs species can also be spread by vegetative reproduction or clonal growth, which is efficient for competition at the growth site but not for wider distribution in the landscape.

Connectivity and dispersal of woody plants

Connectivity describes the possibilities for species to disperse through a landscape mosaic, and is a function of the composition of the spatial arrangement of landscape elements and the movement behaviour of the species of interests (Burel and Baudry, 1999). Connectivity for animal dispersed woody plants is hence related to the specific feeding habitat, territorial and migratory behaviour of dispersing animals, as well as the configuration and composition of the landscape. Bird species may, for example, either prefer to follow landscape corridors for their movement, or be able to cross open areas (Grashof-Bokdam, 1997). Hoarding animals collects nut and acorns and have different strategies for hiding (and finding) these. For wind-dispersed woody species, dispersal is influenced by weight and type of germinule and prevailing wind conditions. For all types of dispersal, distances between sources of propagules and establishment sites are important.

In the field study, different measures of connectivity were tested, based on the assumption that the proximity between a forest and a linear habitat could influence the distribution pattern of woody species. These were: 1) amount of woodland 500 m from hedgerows; 2) distance to nearest woodland from hedgerows; 3) distance to closest wooded landscape element (forest or hedgerows) from a hedgerow. These measures were assumed to correspond with the grain size of the actual landscape, and to match the dispersal processes of the studied species. The tests gave matching results in the response of individual species with some deviation. A general positive relationship was found between occurrence of forest associated trees and shrubs such as *Alnus glutinosa*, *Carpinus betulus* and *Corylus avellana* and the proximity to a forest. An additional test used the proportion of woodland in the neighbouring area as a connectivity measure, by dividing the main study area into two areas with 48 % respectively 19 % of wood cover in each area. In this test, a higher number of animal dispersed species was found in edge habitats in the more open area compared to the more wooded area. In hedgerows in the more wooded landscape, none of the animal dispersed species was observed at a significantly higher frequency, while in the open landscape *Prunus avium*, *Prunus spinosa* and *Sorbus aucuparia* (all animal dispersed) were present in higher proportions. For wind dispersed species the preference differed between individual species, with some forest associated species such as *Alnus glutinosa* and *Carpinus betulus* preferring hedgerows in the wooded area, and *Populus tremula* found in higher proportion in hedgerows in the open area.

Connectedness and dispersal of woody plants

Connectedness refers to the physical links between landscape elements, for example, the density in a hedgerow network. In the study, connectedness was defined as the number of physical connections between two hedgerows, or between a hedgerow and a forest edge. Elements were considered as physically connected if they had less than 5 m gap between them. Four different classes of connectedness were identified (0, 1, 2 and 3-5 connected elements). A clear preference for increasing physical connectedness was detected for four animal dispersed taxa, all with heavy propagules (*Corylus avellana*, *Crataegus* spp. *Euonymus europaeus* and *Quercus robur*. This relates to similar findings of higher activities of dispersing birds or mammals in nodes of a higher density hedgerow network (Fry, 1991, Clergeau and Burel, 1997). Also for one wind dispersed species (*Popolus tremula*) an increasing response to increasing connectedness was found, that might be explained by a greater air turbulence at nodes, which increases the deposit of light wind dispersed seeds.

The principle of a positive feedback loop

A main principle in landscape ecology states that species distribution and landscape structure is linked in a positive feedback loop where landscape structure, defined as the spatial arrangement of different landscape elements, both controls and is controlled by the increase or decrease of species among landscape elements (Forman and Godron, 1986). In the study, animal dispersed shrub species were more associated with wide, dense edges with extended profiles, compared with wind-dispersed species. This could be explained by the combination of the growth characteristics of shrubs and an accumulation of species via animal dispersal interacting with local seed sources. The study also found a higher number of animal dispersal species in hedgerows and a higher frequency of wind dispersed species in forest edges. This relates to the assumption that activities of dispersing animals can be higher in hedgerows than in woodland edges (Fritz and Merriam, 1994).

Conclusion and further suggestions

Studies at a landscape scale of connectivity and connectedness need to consider biological relevance, such as the particular behaviour of dispersal agents, together with the landscape structure. Studies at a site level also need to consider that a broader landscape context can magnify local effects of dispersal of woody species. Increasing landscape connectivity and connectedness of field margins can be used in landscape management to support the regeneration of species-rich edge zones in agricultural landscapes. Further research on the behaviour of dispersal agents, and identification of the different scales at which dispersal processes operate in different landscapes, could contribute to an improved and more sustainable management of wooded edge habitats.

References

Burel, F and Baudry, J. (1999) *Ecologie du paysage.* Editions Tec & Doc.

Clergeau, P. and Burel, F (1997) The role of spatio-temporal patch connectivity at the landscape level. An example in bird distribution. *Landscape and Urban Planning* **38**: 37-43

Forman, R.T.T. (1995) *Land mosaics. The ecology of landscapes and regions.* Cambridge University Press.

Forman, R.T.T. and Godron, M. (1986) *Landscape ecology.* New York. John Wiley.

Fritz, R. and Merriam G. (1994) Fencerow and forest edge vegetation structure in Eastern Ontario farmland. *Ecoscience* **12:** 160-172.

Fry, G. (1991) Conservation in agricultural ecosystems. Spellerberg, I.F., Goldsmith, B. and Morris, M.G. (Eds). *The Scientific Management of Temperate Communities for Nature Conservation.* British Ecological Symposium 31: 415-443. London . Blackwell.

Fry, G. and Sarlöv-Herlin, I. (1997) The ecological and amenity functions of woodland edges in the agricultural landscapes; a basis for design and management. *Landscape and Urban Planning* **7:** 45-55.

Grashof-Bokdam, C. (1997). Forest species in an agricultural landscape in the Netherlands. Effects of habitat fragmentation. Journal of Vegetation Science **8:** 21-28

Herrera, C.M. (1989) Frugivory and seed dispersal by carnivorous mammals and associated fruit characteristics, in undisturbed Mediterranean habitats. *Oikos* **55:** 250-262.

Kollman, J. and Schill H-P. (1996). Spatial pattern of dispersal, seed predation and germination during colonisation of abandoned grasslands by *Quercus petraea* and *Corylus avellana. Vegetatio* **125:**193-205.

Sarlöv Herlin, I.L. (2001) Approaches to forest edges as dynamic structures and functional concepts. *Landscape Research* **26**, no. 1, 27-43

Sarlöv Herlin, I. (2001) How can an understanding of dispersal processes for woody species contribute to the establishment and management of hedgerows? C. Barr and S. Petit. (Eds). *Hedgerows of the world; their ecological functions in different landscapes: Proceedings of the European IALE Congress held at University of Birmingham, September 2001.* 147-156.

Sarlöv Herlin, I.L. and Fry, G.L.A. (2000) Dispersal of woody plants in forest edges and hedgerows in a Southern Swedish agricultural area: the role of site and landscape structure. *Landscape Ecology* **15** (3): 229-242

Sarlöv Herlin, I. (1999) *Edge Habitats in Agricultural Landscapes.* Woody species, landscape ecology and implications for planning. Doctoral thesis. Agraria 202, Swedish University of Agricultural Sciences, Alnarp.

Ancient woodland restoration in the south-east Midlands: four years of learning

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Abstract

In the winter of 1999/2000 the restoration of 7,500 ha of plantations on ancient woodland sites (PAWS) in the southeast Midlands was initiated by the Forestry Commission. Four years on much has been learnt about the process. This paper sets out the original base surveys undertaken into the woodland ecology and archaeology and the lessons learnt. Involving local people has proven invaluable in gaining support for the restoration work in a densely populated area of England. This paper concludes with a brief debate about the pros and cons of clear felling conifers as a means of restoring PAWS on heavy boulder clay, as opposed to undertaking restoration thinning operations.

Introduction

The Forestry Commission's (FC) Northants Forest District launched the Ancient Woodland Project in March 2000. It is an ambitious scheme to restore site native composition and ancient character to all of the FC's PAWS in the southeast Midlands.. The project covers 78 woods, totalling 7,500 ha. This paper is intended as a short summary of the lessons learnt over the last four years.

Start with a Survey

Survey of NVC woodland types and semi-natural classes

As part of the FC's national survey of its ancient woodland sites (Spencer, 2003), Louise Hutchby was employed to survey Northants Forest District in 2000 (Hutchby, 2001). The surveys mapped the woods according to:

- woodland National Vegetation Classification (NVC) sub-community (Rodwell, 1991)
- semi-natural class (describing the extent of native woodland within the plantation).

The percentage area ascribed to each NVC sub-community was:

- 63% W8 (*Fraxinus excelsior – Acer campestre – Mercurialis perennis* woodland)
- 27% W8c (*Deschampsia cepitosa* sub-community)
- 9% W10 (*Quercus robur – Pteridium aquilinum – Rubus fruticosus* woodland)
- 1% W16 (*Quercus spp – Betulus spp – Deschampsia flexuosa* woodland).

From NVC mapping, 90% of the woodland area could be classified as ash (*Fraxinus excelsior*)/ field maple (*Acer campestre*) woodland. This gave a strong indication of which trees were likely to regenerate most freely the original composition of the wood. It did not, however, help identification of what is ecologically important in each wood, or its future management. This point is made strongly by Oliver Rackham in his critique of the NVC 8 category (Rackham, 2003). Although requiring more intensive survey work, the woods perceived as most ecologically interesting in the project area will now be mapped out according to the stand types put forward by Peterken (1993). This will assist with future woodland management planning.

Mapping of semi-natural classes was more interesting, highlighting a significant problem with the way the FC has traditionally recorded its plantation details. The results were as follows:

- 41% semi-natural class 1 (>75% site native broadleaves)
- 9% semi-natural class 2 (50 – 75% site native broadleaves)
- 19% semi-natural class 3 (25 – 50% site native broadleaves)
- 22% semi-natural class 4 (< 25% site native broadleaves)
- 9% felled land awaiting regeneration

There was a much higher area in semi-natural class 1, near native woodland, than anticipated. FC foresters' plantation record systems emphasise planted crop characteristics and thus under-record natural elements, for example, the native under-storey and edge broadleaf/ broadleaf scrub. The semi-natural class survey may thus, in the short-term, be the only reliable means of detailing progress with restoration whilst the longer-term structural problems with the FC's traditional recording systems are ironed out.

Survey of surface visual archaeology

A further survey undertaken for the project was a rapid assessment of the 'surface visual archaeology' of the woodlands (Simco, 2003). This survey was undertaken in conjunction with the county archaeologists following an early plea from Glenn Foard, then County Archaeologist for Northamptonshire, who stated that *'your woodlands are the largest undiscovered repository of archaeology in the county'* (personal communication). This statement turned out to be true, with the range of features discovered extending from Bronze Age cairns to World War II structures; including settlement sites, defensive features and a vast range of woodland management features. The latter included wood and coppice banks, medieval hunting lodges, charcoal pits and iron smelting sites.

Involve local people

Today it is 'politically correct' to involve local people. But do they care and can they help? To support the project a series of consultation events, walks and talks were arranged across the project area between 2000 and 2002. The response was over-whelming, and:

- Yes, they do care about their woods, mostly about access and keeping the woods 'un-developed'. The majority are fascinated by the woods history and wildlife but, generally, most do not 'care' whether the wood is coniferised or not, instead there is a noticeable liking for big trees and a deep love of oak (*Quercus*). This is a key lesson, managers need to sell and explain restoration to local people before work starts.

- Yes the local people can help. Not all, but some know more about the woods than we do, and a surprising number are willing to give up time to help with practical work including ecological monitoring, litter picking and coppicing. Learning from the Woodland Trust and Wildlife Trust experience, the FC has found that getting local people involved is the best tool to gaining widespread and enthusiastic support for the work, and indeed for woodland generally.

The conversion process: getting from conifer plantation to semi-natural ancient character woodland

To clear fell or not to clear fell the non-natives!

The project began with 800 ha of forest land that had been cleared of dead and dying Norway spruce (*Picea abies).* Many of these sites had been cleared of tree cover for more than a year and were vast prairies of coarse grasses such as *Calamagrostis epigejos* and *Deschampsia cespitosa.* Lack of tree cover left the sites' heavy boulder clay soils open to the extremes of summer heat and drought as well as winter cold and waterlogging. The hostile site conditions, coupled with the rampant *Calamagrostis*, were believed to conspire against natural regeneration. This led the local team to conclude quickly that clear-felling was not the way forward for the project. A decision to stop further clear felling was taken during 2000. The 800 hectares which had been felled were mainly (but not exclusively) deer fenced and then put into a monitoring cycle with no further herbicide or planting treatments.

Four years on was this the right decision? Probably yes, but over those four years the subsequent habitat of 'successional woodland' has proven valuable to some unexpected wildlife, not least the grasshopper warbler (*Locustella naevia*) whose call has become common in the project woods most affected by the clear felling. The assumption that natural regeneration would fail in the harsh environment is also baseless. Each site is different, natural regeneration rates have varied from 56 native stems per ha to over 11,000 stems per ha. The majority of sites have in the region of 3,000 stems per hectare, which is considered acceptable at this stage of transition from spruce (*Picea*) plantation to native woodland. Nor have the species regenerating been just birch (*Betula*) or willow (*Salix*), as might have been expected. They include a full range of native trees, including oak and ash as well as the under-storey components of hazel (*Corylus avellana*) and field maple.

Today, many of the trees that had been planted prior to 1999/2000, and maintained with traditional weeding and beat up treatments to 2002, have been out performed by natural regeneration within the same site. The conclusion must be that whilst clear-felling remains a dramatic intervention, both ecologically and visually, it can at least restore native tree cover and structural diversity, in the way of transitional open space.

Restoration or conversion thinning

Restoration thinning is the process by which the unwanted non-native tree species are gradually 'thinned out' of the plantation to give space for natural regeneration of native species to take place. Particular attention needs to be paid to opening up ride edges to encourage the native ride side scrub to colonise the retreating plantation edge. In many ways restoration thinning is the clear alternative to clear-felling. Its benefits include the obvious points that the impact on the landscape and the wood's ecology is minimal. From autumn 2000 onwards restoration thinning became the norm in the project woods.

Four years on, what have we learnt? The main lesson is that restoration thinning makes the plantations vulnerable to wind damage. Several woods were flattened within weeks of receiving a restoration thinning. The worst affected sites have been semi-natural class 4 (the purest conifer plantations) stands in mid-rotation. Other semi-natural classes have had sufficient broadleaf content, or sufficient scrub edges, to make the stands more stable. Is this a reason not to use restoration thinning in well stocked conifer plantations? Again, probably not, for two reasons: firstly, not every stand succumbs to the wind, this seems to be a complex effect of stand dynamics, wind variables and soil specifics which I will call luck; secondly, if the stand does blow, then by treating it as a clear-fell and going through an ecologically exciting successional woodland phase the objective of native woodland conversion can still be met. This wind-blow risk does, however, highlight the need to clear trees from archaeological features in the first restoration thin to avoid windblow causing significant damage.

The other fundamental problem with restoration thinning is that of natural regeneration of non-native trees, notably grand fir and western red cedar. Both of these shade tolerant species will set seed and regenerate under an existing canopy which gives them an advantage over our native trees. If a stand is dominated by grand fir or western red cedar it may be more sensible to opt for clear-fell rather than continually fighting unwanted regeneration with heavy herbicide treatments, which may affect the survival or colonisation of native plants.

The lessons learnt: a summary

- Patience is a virtue and will save you money, natural regeneration is not necessarily an expensive option
- Each site is different, there is no one correct method of restoration
- Local people are a valuable resource to be used wisely and not taken for granted
- Clear-fells are destructive short term but can have unexpected benefits
- Restoration thinning is slow and is not risk free.

References

Hutchby, L. (2001) Northants Forest District ancient and native woodland ecological survey, unpublished. Forestry Commission.
Peterken, G.F. (1993) *Woodland conservation and management (2^{nd} edition)*, Chapman & Hall.
Rackham, O. (2003) *Ancient woodland its history, vegetation and uses in England*, Castlepoint Press.
Rodwell, J. (1991) *British plant communities, volume 1: woodland and scrub*, Cambridge University Press, Cambridge.
Simco, A. (2003) Ancient Woodland Project archaeology. A review of the archaeological surveys in the Northants Forest District 1996 –2001, unpublished.
Spencer, J.W. (2003) *Ancient woodland on the Forestry Commission Estate in England*, Forestry Commission, Edinburgh.

Phytochemistry of Scots pine as a driver of spatial heterogeneity in insect herbivore and broader diversity in a native woodland landscape

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Abstract

We hypothesised that variation in the chemical characteristics of Scots pine trees will affect the distribution and abundance of insect herbivores in a native forest landscape. Because insects play a key role in nutrient cycling this could have consequences for forest soil processes and be a driver of heterogeneity in microbial, mycorrhizal, soil invertebrate and plant species diversity with cascading effects through the food chain. Interception funnels were constructed under Scots pine trees of contrasting phytochemistry to assess the composition of material falling from pine crowns. Material was continuously collected for two years and sorted into plant and insect categories. Significant spatial associations were detected during early summer. The distribution of insect herbivores in early summer was related to the concentrations of β-pinene and two other monoterpenes. However, the overall average distribution of herbivores each year was related to landform, especially altitude, and illustrates the complex terrain of the study site and potential interactions within the community.

Introduction

An understanding of the genetic composition of the structural species of an ecosystem, and the extent of spatial variation in its genetic composition and functioning, is essential for the effective management, conservation and sustainable exploitation of any natural system. For Scots pine (*Pinus sylvestris*) in native pinewoods, a high conservation priority habitat at both national and European levels, monoterpenes are a prevalent component of pine resin and are under strong genetic control (Sukovata *et al.*, 2003) and have been used as biochemical markers to study genetic differences among populations (Yazdani and Lebreton, 1991). Scots pine shows considerable intrapopulation variation in the amounts of the different monoterpene fractions (Hiltunen, 1976; Heyworth *et al.*, 1998).

Monoterpenes have been shown to have a role in attracting (Zagatti *et al.*, 1997) and deterring (Barnola, 1994) insect herbivores. We hypothesised that variation in the chemical characteristics of Scots pine trees will affect the distribution and abundance of insect herbivores in a native woodland landscape. Insects play a key role in nutrient cycling and this could have consequences for processes in woodland soil and be a driver of heterogeneity in microbial, mycorrhizal, soil invertebrate and plant species diversity with cascading effects through the food chain.

Methods

A substantial remnant of native pinewood in Deeside, NE Scotland was used as the study site. Needles were collected from 150 Scots pine trees and analysed to determine monoterpene composition of individual trees. A subset of 45 trees was selected to represent the monoterpene variability identified in the larger sample. We constructed five interception funnels beneath the canopy of each tree to assess the composition of material falling from the crowns. Material was continuously collected for two years and sorted into plant and insect categories. Insects were identified and assigned to one of seven feeding guilds (Moran and Southwood, 1982), with phytophages further divided into chewers and suckers.

Semivariance models were fitted to the phytophage data for all tree locations by individual date. The semivariance model was used to interpolate values for each arthropod group to the cells of a grid covering the study area. A forward, stepwise variable selection and multiple regression was performed to identify significant relationships between insect numbers and 11 monoterpenes along with locational and structural characteristics of each pine tree.

Results and discussion

The number of phytophages was generally low but significant spatial associations were detected during early summer when populations were establishing (Figure 1). No one taxonomic group prevailed, instead different phytophagous groups dominated as the summer progressed; lepidopteran caterpillars in May, dipterous larvae in June and August, and aphids (Hemiptera: Aphididae) in July. Semivariance models indicated significant spatial association at distances of 0–0.5 km in May and June but random distributions across 0–2 km in July and August (Figure 1).

Table 1: Factors relating to insect phytophage abundance (all significant at $P < 0.05$)

Taxon	Coefficient of determination (R^2)	Date	Fitted Variable	Trend
Phytophages	0.12	Combined	Altitude	-
	0.38	14/05/02	β-pinene	+
Chewers	0.11	Combined	Altitude	-
	0.38	14/05/02	β-pinene	+
Aphids	0.53	Combined	Ants	+
			Tree height	+
			Tree density	+
			Height of lowest branches	-
			Terpinolene	-
	0.40	16/09/02	Ants	+
			Tree density	+
			α-terpinene	-

Total phytophage numbers and specifically chewers, correlated negatively with altitude but positively with levels of β-pinene (Table 1). Aphids were associated with the presence of ants and a number of physical factors. The abundance of aphids was negatively correlated with two monoterpenes; terpinolene and α-terpinene. Monoterpenes seem to affect the distribution patterns of insect herbivores in early summer but the overall distribution each year is more strongly related to landform.

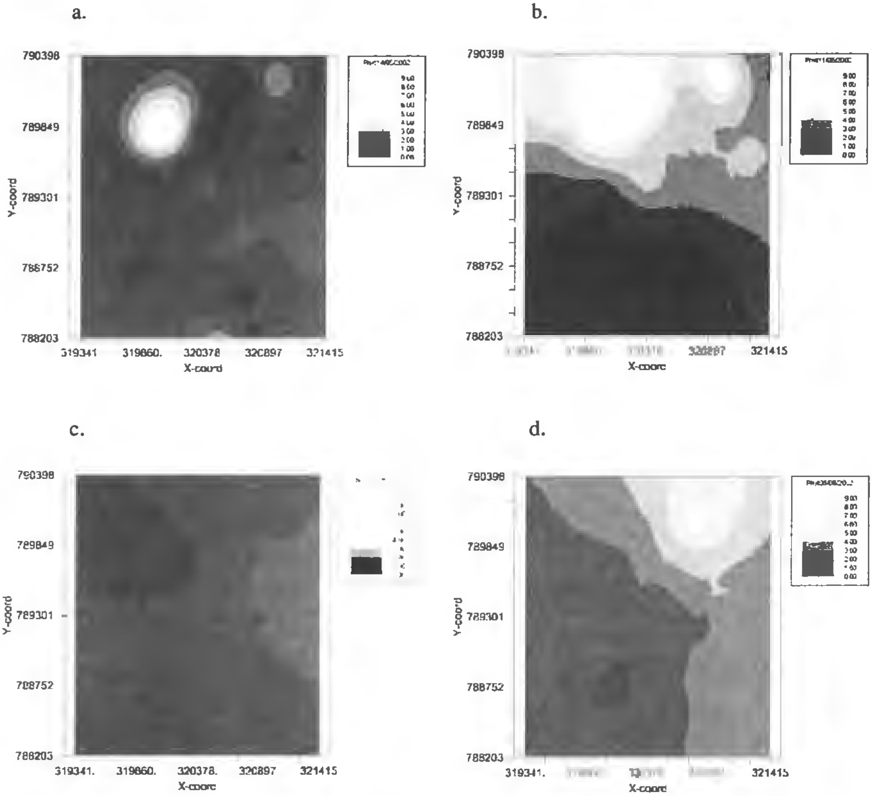

Figure 1. Spatial distribution of insect phytophages across a native pinewood remnant in Deeside, NE Scotland. The interpolation maps are based on semivariance models at each time: a. May ($R^2 = 0.49$), b. June ($R^2 = 0.83$), c. July ($R^2 = 0.04$) and d. August 2002 ($R^2 = 0.15$). Eastings and northings represent Ordnance Survey national grid references.

The preliminary results from this study highlight the complexity of potential interactions between insect herbivores and their environment. Plant chemistry is known to be one of the factors influencing the distribution and abundance of insects (Zangerl and

Berenbaum, 1993). We have demonstrated an association between certain monoterpenes and insect herbivores early in the year. The observed positive correlation with β-pinene and negative correlation with terpinolene and α-terpinene agreed with earlier findings (Sukovata *et al.*, 2003; Park *et al.*, 2003). However, the influence of landform on phytophage distribution is not surprising because of the variable terrain associated with the remnants of native pinewood in Scotland. Further data and analyses will be used to estimate the magnitude of effect of pine tree chemistry compared with other factors on the distribution and abundance of insect herbivores.

Acknowledgements

We thank the Scottish Executive Environment and Rural Affairs Department for funding and Balmoral Estate for access to the study site.

References

Barnola, L.F; Hasegawa, M. & Cedeño, A. (1994) Mono- and sesquiterpene variation in *Pinus caribaea* needles and its relationship to *Atta laevigata* herbivory. *Biochemical Systematics & Ecology* **22**: 437-445.

Hiltunen, R. (1976) On variation, inheritance and chemical inter-relationships of monoterpenes in Scots pine (*Pinus sylvestris* L.). *Ann Acad. Scient. Fenn., ser A IV* **208**, 54pp.

Heyworth C, Iason GR, Temperton, V, Jarvis, P, & Duncan AJ. (1998) The effect of elevated CO_2 concentration and nutrient supply on carbon-based plant secondary metabolites in *Pinus sylvestris*. L. *Oecologia* **115**: 344-350.

Moran, V.C. & Southwood, T.R.E. (1982) The guild composition of arthropod communities in trees. *Journal of Animal Ecology* **51**: 289-306.

Park, I.K; Lee, S.G; Choi, D.H; Park, J.D. & Ahn, Y.J. (2003) Insecticidal activities of constituents identified in the essential oil from leaves of *Chamaecyparis obtusa* against *Callosobruchus chinensis* (L.) and *Sitophilus oryzae* (L.). *Journal of Stored Products Research* **39**(4): 375-384.

Sukovata, L; Kolk, A; Jaroszynska, J; Krajewska. U; Purzynska, A. & Isidorov, V. (2003) Host-tree preferences of the pine moth (Lepidoptera: Lasiocampidae) and pine beauty moth (Lepidoptera: Noctuidae) larvae in relation to needle quality. *Proceedings: Ecology, Survey and Management of Forest Insects*, Poland, 1-5 September, 2002 (eds. M.L. McManus & A.M. Liebhold), pp. 98-106.

Yazdani, R. & Lebreton, P. (1991) Inheritance pattern of the flavonic compounds in Scots pine (*Pinus sylvestris* L). *Silvae genetica* **40**: 57-59.

Zagatti, P; Lemperiere, G. & Malosse, C. (1997) Monoterpenes emitted by the large pine weevil, *Hylobius abietis* (L.) feeding on Scots pine, *Pinus sylvestris* L. *Physiological Entomology* **22**: 394-400.

Zangerl, A.R. & Berenbaum, M.R. (1993) Plant chemistry, insect adaptations to plant chemistry, and host plant utilization patterns. *Ecology* **74**(1): 47-54.

Pan-European ecological networks and European target species

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Introduction

Biological diversity is highly dependent on the quality, quantity and spatial cohesion of natural areas or habitats. Loss of habitat (in quality, and quantity) and loss of spatial cohesion results in fragmentation. Fragmentation has decreased biodiversity in the highly urbanized area of Western Europe and due to fragmentation many species have already disappeared locally or may disappear in the future.

A solution for fragmentation is the protection or development of ecological networks. The ecological network links nature areas by means of corridors and small habitat patches. Coherent ecological networks are of cardinal importance to maintain and support populations.

International policies

The development of ecological networks is part of global and European policy. The ecological network was officially recognised as an important strategy for biodiversity conservation in the Pan-European Biological and Landscape Diversity Strategy (PEBLDS). This strategy calls for the development of the Pan-European Ecological Network (PEEN). The EU Habitats Directive also acknowledges the importance of landscape elements that enhance connectivity.

European policies such as the Bonn and Bern Convention, the Birds and Habitats Directives, oblige member states to take effective measures in conservation and management of the listed species. The species included in this poster are examples of protective measures for these European species.

Different types of corridors

Corridors can differ in shape, in function or in size. Shapes of corridors can be divided into three types; line, stepping stone or landscape corridors (Figure 1). Invertebrates use what may be regarded as a fourth ‘sub-type’: line corridors with nodes.

Corridors are necessary for dispersal, migration and commuting. A *dispersal corridor* is defined as ‘a corridor that links two habitat patches with each other and has a length that is less than the maximum dispersal distance of the species’. A *migration corridor* is defined as ‘a corridor used for movement from one resource area to another (e.g., from breeding to wintering grounds)’. Mostly, this type of corridor will consist of a set of areas used during migration as 'stopover' places (e.g., for waterfowl and waders). A *commuting corridor* is defined as ‘a corridor that links areas, within a home range or territory of a species, that have different functions’ (e.g., foraging and breeding areas or resting/sleeping and foraging areas, mostly daily movements between these areas).

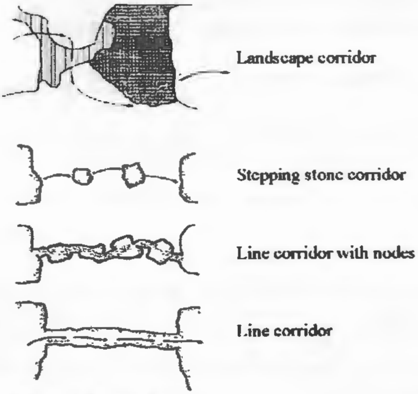

Figure 1. Shapes of corridors

These posters demonstrate how fragmentation problems at a Pan-European scale level for specific species could be resolved. They provide some practical solutions for the development of ecological networks in Europe at different scale levels, for the species listed in Figure 2. Real forest species presented in this poster are the brown bear (*Ursus arctos arctos*), stag beetle (*Lucanus cervus*), and lynx (*Lynx lynx*).

Shape \ Function	**Dispersal**	**Migration**	**Commuting**
linear corridor		Atlantic Salmon	
linear corridor with attached nodes	**Stag Beetle**		Large Copper
stepping stone	**Lynx** Yellow-Legged Dragonfly	Brent Goose Yellow-Legged Dragonfly	
landscape mosaic	**Brown Bear** Large Copper	**Brown Bear**	**Brown Bear**

Figure 2. Shapes of corridors and use by species

For the brown bear we show that landscape corridors in Abruzzo (Italy) may facilitate migration but also foraging, and commuting of the species. Corridors, if properly situated, may enlarge and stabilise further the population of brown bear.

The stag beetle requires old dead trees and tree stumps. Moribund deciduous trees can strengthen ecological networks. Breeding places in 'nodes' less than 2 km apart will facilitate dispersal.

The lynx needs proper corridors with stepping stones, to facilitate its increase in range, from traditional strongholds in Northeast Poland to Northwest Poland, from Western Poland towards the Harz in Germany, and between France (Vosges, and Jura) towards Belgium.

Use of computer visualisation in woodland landscape analysis: available tools, practical issues and potential applications

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Abstract

Woodland managers are now expected to interpret the potential ecological and landscape implications of their decisions to audiences with differing requirements and levels of technical awareness. This applies in both "traditional" rural forestry contexts and in urban design projects where trees form one part of an integrated landscape and built architectural solution.

Computer visualisation techniques used extensively in architecture and landscape design over the past decade are now becoming available at a cost enabling them to be applied to woodland management. The ability to combine visualisation techniques with spatial ecological information and tree and stand growth models offers the prospect of delivering a higher level of ecological precision and visual predictability in woodland design work.

The poster will highlight some of the tools available, discuss practical issues in modelling wooded environments and identify potential applications of these techniques for an independent forestry and landscape design practice.

Introduction

There is an increased expectation that woodland managers should predict and interpret the ecological and visual landscape implications of their decisions to the public. This applies particularly where foresters are delivering the "woodland components" of integrated architectural and landscape design packages in and around urban areas. Fortunately, computer spatial modelling and visualisation tools are becoming available at much lower entry costs and with a less specialist hardware platform requirement than was previously the case, allowing woodland designers and managers the opportunity to apply them at all scales.

This poster charts the experience of the author as an independent forester and ecological landscape designer adopting a suite of such tools including (a) ArcView GIS for spatial analysis (ESRI UK, Aylesbury), (b) Visual Nature Studio (3D Nature LLC, Colorado) for woodland landscape visualisation, (c) OnyxTree 6 (Onyx Computing Inc, Cambridge, Massachusetts) for the procedural 3D modelling of individual trees and (d) trial Autodesk 3D Viz 4 (Autodesk, USA) for the insertion of "architectural standard" 3D models of built landscape features.

Methods

Landscape terrain

For landscape-scale 3D visualisations over significant topographical relief it is essential to purchase a "DEM-tile" (digital elevation model) from the Ordnance Survey, covering the area of a project. This can be imported as the base platform for the scenes. However, the significant costs for purchase and maintenance of these OS tiles in the UK are an issue for adoption by the small-scale user. For close-up detailed scenes or those on simple relief, an alternative approach is to "sample" a matched surface panel from terrains created randomly using the VNS Terrain Generator mode.

Vegetation: ecosystems and ecotypes

Positioning

VNS allows for different modelled ecosystems (consisting of overstorey and understorey ecotypes) to be positioned over the 3D terrain surface using four methods (a) manually-specified vectors, e.g., for individual trees in a parkland landscape, (b) "rules of nature", e.g., pine-birch on north-facing slopes below 500m elevation, (c) "colour maps", e.g., recognition from draped air photos or raster maps and (d) using attribute date from a "draped" vector GIS theme, e.g., showing climate, soil quality or wind-throw hazard information.

Ecosystem population: using 2D image object modelling

For woodland visualisations where the observer is "distant", i.e., more than a few tree heights away from the woodland edge, it is possible to use planar images of trees and plants acquired by digital or film+scan photography to populate the modelled ecotypes. Trees and shrubs are photographed and the background removed in a photo-editing programme. Specified assemblages of many such images are used to create ecotypes, e.g., for a Caledonian pinewood one would have several photos of individual granny pines, silver birches, aspens, junipers and bilberry bushes, with heather ground cover applied as a "flat panel" texture. Random positioning is used to avoid any "clonal forestry" repeating patterns.

Ecosystem population: using 3D procedural modelling

For close-up photo-realistic woodland visualisations where the observer will "walk through the woods" or "go round behind the trees" one must abandon the photography-based approach and import 3D procedurally modelled trees. Fortunately OnyxTree Pro 6 allows for these to be created in an automated manner, rather than by manual construction of trees from modelling primitives. The results of initial experimentation with this package produced very realistic British trees, again randomisation is used to avoid artificial regularity of branching pattern, leaf angle etc.

Rendering and natural surface textures

When using 3D modelled plants as opposed to photographs, there is a need to apply naturalistic surface textures, e.g., lichen covered bark. This can be accomplished by cloning photographed texture patches from the field or by using the procedural texture modelling module within VNS. Each has its advantages and disadvantages, depending on the scene. Similar approaches are used to describe rock and soil surfaces for application to 3D terrain.

Animation and growth modelling

VNS enables all the simpler forms of camera animation; walk through, fly over, zoom, pan, bank etc. However VNS also allows most numerical parameters of a scene to be animated using a "virtual time lapse" approach, for example; height growth of trees over a rotation, species composition of a developing forest stand, seasonal flushing of tree leaves and flowering of plants in the ground vegetation; the possibilities are essentially unlimited. Most users will specify the animated parameters manually, e.g., show Scots pine as 15m tall after 20 years, 25m tall after 40 years and then animate on a straight-line basis between these "key frames". Similarly the proportion of a late-successional tree species or the percentage ground cover of a shade intolerant ground plant would be specified at two stand ages and linearly interpolated.

However it is possible to automate the time-lapse animation by reference to a mathematical model. The simplest level of this would be to use yield table stand growth models to control the animation of even tree heights in a regular conifer stand, rather than a straight line interpolation. Next, site quality information could be adopted from a draped GIS theme to specify the appropriate stand growth curve to animate for each tree species at any point in the modelled landscape. Finally, spatial wind climate information could be used to position and time-specify a progressive switch to partially windblown overstorey ecotypes. Ultimately, much more demanding applications would see the species composition and dimensional structure of an irregular mixed species stand "time-lapse" animated using the output from a single-tree growth model of the type now being developed for application to continuous-cover forestry. VNS has the capability to model shade cast by, and received by, tree objects, which could provide the basis for "light environment specified" animation of individual component trees.

Scene building: inclusion of non-vegetative 3D objects

VNS incorporates specific modules for the following non-vegetative 3D components of natural environment scenes; (a) rocks, stones and soils, (b) skies, moon and stars, (c) sun, including diurnal illumination and shadow management (d) snow cover and (e) lakes, ponds and rivers. There is also the scope to 3D model simple built features such as roads, paths, cabins, pylons and simple "block" buildings. However, there may often be a desire for sophisticated photo-realistic visualisations of 3D buildings for architectural design purposes (e.g., a mediaeval hunting lodge within a historical wood-pasture landscape or a new architect-designed house set within a 20 year "roll forward" of a landscape-designed woodland garden). For these, 3D models of the built features should be created in a solid 3D modelling package such as Autodesk 3D Viz and imported for texturing and rendering within the VNS environment. Where subcomponent animation is needed (e.g., a wind-turbine rotor) Autodesk 3D Studio Max offers the greatest flexibility for construction of suitable animated 3D objects.

Delivery of Outputs

The work described above is carried out within the author's practice using a high-end double processor desk-top work station platform which gives acceptably short scene rendering times even for demanding and complex still scenes and animations. Including the various software, this required an investment of some £7-10k to install. However, from the perspective of an independent consultant it is important to be able to deliver the outputs of 3D visualisation work effectively to third-party clients with a range of levels of in-house computing facilities.

As with earlier 2D GIS mapping work, 2D "still projections" of rendered scenes can be exported as conventional image files (.tif, .jpg etc.) for inclusion in printed report documents, poster displays and PowerPoint screen presentations. Predetermined animated sequences can be exported as QuickTime or similar multi-media video files, which can be viewed on an increasing proportion of clients' standard desk-top PC's. If an on-the-spot interactive capability is required, then one needs to take a laptop computer running VNS to the client, depending on the level of manipulation "on-the-spot" that is required, this may need to be a high-end laptop of the mobile workstation type to give a satisfactory rendering performance.

Many applications of landscape visualisation involve some form of public consultation or public participatory processes, classically the "village hall meeting" or planning committee. A variety of levels of projection for a multiple audience can be achieved including; (a) conventional LCD projection onto a single flat screen, (b) projection onto 3 angled flat screens using 3 LCD projectors, this gives a satisfactory level of immersion but takes some hours to set up and align for each location and (c) projection onto a continuous curved screen in a purpose-designed immersion/ IMAX theatre which can be fixed or mounted on a lorry trailer. The latter option gives the most satisfactory performance, but is clearly the most costly.

Potential Applications

The convenience and relatively low capital costs involved in applying the methods described allow them to be potentially made available to a far wider range of project types and client sizes than the pre-existing market for bespoke forest landscape design simulations, which has essentially been restricted to large scale public and private sector forestry planning projects. There are four main application sectors envisaged in terms of my practice's involvement:

- Landscape and forest scale applications in "conventional" rural forestry settings such as establishment, harvesting planning, restocking and conversion to CCF systems
- Large-scale applications where woodland creation and management form part of multi-purpose landscape planning together with other forms of development, e.g., forest habitat networks, new wood-pastures/ wildwoods and major regional planning projects in wooded settings, such as those currently envisioned for housing in the southeast
- Smaller-scale application for bespoke woodland solutions in association with architects and landscape designers on commercial developments and private housing projects
- Woodland related multi-media content generation, e.g., virtual scene-setting and planning for the film+ television sectors, publishing and on-site interpretation facilities

Poster abstracts

Woodscape: delivering ancient and native woodland restoration in south east England's public forests

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Abstract

Following an extensive survey of the ancient woodland resource on the Forestry Commission's estate, which commenced in 1999, each of the 11 Forest Districts in England have been actively engaged in rolling out a major programme of native woodland restoration. Of the 53,128 hectares of ancient woodland occurring in the English Forestry Commission Districts some 14,346 ha or 27% is located in the forests and woodlands of the South East England Forest District. Many of these ancient woodland sites are deemed nationally important and include such renowned sites as the Forest of Bernwood and Chiddingfold and Orlestone Forests.

The survey accurately mapped the extent of ancient semi-natural woodland (ASNW) and plantation on ancient woodland sites (PAWS). It also provided maps of the National Vegetation Classification (NVC) communities and ranked PAWS sites according to their semi-natural class. This information is used by foresters to guide their planning and operational activity.

Operational experience has demonstrated how each site offers a unique set of challenges. The Forestry Commission is demonstrating through its efficient management of the public estate that landscape scale habitat restoration can be undertaken in a cost-effective manner making sensitive use of mechanised harvesting machinery. The process has however raised a number of interesting questions:

- Is wholesale removal of plantation conifer a wise economic decision? Retention of a proportion of productive conifer on ancient woodland sites can help to offset the costs of managing the ancient woodland resource
- Natural regeneration versus site native restock; many stands thinned of their non-native element retain few native components and will demand enrichment
- At what rate do we restore sites? Rapid change can do more harm than good e.g., wealden ghyll woods
- Diversification of secondary woodland versus semi-natural habitat restoration (e.g., beech woods on former downland sites)
- In some landscapes there may be very sound ecological reasons for retaining sustainable plantation forestry, e.g., to benefit red squirrels, raptors and heathland birds
- At what intensity should we monitor the restoration process?
- Resourcing the control of invasive species, e.g., rhododendron, western hemlock and red cedar
- What woodland management techniques can we adopt to ensure the survival of priority species in the new native woodlands of the 21st century (e.g., coppice dependent species such as dormice and pearl-bordered fritillary butterfly)?

Mapping a future for ancient trees

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Abstract

Ancient trees are a vital part of our natural and cultural landscape. It has been estimated that Britain may be custodian of many of northern Europe's most ancient trees. They are both host to a unique array of wildlife and also give us an insight to the lives of past generations of people in ways that no other part of our natural world is able.

The Woodland Trust, the Ancient Tree Forum (ATF) and the Tree Register wish to see this priceless legacy conserved for the benefit of all in our society. It is important that there is no further avoidable loss of ancient trees through development pressure, agricultural clearance, mis-management or poor practice. The Trust and the ATF want to see the development of a succession of future ancient trees, properly identified, protected and managed as part of a new cultural landscape, in sympathy with modern society.

Ancient trees have earned their place in our heritage; almost all ancient trees at one time worked for a living, providing generation upon generation of our ancestors with renewable resources of firewood, ships timbers, building materials and animal fodder or as features in historic landscapes. Some have been part of our landscapes for hundreds, even thousands of years and have been havens of peace and tranquillity in a changing world. As relicts of the past, they should be akin to Scheduled Ancient Monuments for their importance to our heritage. And yet unlike structures of brick and stone these living edifices are virtually unprotected. We do not know where they all are, how fast they are disappearing or if there will be future generations to take their place.

The Tree Register holds the greatest number of records of ancient trees and a few locations across the country such as the Corporation of London's Ashtead Common, have started to map their trees. But our records are far from complete. The Woodland Trust is working with the Tree Register and the Ancient Tree Forum to pull together existing records and recruit volunteers to help us create a national dataset of information on ancient trees.

A comprehensive register and map would help us to change people's hearts and minds, so they can see the importance of the UK's ancient trees in the landscape and in our everyday lives. We would be able to identify where the most important trees and concentrations of trees are located, places where the wildlife associated with ancient trees has the greatest potential to survive, and lobby for their protection. It would help us to monitor current threats and losses, so we could lobby for greater sustainability of this vital historical and biological resource. Mapping this extraordinary asset is a crucial step towards a more enlightened approach to ancient trees.

Fragmentation effects on the state of conservation of forest and shrub ecosystems in "Sierra de Guadarrama" future National Park (Madrid, Spain)

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Abstract

Guadarrama Mountain Range has a high ecological value due to the convergence of particular biological, physical and cultural processes. Natural resources planning requires knowledge of the environmental determinants that characterise this area; altitude, ecological corridors, discontinuities, local gradients, etc. The characteristic vegetation, *catena*, of the study area grades from low to high altitude through holm oak forest (*Quercus ilex* subsp. *ballota*), oak forest (*Quercus pyrenaica*), pine forest (*Pinus sylvestris*) and high mountain shrub (*Cytisus oromediterraneus*). The study is focused on these four vegetation units. The level of fragmentation due to transport infrastructure is measured in relation to these vegetation units, as well as in terms of its effects on the associated fauna.

Fragmentation is understood as a dynamic process in which the habitat is being progressively reduced into smaller fragments. The principal consequences are decreasing area and increasing isolation of fragments. Fragmentation also limits exchange of individuals between populations and recolonisation of fragments. This provokes reduction in population size and an increase in the permeability of fragments to surrounding habitats, inter-fragment distance and the perimeter:area ratio. This implies negative edge effects that could impact on populations and make difficult the exchange of matter and energy with the progressive loss of species.

The effects of fragmentation on populations vary due to spatial distribution patterns and peculiarities of landscape history. A model based on measurement of several parameters has been developed, including quantitative and qualitative variables such as species richness, presence of endemic or threatened species, etc. This allows identification of which areas have the highest ecological value and are of greatest conservation importance. It also enables determination of those areas where fragmentation must be avoided in future and areas where defragmentation measures should be targeted. Finally, mitigation measures are proposed to preserve ecological processes and protect the natural values associated with them.

Studies into the application of a landscape-scale management framework for woodland in Wales

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Abstract

The Countryside Council for Wales (CCW) is developing a landscape-scale management framework for woodland. Different management treatments produce different structural types of woodland, which often benefit different aspects of biodiversity. The framework promotes the co-ordination of management within ecologically linked areas, so that a wide and suitably located structural diversity develops. This framework approach is being applied theoretically to example areas in southern Wales (Gwent and Gower) and this paper reports on the progress of these case studies.

Digital maps of woodland cover based on CCW's Phase 1 Habitat Survey and Ancient Woodland Inventory were prepared for each case study. Where available, provisional information on existing and past management for each woodland polygon was collated from survey reports and management plans and entered into the GIS database. Four basic management treatments were considered in the framework: minimum intervention; managed high forest; coppice; and wood pasture or parkland. This exercise provided an opportunity to assess the practicality of this theoretical classification. Information on possible constraints on management, including both economic aspects (e.g., productive forestry) and presence of rare species was incorporated. The spatial relationships of management types were assessed, and the extent to which recommendations for implementation or changes in management to optimise the overall structural diversity and conditions for rare species were explored.

The framework provides a way of identifying a suitable structure of any given wood. As such it helps set conservation objectives, which are essential for condition assessment, monitoring and reporting. These wider applications are discussed. The relationships of the framework to the development of a forest habitat network in Wales are also considered.

Targeting the re-creation of woodland habitats in a fragmented landscape

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Abstract

The Kent Lifescapes Project, part of English Nature's Lifescapes Programme, aims to develop an integrated vision of landscape planning across the entire county. As part of this programme, a series of robust ecological rules have been developed for targeting re-creation of priority habitats under the Kent Biodiversity Action Plan, including a series of woodland types. The rules take account of the make-up of the landscape, including the current distribution of habitats and protected areas in the county.

This poster sets out the method for targeting woodland re-creation in Kent and shows the results of the analysis at both a local (field) scale and at the more strategic scale of the Landscape Description Unit. A discussion of how these results are being used to inform land use planning at both scales across Kent sets the work in its practical context.

The application of landscape analysis to identify priority woodlands for red squirrel (*Sciurus vulgaris*) conservation in Scotland.

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Abstract

A need to address concerns about habitat loss and fragmentation and the implications for species with specific habitat requirements have driven the development of landscape scale conservation strategies, and related GIS tools. For many priority species conservation strategies at the landscape scale involve the identification of priority areas for de-fragmentation of habitat to increase connectivity between sub-populations and increase metapopulation viability. However, habitat de-fragmentation is not always the most appropriate or suitable approach to ensure future population viability, particularly where it allows competing or predator species access to less competitive or vulnerable, priority conservation species. A good example is the continuing displacement of the red squirrel throughout its native range in the UK, by the introduced alien grey squirrel (*Sciurus carolinensis*).

Since their introduction in the late 1800's grey squirrels have displaced the native red squirrels throughout much of their range. This is due to both competition and the presence of parapox virus disease, which is fatal to red squirrels but not to greys. Red squirrels do appear to be able to compete more effectively, however, in pure conifer habitats or areas with few large-seeded broadleaves. Hence large conifer forests (2000+ ha) surrounded by habitat unsuitable for grey squirrels and with no large-seeded broadleaves have been identified as priority habitat for red squirrel conservation (Pepper and Patterson, 1998).

Scotland presently supports the largest proportion of the UK's red squirrel population. With red squirrels increasingly marginalised from deciduous habitat by encroaching greys, conservation efforts have focused on identifying large conifer blocks as red squirrel refuges. Reynolds and Bentley (2001) developed a set of rules for prioritising red squirrel habitat based on; woodland size, composition, management and defendability against grey squirrel incursion. Their approach has been applied, using GIS analysis techniques, to Scotland (excluding Dumfries & Galloway and the Borders) to provide a strategic overview of red squirrel distribution and the location of suitable woodlands where management for red squirrels may be considered a priority. All woodlands with over 200 ha of contiguous conifer cover and less than 5% broadleaves, and which fell within the red squirrel range (identified by 10 km^2) but which contained no grey squirrels were initially identified. Detailed habitat composition data for these woodlands, and the surrounding area, was obtained through meetings with regional representatives, questionnaires and telephone interviews. The woodlands were then scored for 12 factors to provide a ranked list of priority woodlands. One hundred and twenty seven woodlands fulfilled the selection criteria. These were primarily found in Grampian, Highland, Tayside and north Strathclyde. The supporting factors

identifying priority woodlands varied between regions; 'distance from greys' was a major influence for the Highland region and strong 'landscape defendability' was the main factor in north Strathclyde. Three key areas where control of grey squirrel incursion should be a priority were highlighted; the Dee and Don valleys, the area surrounding Inverness and the heads of Loch Long and Loch Awe.

Reference

Pepper H. & Patterson G. (1998) Red Squirrel Conservation. Forestry Commission Practice Note 5, Forestry Commission, Edinburgh.

Reynolds J. & Bentley S. (2001) ***Selecting refuge sites for red squirrel conservation. Unpublished paper to the England Squirrel Forum.***

Investigating long term changes in woodland bird populations

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Abstract

Populations of breeding woodland birds in Britain are tracked by the UK Government's headline indicator for wild birds. The latest indictor (December 2003) shows a decline in woodland bird populations of around 20% in the last 25 years. Furthermore, several species have been added to the Red and Amber Lists of Birds of Conservation Concern within the last year on the basis of declines detected by the Breeding Bird Survey (BBS) and its predecessor the Common Bird Census (CBC). There are certain limitations to the data used to generate these alerts that require caution. Firstly, geographical coverage is not uniform. Changes happening in the south-east of England, where coverage is good, may mask those taking place in the north and west (although the BBS improves on the CBC in this regard). Secondly, for some key woodland species sample sizes have become so low that the reliability of the estimates of population change is questionable.

Seven leading hypotheses explaining the population declines have been highlighted in a recent study (Vanhinsburgh *et al.*, 2003) that reviewed a wide range of possible causes; from small fine-scale trends in habitat quality (such as browsing pressure and nest site availability) to large-scale environmental trends (such as climate change and air pollution), in addition to landscape-level processes such as habitat loss / fragmentation and changes to the agricultural matrix surrounding woodlands. Current on-going monitoring systems (e.g., BBS) are not undertaken in sufficient detail to allow us to assess these possible causes, so in 2003 work started on a national re-survey of woodland birds funded by the Department for Environment, Food and Rural Affairs, English Nature, the Forestry Commission, RSPB, BTO and the Woodland Trust. This will allow us to make preliminary tests of the hypotheses whilst providing more reliable estimates of population changes than are currently available.

The survey comprises revisiting over 350 sites for which historical data on bird abundance are available. BTO fieldworkers are revisiting former woodland CBC sites that were covered between the 1960s and 1980s and RSPB fieldworkers are revisiting woodlands surveyed in the 1980s using point counts. Detailed measurements of habitat structure were taken in the original survey of RSPB sites which will be repeated alongside the bird counts to enable us to look for correlates between changes in bird populations and changes to habitat. New measurements, such as detailed examination of deer impacts and squirrel numbers, have been introduced which we will attempt to relate to changes in bird numbers. The sites being surveyed provide a good geographical spread across England, Scotland and Wales so it will be possible to examine regional differences in population changes, which may be a key issue. The application of a GIS will make it possible to relate bird population changes to characteristics at the landscape level, using datasets such as the Forestry Commission's National Inventory of Woods and Trees and Land Cover Map 2000.

The findings from this survey will provide a much more substantial basis for planning the conservation of British woodland birds and setting the priorities for future research.

Reference

Vanhinsbergh, D.; Fuller, R.J. & Noble, D. (2003) *A review of possible causes of recent changes in populations of woodland birds in Britain.* BTO research report no. 245.

Pollinator turnover in ancient woodland fragments

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Abstract

It is widely held that fragmentation of all habitat types is a major threat to biodiversity. While many examples of grassland studies can be found in the literature, few address the impact of habitat fragmentation upon the flora and fauna of ancient semi-natural woodlands and none from the plant-pollinator perspective over geographical scales. With more than 78 threatened species, this habitat is a high conservation priority.

My project will examine the impact of habitat fragmentation upon the life history of ancient semi-natural woodland indicator and non-indicator plants. Primarily focusing on pollination, I will be analysing the implications for floral reproductive success, and how, in turn, plant demography impacts upon plant-pollinator interactions. These aspects will be examined in relation to local and landscape processes, agriculture, woodland management, and climate.

The ecology and conservation potential of "clough" woodlands within the dark peak natural area of the peak district national park, UK.

I. The woodlands of the dark peak: creation of a natural area woodland GIS

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Abstract

This study addresses the conservation of the Upland Oakwood habitat at the Natural Area scale by examining the landscape ecology of the woodland resource. Local Biodiversity Action Plan (LBAP) targets have recently been set for the conservation, restoration and expansion of Ancient Woodland Sites and the expansion of woodland within moorland fringe valleys known locally as "cloughs". These targets were originally based upon an assessment of the Ancient Woodland Inventory (AWI) data but did not identify priority sites. A Geographic Information System (GIS) has been created utilising available landcover data combined with fieldwork to enable an accurate assessment of the composition of woodlands within the Natural Area and the presence of priority LBAP habitats. Surveys have recorded the woodland structure, composition and management of compartments on Ancient Woodland Sites. Woodlands requiring fieldwork were identified within the GIS to record plantation or semi-natural status, ensuring all woods were recorded to Phase 1 Habitat Survey level. The study identified this as the minimum level of woodland classification allowing patch and landscape characteristics to be based on biologically relevant levels of detail and enabling areas of LBAP priority habitats to be identified. A further range of data at the site and landscape scale will be calculated within the GIS.

The current stage of the study involves the creation of the woodland GIS describing the status and distribution of the woodland network. The comprehensive nature of the woodland GIS allows an investigation into the effects of data availability and the accuracy of GIS-based data as indicators of site conservation value, on the development of conservation strategies at this scale. The different assessments of woodland cover, composition and patch characteristics produced from alternative data sources will be examined and the implications for development of conservation strategies discussed. This will include a detailed description of the woodland resource of the Dark Peak. The accuracy of the AWI will be examined together with the implications of the varying methods of combination of digital AWI data and National Inventory of Woodland and Trees data to define Ancient Woodland composition at this scale.

Future stages of the study will develop a GIS scoring system to identify priority LBAP sites based on a combination of values calculated in the GIS, recorded during fieldwork and resulting from classification of the landscape matrix. Comparisons will investigate the implications for the resulting network of prioritised sites of basing landscape strategies on different data sources. A further stage will investigate any relationship between Ancient Woodland quality indicator values recorded at the site scale through fieldwork and data collected within GIS to investigate the accuracy of GIS based assessments of site quality using landscape data.

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