Strategic woodland conservation planning: landscape ecology, landscape assessment and Geographic Information Systems

A case study examining habitat quality modelling and the prediction of Upland Oakwood biodiversity within "clough" landforms of the Dark Peak Natural Area, Peak District National Park, UK

VOLUME I

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ACRONYMS AND GLOSSARY

ArcView	GIS software package
ASNW	Ancient semi-natural woodland
AW	Ancient woodland
AWInv	Ancient woodland inventory
AWIS	Ancient woodland indicator species
BAP	Biodiversity Action Plan
Dissolve	ArcView command which merges polygons with the same values in a classification, into a single polygon
EN	English Nature (now Natural England)
ESA	Environmentally Sensitive Area
FC	Forestry Commission
GIS	Geographic information system
Landcover map 1990	Digital landcover map of the UK, created using remote sensed satellite data
Landcover map 2000	Digital landcover map of the UK, created using remote sensed satellite data, increased data accuracy and resolution compared to the Landcover map 1990
LBAP	Local Biodiversity Action Plan
Leptokurtic	Distribution, e.g. of species dispersal events, where most instances occur at short distance but rare data point occur at very long distance creating a distribution with a very long "tail" to the distribution
NT	National Trust
NIWT	National Inventor of Woodland and Trees (Forestry Commission survey)
OS	Ordnance survey (UK mapping organisation)
Patch Analyst	Habitat analysis extension to ArcView 3.2 GIS
PAWS	Plantation on Ancient Woodland Site
pdAWI	Ancient Woodland Inventory updated for the Peak District area by map analysis
PDNPA	Peak District National Park Authority
Phase 1 survey	Standard UK fieldwork habitat survey classification methodology
Phase 1 data	Data classified within the alphanumeric classification system of the Phase 1 habitat survey
Resolution	Accuracy of information, relating to GIS data
SAC	Special Area of Conservation
Sliver	Small, usually linear polygons that do not represent intended "data" but represent error polygons.
SSSI	Sites of Special Scientific Interest
TPI	Topographic Position Index
Union	ArcView command combining the extent and data contents of two previously separate GIS themes
upAWI	Updated Ancient woodland Inventory

Abstract

Research assessed the ecology, conservation interest and restoration potential of Ancient Woodlands at the Natural Area scale, as a model for examination of biodiversity-abiotic data associations. Biodiversity indicators including richness, composition and structure were collected. Biodiversity scores were developed directly from the indicators in addition to ordination, detailing associations and clustering between indicators and allowing further analysis of biodiversity-abiotic variable associations in reduced dimensions. A woodland GIS was constructed including classification of the landscape matrix, incorporating modelling of native semi-natural woodland "clough" landform topography zones. Analysis showed abiotic, GIS collated, woodland patch and landscape data to be associated with biodiversity levels. Habitat type and within-patch habitat quality were significant predictors of biodiversity levels within theory developed sequential multiple regression models (r^2 = .37 to .72). Most variance was explained by patch-level variables (habitat type, area and within-patch habitat quality), with lower levels explained by landscape-level connectivity, once patch-level factors had been included in models. However several regional trends remained. The models showed significant interaction occurred between effects of patch area and within-patch habitat quality. Examination revealed that while within-patch habitat quality was consistently associated with higher biodiversity levels, patch area showed a contradictory relationship when examined among the biodiversity ordination scores. Small, but topographically diverse, woodlands occurred which had high biodiversity levels for their unit area. Analysis indicates that in upland areas woodland patch biodiversity may successfully be predicted by use of woodland habitat type and within-patch habitat quality levels (topographic diversity and presence / distance to watercourses). As a case study a GIS model was used to map predicted woodland biodiversity as areas of conservation priority for Upland Oakwood conservation, restoration and creation, within the Dark Peak Natural Area, using targets set by the Local Biodiversity Action Plan, illustrating the use of the method in strategic conservation planning.

PART I: Introduction

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Chapter 1
Introduction and research aims

Dark Peak Clough woodlands: the conservation of Upland Oakwoods at the landscape scale

1.1 Introduction

1.1.1 Habitat fragmentation and woodland conservation

Woodlands are prominent features in the English landscape, their location and form showing the long history and strong influence of man's management activities. Native semi-natural woodlands, in addition to their scenic value and contribution to local landscape character (Bell, 1998), have a high importance for nature conservation, supporting a wide range of species (Peterken, 1996, Peterken, 1977a). In particular Ancient semi-natural woodland (where sites have known longevity since at least 1600AD) are of considerable importance for UK woodland conservation (Marren, 1992, Spencer and Kirby, 1992, Thomas et al., 1997, Peterken, 1996, Rackham, 2003, Peterken, 1977a, Kirby and Goldberg, 2005) and are often the focus of conservation schemes (Chapter 5). Native semi-natural woodlands currently occupy positions to the margins of man's influence, on a landscape-scale occurring in areas such as the moorland fringe (Milsom et al., 2002), but also surviving within more intensively managed landscapes along steep banks and stream valley features, variously termed denes, ghylls, dens and cloughs (Burnside et al., 2006). Over time, and with changing agricultural and forestry land use this native semi-natural woodland resource has become a scarce and fragmented habitat (Rackham, 1999, Rackham, 2003, Peterken, 1993, Peterken, 1977b).

Studies examining the effects of habitat fragmentation, characterised by increasing isolation between habitat patches and decreasing patch size, in addition to reduction of overall habitat area, have shown negative effects on species populations, especially habitat specialists (Andren, 1994, Fahrig and Merriam, 1994, With and Crist, 1995, Saunders et al., 1991). Detrimental effects have been predicted (Fahrig, 1997, Gardner et al., 1987, Sondgerath and Schroder, 2001, With et al., 1997) and shown by observational studies (Mouflis and Buckley, 2004, Bennett and Radford, 2004) to be worse when the habitat is at low levels in the landscape, below the percolation threshold (Chapter 4). Percolation theory has noted certain arrangements of patches result in "thresholds" where a majority of patches are inter-connected and processes, such as dispersal, may cross the landscape with greater ease, or as if it were a single patch (Farina, 1998, Forman and Gordon, 1986, Andren, 1994). Particularly strong fragmentation effects may occur for a variety of species when a habitat occupies less than 10-20% cover (Andren, 1994).

Current national woodland cover averages 8.6% (Anon, 2005a), although higher figures occur in local landscapes (Reid et al., 1996), or where particular forms of woodland habitat are locally clustered (Kirby and Thomas, 1994). Within this total cover lower levels of native semi-natural woodland occur. Previous studies and interpretation from regional cover figures suggest likely national cover of 3-4% semi-natural woodland (Thomas et al., 1997, Anon, 2005a, Gkaraveli et al., 2001, Latham et al., 2004, Peterken, 2002a). Semi-natural woodland cover is thus well below the predicted 10-20% percolation threshold in many landscapes, and is predicted to show strong fragmentation effects. The habitat has been fragmented for long periods of time, such that active fragmentation effects may be most relevant to future conservation / restoration efforts than to current active impacts on existing quality. However recent research has shown that woodland cover remains dynamic with recent losses and gains occurring across the British landscape (Petit et al., 2004b).

Despite long-term woodland decline, woodland conservation activities have a long history from early Forest Law within deer parks (Rackham, 1999) and post-war forestry conservation initiatives (Kirby, 2003a) to the more recent instigation of the Nature Conservation Review (Ratcliffe, 1977) and creation of Sites of Special Scientific Interest (Kirby and Solly, 2000), in addition to many charity led individual nature reserves. However only the most prominent, largest or most important sites are typically protected by formal designation, while other sites are subject to the changes in the fortunes and management practices in the broader countryside (Kirby and Solly, 2000, Marren, 1992). Indeed even within "ancient woodland" sites, acknowledged as generally the finest examples of native woodland remaining, only 14% are formally protected within SSSI reserves (Thomas et al., 1997), and many sites remain subject to recent losses and management impacts (The Woodland Trust, 2000a). Beyond designated areas government subsidies (Forestry Commission, 2006, DEFRA, 2006) may support management of sites within private ownership, but are dependent on individual landowners taking up such schemes voluntarily, and therefore are often spread randomly through the landscape.

1.1.2 Landscape ecology, woodland context and woodland conservation

Insights gained from both landscape ecology theory (Forman and Gordon, 1986, Forman, 1995a, Forman, 1995b, Turner, 1989) (Chapter 3) and habitat fragmentation studies (Andren, 1994, Fahrig, 2003, Collinge, 2000, Collinge, 1996) (Chapter 4) indicate that conservation sites cannot be considered in isolation from their surrounding landscape. Habitats such as woodland are affected by the way the "landscape matrix" affects species movement, dispersal or colonisation and by potentially negative "edge-effects" of operations such as fertiliser or pesticide use within adjacent habitats (Hansson and Angelstam, 1991). The species composition of woodland can alter with the form of landscape matrix surrounding sites (Sisk et al., 1997), while the overall landscape configuration can affect movement of species between sites and relative abundance or breeding success within individual woodland patches (Hinsley et al., 1994). Such studies show the importance of, and interaction between landscape characteristics such as patch size, isolation and the resistance of the landscape matrix. The species composition and relative conservation interest of woodland patches may be determined by a mix of influences from landscape scale (Chapter 4) and within-patch woodland ecology (Chapter 5) characteristics. A number of landscape ecology studies have measured relevant patch and landscape features, using landscape metrics, and examined their effect on woodland species ecology (Chapter 4).

An awareness of the importance of landscape ecology in addressing the effects of habitat fragmentation has increasingly informed woodland conservation and landscape planning (Ferris et al., 2000, Watts et al., 2005). Landscape context is important in affecting conservation value, and woodland sites, including those managed under government conservation grants, are affected by their location. The often random location of grant schemes chosen by landowners may not lead to the most effective use of conservation resources. Theories extended from island biogeography and metapopulation theory indicate that important aspects in the conservation of sites are proximity to other reserves, size and shape of reserves and the structure and land-use of surrounding habitats (Diamond, 1975, Forman and Gordon, 1986, Margules and Usher, 1981, Margules et al., 1988, Pressey et al., 1993, Kirby, 1995) (Chapter 4).

1.1.3 Strategic, landscape-scale woodland conservation planning

Applied conservation planning research has shown that maximum conservation gain can be achieved by considering the relative condition and importance, not only of individual sites to be conserved, designated, or created, but also of the surrounding landscape, isolation and connectivity levels, and the likely impacts these may have on conservation value (Opdam et al., 2006, Margules, 2005, Kangas, 2005, Opdam et al., 2002, Hawkins and Selman, 2002). Concerns for landscape-scale woodland conservation have become elevated due to the effects of agricultural intensification in the landscape matrix in addition to potential effects of climate change (Peterken, 2002b, Dudley, 2001). Landscape ecology provides the framework by which the impact of landscape structure, measured through an analysis of patches, corridors and matrix can be assessed on species populations. Conservation biology and landscape planning, utilising Geographic Information Systems (GIS) and ecological modelling methods can take such relationships and use them in the design of conservation networks, ensuring optimum use of conservation resources. The consideration of the relative importance of site and landscape-scale effects is particularly important in planning that increasingly addresses not only conservation (preservation) but also habitat restoration and creation (Sheail et al., 1997, Young, 2000, Honnay et al., 2002a). Such plans are promoted by policy (Biodiversity Research Support Project, 2000, United Nations Secretariat of the Convention on Biological Diversity, 2001, The UK Biodiversity Steering Group, 1995a, The UK Biodiversity Steering Group, 1995b, Forestry Commission, 1998, Department of the Environment, 1994, Anon, 2005b) and allow a strategic assessment to be made of a habitat resource, and to consider how past losses and current fragmentation levels can be addressed by setting targets for conservation of existing sites, restoration of formerly occurring sites and the creation of new habitat. Key questions raised in the formulation of such plans are: what is the extent and condition of the existing habitat resource?, where are the most important sites?, where are the areas of highest biodiversity?, what is the structure of the habitat network?, where can priorities for woodland conservation, restoration and creation be located such that current biodiversity levels are maintained and future biodiversity levels, across the entire habitat network are maximised?

Conservation policy initiatives increasingly explicitly state conservation goals as addressing habitat fragmentation levels, and landscape structure, in addition to simple area creation or conservation targets (Forestry Commission, 1998, Peak District National Park Authority, 2002, Anon, 2005b). The potential of landscape ecology to inform forest management has been recognised at a number of scales (Zavala and Oria, 1995, Boutin and Hebert, 2002, Baskent and Keles, 2005), and key areas in addition to landscape ecology are the use of GIS for interpreting such planning. Research has begun to examine how these aspects of woodland conservation can begin to combine assessment of site value with assessment of the local woodland landscape or woodland network.

1.1.4 Biodiversity surrogates and woodland conservation planning

The creation of Biodiversity Action Planning (BAP) targets (The UK Biodiversity Steering Group, 1995b) raises challenges of how to implement such recommended actions and identify conservation priorities and sites for action. Because biodiversity planning exercises rarely have accurate assessments of the extent or condition of the entire habitat resource, and never have access to full biodiversity censuses within existing sites or potential restoration or creation sites, various forms of biodiversity surrogate and indicator measures must be used. Surrogate and indicator methods rely on the collection and use of easily measurable data that are themselves indicative of broader, but unmeasured, biodiversity levels (Niemi and McDonald, 2004, Caro and O'Doherty, 1999, Caro et al., 2005). Various indicators have been suggested and utilised (Ferris and Humphrey, 1999, Noss, 1999, Gustafsson, 2000, Lindenmayer et al., 2000, Wilson et al., 2001, Peterken, 1974, Uliczka and Angelstam, 2000, Peterken, 2000a, Rose, 1999, Spencer, 1990, Miles and Miles, 1997). Initial research used existing survey data, or rapid site surveys to collect species presence data, from which analysis allowed identification of species particularly indicative of wider biodiversity presence, for potential use in additional sites or areas, examples being ancient woodland ground-flora and avian guilds (Peterken, 1974, Mikusinski et al., 2001, Rose, 1999). Recently abiotic biodiversity indicators, inspired by landscape ecology theory, have also been used. These include the use of patch area, shape and inter-patch isolation (Lee et al., 2001b, Nikolakaki, 2004, Thompson et al., 2001b, Nikolakaki, 2001), or of total area of habitat in the landscape to indicate likely biodiversity levels in the habitat (Peterken, 2000b, Peterken, 2002b, Peterken et al., 1995). These indicators are based on abiotic patch or landscape conditions, rather than species presence and are easily obtainable from remote sensed data and are applicable for rapid use and manipulation within GIS. Such indicators may be used directly, justified by theory or expert opinion, or relevant abiotic values or thresholds may be derived from examination of species-area associations or complex assessment of focal species land-use requirements (Lambeck, 1997, Watson et al., 2001, van Rooij et al., 2004). The use of either of these two types of indicators (abiotic and biotic) allows biodiversity planners to minimize data requirements while aiming to map or quantify relative current or future biodiversity levels across a landscape. Key areas in the development of these studies was the move from structural assessment of landscape to functional assessment (Ray et al., 2004b, Belisle, 2005, Murphy and Lovett-Doust, 2004, Adriaensen et al., 2003), the debate arising over the value and rigour of the use of biological species (Armstrong, 2002, Caro, 2002, Lambeck, 2002, Lindenmayer et al., 2002), and of how to accurately capture multiple species use of a habitat at multiple, appropriate scales (Wiens, 1989, Holland et al., 2004, Humphrey et al., 2004, Cushman and McGarigal, 2002, Vos et al., 2001, Cushman and McGarigal, 2004). Therefore a wide range of methods and implementation can be seen in the literature, assessing habitat patch and landscape abiotic form, or indicator species presence, and linking these to relative perceived or planned conservation priorities.

1.1.5 Current UK initiatives in woodland conservation assessment and planning

Landscape ecology theory and biodiversity indicators have now been widely used in UK woodland conservation planning, resulting from academic research (Lee et al., 2002, Thompson et al., 2001b, Bailey et al., 2006, Bailey et al., 2002), and from applied studies within conservation organisations (Smithers, 2000, Kirby, 1995, Kirby and Reid, 1997, Good et al., 1997, Buckley and Fraser, 1998, Peterken et al., 1995, Good et al., 2000) (Chapter 6). With the development of Local Biodiversity Action Plans woodland conservation planning has been undertaken at county and regional levels (Peak District National Park Authority, 2002). Early UK woodland conservation work undertook simple assessments of conservation priorities within Natural Areas, on a national level (Reid et al., 1996, Kirby and Reid, 1997). Natural Areas are units of relatively homogenous landscape character considered suitable for landscape planning (English Nature, 2005, Hamilton and Selman, 2005). More recent research has involved the identification of zones of land-use planning smaller than Natural Areas within which relative priorities can be identified for woodland conservation (Anon, 2005c, Latham et al., 2004, Latham, 2003).

Strategic spatial woodland conservation activities can result in woodland conservation sites and wooded landscapes with positive conservation features, such as enhanced connectivity compared to landscapes addressed by random owner uptake of available conservation schemes (Buckley and Fraser, 1998, Good et al., 1997, Thompson et al., 1999b). Additionally research showed that consideration of the form and context of local landscapes within woodland creation scemes could result in sites with higher biodiversity levels being created for lower economic costs, than sites created in less ecologically favourable areas in the landscape (MacMillan et al., 1998). Active strategic conservation planning therefore has both biodiversity and economic cost-benefit implications.

Recent research has begun to identify suitable methods for woodland landscape-scale conservation planning, principally utilising GIS to combine existing data sources, to interpret relevant biodiversity indicators and incorporating landscape ecology theory and spatial planning strategies (Purdy and Ferris, 1999, Gkaraveli et al., 2004, Latham et al., 2004, Thompson et al., 1999c, Thompson et al., 2001b). This range of research however has used a variety of methods and biodiversity indicators. Studies have utilised existing biological records from woodland sites based on previous fieldwork (Lee et al., 2002), have examined national species and woodland datasets (Bailey et al., 2002), used umbrella or focal species methods (Nikolakaki, 2004, Ray et al., 2004b, Bani et al., 2006), have collected additional biological records (Honnay et al., 1999a, Honnay et al., 1999b, Jacquemyn et al., 2003) or have focused on abiotic, patch, environmental or topographical aspects of the woodland network in order to prioritise sites (Lee et al., 2001b, Thompson et al., 2001b). Often in UK examples of these prioritisation systems woodland type or designations have been utilised as representative of levels of habitat quality, although these have not always been justified or based upon literature (Purdy and Ferris, 1999, Gkaraveli et al., 2004). Broad woodland site characteristics have been assumed rather than measured in such analysis. No UK studies have justified their scoring methods based on thorough assessment of existing woodland networks and analysis of habitat–environment–structure relationships in established or ancient wood sites.

1.1.6 Incorporating habitat quality directly in woodland conservation planning

The range of research conducted on woodland conservation planning methods suggests mixed support for the types of active woodland conservation planning currently being implemented by organisations such as the Forestry Commission, Woodland Trust and English Nature (now Natural England). Conservation resources are limited and therefore cost-effective methods of planning or biodiversity assessment using indicators are desirable, but work must ensure they are also suitably justified and reliable. Key research areas emerging from recent studies (Chapters 3, 4, 6) are functional connectivity assessment issues, multi-species assessment and patch quality affects. Analysis can be criticised if it takes a too abstract assessment of woodland landscape where e.g. patch abiotic size and isolation conditions are used without assessment of within-patch quality or landscape resistance / functional connectivity effects. Additionally generalised relationships derived from theory, expert opinion or from previous research may not necessarily apply in the landscape in which conservation planning is being undertaken. Woodland is a complex, structurally diverse habitat which is much affected by human management. Previous research has highlighted the likely importance of within-patch woodland quality (diversity, heterogeneity) to woodland biodiversity, e.g. for flora (Peterken and Francis, 1999, Peterken and Game, 1984, Bastin and Thomas, 1999), while such within-patch conditions are often included in exploratory studies of avian fauna biodiversity in woods (Hinsley et al., 1994, Bellamy and Hinsley, 2004, Bellamy et al., 1996a, Mason, 2001). Previous works examining woodland biodiversity prediction have noted that inclusion of further variables such as topography and woodland structure, and therefore relating to habitat quality, could usefully be investigated in future studies (Thompson et al., 2001b). Such research suggests that within-patch woodland quality can be related to flora, fauna and management effects.

Abiotic woodland patch conditions (patch area or shape) may simply be indicative of within-patch habitat quality / diversity, which itself it linked to woodland biodiversity levels (Peterken and Game, 1984, Honnay et al., 1999b), while the supposed poor biodiversity quality of small woodlands has been questioned (McCollin et al., 2000, Hinsley et al., 1994, Peterken, 2000b, Dolman and Fuller, 2003, Gotmark and Thorell, 2003, Lawesson et al., 1998, Honnay et al., 1999b, Young et al., 1996). Complexly shaped or larger woodland are expected to encompass a wider range of soil types and environmental conditions, factors which are causally associated with higher recorded biodiversity under such conditions due to the diversity of niches and resources. Diverse soil and hydrology conditions and favourable management regimes, associated with topography factors may be important in dictating biodiversity levels (Peterken, 1974, Dzwonko and Loster, 1992, Stahle and Chaney, 1994, Therrell and Stahle, 1998, Larson et al., 2000, Coroi et al., 2004). This relationship is expected to be especially strong when additional biodiversity indicators, including management and woodland structural effects are included beyond species richness. Because abiotic patch conditions such as woodland size are frequently used in biodiversity planning this interaction between within-patch quality and patch abiotic conditions is of particular interest. If such habitat quality measures were able to be directly measured they could potentially prove more suitable for use in woodland biodiversity planning then simple patch shape and area measures.

1.1.7 GIS based woodland conservation planning and collection of patch abiotic data

The application of conservation planning at larger spatial scales requires a reduction of woodland biodiversity systems to more simple levels, capable of classification or modelling to aid their interpretation or use in landscape planning. Such simplifications may utilise pure landscape ecology theory, assessment of habitat cover or the use of abiotic indicators. It remains unclear what mix of abiotic patch data, landscape data and within-patch quality data may most influence woodland site biodiversity. Such affects are also influenced by the measures of woodland biodiversity used. However there are strong indications of the importance of within-patch habitat quality in its association with diverse niches and management levels. How such combined effect data are collected and analysed is a challenge. The types of woodland site, and broader woodland networks envisaged and created by action planning and how they fit into local landscape character, or reflect existing woodland form is also an important factor, requiring incorporation of landscape assessment and mapping abilities.

GIS is recognised as a suitable study medium and method for implementation of conservation planning. GIS has a long history of use for woodland planning due to the ease of distinction of woodland habitats from adjacent habitats in remote sensed data (Brown and al, 1994, Petit et al., 1996, Hargis et al., 1999, Twery et al., 1991, Moore et al., 1991, Naesset, 1997, Baskent, 1996, Rickers et al., 1995). Increasing data accuracy and inclusion of model algorithms in GIS packages allows an increased range of abiotic data to be recorded from within woodland patches. With high resolution Digital Terrain Models (DTM) data and accurate vector habitat data a variety of within-patch conditions can be collated which may be indicative of within-patch woodland habitat quality / diversity. Indeed research has examined the potential to use GIS to model detailed site conditions such as soil types and hydrology across landscapes (Russell et al., 1997, Mummery et al., 1999, Thomas et al., 1999, Ryan et al., 2000, Mann et al., 1999, Lark, 1999). Research questions arise as to whether such measures are sufficiently accurate to allow their use in conservation planning and whether they hold more use than landscape ecology inspired spatial patch configuration and patch abiotic variables. Considerable research interest lies in whether such patch-level measures are more indicative of biodiversity levels for planning than local landscape-level factors. If GIS collated within-patch variables prove to accurately model deterministic factors driving woodland biodiversity and influencing management, then they hold promise for use in assessments of both current biodiversity levels, remnant levels for restoration and of the potential future levels able to be realised at woodland creation sites. Such GIS based analysis would then meet the requirements of conservation organisations for cost-effective conservation planning methods.

1.2 Research aims, objectives and research model

1.2.1 Research problem statement

Landscape-scale woodland conservation strategies require the use of GIS and surrogate biodiversity criteria to undertake rapid and efficient biodiversity planning. In England limited research exists with which to examine broader woodland biodiversity beyond assessments of woodland ground-flora and woodland avian fauna. Woodland biodiversity indicator research in the uplands is particularly limited. Several strands of research have utilised raw abiotic data to undertake woodland biodiversity planning. However several studies indicate simple patch values such as woodland size may not be well linked to biodiversity levels and instead may be driven by underlying factors of habitat quality. Woodland biodiversity may be strongly affected by patch habitat quality, across a number of biodiversity groups, and such affects may be linked to management intensity levels, or woodland occurrence, especially in the uplands. It is unclear whether woodland landscape-scale conservation strategies could be more effectively implemented by direct measurement and assessment of such features indicative of within-patch internal habitat quality, as a surrogate for patch biodiversity, rather than patch area and shape (or additional landscape structure variables) and the affect this may have on subsequent prioritisation of conservation areas. The relationships of woodland site biodiversity to such GIS collected variables have not been well studied in the UK. Current conservation and restoration schemes make poor use of the predictive power of woodland habitat type and patch habitat quality for indicating internal patch conditions. The difference in internal woodland habitat conditions and values between broad woodland types are poorly known.

1.2.2 Research focus

This study focuses on the examination of applied woodland conservation planning, at the landscape scale, investigating the combination of woodland landscape character assessment, descriptive GIS development, and the predictive modelling of woodland site habitat quality. The research addresses these areas through a case study of a specific woodland habitat type: Upland Oakwoods and native clough woodland within the Dark Peak Natural Area, Peak District National Park, UK (Section 1.4 and Chapter 2). The results of the research are used to formulate a local woodland conservation strategy, prioritising the allocation of conservation resources to woodland conservation targets set locally by the Peak District Biodiversity Action Plan (Chapter 11).

1.2.3 Research Aim

The thesis research aim is:

> To implement a woodland conservation strategy, addressing woodland conservation, restoration and creation, through analysis of the potential of landscape ecology theory and GIS collated abiotic woodland data to assess woodland biodiversity value at the woodland site, network and landscape scale
>
> - To assess the extent to which woodland habitat quality can be modelled and incorporated into conservation plans using GIS and remote sensed data
> - To use Upland Oakwoods within the Dark Peak Natural Area as a case study
> - To utilise Ancient Woodland sites and multiple site biodiversity indicators as the research model within which to examine associations between biodiversity and abiotic conditions

1.2.4 Research Objectives

A number of objectives arise from the research aims.

Literature review

- To review landscape ecology and landscape planning theory
- To review the effects of habitat fragmentation on woodland and woodland species
- To review factors driving woodland patch biodiversity
- To review woodland conservation and restoration methods at multiple scales
- To review associations and characteristics of woodland specialists, and species typical of Upland Oakwood and Ancient Woodland sites
- To review current woodland restoration strategies being applied at the landscape-scale
- To review the analytical opportunities utilising GIS data to formulate woodland conservation strategies

GIS creation, Dark Peak woodland and clough landscape assessment

- To determine suitable GIS and fieldwork data collation methods for researching woodland conservation at the landscape-scale
- To construct a GIS holding detailed information on the current Dark Peak woodland resource, topography, hydrology and habitats representing opportunities or constraints to woodland development
- To undertake a woodland based landscape character assessment of the Dark Peak
- To carry out a detailed analysis of the Dark Peak woodland resource, its landscape characteristics, composition and associations between ecological and topographical features
- To classify landscape zones characteristic of native semi-natural clough woodland occurrence: "potential clough woodland zone"
- To determine if available digital woodland and landcover data are sufficiently accurate to allow a rapid "Phase 1" equivalent habitat assessment at the Natural Area scale

Woodland biodiversity, conservation value, habitat quality and abiotic diversity

- To develop a new scoring system to summarise woodland site conservation value or restoration potential from multiple site-collated field data
- To assess the current factors associated with, or driving, woodland biodiversity and woodland structure in Dark Peak Ancient woodland sites
- To analyse the associations between woodland classification, quality, habitat heterogeneity, and woodland biodiversity at the site and landscape scale

Woodland landscape conservation strategies

- To examine some of the assumptions and developments within current UK ancient woodland and woodland conservation planning:
    - The use of abiotic patch values to plan spatial conservation strategies
    - The call for use of structural assessment within woodlands
    - The increasing use of AWI, NIWT or Phase 1 habitat data to plan conservation
- To evaluate which abiotic / GIS / landscape factors can be used to develop landscape-scale conservation strategies
- To develop a new method for incorporating woodland site habitat quality assessments into landscape scale conservation strategies
- To create a new woodland conservation strategy assessment system developed from analysis of woodland quality data in addition to use of abiotic patch and landscape value surrogate assessments
- To map the location of priority areas for woodland conservation, creation and restoration

1.2.5 Research Model

The objectives will be met by conducting research within a defined research model. It is impractical to examine all potential factors and therefore a number of assumptions must be made within the study methodology and analysis. The research model is based upon the following areas.

- GIS is a suitable medium within which to examine data and conduct planning at the landscape scale
- Data collection and modelling at multiple scales / distances can be used to increase the relevance of research application to multiple species groups
- The Ancient Woodland network, with known longevity and mapped semi-natural or replanted status is a useful model within which to examine local associations between woodland biodiversity, habitat quality and abiotic conditions

Ancient woodland sites, and their categorisation as ancient semi-natural (ASNW) or replanted ancient woodland (PAWS) allow study of the interaction between abiotic and biological diversity factors. At long established semi-natural sites it can be assumed that semi-natural landscape and site affects remain highly influential on resulting biodiversity levels. Within replanted PAWS sites human management impacts and canopy type are key driving factors in altering biodiversity levels. At such long established sites the environment / landscape / species links are presumed to be high. The ancient woodland system is therefore a suitable model from which to extend a defined conservation strategy to the general woodland cover within a Natural Area.

1.3 Research framework: Project axioms, past research limitations and research postulates

1.3.1 Research axioms

Following development of research objectives and consideration of the research model, the project utilised a number of axioms considered to be established, on which the research was based (Table 1.1). See Chapters 3-6 for the literature review used in the derivation of these axioms.

Table 1.1
Research Axioms

Dark Peak woodland and clough woodland landscape assessment

- Landscape mosaic model is a suitable method for ecological study of landscapes
- Utilisation of remote sensed or GIS collated ecological data can provide highly cost efficient mechanism for conservation planning versus costly fieldwork data collection
- Topographic position and slope angle are directly related to likelihood / potential of agricultural improvement in the uplands

Woodland conservation value, habitat quality and abiotic diversity

- Landscape composition and form impacts on ecological value and function, affecting species populations within habitat patches (although the form of individual species populations may range from isolated to metapopulations depending on species and scale)
- Assessment of functional landscape measures, e.g. connectivity will be more realistic and useful than simple structural assessments
- Ancient woodland sites are hotspots of woodland biodiversity and will be the focus of woodland conservation in a landscape
- Ancient woodland sites are good study system within which to examine associations between biodiversity values, abiotic values and within-site topography due to their longevity of woodland conditions
- Woodland site fauna richness / composition will be largely determined by woodland botanical composition, structure and diversity

Woodland landscape conservation strategy

- Habitat fragmentation is detrimental to species populations
- Landscape attributes can be used to predict species or community values within woodland patches
- Landscape ecology theory can be successfully used to develop spatial landscape scale conservation plans
- Landscape spatial planning for conservation is beneficial compared to random placement of reserves or sites under conservation management agreements
- Landscapes scale conservation strategies can be successfully devised from information extracted from landscape ecology theory, landscape assessment or species or habitat modelling
- Woodland landscape conservation strategies should address the connectivity of sites
- Conservation strategies applied at the cultural landscape scale will encompass a wide range of species populations forms potentially including metapopulations but are also likely to include fully mixing and isolated populations
- Area based metrics are assumed to be preferable to isolation based metrics to predict biodiversity-connectivity effects
- Indirect and direct gradients and variables can be used in GIS of model species / habitat planning and have been shown to be accurate predictors in a number of woodland studies
- As spatial and temporal scales become larger the importance and impact of functional rather than structural isolation becomes more important

1.3.2 Research justification: previous research requirements, gaps and limitations

The literature review (Chapters 3-6) noted a variety of research approaches relevant to woodland conservation planning at the landscape scale. Previous studies have highlighted a range of research areas requiring further investigation (Table 1.2). The current project will utilize a combination of previous research techniques and will address some of their shortfalls. Some of the key limitations to the previous research are;

- A lack of accurate data on the location and classification of the woodland resource
- Application of generalized landscape ecology "rules" without defining the exact scale of the process occurring or the desired outcome for the species or habitats in question
- A lack of testing of the suitability of abiotic or landscape attributes as substitute measures for other biodiversity features

- A lack of testing of the level to which priority scoring methodologies and the datasets used affect the results of the planned network

There is therefore a need for:

- The collection of a standard level of data to quantify the woodland resource at the Natural Area landscape scale
- Analysis to indicate the relative levels of accuracy of the currently available data sources used to estimate the woodland resource at the Natural Area scale
- A detailed investigation of landscape structure and woodland resource levels when defining the goals of a conservation strategy, the scale at which relevant processes are occurring and the landscape and woodland attributes that would most beneficially be altered
- Further identification of useful biodiversity indicator groups, including structural measures
- Priority scoring to be based on indicators of biological diversity or habitat quality at the site scale, allowing an examination into whether landscape scale / GIS indicators could be used to infer these values with a sufficient level of accuracy at the landscape scale

Table 1.2
Potential areas requiring further research investigation, identified from previous woodland research publications

Landscape ecology and landscape planning research requirements

- Development of methods to address species-landscape interactions across multiple scales
- Development of a range of suitable surrogate / indicator species for use in UK woodland planning
- Further development of functional connectivity rather than structural connectivity assessments
- Increased use of patch quality and heterogeneity values in landscape planning scoring methods
- Investigation and quantification of the cost-benefits of using surrogate or indicator species in landscape planning in order to more clearly define the assumptions and limitations of their use
- Identification of the abiotic patch and landscape factors most predictive of UK upland woodland diversity
- Identification of the relative importance of within-patch vs. patch abiotic and landscape values in UK woodland biodiversity

Habitat fragmentation research requirements

- Further development of methods to quantify the impact of habitat fragmentation on species across multiple scales
- Further clarification of the extent to which modelled / predicted impacts of fragmentation in neutral / artificial landscapes can be predictive of real world fragmentation situations
- Further clarification of which landscape metrics are most predictive of fragmentation effects in different species groups

Ancient woodland sites and woodland conservation ecology research requirements

- Assessment of the use of the Ancient Woodland Inventory (AWI) as a data source for conservation planning
- Identification of the extent to which Ancient woodland diversity, classification or structural features may be predicted from abiotic or topographic data
- Further clarification of the link between ancient woodland site occurrence, management and topography
- The extent to which current recommendations for AW and PAWS conservation can be developed into a spatial strategy

Woodland landscape conservation strategies research requirements

- Potential for incorporating functional rather than structural connectivity within strategy formulation
- The incorporation of assessment of woodland habitat quality within landscape assessment and habitat modelling approaches
- Investigate the potential to incorporate measures of habitat quality and structure in scoring systems
- Investigate and develop appropriate relationships between the scale of investigation and scales of ecological action and scales of assessment – in particular assessing the potential for multi-scale assessment methods
- Investigate the potential for application of alternative surrogate system than species based
- Investigate the potential to extend scoring strategies to separately value sites for conservation, restoration and creation
- Research into development of suitable abiotic factors / data that are drivers of woodland biodiversity, for use in strategy planning
- Further investigation of the extent to which small woods may be being under valued in current woodland strategies

Previous woodland research has been undertaken using a variety of data sources used to quantify the woodland landscape to inform the development of conservation strategies. These have ranged from the simple identification of Ancient Woodland status to broad woodland classification categories and site-based species diversity assessments. This range of data has often resulted from research not specifically designed to collect data for such purposes, or to be extrapolated for use at additional scales. Therefore the impact of availability of woodland classification data on the use of data for planning is of conservation interest. Frequent use is made of the Ancient Woodland Inventory (AWI) as a data source, upon which to base plans (Purdy and Ferris, 1999, Gray and Stone, 2003). However this is now becoming dated, there are issues over its accuracy (Forrest, 2001, Pengelly and Heath, 2003), and therefore its use in such schemes requires investigation. In addition to the AWI the UK has a rich supply of biological data (Griffiths et al., 1999), and various woodland data from site designations (SSSI, Nature Reserves) to categories of canopy or habitat classification (Phase 1, NIWT, Landcover Map 2000, Landcover Map 1990) have been used to assign values to woodland sites for use in landscape planning. Some studies have addressed automatic / remote classification of woodland from combining different data sets in order to classify AWI sites, and this requires further work (Pryor and Smith, 2002, Purdy and Ferris, 1999). The justification for these relative values and their accuracy for use in such planning require investigation. Much of the applied woodland conservation research has been conducted in lowland landscapes where set-aside or arable land is a key target for woodland creation and expansion and further applied research is required to examine the relative priorities in upland environments. The use of simple cover thresholds, such as the 30% cover target (Peterken, 1999, Peterken, 2002a, The Woodland Trust, 2002), or of individual patch area or isolation targets (Peterken, 2002a, Peterken and Francis, 1999) may be too simplistic for use in strategy planning. Therefore increased use of site habitat quality and structural / heterogeneity features needs to be made. Previous research has been limited in assuming patches of particular size or isolation are of similar value, ignoring relative differences in patch quality. In assessing such effects a broader assessment of patch diversity levels, beyond simple species richness is required, incorporating structural and composition factors (Ferris and Humphrey, 1999).

Key areas requiring further research are progressing methods to interpret and address species–landscape interactions at multiple scales and to incorporate assessment of habitat quality in landscape planning and landscape, or focal species assessment methods. The use of surrogate methods represents a trade-off between accuracy and levels of investment required; such limitations need to be fully addressed. It is of particular interest to know how diversity factors may be linked to structural and abiotic patch factors to allow rapid assessment of woodland sites, for example for use in producing a spatial version of the recent methodology identified for assessing the restoration potential of PAWS (Pryor et al., 2002). Outcomes of such work, including structural indicator based assessments can allow identification of key woodland biodiversity indicators. Where previous studies have often been based on broad woodland canopy classifications, the current study will enable a single Natural Area to be analysed in terms of woodland composition, fragmentation and size distribution, for each of several woodland canopy types. This data will be of interest in examining the proportions of woodland types within a single Natural Area and in considering the outcomes of other studies based purely on more limited woodland data. Data collection of this intensity will not be practical for conservation organisations to collect in order to formulate landscape based strategies. However the analysis conducted within the project in examining the relationships between field data and GIS based landscape attributes will allow useful indicators to be derived at this study scale.

1.3 Research postulates

Following the clarification of axioms and of potential research areas highlighted by previous research, detailed research postulates were derived (**Table 1.3**). These are discussed in later chapters, and where relevant were tested using null hypothesis with suitable data sources and statistical tests.

Table 1.3
Research postulates

GIS creation, Dark Peak woodland and clough landscape assessment

- Abiotic characteristics differ between woodland habitat types: there is an association between woodland habitat type and site topography and abiotic factors
- Semi-natural woodland location is strongly determined by topography
- The ancient woodland inventory (AWI) is an accurate digital dataset suitable for use in woodland assessment

Woodland biodiversity, conservation value, habitat quality and abiotic diversity

- Woodland biodiversity indicators can be compiled into a score representative of conservation value or restoration potential
- Biodiversity levels differ between ASNW and PAWS, and will be higher in ASNW sites and will differ between PAWS habitats, ranked from higher levels in broadleaved plantation to mixed plantation and lowest levels in conifer plantations
- Biodiversity can be represented by a reduced number of individual biodiversity surrogates that reflect the broader levels present
- Due to the abundance of potential abiotic variables a reduced range of factors will be able to be extracted by analysis to succinctly define woodland patch abiotic character
- The order and effect of accessibility, topography or woodland size on AW sites conversion (ASNW to PAWS) is unknown and therefore exploratory analysis will be undertaken of the similarities and differences in abiotic conditions between AW habitats and any trends in such conditions between the habitats
- Woodland patch biodiversity / structure values are associated with woodland abiotic variables
- Woodland patch biodiversity indicators can be predicted from abiotic surrogate / indicator groups
- A limited range of patch abiotic variables are strongly associated with patch biodiversity levels
- Conservation value / restoration potential can be predicted from surrogate / indicator groups – biological, landscape, or patch based abiotic
- Patch area and shape are surrogates for features of within-patch habitat diversity (topography, soils, watercourses etc) and therefore are redundant when these within-patch features are used to explain patch biodiversity
- Within-patch abiotic variables are more determinant to internal biological features of woods than landscape based factors that may relate to colonisation / isolation factors
- The combined site field based summary "conservation value / restoration potential" score will be more closely associated / able to be more accurately predicted from within site habitat indicators than from external abiotic / environmental variables

Woodland landscape conservation strategy

- Woodland habitat type e.g. "Phase 1 habitat survey" are useful for landscape woodland conservation strategy as they are reliably indicative of within-patch conservation value
- Woodland type and management can be used as indicator of broad interest level in surrogate planning
- A limited range of woodland "biodiversity indicators" can be selected that are indicative of overall site value, reliable / repeatable and thus suitable for use in planning / assessing woodland conservation
- Selected patch abiotic and landscape values (e.g. "landscape metrics" or within-patch "habitat quality" indicators) can be selected that are suitably indicative / reliable / repeatable of within-patch ecological value / restoration potential to be of use in conservation planning
- Effective woodland conservation strategies cannot be designed unless they sufficiently account for within-patch habitat quality
- Woodland habitat type and habitat quality can be used to predict overall woodland biodiversity levels, and thus be used in conservation planning
- The assessment and inclusion of relative habitat quality measures is critical to the production of effective landscape scale conservation plans

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1.4 Case study: Upland Oakwoods in the Dark Peak Natural Area

1.4.1 Introduction

The research will be undertaken using a case study of the conservation of a woodland habitat, Upland Oakwoods, at the landscape scale within the Dark Peak Natural Area, Peak District National Park. Survey work will examine ancient woodland sites, while a GIS will be constructed to analyse the landscape and woodland resource.

1.4.2 Upland Oakwoods strategic conservation

Upland oak woodlands were one of the initial habitats identified during the first tranche of the UK Biodiversity Action Plan process (The UK Biodiversity Steering Group, 1995b) (Table 1.4). In order to implement such plans the production of Local Biodiversity Action Plans (LBAP) has been encouraged (Department of the Environment, 1997). A key theme of such targeting has been prioritisation of actions within Natural Areas (The UK Biodiversity Steering Group, 1995a). These are defined as "biogeographic zones which reflect the geological foundation, the natural systems, processes and wildlife in different parts of England, and provide a framework for setting objectives for nature conservation" (The UK Biodiversity Steering Group, 1995a). Within the Peak District the creation of the LBAP was an integral part of the National Park Management Plan 2000-2005 (Anon, 2000). The strategy includes commitment to produce and implement action plans which recognise distinctive National Park assets, areas of opportunity or concern (Anon, 2000). These plans included: *biodiversity* (to protect and enhance sites and species, and create new habitats where appropriate), and *wilder areas*: to consider whether areas of the National Park should be allowed to "revert to nature" (Anon, 2000). The strategy identifies a number of areas including actions that will "halt the loss and degradation of the Park's special qualities" and "create or enhance features which add to them" (Anon, 2000). The Peak District LBAP was produced in 2002, covering three constituent Natural Areas (Peak District National Park Authority, 2002). Upland oak woodlands were identified as a priority habitat (Table 1.5). The Action Plan includes a "Vision" for the future of the habitat which states;

> "A positive future for upland oak/birch woodland lies with an expanded and inter-connected network of well managed woods spanning the many moorland fringe areas and extending into the enclosed farmlands along cloughs and valley sides to form an integral component of the upland habitat mosaic" (Peak District National Park Authority, 2002).

Table 1.4
Upland Oakwoods action plan targets (The UK Biodiversity Steering Group, 1995b)

Maintain the existing area (70,000–100,000 ha) of the upland oakwood system and improve its condition, by a mixture of management for timber (predominantly as low intensity high forest), as sheltered grazing and minimum intervention.
Expand the area of upland oakwood by about 10%, onto currently open ground, by some planting but particularly by natural regeneration, by 2005
Identify and encourage the restoration of a similar area (about 10 %) of former upland oak woodland that has been degraded by planting with conifers or invasion by rhododendron

Table 1.5
Peak District Local Biodiversity Action Plan Upland Oakwoods Action Plan (Peak District National Park Authority, 2002).

Objective	Target
Maintain extent of upland oak/birchwoods and bring all existing ancient semi-natural woodland on the Ancient Woodland Inventory (AWI) into favourable condition.	Initiate measures by 2005 to bring 300 ha (approximately 30 %) of oak/birchwoods on the AWI into favourable condition, and the remainder by 2015.
Bring priority examples of non-ancient semi-natural oak/birchwoods into favourable management.	Introduce appropriate management regimes by 2010 to bring 100 ha (approximately 10 %) of oak/birchwoods which are not on the AWI into favourable condition.
Convert Plantations on Ancient Woodland sites (PAWS) back to oak/birchwoods where this is a priority	Introduce appropriate management regimes over 80 ha (15 %) of relevant PAWS by 2005, to restore site-native species over appropriate time spans. Review and set a new target for 2005 - 2010.
Reverse woodland fragmentation by creation of new woodland, particularly by natural regeneration. Prioritize the extension/linking of existing ancient woodlands and relic clough woodland	Initiate measures by 2005 to create 200 ha of new oak/birchwood, including at least 100 ha of clough woodland in relic sites adjacent to existing ancient woodland, following current best practice. Review and set a new target for 2005 - 2010.

1.4.3 Upland Oakwoods definition and description

Upland Oakwoods are restricted by geology and climate, and typically occur in the uplands, where they are associated with poor, infertile soils. These woods have an Atlantic distribution and the UK holds a large proportion of the total European cover, leading to their international conservation importance (Rodwell and Dring, 2001). In addition to being a UK Biodiversity Action Plan priority the habitat is also an Annexe 1 habitat under the Conservation (Natural Habitat, &c.) Regulations 1994 (Statutory Instrument 1994 No. 2716) (European Community Directive 92/43/EEC on the Conservation of Natural Habitats and Wild Fauna and Flora). Within the UK Habitat Action Plan "Upland Oakwoods" are described as:

> "characterised by a predominance of oak (most commonly sessile, but locally pedunculate) and birch in the canopy, with varying amounts of holly, rowan and hazel as the main understorey species.... The range of plants found in the ground layer varies according to the underlying soil type and degree of grazing from bluebell-bramble-fern communities through grass and bracken dominated ones to heathy moss-dominated areas. Most oakwoods also contain areas of more alkaline soils, often along streams or towards the base of slopes where much richer communities occur, with ash and elm in the canopy, more hazel in the understorey and ground plants such as dog's mercury (*Mercurialis perennis*), false brome (*Brachypodium sylvaticum*), Ramsons (*Allium ursinum*), Enchanter's nightshade (*Circaea lutetiana*), and tufted hair grass (*Deschampsia caespitosa*). Elsewhere small alder stands may occur or peaty hollows covered by bog mosses *Sphagnum spp.* These elements are an important part of the upland oakwood system. The ferns, mosses and liverworts found in the most oceanic of these woods are particularly rich; many also hold very diverse lichen communities and the woods have a distinctive breeding bird assemblage, with redstarts (*Phoenicurus phoenicurus*), wood warblers (*Phylloscopus sibilatrix*), and pied flycatcher (*Ficedula hypoleuca*) being associated with them throughout much of their range... The invertebrate communities ..... support a range of notable species...."
> (The UK Biodiversity Steering Group, 1995b)

A number of distinct Upland Oakwood communities exist, dominated by *Quercus* and / or *Betula*, and have been described within the British Plant Communities series (Rodwell, 1991, Hall and Kirby, 1998) (Appendix 1.1). Each of the individual NVC communities are believed to occur within the Dark Peak, although the bryophyte rich communities have been detrimentally affected by past levels of air pollution (English Nature, 1998; Peak District National Park Authority, 1999). Within the Dark Peak Upland Oakwoods may be found in a variety of locations with differing management. These may occur as open, grazed, remnants on the moorland fringe, as enclosed plantations of natives species and as small pockets of remnant unmanaged woodland accidentally preserved within larger conifer plantations. These Dark Peak woodlands are dominated by a restricted range of trees, typically either semi-natural areas of *Betula pubescens*, *B. pendula*, and high-forest or plantations of *Quercus robur*, *Q. petraea*. Additional species such as *Fraxinus excelsior* and *Alnus glutinosa* are typically restricted to streamsides or flushlines. Associated trees are generally scarce although a range of shrubs may occur. Ground-flora in these woods is often relatively poor. Moorland fringe sites typically hold flora reminiscent of open heath or dominated by carpets of *Deschampsia flexuosa* and mosses. Richer sites hold lawns of *Holcus mollis* and may be joined by species such as *Oxalis acetosella*, *Hyacinthoides non-scripta* and *Viola riviniana*.

1.4.4 Upland Oakwood extent and distribution

The national resource of Upland Oakwoods has been estimated at between 70,000-100,000 ha (The UK Biodiversity Steering Group, 1995b). The community distribution is principally associated with upland and North-Westerly areas (Fig 1.1, 1.2). Jones considered *Quercus* dominated woodland to rarely occur over 300m elevation although examples were noted at 410m and 430m on Dartmoor (Jones, 1959). No comprehensive survey programme, prior to the current study, has examined Upland Oakwood habitats within the Peak District. However, several surveys allow an estimate of the resource to be made (Table 1.6). The Peak District Biodiversity Audit estimated that approximately 2,050-2,200ha remained within the 3 Natural Areas (Peak District National Park Authority, 1999). It was considered that a substantial proportion of this occurred within the Dark Peak (Peak District National Park Authority, 1999), where the majority of remaining semi-natural woodland was considered to be upland oakwood (English Nature, 1998). These Dark Peak woodlands support areas of W4, W10, W11 and W16 NVC communities, while small areas of W17b woodland have also been recorded (English Nature, 1998). The wider Derwent valley area holds up to 40% of the cover of ancient woodland and includes some of the most important sites e.g. Yarncliffe Wood SSSI (Padley) and Abney & Bretton Clough SSSI (Peak District National Park Authority, 1999). Other important areas include Longdendale and the areas around Glossop and Hayfield.

Table 1.6
Estimates of upland oakwood woodland cover within the 3 constituent Natural Areas of the Peak District (reproduced from (Peak District National Park Authority, 1999)

	Ancient woodland within National Park		Ancient woodland outwith National Park		Other semi-natural woodland		Total	
	Sites	Area (ha)	Sites	Area (ha)	Sites	Area (ha)	Sites	Area (ha)
Dark Peak	107	1299	14	129	118	644	239	2072
South West Peak	33	268	26	156	83	288	142	712
White Peak	3	20	-	-	2	10	5	30
Total	143	1587	40	285	203	942	386	2814

Figure 1.1
Published Local Biodiversity Action Plans incorporating an Upland Oakwood action plan
(source http://www.ukbap.org.uk/UKPlans.aspx?ID=1)

Figure 1.2
Distribution of known BAP Upland Oakwood sites (reproduced from (Rodwell and Dring, 2001).

1.4.5 Conservation interest of Upland Oakwoods

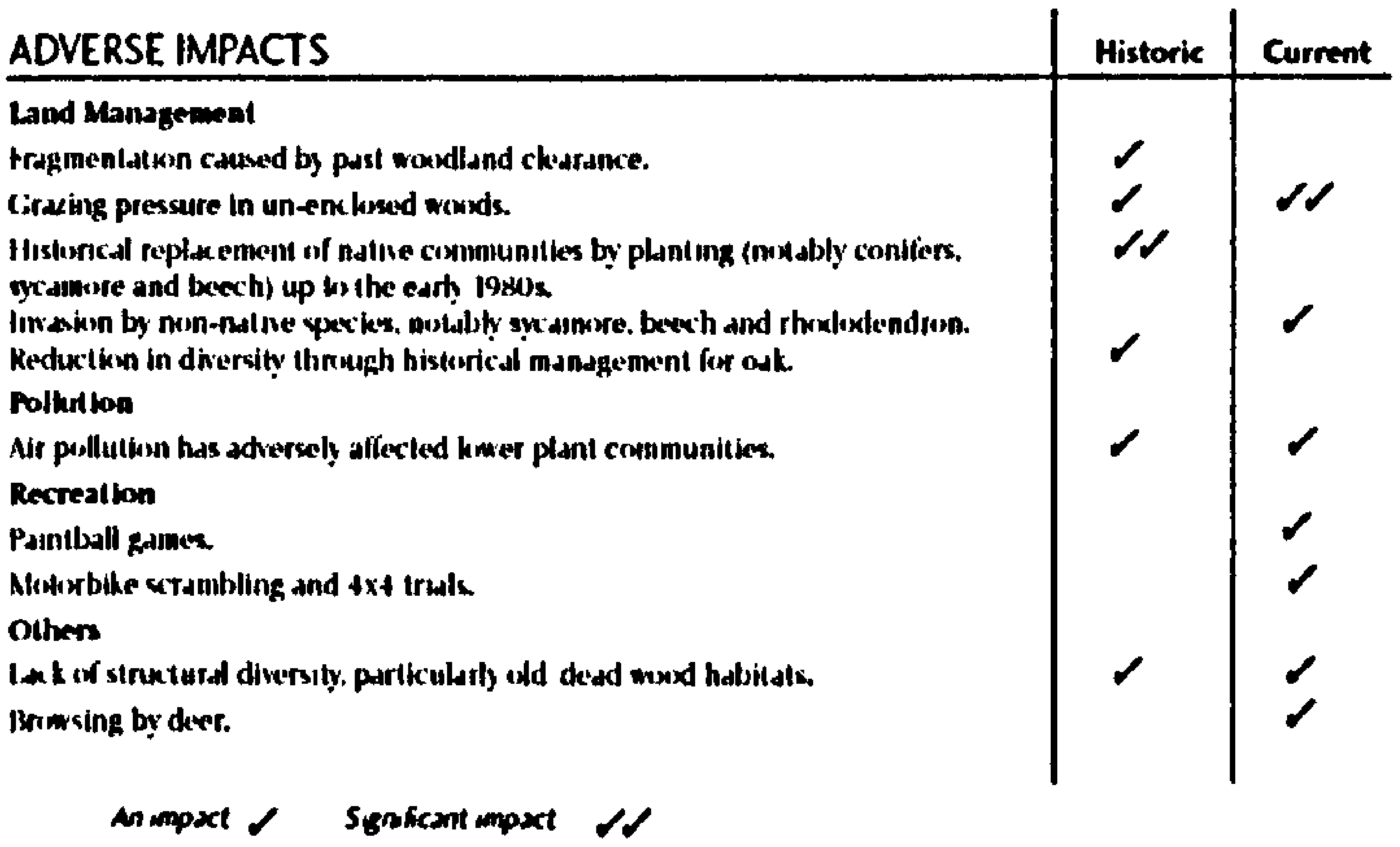
Broadleaved woodland has been identified as the habitat supporting the most dependant BAP species (Simonson and Thomas, 1999). There are 11 priority BAP species for which Oakwoods are the main or joint habitat type, while 35 priority species have been recorded within the habitat (Simonson and Thomas, 1999). Nationally the structural features of broadleaved woodland which support the most priority species are: early successional / open habitats (coppice, woodland gaps, edge and scrub habitats) (922 species), mature high forest (6 species), wet areas (5 species), bare soil / rock (5 species) and oak (5 species) (Simonson and Thomas, 1999). Within the Peak District Oakwoods support a significant number of nationally and locally notable species groups (Table 1.7). The habitat is considered to be of particular importance for its assemblages of dead-wood invertebrates, birds, lichens, bryophytes and bats (Peak District National Park Authority, 1999). Semi-natural woodland has been found to contain more listed scarce invertebrate species than any other individual habitat in the Dark Peak Natural Area (Drake et al., 1998).

Table 1.7
Species of conservation concern occurring within the Dark Peak Upland Oakwood habitat (reproduced from (Peak District National Park Authority, 1999)).

Species groups	National species	Long-list BAP and candidate RDB	Natural Area
Birds	7	25	
Mammals	4	4	
Invertebrates	72		67
Plants		1	11
Lichens			1

1.4.6 Threats and trends in extent

Nationally semi-natural upland oakwoods have declined by 30-40% over the last 60 years, caused by replanting with conifers, clearance for agriculture, or development (HMSO, 1995). The Peak District audit notes the local extent has been in historic decline, Chalmers (1974) showing losses between 1909-1974 ranging from 68% in Padley valley, 48% in Longdendale and 34% around Glossop, to a more modest 8% around Longnor (Peak District National Park Authority, 1999, Chalmers, 1974). Recent management has also led to declines in site quality, with widespread grazing of woods by sheep causing a lack of tree and shrub re-generation and impacting on the quality of the ground-flora (Fig 1.3) reflecting the declines on moorland also seen by overgrazing (Anderson and Yalden, 1981). Areas with long periods of grazing have led to declines in tree cover and caused degradation to scattered tree cover. Such declines have been well studied and several sites now exist where woodlands have been fenced to encourage regeneration (Jarvis, 1960, Piggott, 1983). However many sites on the moorland fringe remain grazed and lack regeneration under the current grazing levels. Other threats include the presence and spread of alien species such as *Rhododendron* which due to heavy shading and competition causes a decline in diversity of native ground-flora and shrub species. There is however some indication that the lichen flora, which can be an important component of these woods, is now improving due to the general increase in air quality over the second half of this century (Peak District National Park Authority, 1999).



ADVERSE IMPACTS	Historic	Current
Land Management		
Fragmentation caused by past woodland clearance.	✓	
Grazing pressure in un-enclosed woods.	✓	✓✓
Historical replacement of native communities by planting (notably conifers, sycamore and beech) up to the early 1980s.	✓✓	
Invasion by non-native species, notably sycamore, beech and rhododendron.		✓
Reduction in diversity through historical management for oak.	✓	
Pollution		
Air pollution has adversely affected lower plant communities.	✓	✓
Recreation		
Paintball games.		✓
Motorbike scrambling and 4x4 trials.		✓
Others		
Lack of structural diversity, particularly old dead wood habitats.	✓	✓
Browsing by deer.		✓

An impact ✓ *Significant impact* ✓✓

Figure 1.3
Adverse impacts on the ecological quality of upland oakwoods in the Peak District (reproduced from (Peak District National Park Authority, 2002)

1.5 Thesis structure

The thesis is presented in the following parts.

Part I Introduction

Part I forms the introduction, including the research model, aims and objectives described above, together with a description of the study area and its vegetation development.

Part II Literature review

Part II present the results of the project literature review where a number of areas relevant to the project aims are reviewed and discussed. These areas are landscape ecology and landscape planning, habitat fragmentation, woodland ecology and woodland conservation methods and spatial woodland conservation strategies.

Part III GIS: methods and analysis

Part III outlines the project methodology involved in the creation of the Dark Peak woodland GIS, the collection of woodland fieldwork data and initial GIS analysis to define the native clough woodland landscape zone and landscape areas with potential for woodland conservation.

Part IV The Dark Peak woodland resource: results and analysis

Part IV of the thesis presents the results and analysis of the woodland resource resulting from the compilation of Phase 1 habitat survey data, and the detailed ancient woodland site surveys. The association between woodland biodiversity indicators and woodland abiotic patch condition is examined using the ancient woodland site data.

Part V Conclusion: Natural Area based woodland conservation strategy

Part V discusses the implication of the analysis of woodland biodiversity and abiotic data associations, and combines this analysis with insights gained from both the phase 1 woodland survey and the landscape character assessment of Dark Peak landscape structure to produce a case study woodland conservation strategy mapping priority conservation areas for Dark Peak Upland Oakwoods.

Chapter 2
The Dark Peak Natural Area

Study area location, context and vegetation history

2.1 Location and extent

The Dark Peak Natural Area covers 860 km^2 (86,000 ha) of the southern tip of the Pennines stretching between the cities of Sheffield and Manchester and was one of 120 "Natural Areas" identified by English Nature and the Countryside Agency (Fig 2.1, 2.2) (English Nature, 2005, Countryside Agency, 2005). Natural Areas are homogeneous landscapes forming bio-geographic zones that are of use in landscape scale planning, rural planning and increasingly, in conservation delivery. The research area forms the northern half of the Peak District National Park, mainly occurring within Derbyshire, with fringing areas of South Yorkshire and Greater Manchester administrative boundaries.

Figure 2.1
Location of the Dark Peak Natural Area (25) among the 120 identified Natural Areas of England (Source (English Nature, 2005)).

2.2 Woodland Vegetation history

2.2.1 Post glacial vegetation development

Following the retreat of the last glaciation a succession of vegetation communities moved northwards with the warming climate from southern and central Europe. Species began to colonise from refugia in the central European mountains from 13,000 BP (Brewer et al., 2002), with research suggesting Oak species arrived from the Iberian peninsula (Cottrell et al., 2002). The colonisation of Britain by successive waves of woodland species was halted by its isolation from mainland Europe by the English channel approximately 7,800 BP (Simmons, 2003). Colonisation and succession led to the development of relatively dense woodland cover across many parts of Britain, except for the higher areas, before extensive modification of the tree cover by man. However there remains debate over the form such woodland communities took at the height of woodland cover, whether the cover in many areas was dense, closed canopy, high forest or comprised a more open and patchy parkland / savannah landscape maintained by large herbivores (Vera, 2000, Kirby, 2003b).

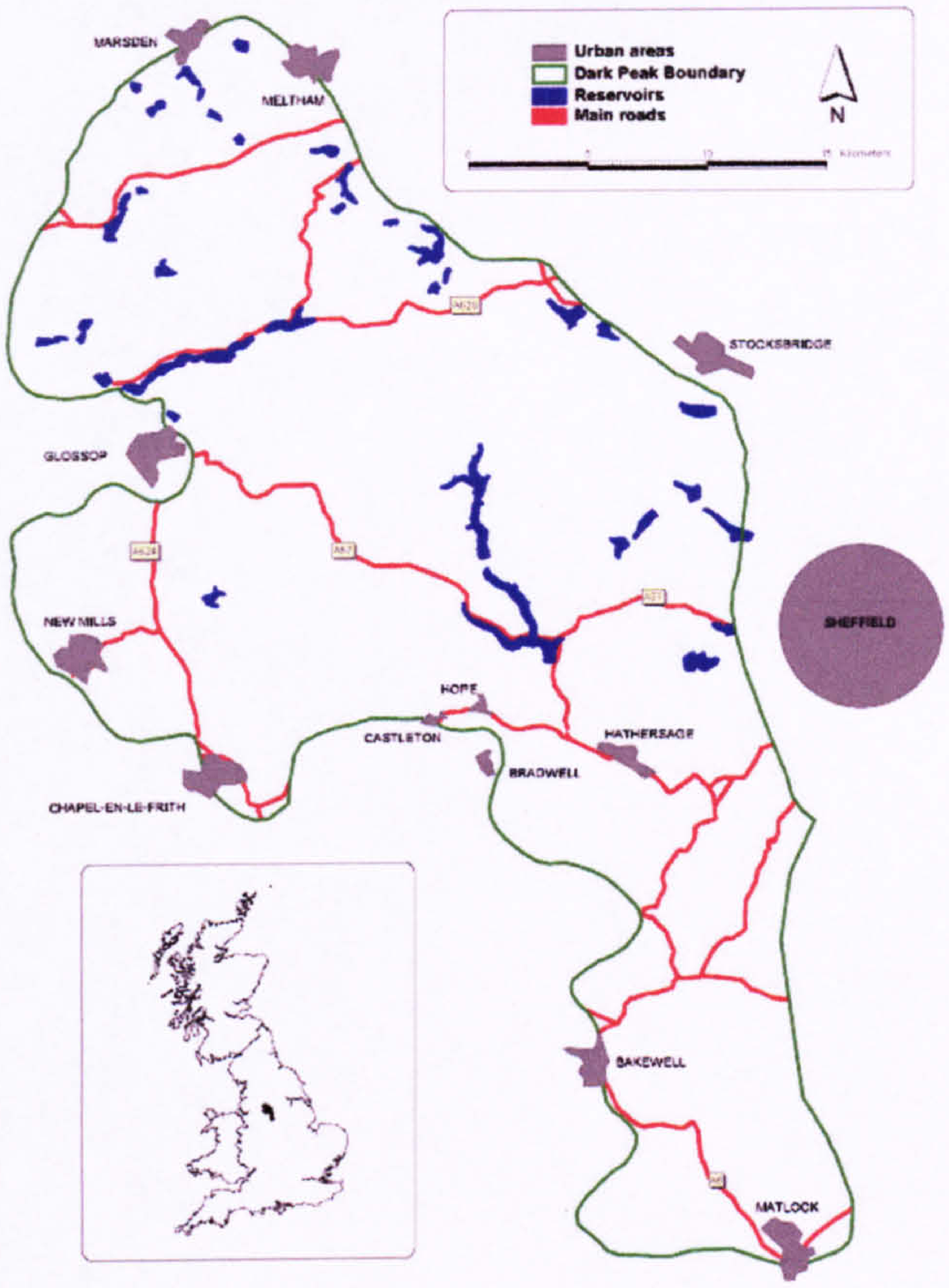


Figure 2.2
Location of the Dark Peak Natural Area

The description of vegetation history for a defined area such as the Dark Peak is problematical due to the difficulty of gaining sufficient volumes of information on past environmental conditions from sufficient sites. Methods normally rely on pollen analysis and the radiocarbon dating of preserved remains, although novel approaches have included the ecological modelling of past environmental conditions to explain woodland colonisation and succession patterns e.g. Spikins (2000). Much research has been undertaken in the Dark Peak and these represent the best data with which to interpret past conditions. Research exists for the central Dark Peak moors / Kinder / Bleaklow / Longdendale areas (Tallis, 1991, Tallis and Switsur, 1983, Tallis and Switsur, 1990, Tallis, 1964a, Tallis, 1964b, Tallis and Switsur, 1973, Anderson, 1982), the Upper Derwent valley (Charles and Craigie, 2005c, Charles and Craigie, 2005b, Charles and Craigie, 2005a) and at lower altitudes to the east and south of the Dark Peak at sites including Stoke Flats, Ringinglow bog and Leash Fen (Long et al., 1998, Hicks, 1971, Conway, 1947, Conway, 1954) and Highlow (Barnatt and Coles, 1991).

Although the Dark Peak was not directly glaciated during the last ice-age the area showed the intense effects of peri-glacial conditions 20,000–15,000 BP. An arctic / tundra vegetation occurred comprising species including *Juniperus*, *Betula nana*, *Armeria maritime*, *Lycopodium spp*, and *Thalictrum* species with grasses and initially in the absence of *Calluna* in a vegetation currently typical of Scottish mountain summits of the NVC type U10 *Carex bigelowii-*

Racomitrium lanuginosum moss-heath (Tallis, 1964a). Following gradual warming into sub-arctic and Atlantic conditions woodland gradually colonised the area with tundra and moorland remaining on the higher exposed slopes. Tallis and Switsur (1990) describe the colonisation "as birch forest was replaced in the lowlands by hazel scrub and then by coniferous forest, so birch colonized the formerly unwooded slopes and with continuing amelioration of climate, moved steadily upslope. Similarly, the hazel scrub and coniferous forest that replaced it were later displaced from the lowlands by invading deciduous forest, and formed zones of upland forest and scrub below the birch" (Tallis and Switsur, 1990). *Betula* and *Salix* scrub was developing on these hillslopes by 9,000 BP and the rapid influx and dominance of tree species pollen, mainly of *Betula* and *Corylus* was complete by 8,000 BP (Tallis and Switsur, 1990) (Fig 2.3). The succession of trees and shrubs therefore occurred over some 2,000-3,000 years before relatively stable sequences of woodland cover had developed. Following these periods of development many areas of the Dark Peak would have held covers of semi-natural Oak/Birch woodland possibly reaching a maximum, relatively stable, forest zonation by approximately 6,000 BP (Tallis and Switsur, 1990). Simmons notes that in the English uplands generally 8,000-7,000 BP was the period of maximum extent of woodland cover (Simmons, 2003). The gradual decline and replacement of this woodland cover then resulted from a complex interaction of changes in climate and the influence of human land management and ultimately farming over long periods from 8,000 BP to the present. The limits to woodland and causes of its decline in many areas were linked to peat formation. Areas of the Dark Peak were never colonised by dense woodland cover because at approximately the same time in which woodland was expanding climate changes led to the formation of peat in moorland situations, with this process occurring earlier in the South Pennines than in other upland areas of England and Wales (Tallis, 1991). This process occurred over a long period from 9,000 BP to 4,000 BP and provided limits to woodland extent by inhibiting dense tree growth where peat and blanket mire formed (Tallis, 1991). The Dark Peak has not undergone a period of total tree cover dominance, in comparison to lower and more easterly upland areas, such as the North York Moors, which were believed to have reached full woodland cover in the mid-Holocene (Innes and Simmons, 2000). More southerly and lower altitude upland areas such as Bodmin Moor were also thought to have been totally covered by dense woodland conditions by 6,500 BP (Gearery et al., 2000).

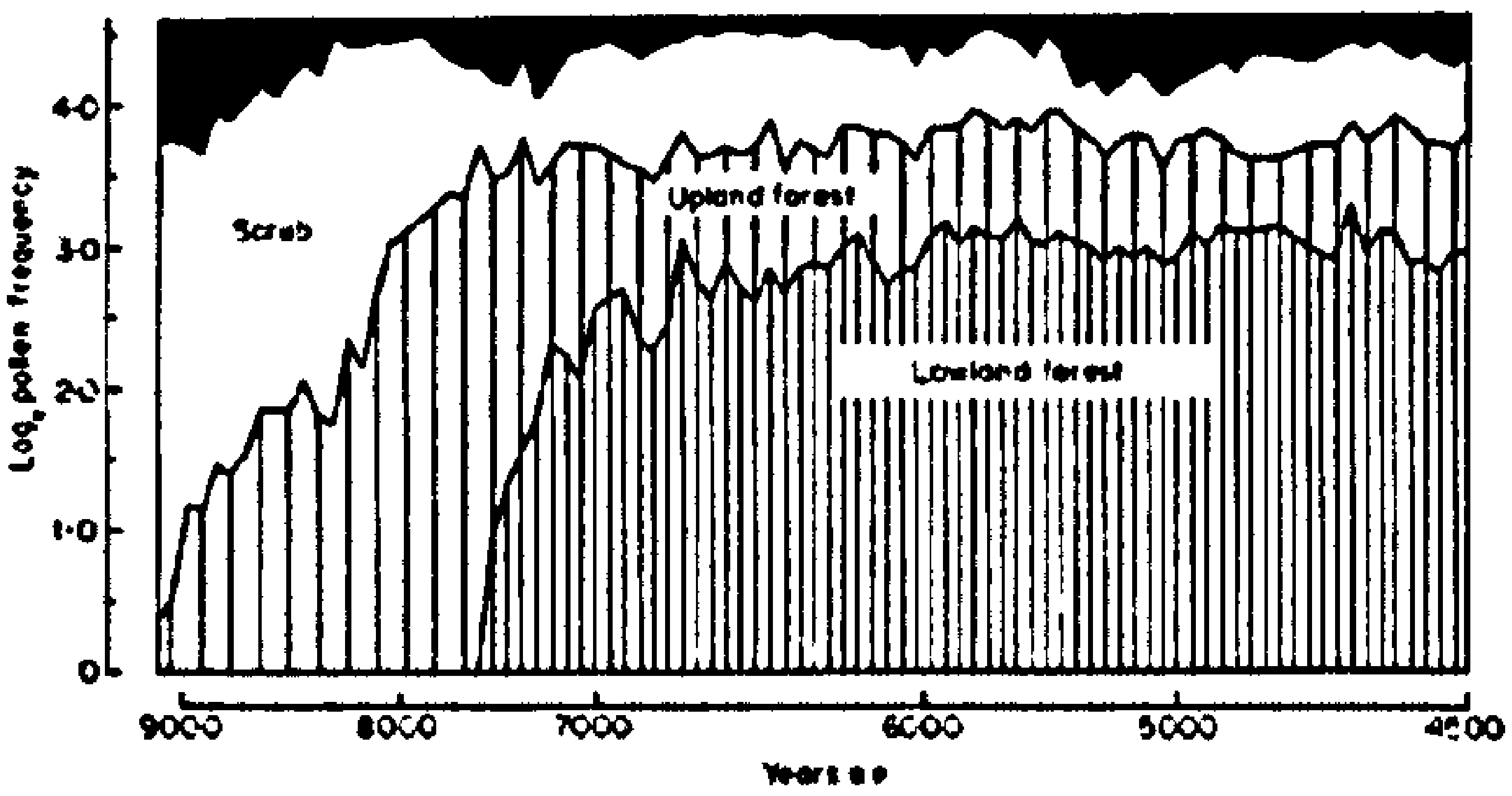


Figure 2.3
Pollen influx diagram from the Dark Peak showing the interpreted colonisation of successive communities of scrub, upland and lowland forest over time. Reproduced from (Tallis and Switsur, 1990).

2.2.2 Increasing human influence from the Mesolithic

Human influence in the Dark Peak is longstanding, with "little doubt that the moorlands were major summer hunting grounds in the Mesolithic" (Tallis and Switsur, 1990). Barnatt and Smith (2004) note that from 6,000 BP the Peak District "was one of the most important areas in Britain", referring to settlement potential due to its rich and light soils (Barnatt and Smith, 2004). While much of this settlement would have been in the adjacent limestone areas of the White Peak, such usage indicates the levels of human habitation that would have occurred in the vicinity of the Dark Peak. There existed a long period of time of several thousands years over which the initial influences by man in these areas was through the use of the landscape for hunting and associated management of vegetation or animal behaviour using fire. Researchers have hypothesised that open glades in uplands woods may have been maintained by humans in order to encourage grazing animals or to make herds easier to hunt (Simmons, 2003). While such glades may have been maintained by fire, workers have noted that in general it is highly unlikely that deciduous woods in England would have burned with "crown fires" rather than ground fires and therefore such fires were unlikely to have been devastating in the sense that fires in dry climates are (Simmons, 2003). Recent discussion of the role of human induced forest fires in the early Holocene has noted the potential for such fires to be localised, cyclical events, occurring along the edges of existing vegetation communities, rather than large-scale catastrophic events, with such effects occurring gradually and cumulatively over long periods of time (Moore, 2000).

Important events in the formation of the current Dark Peak upland landscape include the change to a wetter more oceanic climate 7,000 years ago and the "expansion of upland settlement in the Bronze age c. 4,000 years ago" (Tallis and Switsur, 1983). Therefore while man may have been affecting the extent and composition of areas of the woodland cover by 9,000 BP it was not until c. 4,000 BP that increased direct clearance of woodland for farmland would have become increasingly common. Thus at the period 4,000 years BP while large areas of the Dark Peak at higher elevations would have already been devoid of woodland, lower areas would still have held woodland cover.

Evidence for the past extent of woodland cover, and the elevations to which it occurred have been investigated by analysis of remains of tree species found buried within Dark Peak peat. Studies around Bleaklow have revealed they occurred from three main time periods 7,675-7,000 BP, 5410-4495 BP and 4,340-3,995 BP (Tallis and Switsur, 1983). These remains revealed the presence of past woodland cover and of the conditions in which peat formation was sufficiently rapid to smother tree growth, leading to their preservation. There has been a long history of the interaction of the forest and moorland edge in the Dark Peak. Initially tree cover declined due to the formation of peat when tree growth became unfavourable in the cold and wet climatic conditions on level areas at higher altitude, while in periods of presumably more favourable climate "on at least one occasion trees colonized the shallower blanket peats, particularly those near the margin, and then died and were engulfed by the accumulating peat" (Tallis and Switsur, 1983). The picture of woodland growth and peat formation is thus not simple in these areas. The expansion of tree growth back onto areas of peat was not the demise of these moorland habitats, as tree cover declined when climate or land-use activities changed. The moorland edge was dynamic.

Tallis and Switsur note that continuous forest previously occurred up to approximately 500m (Tallis and Switsur, 1990). Within this zone *Alnus* was prominent at lower altitudes and *Pinus* and *Quercus* at higher altitudes. Above 500m a discontinuous cover of *Betula* and *Salix* may have occurred (Tallis and Switsur, 1990). *Quercus* and *Pinus* were only locally recorded above 500m, while *Betula* were recorded at 595m and *Salix* at up to 610m (Tallis and Switsur, 1983). The upper limit of hillslope forests was related to climate changes and to the effects of soil degradation and man induced burning (Tallis and Switsur, 1990). Pollen records at Robinson's moss indicate 460mn was likely to represent the upper limit of "upland forests" and 425m the upper limit of "lowland forest" communities (Tallis and Switsur, 1990). The woods comprised *Alnus*, *Tilia* and perhaps *Fraxinus* in the "lowland forests" and *Pinus*, *Quercus* and *Ulmus* in the "upland forests" (Tallis and Switsur, 1990). The zonation was "stable" on the hillslopes by around 6,800 BP (Tallis and Switsur, 1990). Major changes then occurred around 5,500 BP when the forests receded downslope, associated with an increase in carbonized material in the peat which continued until after 5,000 BP, with the authors suggesting that repeated burning of the upper forest and scrub may have been responsible for the decline (Tallis and Switsur, 1990). Following this period there was a period of increased colonisation of the shallow marginal peats by *Pinus* and *Betula* between 4,300 and 4,000 BP which may be attributed to warmer climatic conditions allowing growth on the peats or a combination of climate with reduced burning (Tallis and Switsur, 1990).

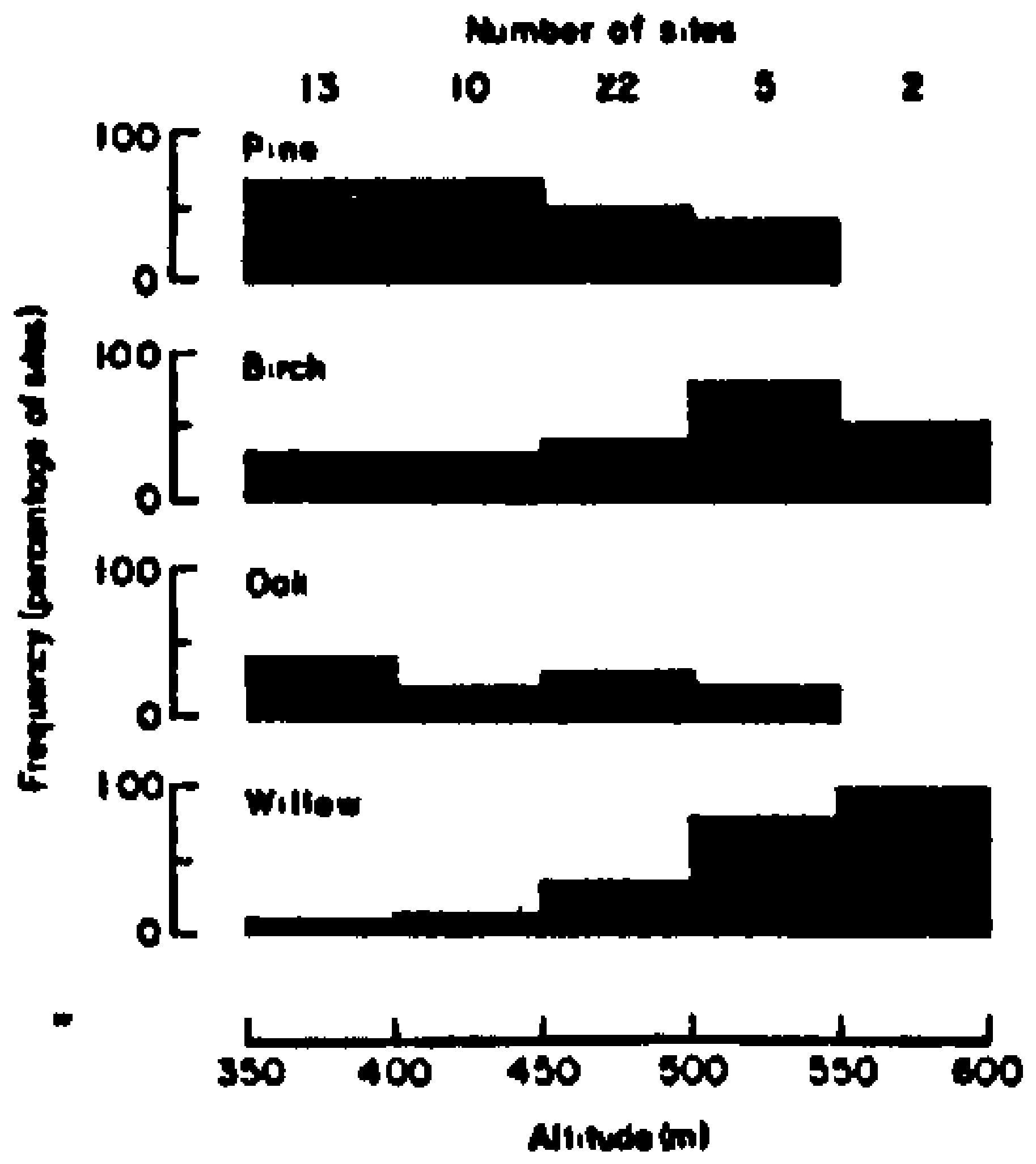


Figure 2.4
Histograms detailing the representation of fossilized tree species in successive 50m from the Dark Peak study area. Reproduced from (Tallis and Switsur, 1983).

2.2.3 Woodland decline and the Bronze Age

As human populations increased the clearance of woodland for agriculture or timber increased. It seems most likely that topography, hydrology and the occurrence of soils types would have subsequently dictated the patterns of human land-use and clearance, with areas of more productive land being preferentially cleared and farmed and the more accessible areas closer to settlements being utilised more intensively for woodland products. This would logically have led to gradual clearance from the major river valleys and floodplains and the level, more accessible areas of moorland plateau or moorland fringe which at certain periods of history have been considered favourable for attempting improvement or arable land-use. Rackham notes that during the Bronze Age "the inroads of civilisation on the wilderness ceased to be merely local. Most wildwood disappeared from high altitudes and river valleys, and inroads were made on some of the heavy soils" (Rackham, 1999). Rackham estimates that half of England may have been cleared of woodland by the early Iron Age 2,500 BP (Rackham, 1999). Woodland in the Peak District would have been actively cleared to create arable farming plots and to expand areas for grazing (Barnatt and Smith, 2004). In areas of lower altitude moorland in the Dark Peak, at Stoke Flats, there is evidence for the occurrence of peat formation after the rapid clearance of tree cover at 2,000–3,000 BP (373 Cal BC – 223 Cal. AD) (Long et al., 1998). Climatic decline in the centuries around 3,000 BP led to the decline of areas of agriculture (Barnatt and Smith, 2004). Evidence from another central / southern site at Leash Fen records tree decline in association with agricultural activity at approximately 2,300 BP (758-66 Cal. BC) (Hicks 1971). Tallis and Switsur indicate woodland clearance events around the Featherbed moss area in the Iron age / roman periods 300 BC–550 AD (Tallis and Switsur, 1973).

2.2.4 Active woodland management

Evidence for the periods between sites with pollen data and more recent map or historic document evidence are limited. Evidence at an excavation of a likely C13th lead smelting hearth in an upland area of the Dark Peak (Upper Derwent valley) revealed charcoal from a range of species, principally *Quercus* and *Betula* with *Prunus* and *Corylus* that were probably taken from mature to semi-mature trees rather than from coppice (Bevan, 1997). This indicates that in the early C13th these moorland fringe cloughs and small valleys probably retained elements of richer woodland covers compared to the reduced range of species which tend to persist in such areas today. Simmons notes that within England much "assarting" of moorland habitats would have occurred in the C12th and C13th where fields were enclosed from the open moorland along the limits of cultivation (Simmons, 2003). Simmons considers that in medieval times in England and Wales the limits of practical and useful cultivation and improvements along the upland fringes was probably between 200m and 350m (Simmons, 2003) implying that few areas above these points would have been actively cleared for agriculture at that time. Within England as a whole woodland cover had reduced from 15% of England in 1086 to perhaps 10% in 1350

(Rackham, 1999). Interpreted pollen diagrams around Kinder and Bleaklow show evidence of heavy woodland decline and clearance probably due to increased sheep keeping in the C12th (Conway, 1954). A combination of environmental and social changes in the C14th led to decreased arable agriculture in the Peak District and increased livestock rearing (Barnatt and Smith, 2004). Pollen diagrams from the Wessenden and Kinder moors indicate increased use of the moors and associated tree clearance / decline in the C10th, although such exploitation did not occur on an extensive scale until the C16th (Tallis, 1964b). In the Dark Peak removal of native tree cover occurred into historic times with conversion of woodland to agriculture up to the times of the enclosure awards. Examples include the decline of tree cover in the East Moor area from 15,592 ha in 1086 to 999 ha in the late 1700's (Eyre in Chalmers, 1974). Rackham notes that large areas of wood pasture were recorded in Derbyshire in the Doomsday book but that by 1300 such wood-pasture commons had been greatly reduced in size and importance (Rackham, 1999). Ratcliffe considers that most of the forest clearance had already occurred in England, Wales and southern Scotland by 1700 AD (Ratcliffe, 1984). Pollen records at Ringinglow bog near Hathersage indicated that hillslope forests around the site began being destroyed more intensively around 1100 AD and had mostly been cleared by C17th (Conway, 1947).

While substantial woodland clearance would have occurred by these periods these times would also have seen the management of the remaining areas of woodland for a wide variety of forestry and timber products. In more productive or lowland fringe areas such uses would have ranged from the production of coppice, standards and pollards while in more upland areas open grazed pasture woodland may have simply been utilised for the collection of firewood or possible use as feed for livestock (Bevan, 1997, Scurfield, 1999). In these times the management of woodland would largely have consisted of native timber species such as oak, ash and hazel, although the frequency and occurrence of these species within woodland sites would have been substantially altered by patterns of felling and woodland management. Although native species were frequently used, increasingly planting of stock was also undertaken and these may have originated from other provenance areas. Species such as Sycamore (*Acer pseudoplatanus*) although with a relatively long history of introduction to the UK begin to show selected records of woodland planting use in England from the later 17th Century onwards (Rackham, 1999). The increased use of silvicultural methods such as the raising of tree sapling in nurseries and later transplanting into the forest would also have increased from these times (Rackham, 1999). Harris et al (2003) credit the increased use of active forestry methods: site preparation and the use of transplanted seedlings, to the publication of the book "Sylva" in 1664 by John Evelyn (Harris et al 2003). Scurfield (1999) considers that in the C17th the woodlands of the Upper Derwent and Hope Woodlands valleys in the Dark Peak would have been managed to provide coppice and timber and would also have supplied tanbark, animal fodder, grazing and shelter (Scurfield, 1999). Many of the enclosed fields in the Upper

Derwent areas that were not mapped as woodland in mid C17th maps were never the less described as holding high covers of woodland or scrub (Scurfield, 1999). Bevan interprets C17th maps of the Upper Derwent area as holding significant woodland cover, but considers that even higher covers would have existed in the C13th century but had been lost to piecemeal clearance and enclosure by the C17th (Bevan, 2004). Scurfield (1999) interprets the frequent occurrence of the enclosure name "cow hey" for wooded enclosed areas as indicating past usage as areas where cattle may have been kept and supplied with cut foliage and tree branches to provide fodder. Increasingly the woodlands in the Dark Peak would have become important not just for supplying local economies and contributing to farm management but for supplying the needs of the adjacent industrial areas from the C18th (Bevan, 2004). While in southern England there are incidences of the early use of conifers planted in mixtures with native oaks from 1761 (Harris et al 2003) the use of additional non-native timber species in the Dark Peak probably only increased from the late C18th onwards by which time conifers were readily available from nurseries (Simmons, 2003). Early evidence of the planting of conifers in the area are noted by Bevan as the planting of *Larix* species in 1818 and 1861-62 (Bevan, 2004). The maps by Moss (1913) indicate that some of the Ancient woodland sites in the Dark Peak had already been converted to conifer plantation by the late C19th. As the C19th progressed the rate of woodland decline in the area slowed as areas of plantation forestry increased, estates such as Chatsworth undertook planting on a large scale.

The composition of the newer intensively managed woods differed greatly from the woodland that had long been cleared from the Dark Peak. In the periods when native timber such as oak and hazel were required for adjacent industrial areas of Sheffield, Manchester and Derby these oak woods would have been protected from grazing and regularly managed. As the industrial revolution progressed the use of traditional timber products (oak bark for the tanning industry and hazel stakes / poles) declined (Rackham, 1999). With this decline the composition of the Dark Peak woods changed even further. In discussing woodland management in the Peak District Essex noted that "agricultural reports of the late 18th and early 19th centuries recognized the poor standard of woodland management in the area" (Essex, 1990). Evidence from one Dark Peak Oakwood notes the last date of coppicing as 1870 (Yarncliffe wood, Padley) (Piggott, 1983). Following such decline of traditional woodland products, areas would have been converted to high forest plantation, possibly of more productive conifer species while substantial areas of woodland would have passed into periods of decline being left unmanaged or opened up to provide grazing and shelter for livestock. In some areas the conversion of coppice woods to high forest plantations was an intensive undertaking. Many of Derbyshire's woods were converted to high forest in the late Victorian periods the old coppice stools may have been removed or "grubbed up" using machinery such that these sources of past woodland management may often be absent from converted Ancient Woodland sites (Jones and Rotherham, 2000).

2.2.5 Recent woodland management

The use and management of woodland was intensified again following the two world wars and the creation of the Forestry Commission with its remit of increasing the supply of UK timber and reducing dependence on imports. The main intensive loss of woodland and conversion of such sites then occurred after 1945, with Rackham noting that the rate of loss of ancient woodland in the 1950's–1960's was probably unparalleled, as management intensified or clearances were made for agriculture (Rackham, 1999). With resurgence in woodland development, the focus shifted to the creation of large conifer plantations, frequently in the uplands. The scale of such plantations increased into the 1970's and 80's with such activities being criticised at the time due to the inappropriate use of such landscapes driven simply by the availability of large areas of cheap land, viewed by some as unproductive and barren and thus awaiting "development" (Tompkins, 1989). Such past large scale and rapid afforestation caused problems of access, landscape change and damage to wildlife areas in northern England and Scotland (Tompkins, 1989).

Table 2.1
British Woodland cover (Peterken, 1996).

Year	Total woodland area (% land area)	Primary or ancient semi-natural woodland (% land area)
3,000 BC	85	85
1086 AD	15	5-10
1895	4	2
1992	11	1.5

In summarising Dark Peak woodland history, a diverse and zoned cover of woodland communities had developed by approximately 6,000 BP, following which a long period of direct and indirect activities of man led to the almost total loss of woodland cover by recent times. The majority of the Dark Peak has therefore been deforested for very considerable periods of time, while in some areas the loss of extensive areas of native woodland cover and of ancient woodland sites has occurred significantly within the last 300 years with rates of clearance or conversion remaining high into the 1950-70's. Areas of woodland occurring today may still hold similarities to previous woodland cover where conditions have prevented human management or intervention but the majority of sites have undergone periods of direct management and even sites lacking evidence of direct management show the influence of man's activities and environmental change through effects on soil conditions and raised pollution levels. Rackham considers that in England in general the wildwoods "passed away in prehistory and have left neither written record nor legend" (Rackham, 1999) and considers that all woods were managed for centuries except for those on inaccessible areas like cliffs or those formed naturally, but of recent origin (Rackham, 1999). Evidence of such long periods of management are confirmed by the generally impoverished tree and shrub species diversity of modern semi-natural woodland compared to the species thought to exist in the mid Holocene, while many woods now also contain significant covers of non-native species.

It is apparent from the consideration of pollen profiles and descriptions of woodland communities that Peak District woodlands have undergone considerable compositional changes from pre-historic to historic times. While current woodlands are often species poor and impoverished it is of interest to know when this impoverishment may have occurred, and what species may until recently have occurred in these woods. The presence of some species may be difficult to gauge from pollen cores due to low pollen production e.g. *Crataegus*. All the local pollen cores examined have constant presence of *Quercus, Alnus* and *Betula* from colonisation to present day. The representation of additional native species e.g. *Fraxinus* is more sporadic, with this tree having a longer history of restriction to locally more favourable sites, such as valley bottoms and cloughs than the other tree genera. Additional genera including *Tilia, Pinus, Fagus* and *Carpinus* have a long history of presence and decline. While the past presence of *Pinus* and *Tilia* in local woodland is accepted, many pollen diagrams indicate sporadic occurrence into local times, although problems with interpretation of pollen diagrams and the effects of long-distance pollen transfer make it unclear whether locally strong populations occurred into recent centuries or if presence in pollen cores represents transport from more distant populations. Ultimately the species composition and character of the Dark Peak woods changed as their extent declined, but these woods have often held a range of species beyond that seen in many impoverished woodland sits today.

2.3 Landscape, landform and geology

The Dark Peak landscape is "upland" in character containing a large central core of rolling moorland plateau with peaks rising to over 630m. The area forms one of the largest semi-natural wilderness areas in England (Countryside Agency, 2005). The geology is dominated by Upper Carboniferous shales and sandstones of the Millstone Grit Series, laid down around 330-320 million years ago (English Nature, 2005). The Peak District was not glaciated in the most recent glaciation (Devensian) but does show much evidence of the effects of periglacial conditions such as the formation of tors and slope deposits (English Nature, 2005). The geology has led to the formation of typical moorland landscape with extensive upland land-use occurring over infertile areas with peat on the higher ground. Around the moorland fringe transitions to more enclosed lowlands landscapes occur, characterised by dry stone walls, boundary trees and farmsteads. Typical landform features in the area are the gritstone edges where formations of ridges of exposed rock occur at the edges of the moorland plateau and the many moorland and moorland fringe narrow steep valleys known locally as "cloughs" (Fig 2.5). The southern fringes of the natural area are more diverse topographically and contain a range of wider valley, rolling farmland and floodplain features. The Dark Peak holds a dramatic landscape character which is enhanced by long, uninterrupted views on the moorland plateaus, the low levels of habitation and the wild character of the vegetation cover (Countryside Agency, 2005). Additional features are the reservoirs in the larger valley catchments, which are typically associated with large coniferous woodland plantations (Countryside Agency, 2005).

Figure 2.5
Moorland habitats and cloughs © PDNPA

2.4 Climate, land-use, agriculture and soils

Landform, geology and climate define land-use and settlement in the Dark Peak. The upland climate, with a lower period of growth, colder conditions and increased rainfall in comparison to more lowland areas combine with the geology to produce a range of infertile, leached soils. The majority of the soils are classed as very poor (5) or poor (4) on a five point scale of agricultural land capability (Anon, 1988). On the higher moorland plateau thick layers of peat occur (English Nature, 2005). The Peak District forms the most south-easterly area of high rainfall in England (Fig 2.6) although total rainfall levels are not as high as the more westerly upland areas of Dartmoor, Exmoor and areas of Wales. Using data interpolated from Met Office recording stations wide variations exist within the Dark Peak, dependent on topography, with the upland moors holding damper, cooler conditions compared to the fringes of the natural area (Table 2.3).

Table 2.3
Precipitation values for the Dark Peak Natural Area, interpolated from Met Office station readings. (Produced from data supplied from the Met Office, available at http://www.metoffice.gov.uk/climate/uk/averages/index.html).

	Dark Peak		Great Britain	
Precipitation	Min	Max	Min	Max
Total annual	860	1607	507	4134
Days > 1mm	135	190	101	266
Days > 10mm	23	53	9	115

Table 2.2

Dark Peak woodland history, compiled from (Tallis and Switsur, 1983, Tallis and Switsur, 1990, Tallis, 1991, Long et al., 1998, Rackham, 1999, Barnatt and Smith, 2004, Hicks, 1971)

Time period	Environmental conditions and human activity, including woodland management	Woodland composition, location and altitude
0 – 10 BP (1990 – Present)	Temperate/warming climate. Peat erosion occurring. Pollution levels declining. Plantation forestry management continues, many conifer plantations are entering their second or third rotations. Value of softwood conifer timber continues to fall. Multi-purpose forestry now the acknowledged standard.	Increased occurrence of native woodland restoration schemes. Locally reversion of conifer plantation to native moorland habitats are favoured, although situations in which conifer woods have developed high conservation value in their own right exist where conversion "back" to moorland cover is not considered desirable.
60 – 25 BP 1945-1980	Increased recreation pressure and productivity-led land-use encourage improvements of large tracts of landscape, with associated high stocking levels and consequent high grazing pressures. Increased conversion of native woods to plantation cover, with productive conifers favoured. Increased planting of new conifer woods on the upland fringes and plateau.	Creation of plantations causes the average elevation of woodland to increase for the first time in many centuries. Woodland open to grazing become excessively grazed.
200 - 100 BP (1800 - 1900AD)	Industrial revolution in adjacent urban areas. Traditional forestry management declines. Plantation forestry increases, often conifers. Native timber species plantations decline. Active coppicing management has ceased by end of period	Reduction in the cover of native woodland species due to conversions to conifers. Large reduction in the areas of young growth woodland as coppice management ceases and are converted to high forest growth.
300 - 200 BP (1700 - 1800AD)	Increased use of timber for charcoal, tannings industry, and other products locally, although the use of traditional woodland products begins to decline by end of century. Woodland management begins to intensify locally with first creation of more productive plantations of non-natives.	Most native woodland now cleared from all plateau areas and now increasingly restricted to cloughs and valleysides. All woodland well below 500m, all lowland level productive areas now cleared.
400 - 300 BP (1600 - 1700AD)	Climate colder than today. Increased lead and copper industry. Increased use of charcoal. Intensification of woodland management for charcoal production. Peat formation still occurring. Increased enclosure of moorland edge habitats including woodland.	Evidence that significant woodland cover still occurred within upland clough valleys and slopes e.g. Upper Derwent but was gradually reduced by enclosure
900 - 700 BP (1100 - 1300AD)	Warmer climate. Increased clearance of woodland and permanent agriculture, e.g. enclosure of Peak Forest. Increased assarting and enclosure of the moorland edges.	Rate of decline of hillslope forest increases again, although some evidence that woodlands still occurs in upland cloughs and valleys
1500 - 1000 BP (1000 – 500 AD)	Norman / Anglo-Saxon Farming stable or declines. The Royal Forests are enclosed.	Continued gradual woodland clearance.
1600 – 1950 BP (50AD-400 AD)	Romano British First towns begin. Mixed farming becomes common with arable farming now increasingly important	Increased woodland clearance around the Featherbed moss area, Kinder.
2000 – 2500 BP (500 BC – 0 AD)	Iron age Colder and wetter than current climate. Evidence of arable farming on some moorland areas. Peat formation still occurring.	Increased active woodland clearance. Evidence for rapid woodland decline at areas of lower elevation moorland (c.300-400m) Leash Fen and Stoke Flats.
3500 – 4000 BP (1500 - 2000 BC)	Bronze age Warmer than current climate Woodland clearance No peat formation	Woodland clearance is piecemeal and temporary in many areas but overall levels continue gradual decline
4,000 – 5,000 BP	Bronze age Peat accumulation has been rapid and maximum, current extent of peat areas reached by 4,000 BP. Increased direct clearance of woodland for use of land as arable or to increase grazing areas.	Much land below 350-400m may have still held woodland cover. Warm period 4300 – 4000 BP leads to leads to expansion of growth of *Pinus* and *Betula* onto areas of thinner dry peat, subsequently preserved in peat as tree remains. Period of Ulmus decline approx 4,800 BP.
5,000 - 5500B.P	Late Stone age Colder and wetter climate. Farming activity increases. Possible evidence for increased burning of the upper limit of wooded treeline / moorland fringe. Period of rapid active peat development across much of the area, now extending to lower altitude moorland areas with much blanket peat development on flat ground above 450m	Woodland limits recede downslope. Wood clearance activity and woodland now increasingly restricted by peat formation even on some of the lower moorland plateau areas, restricting the upper limits of woodland cover on areas of flatter ground. Possibly still extending up to 500m in places.
5,500 – 6,000 BP	Peak District, including adjacent White Peak becomes important settlement area	Relatively stable zonation of upland and lowland forests, each with their own distinct woodland communities. .Maximum upper limit of woodland slope forests reached, approx 500m, then forests begin to recede downslope.
7,000 BP	Change to wetter oceanic climate. Period of active peat development	*Pinus* pollen declined sharply, partly ousted by *Quercus*
8,000 – 9,000BP	Peat beginning to form on areas of higher ground. Human populations at very low levels. Evidence for linking periods of potential anthropogenic burning on the moorlands to increased levels of peat formation.	Dark Peak largely dominated by scrub and "upland forest" tree and woodland cover except for higher plateau ground. Some initial evidence for use of the areas by man, including possible woodland burning / clearance.
9,000 - 10,00 BP	Mesolithic Ice sheets melt away. Tundra habitats develop. Peat formation begins in localised areas.	Woodland begins to colonise and develop on lowland areas and slopes.

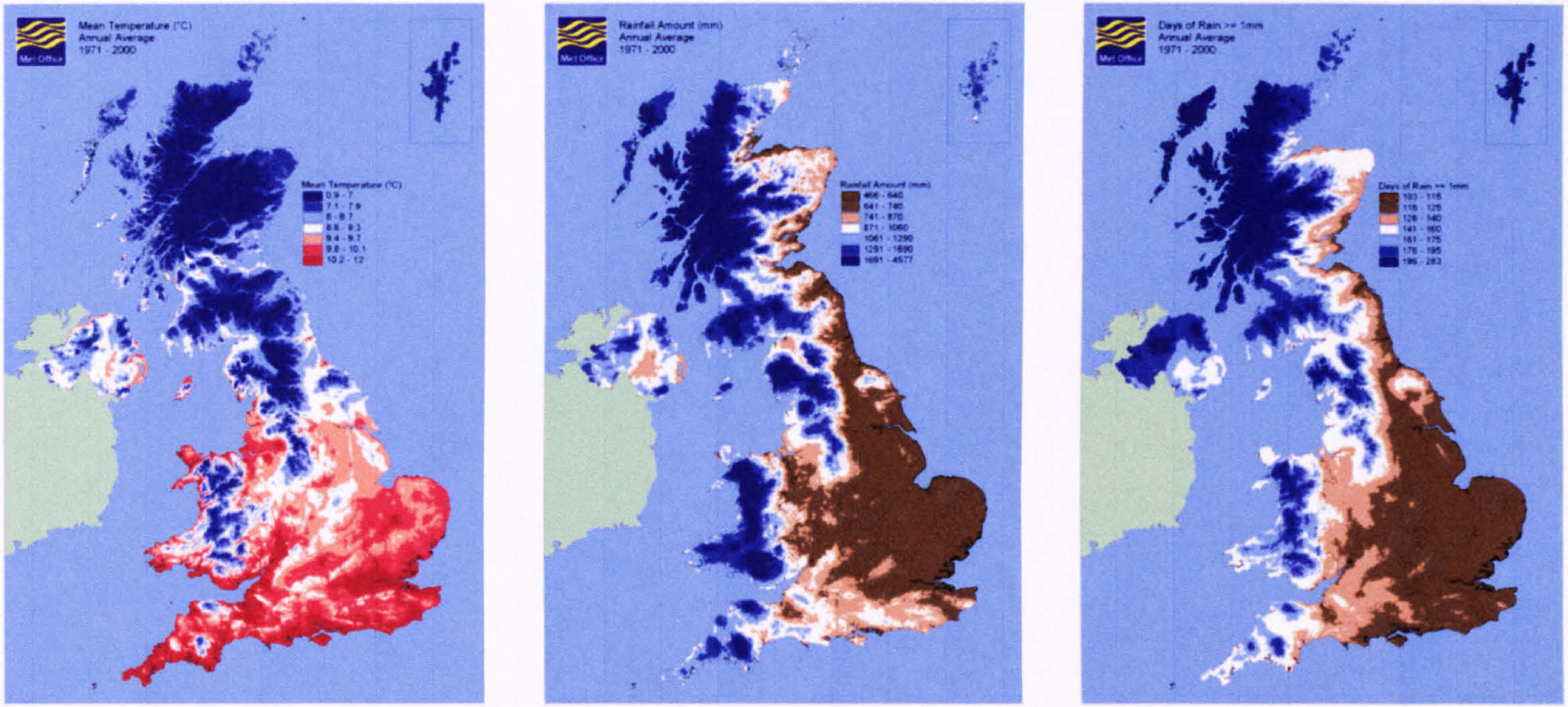


Figure 2.6
Summary climate data for the period 1971 – 2000. (Reproduced from http://www.met-office.gov.uk/climate/uk/averages/19712000/mapped.html).

The highest areas of land within the Dark Peak are unsettled and are farmed in an extensive manner, with land-uses on the moorland plateau being sheep rearing and grouse moor management, which has occurred there since the early C19th (Countryside Agency, 2005). In the enclosed lowland fringe areas cattle and areas of hay meadows and silage fields occur. Although current intensive land management is mainly restricted to the enclosed "in-bye" land along the moorland fringes and within the main river valleys there is much evidence for use of the moorland areas in a more intensive manner in historic times, with varying levels of intensity of use reflecting differing periods of climate or land-use trends as well as being driven by the changing economy of agricultural products. A particular feature of this changing management has been the dynamic nature of the moorland fringe areas, which have continuously undergone periods of expansion and regression into the moorland core (Countryside Agency, 2005)

2.5 Landcover, Fauna and Flora

The Dark Peak holds fauna and flora typical of the English uplands with similarities to Dartmoor and the Scottish borders, but moderated by the strong influence of past levels of management, the effects of the proximity of major industrial centres, and high levels of recreation use. The area supports a wide range of habitats ranging from upland blanket mire to floristically rich grasslands and hay meadows. Large areas of vegetation on the moorlands are dominated by heather (*Calluna vulgaris*), with additional dwarf shrubs including bilberry (*Vaccinum myrtillus*) and crowberry (*Empetrum nigrum*). High moorland areas may be dominated by cottongrass (*Eriophorum vaginatum*) over peat. Fringing grasslands hold rough or semi-improved grassland, species including matt grass (*Nardus stricta*), Fescues (*Festuca ovina, F. rubra*) and common bent (*Agrostis capillaris*). Woodland areas comprise frequent conifer plantations of Scots pine (*Pinus sylvestris* and Spruces (*Picea sp.*) while broadleaved plantations are frequently dominated by beech (*Fagus sylvatica*) or mixtures of beech and sycamore (*Acer pseudoplatanus*). Remaining areas of native woodland comprise areas of Oak (*Quercus petraea*) and Birches (*Betula pendula* and *B. pubescens*). The fauna comprises a variety of upland and upland fringe bird species, typical upland invertebrates and only small and localised deer populations in the main moorland areas. No compiled statistics exist for the landcover of the Dark Peak. Indications of habitat coverage can be gained from levels within the National Park and North Peak Environmentally Sensitive Area (ESA) areas (**Tables 2.4 and 2.5**) and from Phase 1 habitat data (from the 1980's) compiled by English Nature for an area that approximately follows the boundaries of the National Park (**Table 2.4**) (Ecological Advisory Service, 1993). ESA statistics were derived from analysis of the North Peak ESA landcover survey (ADAS, 1997).

Table 2.4
Compiled Phase 1 habitat for the Severely Disadvantaged area of the Peak District (approximately coincident with the National Park area. Reproduced from Ecological Advisory Service (1993).

Habitat	ha	%
Semi-natural broadleaved woodland	2,577	2
Planted broadleaved woodland	1,937	1
Coniferous and mixed woodland	5,908	4
Dense scrub	551	0
Unimproved acid grassland	19,096	12
Unimproved neutral grassland	5,493	3
Unimproved calcareous grassland	2,307	1
Marshy grassland	3,227	2
Semi-improved, improved grass, arable	68,488	42
Continuous bracken	3,880	2
Dry heath and mosaics	16,679	10
Wet heath and mosaics	967	1
Bog	21,828	13
Mire	14	0
Quarries, spoil etc	1,382	1
Urban and amenity	5,312	3
No access / not surveyed / other	4,267	3
Total	163,913	100

Table 2.5
Habitat areas occurring within the North Peak ESA area (amended from (ADAS, 1997)).

Land cover class	ha	%
Heather moor	13,614	25
Bilberry/crowberry moorland	8,233	15
Cotton-grass moorland	5,934	11
Bracken	2,664	5
Eroding moorland	3,494	6
Bare ground	126	0.2
Grass moor	9,021	16
Rough pasture	1,359	2
Permanent grassland	5,018	9
Cultivated land	500	0.9
Woodland and Scrub	3,087	6
Other	1,835	3
Total	54,885	100

2.6 Woodland cover and distribution

The Dark Peak currently holds varying levels of woodland cover across areas of differing landscape character. Figures from the ESA and Peak District National Park indicate covers of between 6% and 7% woodland (Ecological Advisory Service, 1993, ADAS, 1997), while Good et al report 10% total woodland cover, and 7% deciduous in the National Park (Good et al., 1997).

The broad distribution of woodland is influenced by geology and landform. The moorland plateau holds peat cover which is mainly free of tree cover. The lowland fringes and valleys hold alluvium with richer soils and are typically dominated by agricultural leys, hay-meadows or pasture. Most woodland cover therefore occurs on sloping valley sides and "cloughs" where traditionally areas of broadleaved woodland have occurred in these infertile and topographically variable sites, while more recently conifer plantations have arisen. Of the woodland types semi-natural tree cover and scrub are most frequently associated with very steep slopes or large block boulder scree formations.

Perhaps the most distinctive woods are the large areas of coniferous plantation around the Upper Derwent and Longdendale reservoirs, along the A57 snake pass road and around the Matlock moors. There are also significant, although less visible areas of semi-natural woodland occurring in cloughs and along the gritstone edges and ridges. The Dark Peak lies at the phyto-geographical boundary where semi-natural "lowland mixed deciduous woodland" and "upland oak woodland" are likely to be found. Many of the remaining semi-natural woodlands have been heavily modified by both past management and levels of industrial pollution such that they are degraded and classification between these two classes may be difficult. However much of the semi-natural woodland at higher altitudes and on poorer soils tends towards the upland Oakwoods habitat, while areas on the lower upland fringes of the Dark Peak or on areas of richer soils may tend to "Lowland mixed deciduous woodland". Conditions in the higher altitude woods may be markedly different to those occurring in sheltered lowland situations, with Simmons noting a 13 day reduction in growing season for every 100m increase in altitude in the English uplands (Simmons, 2003). The areas of semi-natural upland oak/birch woodlands occupy a range of conditions and are widely scattered around the moorland fringes, differing greatly in their quality and composition. Sites range from highly natural, although secondary and relatively recent, birch dominated woodland to ancient woodland sites dominated by oak, but often with a long history of forest management and therefore complemented by a range of introduced tree species. The current growing conditions in these clough woods on steep, thin soils are relatively harsh with a study in Yarncliffe wood, Padley noting that oaks although only 15-40cm in diameter were 100yrs old (Piggott, 1983). The remaining areas of broadleaved woodland in the Dark Peak occur as areas of broadleaved plantation, often comprising beech or sycamore and occurring around farmsteads or within the managed and enclosed valleysides.

Figure 2.7
Semi-natural clough woodland and moorland edge woodland sites. Photos by the author and © PDNPA

2.6 A protected landscape

The majority of the Dark Peak falls within the Peak District National Park (Fig 2.8) which was designated in 1951 and holds a purpose, defined in the Environment Act 1995, to "conserve and enhance natural beauty, wildlife and cultural heritage" and "promote opportunities for the understanding and enjoyment of the special qualities (of the Parks) by the public". Large areas are also classified as Sites of Special Scientific Interest (SSSI) and Special Areas of Conservation (SAC), affording protection under national and European legislation respectively. The Dark Peak / Eastern Moors SSSI and Southern Pennine Moors SAC designations reflect the national and international importance of the upland heathland and blanket bog vegetation communities and the characteristic flora and fauna these habitats support. These areas were also designated for the occurrence of Upland Oakwood habitats, although these form only a minority of area. The area holds an important location at the northern and southern limit of the extent of a variety of species providing additional conservation interest. The area supports nationally important populations of bird species including golden plover, dunlin, merlin and short-eared owl (English Nature, 2005). A significant area of the Dark Peak is also covered by the North Peak Environmentally Sensitive Area (ESA) designation which exists in order to offer financial incentives to promote agricultural management to benefit areas of higher wildlife and landscape value by encouraging environmentally and wildlife friendly farming.

Figure 2.8
Designations in the Dark Peak

PART II: Literature Review

The following key topics were selected from initial research as being particularly relevant to the development of a Natural Area based woodland conservation strategy. Each chapter undertakes a review of relevant literature with a view to assessing the extent of current knowledge, and where methods, theories and axioms can be incorporated within the current study. The review also highlights areas of ongoing research and where there are gaps in current knowledge. Each chapter concludes with a summary of key issues, including factors relevant to the project analysis and suitable for incorporation within study methods, or where the literature suggested useful future research.

The literature review chapters are:

- **Chapter 3: Landscape ecology and landscape planning**
  A literature review
- **Chapter 4: Habitat fragmentation: landscape structure, composition and ecological quality**
  A review of studies with an emphasis on woodland habitats
- **Chapter 5: Ancient woodland sites and woodland conservation ecology**
  A literature review of ancient woodland, woodland ecology and woodland conservation
- **Chapter 6: Wooded landscape conservation strategies**
  A literature review

Chapter 3

Landscape ecology and landscape planning

A literature review

3.1 Introduction

This chapter describes a range of theories within landscape ecology applied to conservation issues and discusses the various merits of schemes used to prioritise conservation activity at the landscape scale with emphasis on "landscape planning", where conservation activity is co-ordinated across regions and landscapes.

3.2 Landscape Ecology

3.2.1 Landscape Ecology: an introduction

Landscape ecology is a discipline combing ecological study with land-use planning, developing with a broader landscape-planning base in Europe and more closely tied to traditional ecological science in America (**Fig 3.1**) (Wiens, 1997). Landscape ecology and has been defined as the study of the effects of pattern on process (Turner, 1989) and examines how landscape structure affects the occurrence and abundance of organisms. Landscape itself holds numerous definitions, often reflecting the orientation of the study being undertaken. These include: "the configuration of topography, vegetation cover, land-use and settlement pattern which delimits some coherence of natural and cultural processes and activities" (Green at al 1996 in Farina, 1998) and "a heterogeneous land area composed of a cluster of interacting ecosystems that is repeated in similar form throughout" (Forman and Gordon, 1986). In many definitions the cultural or anthropogenic portion of the landscape is recognised. This is critical when addressing change as many driving forces are the result of anthropogenic activities. In modern landscapes it is not appropriate to address the conservation of species or habitats without reference to the cultural and managed landscapes in which they occur.

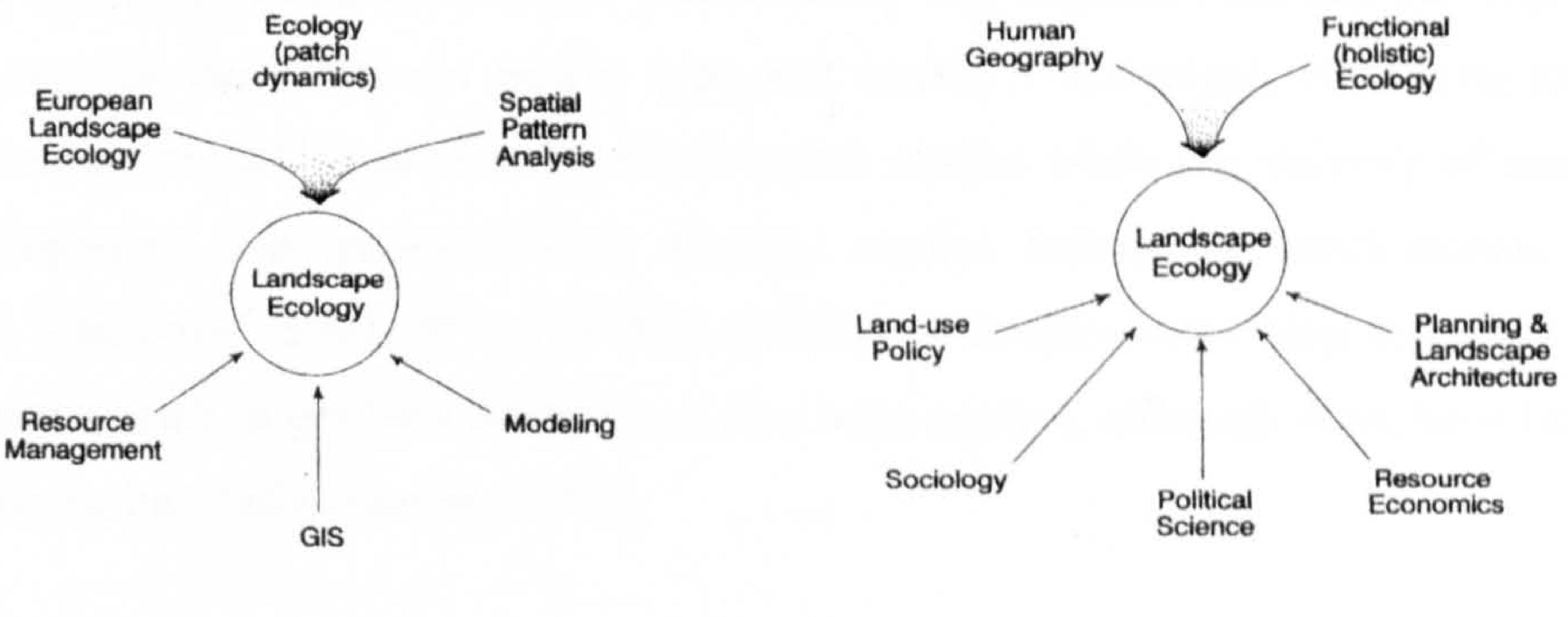


Figure 3.1
Contributions to the development of landscape ecology in Europe (right) and North America (left). Reproduced from (Wiens, 1997).

Landscape ecology provides tools allowing landscapes to be described and studied and may be used in the construction of conservation initiatives. Authors have noted landscape ecology "explicitly spans the spectrum from fundamental research to application" (Mladenoff, 2005). Landscape ecology examines three main landscape characteristics: "structure", "function" and "change" (Forman and Gordon, 1986). Structure is the physical and spatial relationship between landscape elements, function is the way these elements interact, for example the movement of materials, energy or species. These are linked, the structure of a landscape can influence its function, while over time "change" may occur in structure or function (Forman and Gordon, 1986). Landscape elements can be divided: into patches, corridors and the matrix (Forman and Gordon, 1986).

- Patches represent the spatial division of a habitat into separated areas.
- Corridors represent links in the landscape along which flows of resources, species, or allowing for a temporal dimension, habitats can travel (Forman and Gordon, 1986). Habitat corridors may be structural, (when they are similar to an elongated form of patch), or purely functional when no obvious structural corridor exists and virtual corridors occur (Pe'er et al., 2005).
- Landscape matrix represents the landscape context among which a study habitat or species is examined. Where there are clear divisions between habitats the distinction between study patches and the background landscape matrix is obvious. However when intimate mosaics occur or patches show wide ecotones this distinction may be difficult to make.

The treatment of landscape matrix distinguishes between the two main categorical map models. Patch-matrix models stem from island biogeography and metapopulation biology theory and examine patches within a homogeneous and typically inhospitable matrix. The "landscape mosaic model" derives from landscape ecology discourse and attempts to realistically define landscapes, noting that habitat patches will differ along boundaries to different extents, habitat types will share levels of similarity rather than being completely distinct, patches will differ in quality even within the same habitat type and critically that organism use and survival within the matrix may be dependent on matrix type and quality (McGarigal, 2002) (Fig 3.2). The "landscape mosaic model" is prominent within recent studies while the majority of modelling island biogeography and metapopulation biology studies follow the patch-mosaic model (Hanski and Simberloff, 1997, Wiens, 1997). Additional theories describing the landscape in more fluid terms, such as gradient theory have also been applied, although these have been less well used (McGarigal and Cushman, 2005).

Figure 3.2
A strict island biogeographic / metapopulation representation of the landscape (A) and a landscape mosaic representation (B). Black patches represent occupied patches linked by migration, hatched patches are unoccupied, and arrows represent dispersal. The nature of the landscape mosaic may affect dispersal processes. Reproduced from (Wiens, 1997).

A number of landscape patch definitions exist. The "resource patch" represents an area equal to or smaller than an animal's home range within which resources are available (Forman and Gordon, 1986, Farina, 1998). A "habitat patch" is typically a distinct vegetation community that is generally larger than individual home range (Ostfeld, 1992 in (Farina, 1998)). A landscape can be divided into numerous "patches" depending on study species ecology. The definition of patches highlights a central feature of landscape ecological research, that of scale. Definition of patch and landscape extent are dependant on the scale of landscape use by the study species. While this has advantages in allowing the study of patch ecology and process at different scales, it also limits the accurate application of landscape ecology theory to species whose individual ecology is well known. When detailed species ecology information is lacking studies may be limited in their ability to accurately define landscape extents. Inevitably definition of landscape boundaries and scale must be achieved through a combination of knowledge on species ecology and landscape processes. In practice however within many studies, landscapes are rarely defined as smaller than several kilometres width (Forman and Gordon, 1986, Turner, 1989, Forman, 1995b).

A central assumption of applied landscape ecology is that study scale must relate to the organism or habitat being studied. However scale refers to two areas, firstly to the scale at which the study/research is undertaken and secondly to the scale of perception or relevance to the study organism (Forman and Gordon, 1986). Key factors are species dispersal ability, perception of the surrounding environment and methods of resource use. Highly mobile, wide ranging species such as birds can be investigated at scales easily perceivable by humans. However when immobile or poorly dispersing species are studied the concept of landscape differs greatly from an anthropocentric position. The choice of study scale therefore becomes problematical when examining larger cultural landscapes where the conservation of several species groups may be examined. In these situations, although cultural landscapes may be defined at a scale suitable for management, it must be acknowledged that species will perceive and utilise landscapes to different degrees. Such a study may represent a combination of patch-level and landscape-level features for different species, the landscapes being able to be examined with increasing details becoming apparent at closer scales, like fractals (**Fig 3.34**).

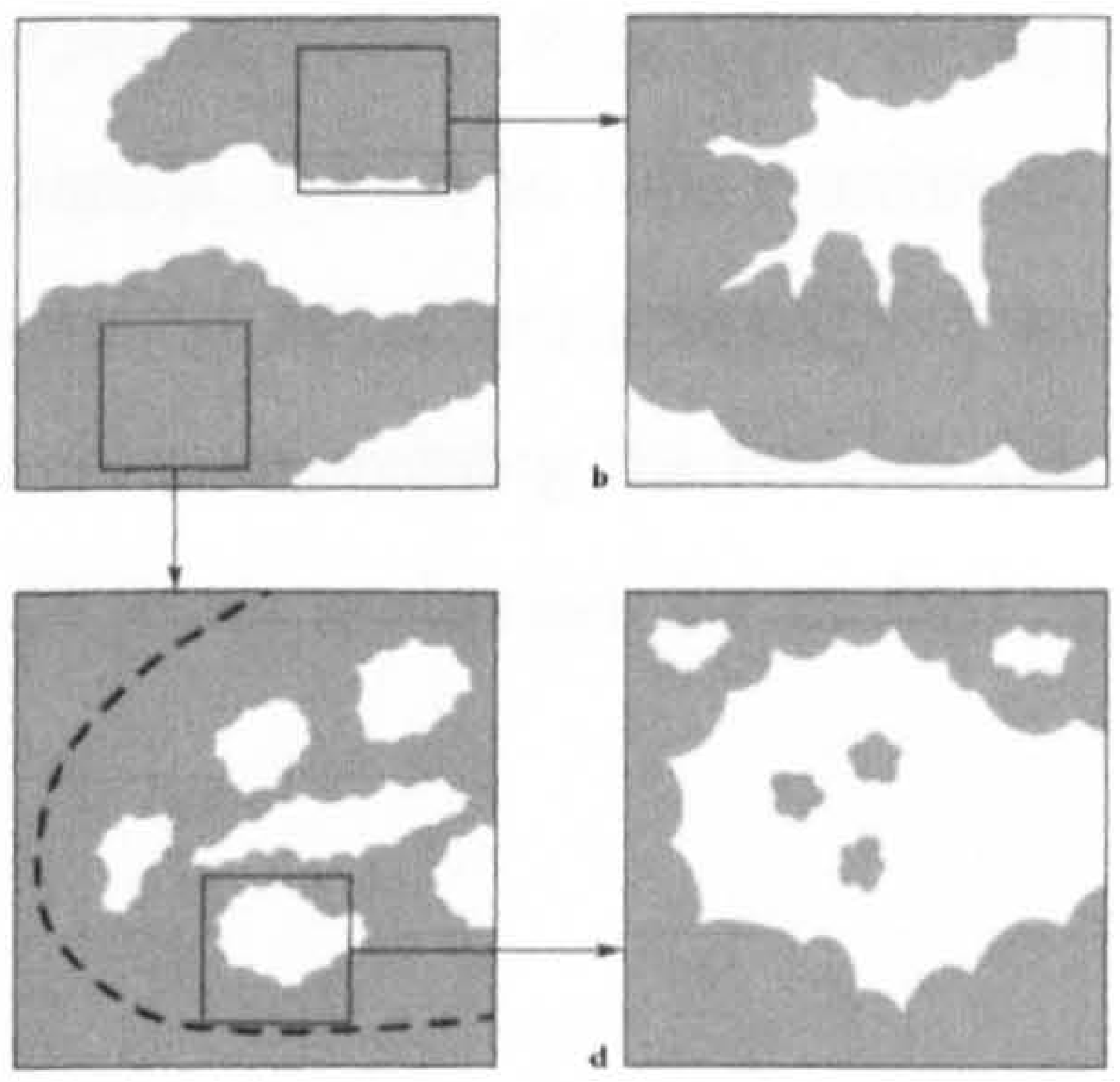


Figure 3.3
Examples of fractal landscapes existing in a woodland context. a= woodland blocks (dark grey) within a non-woodland matrix, b= interior open space within the woodland patch (e.g. grazed areas), c=Windthrow patches within the larger forest patch (e.g. due to topographic exposure), d=Individual trees surviving within larger windthrow gaps, with adjacent areas of individual tree fall within the remaining old growth stands. Reproduced from (Dolman and Fuller, 2003).

Grain is a term relating to study scale and data resolution and has been defined as the finest level of spatial resolution possible given a particular data set, an example being pixel size for GIS raster data (Farina, 1998). Grain has also been defined as the minimum area at which an organism responds to landscape patch structure (Kotliar and Wiens (1990), in Farina, 1998). Critically therefore when data grain is larger than the grain of perception by species then any examination of landscape structure and function will not be able to be accurately modelled. Landscape extent can be defined as the coarsest scale of spatial heterogeneity at which organisms react (Farina, 1998). Therefore while grain defines the minimum areas by which a particular research species are able to perceive the landscape, extent reflects the largest areas or spatial combination of features beyond which species cannot perceive or are not affected by landscape structure. In addition to species perception of landscape grain and extent habitats may show different "resistance" to species movement based on their structural characteristics (Forman and Gordon, 1986). This is affected by boundary discreteness, ability to cross a boundary for the species concerned, hospitability of the habitats and boundary length (Forman and Gordon, 1986).

3.2.2 Landscape Ecology, Island Biogeography and Metapopulation biology

Landscape ecology remains a young discipline. Development of the science has involved the integration of a number of fields of study and the use of theories from traditional ecology, conservation biology and geography. Prominent theoretical frameworks include: hierarchy theory, percolation theory, island biogeography theory and meta-population theory. Additional methods of examining landscapes such as economic geography models have also been proposed but have been less widely applied (O'Neil, 2005).

3.2.2.1 Hierarchy theory

Hierarchy theory is essentially an example of landscape classification and considers systems as components of larger systems that can be scaled, allowing ecosystem processes to be studied in a scaled hierarchical manner (O'Neil, 2005). Examples include the identification of river basins within larger watershed catchments, which in turn contribute to larger regional river systems. Each unit studied can be related to other units based on their hierarchical relationship. This theory has important applications in examining the flow of functions through a landscape where the interrelation of factors of different scales can be studied by examining the overall hierarchy (Farina, 1998).

3.2.2.2 Percolation theory

Percolation theory is concerned with examining landscape function and the flow of resources or species across landscapes is percolation theory. This is based on computer modelling of artificially generated "neutral" landscapes and examines how structure interacts to cover a landscape surface indicating particular spatial arrangements of habitat patches within a landscape will have strong implications for landscape function (Fig 3.4). Percolation theory has noted certain arrangements of patches result in "thresholds" where a majority of patches are inter-connected at which point processes may cross the landscape as if it were a single patch (Farina, 1998, Forman and Gordon, 1986, Andren, 1994). Applications of percolation theory include the modelling of forest fires, pest outbreaks and animal movements.

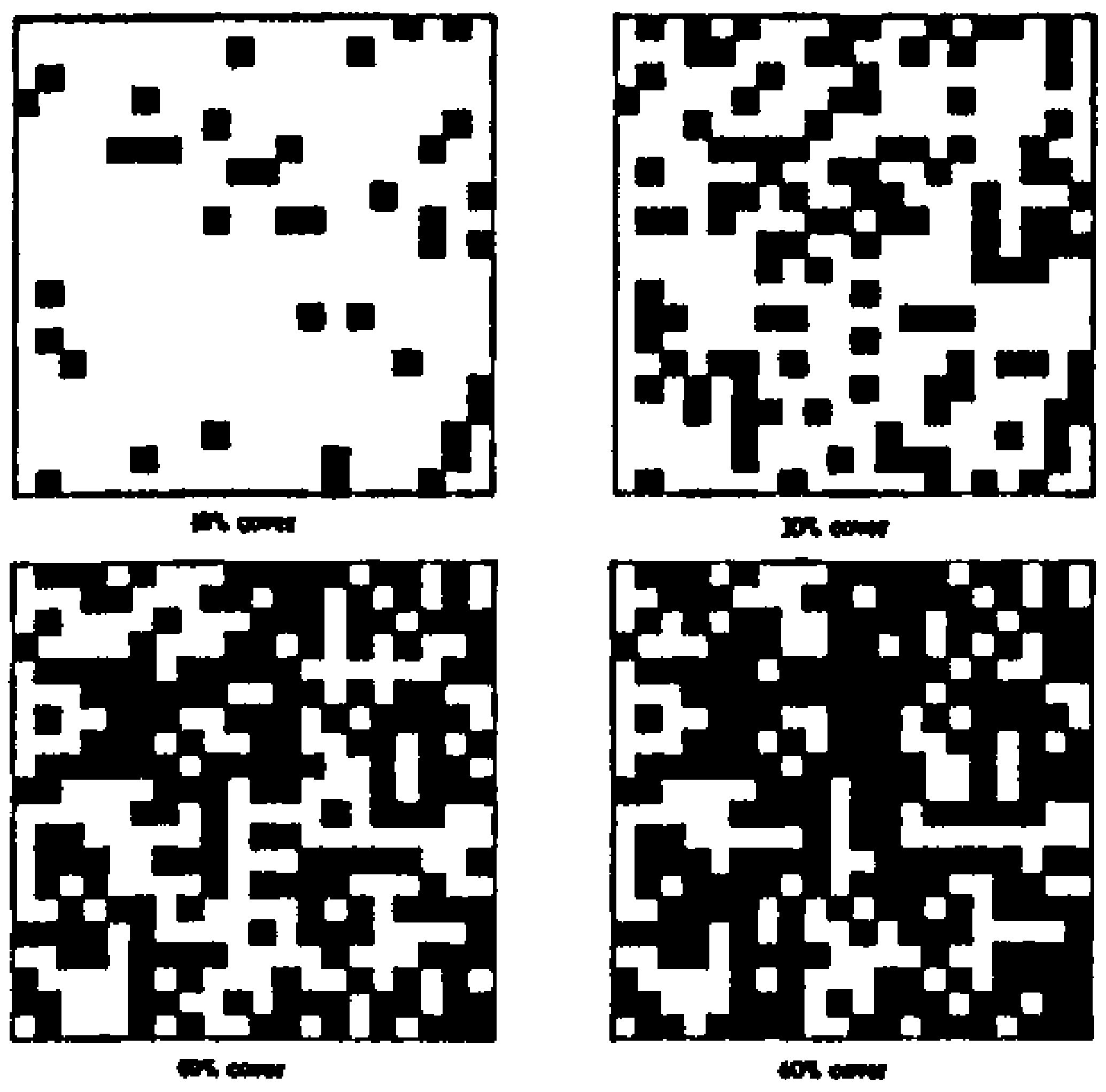


Figure 3.4
Levels of habitat % cover (black cells) illustrating that increased habitat cover reaches a "percolation threshold" when large areas of a landscape are able to be crossed within the habitat type, when high cover of habitat are reached, in randomly generated "neutral" landscape models (The Woodland Trust, 2002)

3.2.2.3 Island biogeography theory

Island biogeography theory has had a large influence on conservation biology and landscape ecology. The theory was developed to explain colonisation and species diversity on newly created oceanic, land-bridge or habitat islands (MacArthur and Wilson, 1967). The theory hypothesised islands developed an equilibrium species richness resulting from an interaction of three factors: island size, age and distance from colonisation sources. Islands closer to colonisation sources held higher numbers of species due to higher rates of colonisation while larger islands held more species due to extinction events being more frequent on smaller islands and the association of larger islands with development of more diverse habitats (Forman and Gordon, 1986). Island biogeography theory has been widely applied beyond the original scenarios from which it developed. Its use encountered problems when applied to landscapes that did not meet original model assumptions. The model assumes suitable habitat patches are isolated from one another by an expanse of hostile habitat where individuals of each species only use one habitat patch, which is bigger than their area requirements, defined as a coarse-grained landscape (Levins 1968 in Andren, 1994).

3.2.2.4 Metapopulation theory

The metapopulation model has many similarities to the island biogeography model, however while island biogeography theory modelled species richness developing on islands receiving immigrants from a permanent source population, Levins's metapopulation model examined single species dynamics within a network of habitat patches (Hanski, 1999). The theory came to dominate landscape ecology and conservation thinking, replacing island biogeography theory, perhaps due to its perceived increased relevance to threatened species in fragmented landscapes (Hanski, 1999). Metapopulations exist when a population occurs in several areas that are fragmented from each other by less favourable, or hostile, habitats but where dispersal still links habitats (Hanski, 1999). The population is a true metapopulation when separate patches are linked by situations of emigration or immigration that occur after the extinction of local populations, when they are re-colonised from adjacent areas. In this situation the whole group of interlinked populations are termed a metapopulation. Populations are affected by the ability to exchange individuals between patches affecting the risk of local extinction and probability of re-colonisation of individual patches. Habitat patches therefore must be large and permanent enough to sustain a population for at least a few generations (Hanski, 1999). The Levins metapopulation model provided insight into mechanisms of extinction in fragmented habitats: as patch density reduces, colonization rates decline and when rates fall below those necessary to compensate for extinctions the metapopulation becomes extinct (Hanski, 1999).

3.3 Conservation biology at the landscape scale

The increasing application of landscape ecology methodologies has resulted in awareness of the importance of landscape context for the conservation of threatened species and habitats. Conservation activities are increasing beyond the site scale (Hawkins and Selman, 2002). There is now an understanding of the need for conservation policies to apply to networks of sites, or to address the conservation of the landscape "matrix" by examining biodiversity in the wider countryside beyond the boundaries of nature reserves and protected areas.

3.3.1 Sites to strategies: conservation at the landscape scale

Conservation has historically shown a reliance on site-based preservation, often on a reactive basis due to perceived threats to species or habitat persistence (Marren, 2002). With the development of larger conservation charities and agencies at national and international scales there has been a move towards more strategic conservation approaches. A variety of organisations are currently undertaking large-scale habitat creation and restoration projects within the UK, including the RSPB, National Trust and the Woodland Trust. National Biodiversity Action Plans are also promoting the conservation and restoration of threatened habitats above levels currently protected within nature reserves and conservation schemes (The UK Biodiversity Steering Group, 1995b). Conservation planning has acknowledged that simple site protection mechanisms are unsustainable to conserve habitats and species. Nature reserves and protected sites do not exist in isolation from their surroundings; the ecological interest of sites may be dependent on the processes or quality of adjacent areas of habitat or the quality of the surrounding matrix. Several interrelated strategies are therefore increasingly being practised by conservation organisations: conservation at larger spatial scales, addressing the configuration of habitats within the landscape by designing networks, and by examining the restoration of landscapes to states when habitat connectivity was higher, or levels of fragmentation were lower. Studies have indicated that conservation at broader scales are beneficial for several reasons, including the ability to address issues beyond the scale at which landscape structure of regional local weather patterns may cause population synchrony in species such as birds which may enhance local extinction events (Bellamy et al., 2003).

Conservation activity can be defined by its intent: fine-filter and coarse-filter conservation dealing respectively with the conservation of genes/populations/species, and communities/ habitats/ecosystems/landscapes (Schwartz, 1999), while additionally ecosystem management focuses on the conservation of processes such as nutrient cycling (Simberloff, 1998). While these aims can be applied at different spatial scales, it would be more likely for species to be conserved at the "site scale" and coarse-filter conservation applied across larger areas (Schwartz, 1999). Schwartz (1999) noted ecological theory suggested conservation at larger scales would be preferred but that both fine-filter and coarse-filter objectives have been met in practice by small reserves.

Landscape ecology has emphasized that patch characteristics are important and these considerations have been applied to conservation planning (Forman and Gordon, 1986). Theories extended from island biogeography and metapopulation theory indicate that important aspects in the conservation of sites are proximity to other reserves, size of reserves, shape of reserves and structure and land-use of surrounding habitat (Diamond, 1975, Forman and Gordon, 1986, Margules and Usher, 1981, Margules et al., 1988) (Fig 3.5). The exact relevance of factors however, and the scale at which they apply, will depend on the species, guild or habitat that is the target of conservation action. An extensive debate developed within the literature, examining the SLOSS or "Single large or several small" issues. Various authors noted the relative merits or limitations of island biogeography theory to aiding nature conservation planning and design (Diamond, 1975, Ovaskainen, 2002, Gilbert, 1980, Higgs and Usher, 1980). Research considered whether the benefits of conserving an area of land would be greater if all effort were placed into a single large reserve or several smaller reserves, covering the same total area. Studies have confirmed that individually reserve size was positively linked to species richness, however several smaller reserves could contain a larger set of total species than an equivalent sized reserve e.g. (Peterken and Game, 1984). These relationships depend on habitat heterogeneity within each site and the relative distance between reserves. Where sites with heterogeneous habitats occurred over sufficient distances to be affected by regional differences in species presence, then a larger number of smaller reserves may be preferable. However, where sites occurred in more homogenous terrain, within a distinct area, then a single larger site may be preferable. A number of additional factors were also examined and found to potentially affect reserve design and location value, such as the extent to which edge-effects may be detrimental (Lidicker, 1999, McCollin, 1998). Additionally the more recent development of the landscape mosaic model has fostered consideration of conservation activity beyond site boundaries addressing the quality of the wider countryside in addition to reserve design considerations (Kupfer et al., 2006, Donald and Evans, 2006, Verbeylen et al., 2003). Early examples of the application of conservation prioritisation strategies include the identification of the Nature Conservation Review sites within the UK (Ratcliffe, 1977), while Sites of Special Scientific Interest (SSSI's) attempt to form a representative network of high quality habitats.

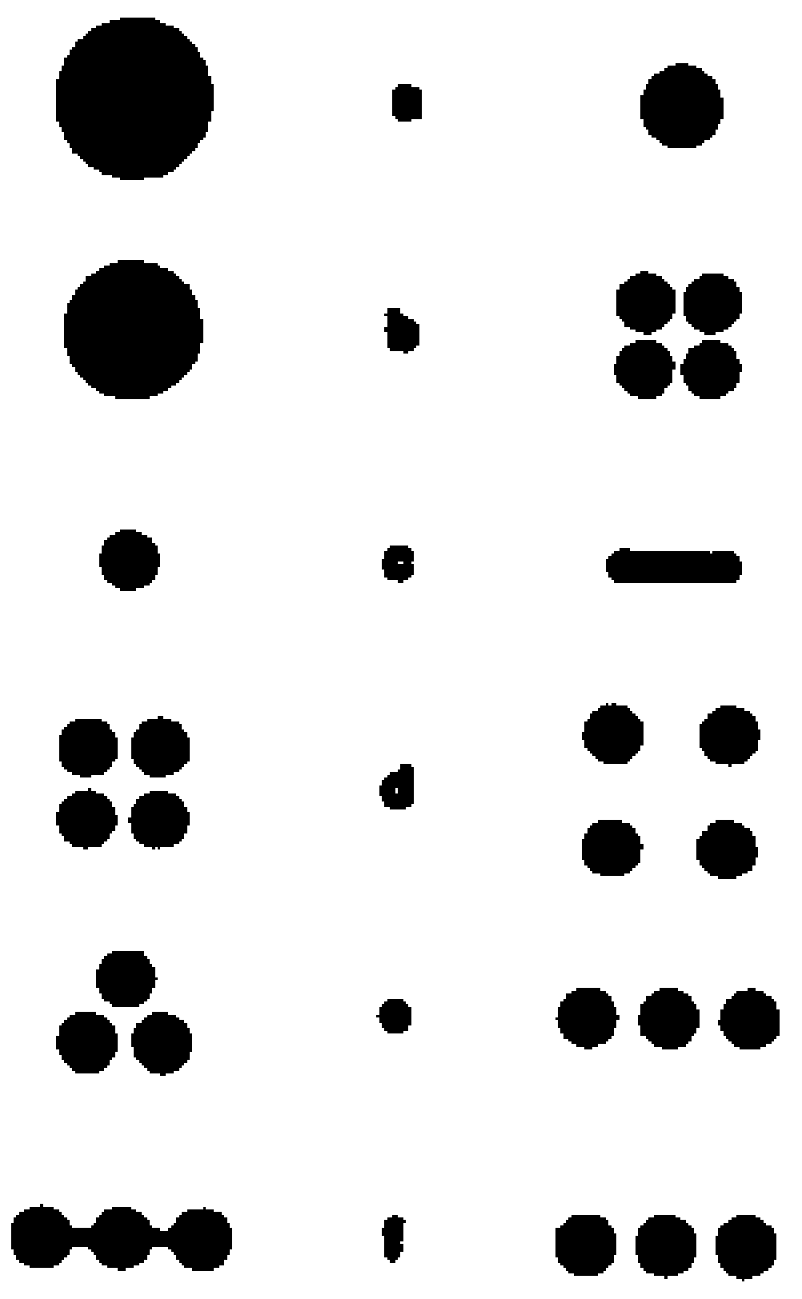


Figure 3.5
Island biogeography rules for nature reserve / refuge design. Designs on the left were considered superior to those on the right. Reproduced from (Hanski and Simberloff, 1997).

3.3.2 A review of methods to select, prioritise or target areas for conservation action

3.3.2.1 Introduction

When conservation is applied across a landscape, or to create a network, then sites, areas of land or alternative conservation options must be chosen from potential lists and prioritised. Conservation planning must assess where the most appropriate areas of habitat to conserve are, in order to achieve maximum benefit. The choice of reserve or restoration network should be well planned, justified and "optimal" (Pressey et al., 1993, Lomolino, 1994, Flather et al., 1997)). Differing approaches may occur where reserve strategies are based on the conservation of existing biodiversity or where strategies aim to undertake habitat restoration, developing future biodiverse environments. Strategies are based on assessments of either current or potential diversity levels. Schwartz (1999) summarised several reserve selection systems based on maintaining or maximising biodiversity levels. These were: flagships, umbrellas, indicators, hotspots, minimum sets, gap analysis and gap analogs.

3.3.2.2 Surrogate and indicators species

The majority of reserve planning methods include forms of rapid biodiversity assessment, for use in landscape planning that rely on surrogate or indicator species. Species may be used as "surrogates" for uncollected information on which conservation action is prioritised (Caro and O'Doherty, 1999). Where the presence (or occasionally the abundance) of individual species are frequently associated with high levels of general biodiversity then they may be used to prioritise areas for conservation, without the need to fully survey an area's biodiversity (Caro and O'Doherty, 1999, Noss, 1990). However the use of such indicators may be problematic and studies examining the supposed association between indicator species presence and biodiversity levels have often failed to correlate diversity in one group within another, possibly due to examining areas at too coarse a scale (Schwartz, 1999, Flather et al., 1997). Therefore while indicator species development in resource management is a popular topic, and has been widely applied, some researchers have expressed concern that relatively few indicators exist that have been adequately tested or validated (Noss, 1999).

3.3.2.3 Flagship and umbrella species

A common use of the surrogate species concept is when the conservation of a habitat is promoted with "flagship species", used to attract conservation funding or public awareness of the conservation needs of a habitat or landscape (Simberloff, 1998). Flagship species may simply be large, well known or emblematic species of a particular ecosystem and may not necessarily fulfil the biological function of other indicator species, instead acting by their relative high public profile or appeal. In contrast "Umbrella species" are utilised where it is believed that conservation activity targeted to areas containing the umbrella species will result in the conservation of associated habitats and species, due to their large home ranges or levels of landscape use (Schwartz, 1999). The use of umbrella species ideally requires detailed knowledge of species area and habitat requirements (Caro and O'Doherty, 1999).

3.3.2.4 Focal species and landscape species

Lambeck, developed a multi-species extension of the umbrella species concept, defining individual "focal species" to select particular landscape attribute states that when combined would meet the conservation needs of the wider community or habitat (Lambeck, 1997, Lambeck, 2002). Lambeck suggested grouping species by categories of persistence "threat" and utilising the most sensitive species (focal species) within each category to define minimum acceptable threat levels allowed to be present in the landscape (Lambeck, 1997). The definitions of these sets of landscape attributes allowed areas of the landscape or patches to be defined that were likely to be highly important for conservation without the need to prove current presence in such patches of any one priority species. Additionally the process allows future landscape forms to be designed or suggested by providing guideline templates on patch sizes and configuration that are likely to be beneficial for the conservation of the community. The theory acknowledged past criticisms of single-species orientated management and noted that because a landscape was planned based on a set of the most demanding species it would also support the majority of other species within the community. The original umbrella species concept allowed areas to be prioritised based on umbrella species presence, conservation boundaries being suggested by the distribution limits of the species. Lambeck's approach was more explicitly spatial and grounded in landscape ecology, enabling particular features of the landscape to be quantified, such as minimum patch area and isolation measures, such that whole landscapes could be designed or altered in order to foster conservation aims. Lambeck noted that while focal species choice will ideally be based on the availability of a full range of auto-ecological data, that in reality for conservation planning this will rarely be the case and best usage of available and estimated data must be made (Lambeck, 1997). The value of this additional surrogate species terminology has been debated (Caro, 2000, Caro, 2002, Armstrong, 2002). The approach extends the selection procedure to multiple species, allowing identification of several conservation concerns in the landscape and avoids the use of generalisation by linking particular species characteristics to landscape structural variables. This approach has now been applied in Australia: (Brooker, 2002, Lambeck, 1997), Europe: (Padoa-Schioppa et al., 2006, Ray et al., 2004a, Ray et al., 2003b, Ray et al., 2004b, Latham et al., 2004) and South America and Africa: (Coppolillo et al., 2004, Sanderson et al., 2002). Brooker termed a suite of focal species a "focal community" and developed a methodology to formalise focal species choice within "ecological neighbourhoods" defining local patch size, spatial arrangement and connectivity where species conservation may be fostered (Brooker, 2002). In discussing the inevitable limitations of focal species selection and landscape planning Brooker noted "in order to develop effective plans for intervention, it is necessary to apply the focal species knowledge in a way that is appropriate to the expectations and goals of the community; that is achievable within the constraints of time and money; is progressive in application so that early efforts can be built on, not wasted; and parsimonious in design so that simple actions translate into complex achievements" (Brooker, 2002). Other authors have defined similar methodologies, the "landscape species approach" has been applied by the American Wildlife Conservation Society to sites in Latin America and Africa (Sanderson et al., 2002). "Landscape species" are noted as useful for highlighting human and conservation land-use conflicts, being defined as species that "use large, ecologically diverse areas and often have significant impacts on the structure and function of natural ecosystems" Redford et al (2000) in (Sanderson et al., 2002). The practice is related to focal species and keystone species planning but aims to define the upper and lower limits of the landscapes to be conserved, the grain and extent, based on local landscape species presence (Sanderson et al., 2002). The method aims to highlight whole landscapes rather than examining individual patches or ecological neighbourhood features. The concept was recently extended by methods designed to make species choice more rigorous, accountable and less arbitrary (Coppolillo et al., 2004).

3.3.2.5 Keystone species

"Keystone" species, has been considered particularly useful, with management for such species thought likely to maintain species richness (Simberloff, 1999, Caro and O'Doherty, 1999). These species have a keystone position in an ecological community, with impacts on a number of other species beyond the levels suggested from their abundance or biomass (Paine, 1969 in Simberloff, 1999). This method has been suggested as uniting the best features of single-species and ecosystem management based conservation approaches and praised for its attempts to understand ecosystem structure and function (Simberloff, 1998). However limitations include the difficulty in defining relevant keystone species without conducting extensive fieldwork and the potential for different keystone species, although valid, to have different requirements, therefore while useful management should not be so driven to increase keystone species at the expense of other species in the habitat (Simberloff, 1999).

3.3.2.6 GAP analysis and map methods

Several spatial conservation-planning techniques are applied following the designation of sites and reserves or to test the potential value of hypothetical reserve networks. GAP analysis assesses conservation activities to test if networks provide optimal conservation benefits (Burley 1988 in Flather et al 1997). The method uses cartographic approaches to compare maps of vegetation or animal distribution against networks of conservation reserves, in an attempt to assess whether networks adequately cover the diversity of a landscape (Flather et al, 1997). Analysis occurs at large scales and therefore typically deals with coarse-filter conservation (Schwartz, 1999). GAP analysis may examine the coincidence of biodiversity "hotspots": areas where there are large numbers of species in different taxonomic groups. The method is therefore reliant on the assumption that "indicator" species used to map the biodiversity hotspots reflect the distribution of total unmeasured biodiversity, and that biodiversity hotspots for different species groups are likely to overlap, both of which are not always the case (Flather et al., 1997).

3.3.2.7 Site ranking, representativeness and complementarity

Following the identification of potential reserve sites within a network or conservation program sites may be ranked to identify priorities. Ranking may be based on factors such as species richness or indexes of representativeness. Ranking based upon species richness is problematical as two sites may hold similar levels of species richness but may differ in their actual species composition, therefore indexes that use a measure of "representativeness" are considered superior to simple species richness (e.g. Margules et al 1988 in Flather et al, 1997). Representativeness indicates the extent to which the range of species recorded within a candidate reserve is typical of the range of species found within a region or target habitat. Although this method is typically based on a comprehensive range of species occurrence data from one or more groups, the method still holds limitations in that it assumes, as with indicator species based approaches, that reserves selected on the basis of representativeness for sampled taxa are also representative for un-sampled taxa (Flather et al, 1997). However the authors noted that results based on studies examining representativeness criteria were more often supported than other measures (Flather et al., 1997). Additional selection methods include the use of complementarity, where the marginal gain that adding another reserve site to a reserve network brings to total conserved biodiversity is assessed between sites in order to select the most efficient reserve network (Margules, 2005). Achieving representativeness in a reserve network can be costly and difficult, especially where habitats are fragmented, and can result in the need to acquire many reserves (Flather et al, 1997). Recent studies considering reserve network creation, based on examination of species data, indicate priorities should target areas not only where species occur, but where they occur in "peak" abundance (Gaston and Rodrigues, 2003). However in order to achieve networks reflecting a species peak abundance considerable sampling effort is required and significantly larger network areas may be needed compared to networks based on simple occurrence data (Gaston and Rodrigues, 2003).

3.4 Landscape planning, GIS and landscape metrics

3.4.1 Introduction

The coupling of landscape ecology with conservation biology enhances the application of landscape planning (Naveh and Lieberman, 1990). This has resulted from the development and increased use of two inter-connected technologies: Geographic Information Systems (GIS) and Ecological Modelling. GIS allows data to be manipulated, compiled and queried at the landscape scale. Ecological modelling allowed analysis to define environmental variables affecting species and through modification of these variables to understand what changes in species distribution or abundance may occur. GIS research has often leant towards the practical examination of conservation problems and has tended to simplistic assumptions about species-environment relationships, typically investigating habitat conservation, while much ecological modelling has concentrated upon species.

The goals of landscape planning may be various, likely aims being the creation of landscapes within which natural processes maintain species diversity. However it is often recognised that ongoing and long-term human intervention may be required (Boothby, 2000). Landscape planning has been considered to be "most worthwhile where it tackles the processes that generate and control landscape change" (Boothby, 2000). Areas of planning may therefore investigate the cultural, social and economic factors that cause such change. Boothby (2000) summarised that "the science of landscape planning can never be a purely technical matter, but one which also takes account of (i) the practical difficulties of reconciling legal (statutory planning) and ecological viewpoints, (ii) structural impediments and (iii) the continuous creation of landscape as a cultural product". The increased availability of appropriate technology and ecological knowledge have therefore placed "the landscape planner in the role of integrator between policy, economy and ecology" (Jongman, 2002). Such activities may then be applied to regional or local mapped landscape zones within such landscape planning programs (Hamilton and Selman, 2005, English Nature, 2005).

3.4.2 GIS Geographic Information Systems (GIS)

The development and use of Geographic Information Systems (GIS) in the geographic and biological sciences has increased rapidly within recent decades. A GIS consists of computer hardware and software where a variety of visual, topographic and land-use data can easily be accessed, interpreted and analysed (Johnston, 1998). GIS data are geo-referenced to a common co-ordinate system and hold attribute information allowing the investigation of relationships between datasets (Johnston, 1998) (Fig 3.6). GIS is recognised as a key medium for the study of ecosystems and many examples exist in the literature (Dettmers and Bart, 1999, Swetnam et al., 1998, Peccol et al., 1996, Purdy and Ferris, 1999, Nikolakaki, 2004).

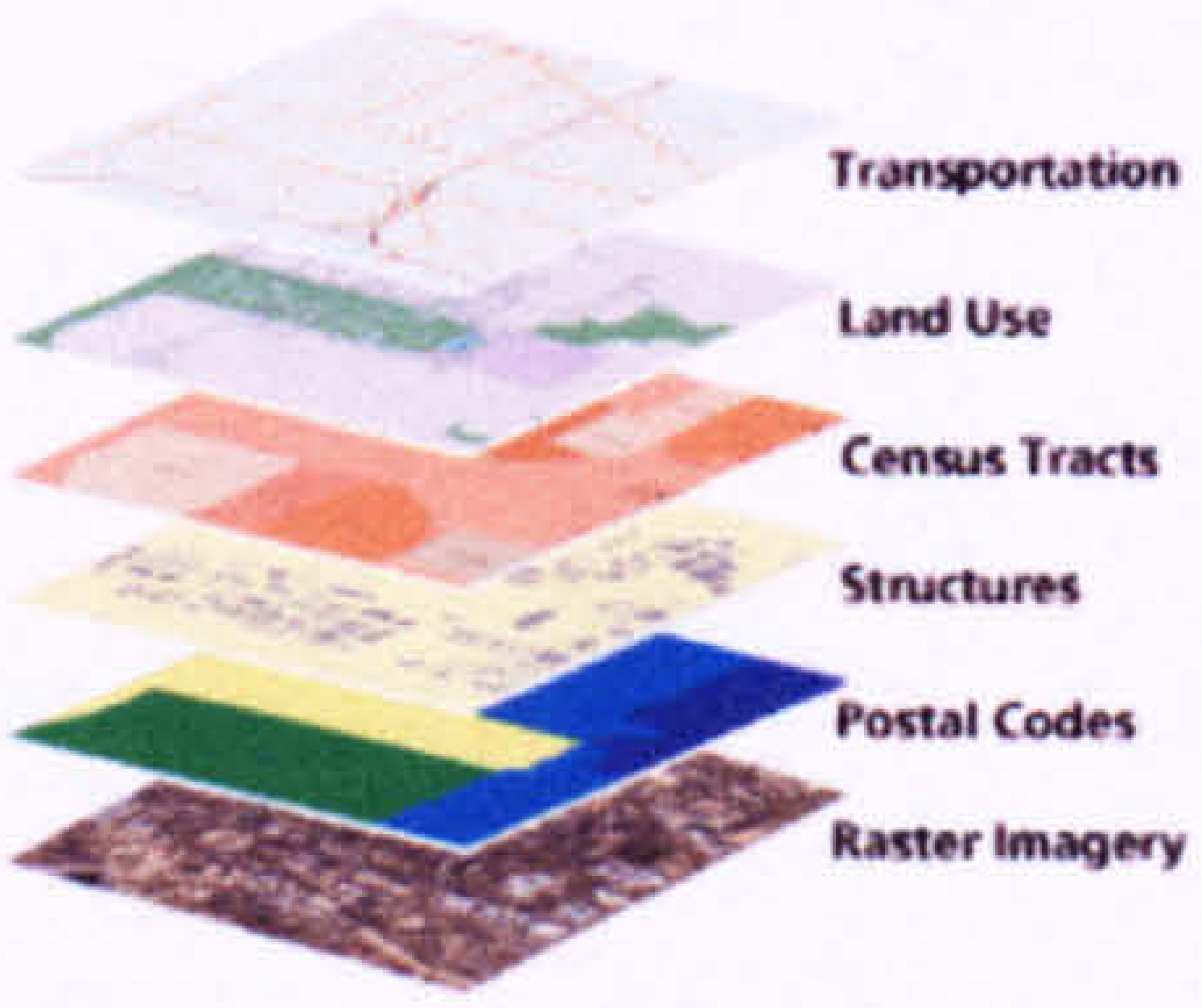


Figure 3.6
Types of data able to be overlaid within a GIS. Reproduced from GIS.com

GIS combines database technology with spatial and visual analysis and mapping tools. Within ecology GIS may be utilised within areas including the visualisation and communication of research results, for the audit, inventory and analysis of data and within the field of prediction, modelling and decision making (Wadsworth and Treweek, 1999). Within a GIS spatial querying and manipulation can allow a set of spatial circumstances to be extracted for use in management planning. Most commonly this can be by the combination of different datasets such as geology, topography and land-use (Anon, 2002b). This is invaluable in providing rapid solutions to landscape management queries (Fig 3.7).

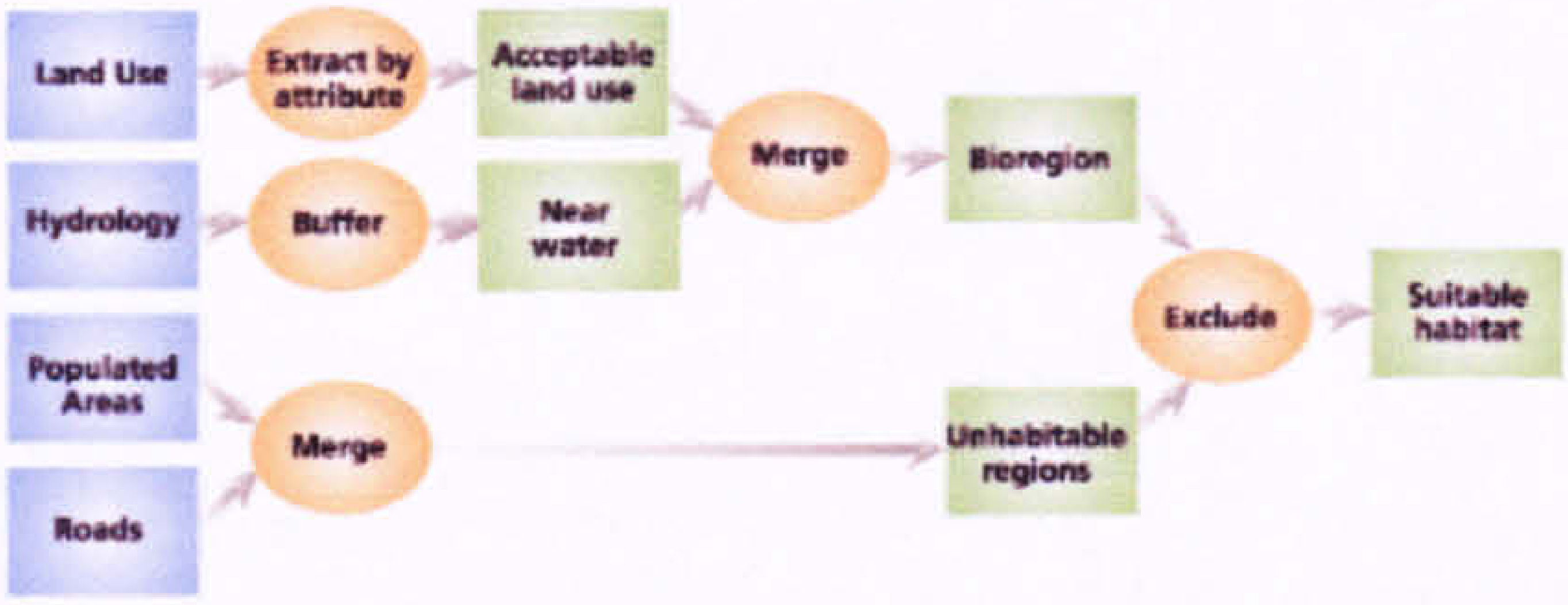


Figure 3.7
Example of GIS data manipulation and combination methods. Reproduced from GIS.com

Within the UK GIS have been developed to target habitat restoration or creation schemes (Gkaraveli et al., 2004, Thompson et al., 1999c, Thompson et al., 2001b), while some studies have developed specialised GIS interfaces allowing visualisation of altered landscape management scenarios, facilitating understanding by stakeholders and landowners (Brown et al., 1998, Garcia and Armbuster, 1997, Dolman et al., 2001, Hehl-Lange, 2001). GIS studies are most typically based on real landscapes utilising the patch-matrix or landscape mosaic models, however some studies have utilised hierarchy theory (Palik et al., 2000). Data sources used typically comprise land-use data (Purdy and Ferris, 1999, Latham et al., 2004), combined with topographic data, principally elevation and its derivatives: aspect, hillshade and slope, or novel applications, such as depth-in-sink (Antonic et al., 2001). Topographic features may be utilised as environmental variables themselves or may be used to estimate soil hydrology character through wetness indexes (Russell et al., 1997, Mummery et al., 1999) or be used to model soil formation processes (Ryan et al., 2000).

3.4.3 Ecological modelling and predictive habitat models

A model consists of a mathematical abstraction of a real world situation. Modelling therefore allows outcomes to be extrapolated into future or novel environments where the model may have predictive power. Ecological modelling can be of use in testing which properties of hypothetical or modified landscapes may be of importance for conservation. Two types of ecological model occur, descriptive models that condense empirical facts into concise models but don't reveal underlying ecological functions and simulation models which attempt to exactly model the ecological processes and so reveal more of the ecological factors in action (Wissel, 1992). Predictive species–habitat models or "habitat suitability" models extract species-use information from a landscape, quantify the information within a model and apply the relationship back to landscapes to be predictive, such models are therefore often empirical (Guisan and Zimmermann, 2000). These models link species distribution, presence or abundance data to environmental variables, and formulate rules for landscape use. Model outcomes then map species' potential or realised niche (Austin et al., 1990, Birnie et al., 2000, Westman, 1991). In addition, models have also been developed for "predictive vegetation mapping" where the environmental variables determining habitat presence or quality are investigated (Franklin, 1995). The formulation of such models has been described as "static" since they relate distributions to current environmental situations and do not take into account changing environments or dynamic ecosystem processes (Guisan and Zimmermann, 2000). They may still retain predictive power when applied to altered static environments, but cannot model dynamic processes. They may however require re-testing or re-calibration in novel situations to re-determine their predictive ability.

In species based models, ideal variables are those that represent casual factors affecting species distribution and abundance (Guisan and Zimmermann, 2000). However such data may be difficult to measure across large areas and may be gained from sources that are themselves models, such as interpolated temperature maps. A partial solution is the use of "indirect gradients" (Guisan and Zimmermann, 2000). These are defined as "variables that have no direct physiological relevance for a species" such as slope aspect, topographic position, geology, but can allude to or replace the measurement of the actual variable that affects a species and through simple rule associations can be used in modelling instead of the original "direct" or "resource gradient" (Guisan and Zimmermann, 2000).

Table 3.1
Recommended stages in construction of a predictive species distribution model within a GIS. Reproduced from Tucker (1997) in (Wadsworth and Treweek, 1999)

Stage	Model construction
1	Collate the potential habitat preferences of target species from the scientific literature
2	Link these habitat preferences with the environmental data held in the GIS directly or by deriving "surrogate habitat variables". These are variables that can be readily extracted from the GIS, which are the nearest representation (in ecological terms) to the true habitat characteristics preferred by the species, as selected in stage 1.
3	Assign conditional probabilities to the habitat variables held in the GIS, given the presence or absence of the species
4	Calculate the prior probability of the species being found anywhere in the landscape, irrespective of habitat
5	Assess the extent of independence of predictor habitat variables
6	Perturb the conditional probabilities associated with each GIS habitat variable over the range of possible joint probabilities if predictor variables are non-independent. Calculate the posterior probability of the species being found, given a particular suite of GIS variables at each point in the landscape, for each perturbation.
7	Buffer around areas of habitat that are avoided by the species (e.g. urban areas, afforested areas). Remove from the map predicted areas of habitat that are too small to form viable territories
8	Output final GIS maps of predicted species distribution.

A wide variety of environmental variables are utilised for modelling studies, frequently stored within GIS. These include: field surveys, observational studies, printed or digitised maps, remote sensing data (aerial photographs and satellite images) and maps obtained from GIS-based modelling procedures (Guisan and Zimmermann, 2000). Digital Elevation Models (DEM) are frequently used and are considered one of the most accurate data sources, but often do not have the most predictive potential (Guisan and Zimmermann, 2000). Species or habitat distributions may be related to underlying environmental variables and modelled using a number of statistical methods: generalized regressions, classification techniques, environmental envelopes, ordination techniques, bayesian approach and neural networks (Guisan and Zimmermann, 2000). The model outputs are then frequently extrapolated to the wider study area where they are typically presented in map form indicating: probability of occurrence (e.g. logistic GLMs), most probable abundance (e.g. ordinal GLN), predicted occurrence (e.g. CCA) or most probable entity (e.g. from hierarchical considerations) (Guisan and Zimmermann, 2000). Methods for the ecological modelling of species distributions are widely available (Table 3.1)

3.4.4 Landscape Metrics

Integral to modelling and research at the landscape scale has been the development of a number of "landscape metrics" used to describe, quantify and study landscape structure and form. These are calculated from GIS categorical data and may be used directly in landscape analysis, or incorporated into ecological models. These have been particularly important in the study of habitat destruction, Schumaker (1996) noting that "much of the effort to integrate conservation biology with landscape ecology has concentrated on the development of better methods for quantifying habitat fragmentation". Metrics range from simple statistics indicating the number and size of patches, to complex metrics defining composition and configuration. A wide range of metrics have been developed within the literature: (He et al., 2000, Jaeger, 2000, McGarigal et al., 2002, Schumaker, 1996, Li and Reynolds, 1993, Gustafson and Parker, 1994). A number of researchers have examined their relative merits and behaviour under different scenarios (Gustafson and Parker, 1992, Neel et al., 2004, Ritters et al., 1995, Hargis et al., 1998, Gustafson, 1998), and their application within different landscapes types or with different species dispersal characteristics (Tischendorf, 2001). The use of such metrics has been promoted through their inclusion within the Fragstats software (McGarigal et al., 2002) and the Patch Analyst ArcView extension (Rempel and Carr, 2003), which adds some of the functionality of Fragstats to a GIS environment. A range of additional software are also available: r.le (Baker and Cai, 1992), LEAP II (Oullette, 2000), IAN (DeZonia and Mladenoff, 2004) and APACK (Mladenoff and DeZonia, 2004). Although additional methods such as graph theory have also been applied to landscape quantification (Roberts et al., 2000, Cantwell and Forman, 1993), the use of metrics remains prominent.

Landscape metrics occur within two main categories: composition and spatial configuration (McGarigal, 2002, Gustafson, 1998). Composition metrics examine the occurrence of patches in terms of their frequency, variety and area without referring directly to spatial configuration. Composition metrics include: abundance of each habitat/class, the number of habitat/class patches and measures of evenness and diversity (McGarigal, 2002). Spatial "configuration" measures the spatial occurrence of patches across a landscape, beyond simple descriptions of patch occurrence and total area. These metrics include: patch size distribution, shape complexity, core area, isolation, contrast, dispersion, contagion, subdivision and connectivity.

However the extent to which metrics can truly represent landscape "spatial arrangement" e.g. by contagion has been criticised (Hargis et al., 1998).

Metrics may be reported as their raw distribution, mean values or as area weighted means at three different levels: patch metrics refer to metrics calculated for individual patches, class metrics refer to metrics that are summarised in some form to represent the value for all patches of a particular habitat class across a study landscape while landscape metrics refer to values relating to the entire study landscape (McGarigal et al., 2002). All metrics within these levels may be compared between different landscape types while patch and class level metrics can also be compared between different habitats or patches within the same landscape.

3.4.5 Landscape character assessment

Landscape Character Assessment has developed within the field of landscape architecture and involves the identification and mapping of dominant landscape aspects, and is utilised in areas such as environmental assessment, landscape planning and conservation. The approach has originated from a largely visual assessment methodology and although incorporating elements of ecological and environmental data remains an anthropocentric and experiential based assessment process (Swanick and Land Use Consultants, 2002). The approach has been widely applied within England to map 120 Natural Areas, defined as areas of relatively homogenous landscape character (English Nature, 2005). While the Landscape Character Assessment methodologies are biased towards aspects of human perception by their intended usage, they do incorporate ecological factors defining different landscapes, such as geology and soils and are considered suitable units for landscape planning. Authors have noted that while the analysis is apparently objective, that the approach concentrates on aesthetics and design and remains "value laden and subjective" (Dolman et al., 2001). The authors also note such techniques are generally focused on "the reactive preservation of relatively recent cultural landscapes rather than enhancement through visionary change" (Dolman et al., 2001). Such limitations may be addressed where current landscape character assessments are integrated with information from historic landscape assessment (Anon, 2002c) and for "future landscape modelling" as in the English Nature Lifescapes and Living Landscapes projects (Griffiths et al., 2004a). There remains further scope for integration of landscape character assessment with landscape ecology and landscape planning, an approach that has been taken in several recent studies (Griffiths et al., 2004b, Anon, 2005c).

3.5 Discussion: landscape ecology and landscape planning

3.5.1 Introduction

Landscape ecology provides a framework by which the impact of landscape structure, measured through an analysis of patches, corridors and matrix can be assessed on species populations.

Conservation biology and landscape planning, utilising GIS and ecological modelling methods can take such relationships and use them in the design of conservation networks, ensuring optimum use of conservation resources, at the landscape scale.

3.5.2 Landscape ecology theory and applied research models

A variety of theories have been proposed by which landscape structure may affect populations, the most prominent of which has been the development of metapopulation theory and its extensions. These theories have provided a conceptual basis by which species-landscape effects can be tested. However where populations cannot be assumed to occur in discrete areas, linked by migration, then metapopulation theory cannot be applied. The smaller and more distinct local breeding populations are, the more applicable the theory is likely to be (Hanski and Simberloff, 1997). Much observational support for metapopulation dynamics comes from relatively short-lived populations of small species in patchy and unpredictable environments (Harrison and Taylor, 1997). The concept therefore has most use if assumptions are fulfilled that the occupation of a patch is related to its size and the reoccupation of patches are related to the distance to occupied patches (Andren, 1994). In classic metapopulations the probability of metapopulation persistence increases with the number of local populations and number of patches (Harrison and Taylor, 1997). Unfortunately in order to be applied in modelling studies metapopulation theory typically simplifies landscape processes and assumes matrix habitats are hostile (Hanski, 1999). This holds problems for realistic studies aimed towards applied landscape planning. Some model developments however have constructed spatially explicit models to reflect more realistic landscapes and account for differences in patch area and configuration (Hanski and Simberloff, 1997).

In considering the application of research models, researchers have noted that in reality many populations are unlikely to meet the strict requirements of classic metapopulation theory. For example habitats may be created or destroyed in dynamic processes rather than species becoming extinct, but leaving patches intact (Harrison and Taylor, 1997). A range of modified metapopulation concepts have therefore been developed. Source–sink population dynamics theory addresses the occurrence of sub-populations in patches that differ in habitat quality (Pulliam, 1988). A source population being defined where births exceed deaths and emigration exceeds immigration and sink populations having negative balance between offspring and death where juvenile production does not compensate for adult mortality. Therefore source patches are constantly exporting individuals and sink patches are constantly absorbing them. The source-sink model is closely linked to the metapopulation concept as the quality of patches are linked to patch size. However in short-term studies populations may still be responding to previous land-use or competitive changes and the definition of source or sink populations may be difficult. The terms are useful in highlighting that the presence and size of a population in a patch may not give an accurate idea of the environmental condition present (Hanski, 1999).

Several studies have extended the source-sink terminology and recognised additional groups. Pseudo-sinks occur where habitats are sub-optimal but have an artificially high population due to immigration exceeding births and deaths. While a true sink population becomes extinct if immigration declines, in pseudo-sinks the population will only decrease if immigration does not occur (Watkinson and Sutherland, 1995). The habitat retains sufficient resource quality to maintain a low level of population, but the levels actually seen are enhanced by high immigration rates. Source-sink and pseudo-sink theories highlight how populations in one "patch" can be affected by habitat quality in adjacent patches, with matrix hostility or resistance affecting migration levels and hence population size, density and persistence. The important factor is that patch quality must be recognised in addition to the location, size and configuration of the patches.

Studies based on the metapopulation model remain important theoretically but may be limited due to their dependence on the patch–matrix representation of landscape, and their level of simplification and abstraction from reality (Wiens, 1997). The application of more realistic landscape mosaic models is particularly important in allowing landscape ecology to address practical conservation issues (With, 2005). The level to which landscape ecological models and theories can be applied to species to achieve their conservation depends on characteristics of population connectivity, dynamics and landscape structure. In real landscapes it may be very difficult to identify if populations occur in metapopulations, patchy populations, source-sinks, mainland-island or as single fully interbreeding populations (Dolman and Fuller, 2003). In particular, metapopulations may be difficult to identify in plant species due to their long-term persistence in patches, and the difficulty in observing migration (Ehrlen and Eriksson, 2003, Freckleton and Watkinson, 2002). Rather than true metapopulations most woodland plant species probably exist as either patchy, isolated or mainland-island systems (Dolman and Fuller, 2003). Patchy populations occur where local aggregations are linked by continuous dispersal and "mainland-island" systems occur where small ephemeral populations are repeatedly re-established by colonisation from nearby persistent large populations" (Dolman and Fuller, 2003). Isolated populations may be truly cut off from other similar populations, and may show long-term persistence, where extinction events would therefore form part of regional population decline, not a balanced meta-population process (Dolman and Fuller, 2003). Isolated populations may show "remnant population dynamics" in plants, where species may persist in small remnants and are able to expand when conditions change but are not linked to other populations (Eriksson, 1996). Such dynamics may occur through clonal growth or through long-lived seed banks, rather than traditional dispersal or gene flow (Eriksson, 1996). In considering real-world plant or animal populations, regional populations may exhibit several different population forms across their range. Within closely-spaced habitat patches a species may occur as a fully interbreeding population, while at more distant and isolated patches the population may begin to show metapopulation effects, with extinction and re-colonisation events, or may show rescue effects through immigration (Fig 3.8 e) (Harrison and Taylor, 1997). A population showing such a mixed structure would still exhibit strong effects of patch structure and dispersal characteristics on its regional distribution and population persistence (Harrison and Taylor, 1997), although its exact dynamics may not easily be modelled by any one clear landscape based theory, due to the diversity of population forms.

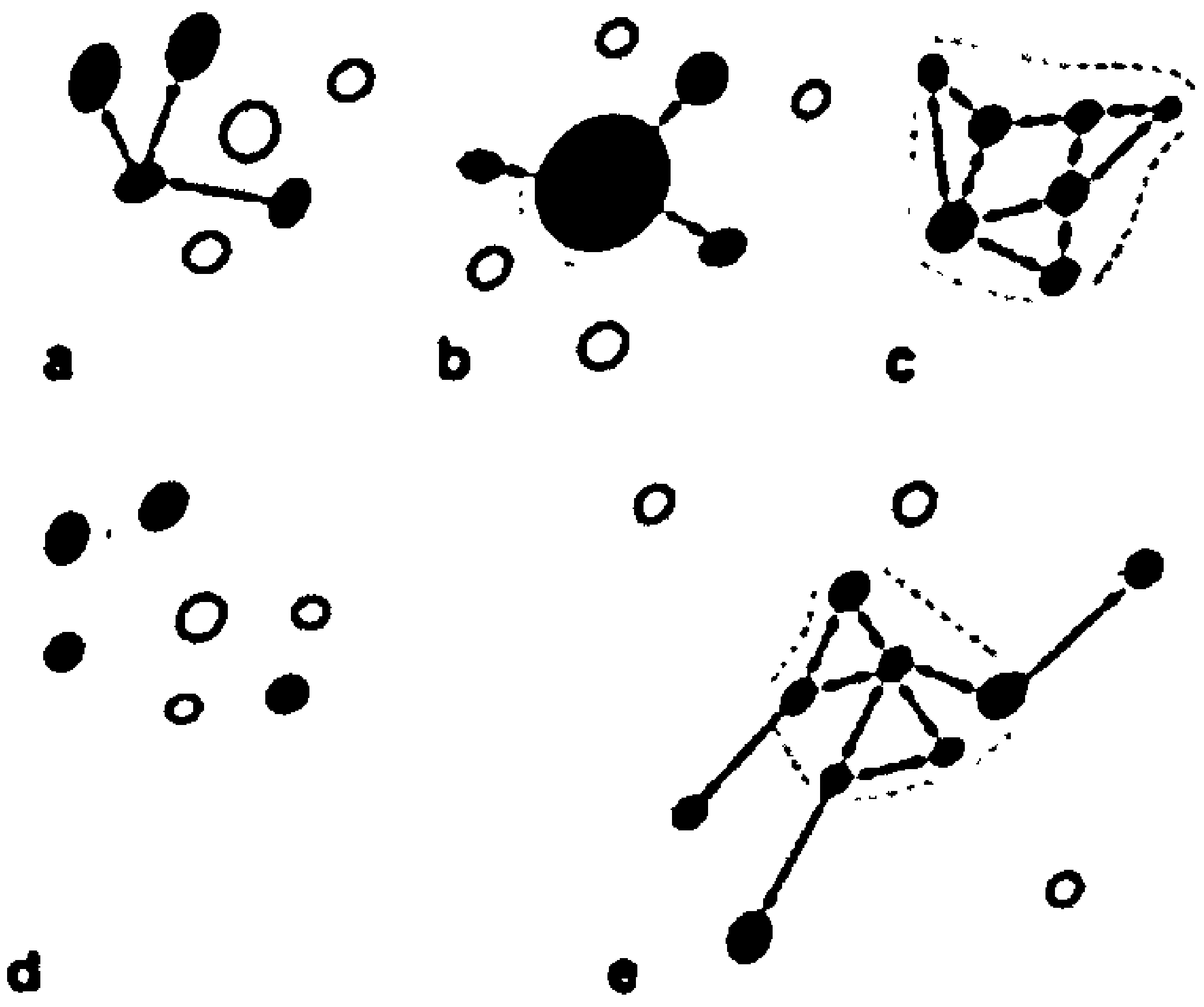


Figure 3.8
Types of metapopulation. Dark circle = occupied habitat patches, open circle = empty habitat patches, dotted lines = local population boundary, arrows = dispersal. a = classic Levins, b = mainland-island model, c = patchy populations, d = non-equilibrium metapopulation, e = intermediate between other options. Reproduced from (Harrison and Taylor, 1997)

The use of particular landscape ecology models for applied conservation planning must be clearly considered in relation to potential population forms and research scales. In very disconnected landscapes it may be preferable to treat patches as effectively separate, and only to consider basic structural variables such as patch size and isolation (Wiens, 1997). However in landscapes with patches closer to the percolation threshold additional features such as the spatial arrangement of habitat and the way in which species cross boundaries between habitat types becomes more critical (Wiens, 1997). While differences exist between the exact effects hypothesised by different theories of population structure most agree that so long as populations are not entirely isolated, and show forms of linkage then populations or sub-populations will be affected to varying degrees by patch size, isolation and the form of the landscape matrix, although the timescales, extent and exact form of such effects will differ between population and species types.

3.5.3 Conservation biology at the landscape scale

Conservation is increasingly based on the conservation of networks of sites or reserves and consideration of biodiversity levels within the landscape matrix rather than focusing on individual sites. Inspiration from landscape ecology studies has focussed attention on the interactions between remnant fragments or reserves in maintaining biodiversity. Acknowledging the fact that conservation resources will always be limited and conservation activity will be based on limited knowledge, a variety of prioritisation methods have been proposed with which to examine potential priorities when examining reserve networks and multiple conservation sites at the landscape-scale. While methods may be based on detailed assessments of site biodiversity, where available, most methods rely on various biodiversity surrogates. Methods may use the presence of species or groups of species to indicate conservation priority based on likely current or potential biodiversity levels (umbrella, flagship, biodiversity indicators) while more recently methods have been developed that attempt to utilise landscape structure factors with which to prioritise action. Landscape structure may be directly linked to conservation importance by the use of focal species whereby landscape structural design attributes enable sites and networks that are hypothesised to be important for focal species to be identified.

Debate remains over appropriate surrogate methods. The applicability of the focal species approach has been challenged (Lindenmayer et al., 2002), although the majority of the reasons for this challenge were refuted by Lambeck as applying to all other data-limited surrogate conservation methods (Lambeck, 2002). Lindenmayer highlighted the problems that lack of available data poses to selection of focal species (particularly dispersal and area requirements), and suggested it was preferable to apply a mix of conservation strategies to account for any potential failings of individual methodologies, potentially combining landscape species and focal species with other strategies. (Lindenmayer et al., 2002). The authors concerns stem from the previously high uptake by conservation organisations of island biogeography reserve design methodologies, before these were fully tested and validated (Lindenmayer et al., 2002). Critically Lindenmayer et al noted that use of the focal species method may lead managers to assume that management will conserve all other biota within the area, noting such aims may be unattainable. Their argument was therefore related to the claims of the method as being suitable for conservation of multiple species by using focal species to target features for landscape planning. Additional criticisms have noted that focal species may have conflicting needs for landscape elements or spatial configuration making identification of "priority" patches or neighbourhoods, or of future landscape configurations, problematical to choose (Westphal and Possingham, 2003). Work has recently noted the importance of considering ecological effects across multiple scales. Therefore recent extensions of the focal species concept, incorporating "generic focal species" with standardised life-history traits (Ray et al., 2004b, Latham et al., 2004) hold potential for analysis across multiple scales, for a range of generic species traits, for which current species profiles may not exist, or which may have already become extinct in a landscape. Such multiple generic focal species analysis can then allow a wide range of potential conservation strategies to be applied across landscapes to quantify functional connectivity, and biodiversity levels in comparison to existing structural patch presence. Additionally many current methods fail to explicitly consider the underlying ecological processes creating biodiversity (Flather et al, 1997). Sites may be prioritised based on current high levels of biodiversity but the process that created these high levels of biodiversity may not be incorporated within the reserves, leading to long-term declines. Potential therefore exists within GAP analysis targeting to include assessment and prioritisation of the factors contributing to underlying ecological processes, by incorporating soils, geology and topographic variables into prioritisation schemes (Angermeier and Kerr, 1994 in Flather et al, 1997). Such approaches could be complimentary to the use of the multi-species focal species approaches forming a complex of approaches to be applied to an area, as recommended by several authors (Lindenmayer et al., 2002, Sanderson et al., 2002). Similar to such concerns of deterministic processes, researchers have recognised the dangers of basing conservation schemes too exclusively on the current range of species present at a site or landscape, where past levels of diversity may be missed. Webb (1989) noted that in many cases relating to the conservation of remnant fragments, that conservation concern is directed towards "a particular set of species, often those characteristic of a community" which Webb termed "representative diversity" (Webb, 1989). Such concerns could apply whereby certain potentially occurring focal species are already absent from a landscape and thus are not addressed. Notably some studies have attempted to include consideration of the requirements of extinct species when designing conservation networks, allowing for potential re-introduction back into a restored future habitat (Ratcliffe et al., 1998). This issue is also relevant to Peterken's distinction between future-natural, current-natural and present-natural woodland types where it is entirely possible that the expected woodland community to develop at a site may differ from current woodland conditions due to ongoing succession and compositional factors (Peterken, 1996). This issue also highlights the problems that exist between the prioritisation of existing areas for habitat conservation based on inferred or recorded habitat quality and the selection of degraded areas for enhancement based upon hypothesised potential future levels of diversity. These discussions note that examining a community as it now occurs is simply a snapshot of that community in time. Therefore such analysis is useful for assessing the levels of biodiversity currently present, but by examining underlying processes driving biodiversity, potential future levels may also be able to be examined. The further investigation of links between habitat quality, biodiversity and deterministic abiotic factors is thus believed to be useful in landscape planning studies.

Many of the conservation planning methods available may be inspired by landscape ecology theory but mostly just acknowledge the axiom that landscape structure will affect biodiversity than follow any particular exact theory of population form. Where detailed species information is available on sites and regional species-pools conservation prioritisation by representativeness, complementarity or ranking of richness may be appropriate. Some indicator species may be useful, but they rely on the indicators chosen being suitably associated with general biodiversity levels. In situations lacking in species information then focal species and generic focal species prove useful. However measures may be heavily biased by the scales at which they are applied and for generic focal species, by the species profiles considered. Even approaches based on landscape ecology assessment of patch sizes and isolation, assumes species-environment usage similar to the definition of generic species profiles by the way in which these relative patch size and isolation distances are valued against potential diversity effects. Key issues are therefore the consideration of conservation planning at multiple scales which would allow an assessment of a wider range of effects than using real focal species and may not be as biased to existing species presence, allowing potential future biodiversity levels to be incorporated.

A range of conservation prioritisation methods exist, varying depending on conservation intent and the availability of data. Several landscape ecology based approaches hold potential for the quantification of potential future interest developing on restoration or creation sites, with generic focal species able to highlight potential usage by a wide range of hypothetical species groups. However most methods suffer from limitations of application scale, notably that even when generic profile species are chosen extremes of profiles such as very short or long distance dispersal or small and large habitat area requirements may not be incorporated due to focusing on typically anthropocentric application scales. Deterministic biodiversity analysis, through ecological modelling and the use of landscape metrics, with increased analysis of abiotic biodiversity links at a range of scales, perhaps inspired by generic focal species profiles provide much scope for future research.

3.5.4 Applied landscape planning (GIS, landscape metrics and ecological modelling)

The combination of conservation biology, landscape ecology theory, GIS and ecological modelling has resulted in landscape planning to define, map or describe landscape features and configuration that are beneficial to biodiversity levels. Such planning goes beyond that traditionally considered within conservation biology reserve network design in allowing a wider examination of landscape features and attempting to truly incorporate ideas of future landscape change and design that may result in enhanced biodiversity levels. Landscape modelling has incorporated a range of GIS and ecological modelling systems and utilised a wide range of landscape metrics. While many studies may be limited by the species information required to parametise models, other approaches such as predictive habitat modelling and planning utilise broad landscape metric approaches and therefore are suitable for use in data-poor situations.

The relevance of landscape planning to conservation is defined by the relative values and importance given to aspects such as the scope of the planning project aims, study scales and the levels of fieldwork or data collection available with which to try and define biodiversity – landscape associations. Additionally while the use of GIS and landscape metrics and of ecological modelling to examine such relationships is standard practice the exact techniques, methods and metrics chosen are rather variable. Landscape planning should utilise at its core an understanding of species ecological requirements and be applied at scales where the relationships between species-environment use and biological and physical processes can be addressed (Boothby, 2000). The successful application of landscape planning however is not entirely an ecological process, requiring clear links to policy and appropriate social and economic situations to allow its implementation (Boothby, 2000).

Landscape planning may utilise landscape ecology theory and methods, but may be implemented by a range of non-academic professions, from landscape architects and planners to ecologists. The gap between research and application has been noted by several authors. The development of ecology from a "reductionist" science to one where it may be applied within landscape planning and decision making situations has been noted as a major challenge (Margules, 2005). For landscape planning to be effective therefore it must be a useable option for a range of professions. Currently the levels of knowledge required for large scale landscape planning are not typically available within organisations such as local authorities, limiting its application (Boothby, 2000). The key balance within such studies is then the relationship between ease of applied landscape planning research, its implementation with e.g. conservation organisations, and the reliability with which such planning can be applied.

Ecological modelling of species distributions are widely applied. Models are used in restoration ecology, where they quantify species-environment usage, and can inform habitat patch qualities that may be of value to consider when designing restoration schemes. Examples being the identification of patches size or isolation distances statistically most likely to result in species presence or breeding use (Hinsley et al., 1994). Additionally where relationships are robust such models may be able to predict future levels of landscape use, when landscape composition is altered, for example following further habitat destruction or habitat restoration (Swetnam et al., 1998). While the number of modelling studies continues to grow there has been some criticism of these studies. These note the frequent lack of appropriate testing of model outputs which when carried out often show low correlations of correctly classified outputs (Beutel et al., 1999). One critique of habitat suitability models found all studies examined held some level of deficiency, many showed poor evaluation of data inadequacies and how variability could affect the final outputs (Roloff and Kernohan, 1999). Many model assumptions may suffer from unknown sampling errors in source data, problems of unreliability of data collected on processes that may be scale dependant and therefore conservation decisions based on such models may not be optimal (Conroy and Noon, 1996). In a review of species-environment "habitat suitability" modelling Beutel et al (1999) proposed that rather than predicted species distribution, that the underlying habitat quality for the species should be modelled in order to reduce the number of model assumptions, increasing its potential accuracy. However this uncertainty remains in the way researchers interpret the modelled habitat quality on potential species presence/occurrence or breeding success (Beutel et al., 1999). Ultimately the accuracy of such models are limited by the data used in their construction. For example where data collected on landscape usage do not differentiate between patch quality (as opposed to patch size and shape) the distribution effects influenced by differences in patch quality will not be accurately modelled. Model accuracy is also affected by data resolution. Research for example has shown increased accuracy in studies using higher resolution DEM data (30m compared to 80m) (Bolstad et al., 1998). Other criticisms of modelling studies apply where the scale of study is not set by clear reference to species ecology and where simple presence / absence data are extracted from inventory or atlas studies using data for purposes for which it was not designed (Conroy and Noon, 1996). Potential solutions to such model limitations include grouping study species by life history traits such as guilds within which data should be based on known features that enable relationships to the landscape to be scaled (Conroy and Noon, 1996).

Within landscape planning studies the way the landscape form is measured and represented in ecological models or biodiversity interest is critical. Importantly studies using modelled species dispersal have confirmed that several landscape metric measures are associated with dispersal characteristics in landscapes, indicating they can be used as surrogates for the potential of landscapes to support particular dispersal processes (Tischendorf, 2001). However there is a danger that due to the wide availability of landscape metrics that forms of the landscape are able to be measured and modelled when the actual function or effect of these metrics may be difficult to discern. Therefore it is important to assess suitable metrics from the literature, or though selection to test particular theory driven postulates. While a wide variety of empirical studies have shown species-environment effects relating to particular landscape metric values such as patch area or isolation many of the studies examining metric use have used neutral landscape models and models of artificial species migration dispersal, in order to allow landscape features to be varied and studies. A central hypothesis of landscape ecology is that landscape pattern will affect species movement and dispersal and thus such computer generations can allow different combinations of landscape pattern to be modelled on potential species movement ability.

The choice of relevant landscape metrics for studies is complicated by a large number of metrics being correlated with each other (O'Neil et al., 1999, Hargis et al., 1998, Ritters et al., 1995). Studies have examined such associations and attempted to define a reduced list of key metrics accounting for landscape variations. Ritters (1995) defined 6 metrics considered suitable for use, these were: average perimeter-area ratio, contagion, standardised patch shape, patch perimeter-area scaling, number of classes, and large patch density-area scaling. O'Neil et al conducted a similar assessment and recommended the use of 5 metrics: average perimeter-area ratio, contagion, relative patch area (average ratio of patch area to the area of an enclosing circle), fractal dimension and the total number of land cover types present (O'Neil et al., 1999). Research examining dispersal success (based on real forested landscape forms) found the best metric predictors to be: core area, area-weighted shape index, area-weighted perimeter-area ratio and patch cohesion (Schumaker, 1996). The proximity index was found to be an effective measure of patch isolation with use being recommended for landscapes with patches below the percolation threshold – landscapes with relatively few focal habitat patch total cover (Hargis et al., 1998). Some studies have criticised use of particular metrics. Schumaker (1996) criticised the use of fractal dimension and contagion noting that most evidence for the use of these indices come from artificial model landscapes which may not accurately reflect real landscape conditions. Neel et al examined the behaviour of class-level landscape metrics within neutral landscapes and noted several metrics were highly non-linear and so unsuitable for linear related statistical analysis, also noted problems may occur with some metrics when the focal habitat being examined is very rare in the landscape (Neel et al., 2004). Others have considered habitat area, number of habitat classes, proportion of dominant habitat, number of polygons, polygon size variance and elevation range were important landscape pattern metrics for landscape managers (Giles and Trani, 1999). These researchers noted that elevation range can be very influential as an independent variable (Giles and Trani, 1999). Metrics can suffer limitations due to the effects of multiple small patches within landscapes, which can cause problems measuring landscape connectivity, although these will differ with study species being examined and can partly be overcome by the use of area-weighted metrics, or by reporting full metric statistics (e.g. standard deviation in addition to means), when examining class metrics (Schumaker, 1996, Hargis et al., 1998). New metrics have been proposed that are less sensitive to small patch size (Jaeger, 2000). Studies have also indicated that thresholds may occur in metric behaviour with varying levels of habitat cover (Hargis et al., 1998, Tischendorf, 2001) and that correlations between metrics and modelled ecological process tend to be higher in landscapes with lower covers of focal habitat, supporting the theory that after the percolation threshold additional landscape structures factors are important beyond habitat area (Tischendorf, 2001). There may be problems in generalizing relationships between landscape patterns and ecological processes including the existence of thresholds, nonlinearity in responses, ambiguity of interpretation and sensitively to spatial resolution (Tischendorf, 2001). Metric studies have therefore indicated, as Andren (1994) suggested, that in landscapes with higher amounts of habitat the main effects on species and populations will be due to the amounts of habitat rather than its configuration, perhaps suggesting these metrics should not be used in such landscapes. The thresholds at which change occurred within the Tischendorf (2001) study were between 30% and 50% of total habitat cover. The author noted that these were related to the percolation thresholds, but also that the analysis based on neutral landscape models were likely to underestimate the predictive potential of landscape indices.

Landscape metrics should be carefully chosen for use in landscape planning studies, with particular combinations of metrics being unnecessary due to inter-correlations, but a sufficient range being required with which to measure landscape characteristics. Studies should assess their relevance to target landscape and species, while analysis of the inter-association of measured metrics should be a key stage in landscape ecology studies, before such metrics are linked to biodiversity variables. In summary the metrics suggested by various studies as providing useful and relatively concise description of landscape process, suitable for consideration in studies include: number of patches, amount of habitat, patch cohesion, area weighted perimeter-area ratio, area weighted shape index, area weighted fractal index, mean nearest neighbour index, area-weighted mean nearest neighbour index, habitat edge, core area and proximity index (Tischendorf, 2001, Schumaker, 1996, Hargis et al., 1998, Ritters et al., 1995). The most robust metrics therefore remain the metrics reflecting information on the nature of individual patches and their immediate surrounding. When class level metrics are being investigated the area weighted forms allow the influence of small irrelevant patches to be minimised. However when patch level metrics are being examined the full range of individual metrics, indicated by their range should be utilised.

3.6 Chapter Summary

Landscape ecology theory and applied research models

- Landscape ecology provides a research framework for conservation landscape planning
- Key factors are the study of patches and the landscape matrix, with the landscape mosaic model being a realistic format within which to undertake landscape ecology studies
- Issues of functional connectivity, patch contrast and / or landscape resistance are important, in addition to patch size, and increasingly, patch quality
- While metapopulation theory is prominent in many studies, research has shown many populations are unlikely to occur as strict metapopulations
- Even where species exhibit mixed population forms, landscape structure will often influence species
- It may be useful within research, instead of focussing on population-ecology perspectives on the landscape / species link, to instead examine deterministic biodiversity factors at landscape and within-patch scales
- Immobile species may occur in remnant populations where patches are particularly isolated and below a connectivity threshold they remain as isolated populations within-patches
- Study landscape boundaries and extent may represent cultural or historic influenced management areas and therefore show anthropocentric scales for conservation implementation purposes
- For analysing or modelling landscape ecology effects study scale should match target species or species groups, e.g. with local neighbourhood connectivity calculations
- Study scale should be appropriate to the organism and habitats in question

Conservation biology at the landscape scale

- In order to address the acknowledged link between landscape form, structure and biodiversity, conservation biology has increasingly targeted action towards the landscape scale, examining multiple habitat patch effects and assessing the potential benefit of corridors and reserve networks
- Informed landscape ecology studies can provide insight into the design of conservation networks and reserve planning
- When many sites are being compared, different conservation networks could be created and conservation biology analysis attempts to investigate which potential network is most optimal and effective in conserving species or biodiversity. Success is measured by examining cost-benefit factors, ranking of site importance, and using various surrogate species methods
- When detailed sites surveys are unavailable landscape ecology methods may be used to target conservation planning action
- Surrogates and indicator species may be used in landscape planning but require careful selection
- Surrogate species methods attempt to minimise data collection by using data for indicator species to represent broader biodiversity in other species groups. Most surrogate methods therefore still rely on collection of this , albeit limited, survey data
- Methods have been proposed to use focal species and landscape species characteristics, which may be extracted from the literature, expert knowledge or from surveys to determine species landscape use characteristics (patch size, isolation) than can be used to assess and plan conservation networks directly by examining landscape structure rather than requiring detailed field collected species data.

- While focal species methods typically require some level of species data other methods, such as application of landscape ecology rules, and development of "generic focal species" for conservation planning simply use classified landscape use characteristics (patch size, isolation etc) that are generalised for particular species groups or are hypothetical in order to assess and plan networks by examining landscape structure without requiring field collected species data
- Benefits of using generic focal species, or landscape characteristics for planning are that they allow future landscapes or levels of species use to be examined that are not linked to current species presence / biodiversity levels: generic species profiles or extinct species characteristics can be used to enhance networks to encourage future biodiversity
- Much work remains to be done on linking broader biodiversity levels to either indicator species presence or landscape structure characteristics
- Habitat quality and within-patch diversity are key areas needing more research
- Examining deterministic patch biodiversity factors could potentially allow future biodiversity levels to be planned rather than just assessing current levels using indicator species
- Potential exists to use an intermediate method between the use of indicator species and the use of arbitrary landscape characteristics where the presence of multiple biodiversity indicators is linked to driving deterministic abiotic factors by assessing the link and relative importance between within-patch quality and landscape level / connectivity effects
- Key areas of investigation for many habitats are the extent to which broader landscape context and connectivity, or within-patch habitat quality features are most important in driving biodiversity levels

Applied landscape planning (GIS, ecological modelling and landscape metrics)

- GIS is a suitable format in which to undertake landscape ecology research and implement conservation planning
- Landscape metrics and ecological modelling are key tools with which to conduct landscape research within GIS
- Ecological modelling can be undertaken in GIS, using landscape metrics to derive links with biodiversity and habitat quality, of use in landscape scale conservation planning
- Applied conservation using GIS and metrics must carefully assess data source reliability and appropriate study scales
- The most useful landscape metrics are functional metrics such as connectivity and metrics less affected by changes in size or scale between patches
- In landscape studies a wide range of metrics should be collected and then assessed for redundancy due to likely high correlation levels
- If deterministic biodiversity or patch quality factors can be extracted for landscape planning that do not rely on measured species indicator presence, they will be useful in conservation due to reduced need for fieldwork surveys
- For the results of ecological models to be applied they should be justified and reliable by either being based on theory driven tests of relevant tested variables, or be confirmed by subsequent studies in additional test landscapes

Chapter 4

Habitat fragmentation: landscape structure, composition and ecological quality

A review of studies with an emphasis on woodland habitats

4.1 Introduction

This chapter reviews landscape spatial and structural factors, associated with habitat fragmentation. The review is conducted within the framework of conservation, and considers the factors that affect woodland site conservation, restoration or creation potential, within a wider woodland network. The chapter begins by examining the definition and effects of habitat fragmentation as indicated by landscape ecology and current theoretical and modelling studies. The chapter then reviews the methods employed to study the effects of habitat fragmentation, and then reviews past research, concentrating on woodland habitats. The major species groups examined are woodland birds and ground-flora due to their frequency within the literature. These groups are normally assessed by species richness, as the most frequently used measures of biodiversity within such research, although abundance and species / absence measure are also encountered within studies. The chapter ends by discussing key findings.

4.2 Habitat fragmentation: definition, causes and effects

4.2.1 Definition and description

Habitat fragmentation is a key concept within landscape ecology and is utilised to describe the multiple effects associated with habitats undergoing both a decline in area and increased isolation, for example as a result of anthropogenic landscape change (Andren, 1994, Saunders et al., 1991) (Fig 4.1). The process is complex, and is often examined using landscape metrics (Section 3.4.4) to link the effects of landscape structure to population dynamics. Separate, but related, factors interact in such studies. An initial set of factors are associated with the physical characteristics of a fragmented habitat; patch configuration, size and shape. A second set are associated with the ecology of the species. A third key element is the landscape matrix and how this relates to habitat patches and species ecology. Research examining habitat fragmentation, that aims to reduce its deleterious effects, allows application of landscape ecology theory through the practice of landscape architecture and planning (Collinge, 1996).

Table 4.1
Landscape characteristics studied in relation to habitat fragmentation.

Habitat Patch	Landscape	Population	Species ecology
Patch size	Total area of habitat	Population size	Colonisation distances
Patch shape	Patch configuration	Population distribution	Movement ability
Patch isolation	Matrix hostility	Population extinctions rates	Habitat preferences
	Matrix composition	Population colonisation rates	Species life history traits

In his review Andren (1994) showed that with progressive fragmentation of a habitat three principal changes occur: the area of habitat reduces, average patch size reduces and patches become more isolated. Two extreme examples of fragmented landscapes have been identified. A "coarse-grained landscape" or "geographical fragmentation" occurs where a focal habitat persists as dominant within the landscape but is broken into several patches (Farina, 1998) (Fig 4.2). Examples include extensive virgin forest fragmented by fire breaks or roads. In contrast a "fine grained" or "structured" landscape comprises an intimate mixture where the habitat has been split into many fragments which are scattered through the matrix and represent only a minor component of the landscape (Farina, 1998). Examples include small pockets of native woodland remaining within extensive conifer plantations or among arable farmland. It is however important to recognise that the same landscape may be perceived by one species as fine-grained and another as coarse-grained, depending on species perception and scale of landscape use and mobility. The study of fragmentation is therefore generally reduced to an examination of the interrelation between matrix and patches (Wiens, 1989).

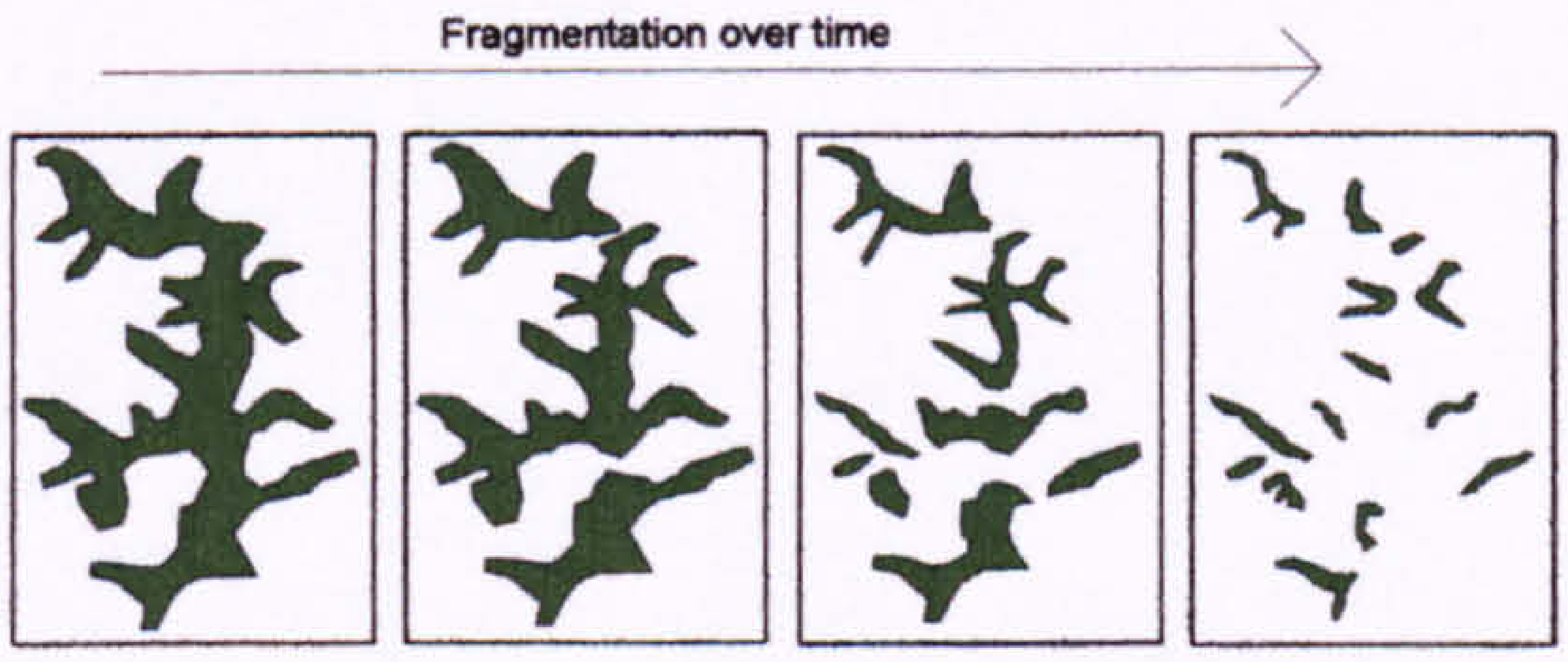


Figure 4.1
Fragmentation of a riparian woodland network. Number of fragment patches increases, patch size decreases and patches become more isolated as fragmentation progresses.

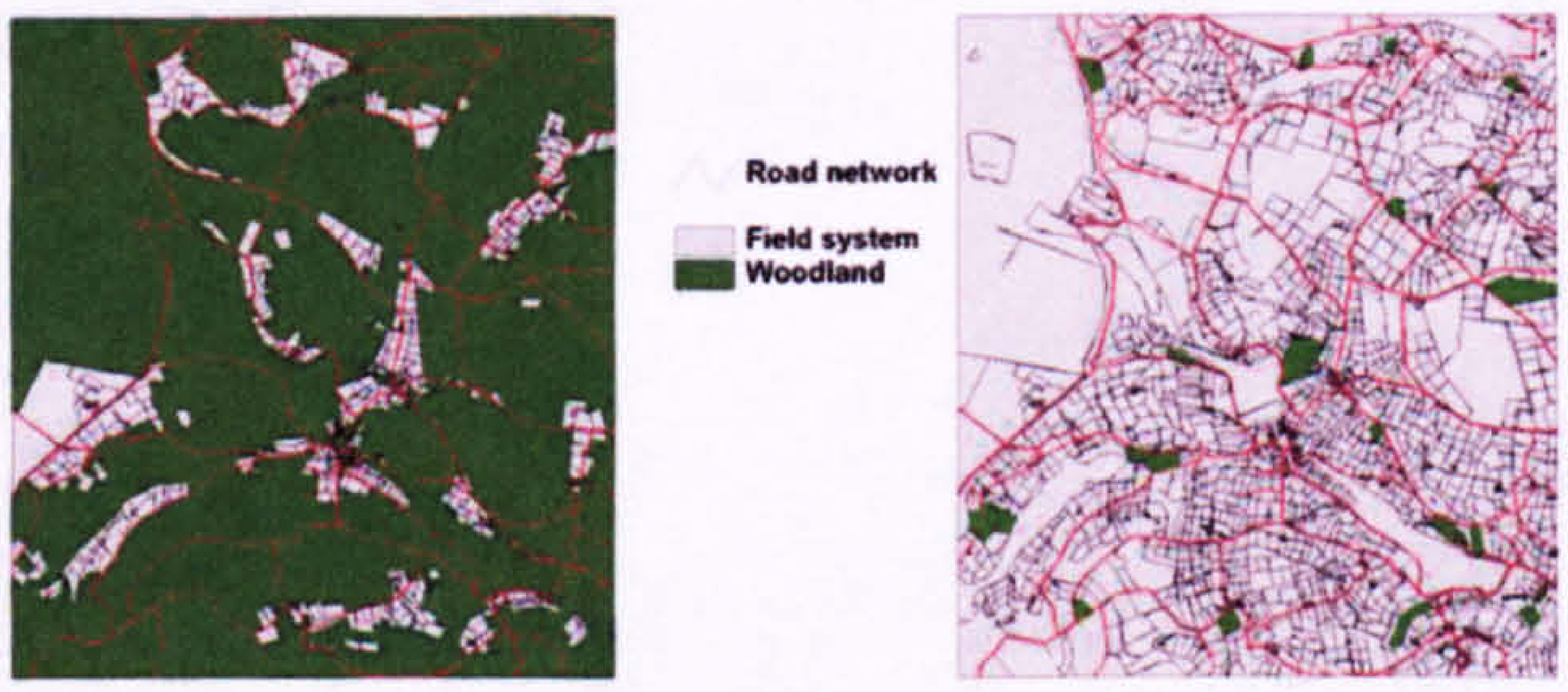


Figure 4.2
Coarse grained (left) and fine grained (right) examples of woodland fragmentation.

4.2.2 The study of habitat fragmentation

That habitat fragmentation is detrimental to conservation interests is generally accepted, with sufficient evidence existing for fragmentation being a cause in the decline of species for further habitat fragmentation to be opposed (Kirby, 1995, Heywood and Iriondo, 2003, Young et al., 1996, Terborgh, 1992, Saunders et al., 1991). A variety of research has attempted to quantify fragmentation effects. Analysis of current associations between species richness / occurrence and landscape structure can reveal where species are affected by the composition and configuration of their environment, and where habitat fragmentation may be having an impact. These "habitat fragmentation", "species-environment" or "landscape use" studies have used various approaches to ascertain the effects that current landscape structure and past or future levels of habitat fragmentation may have. These studies have taken three approaches to quantifying landscape structure effects: observational, experimental and modelling, each with benefits and limitations.

4.2.2.1 Observational studies

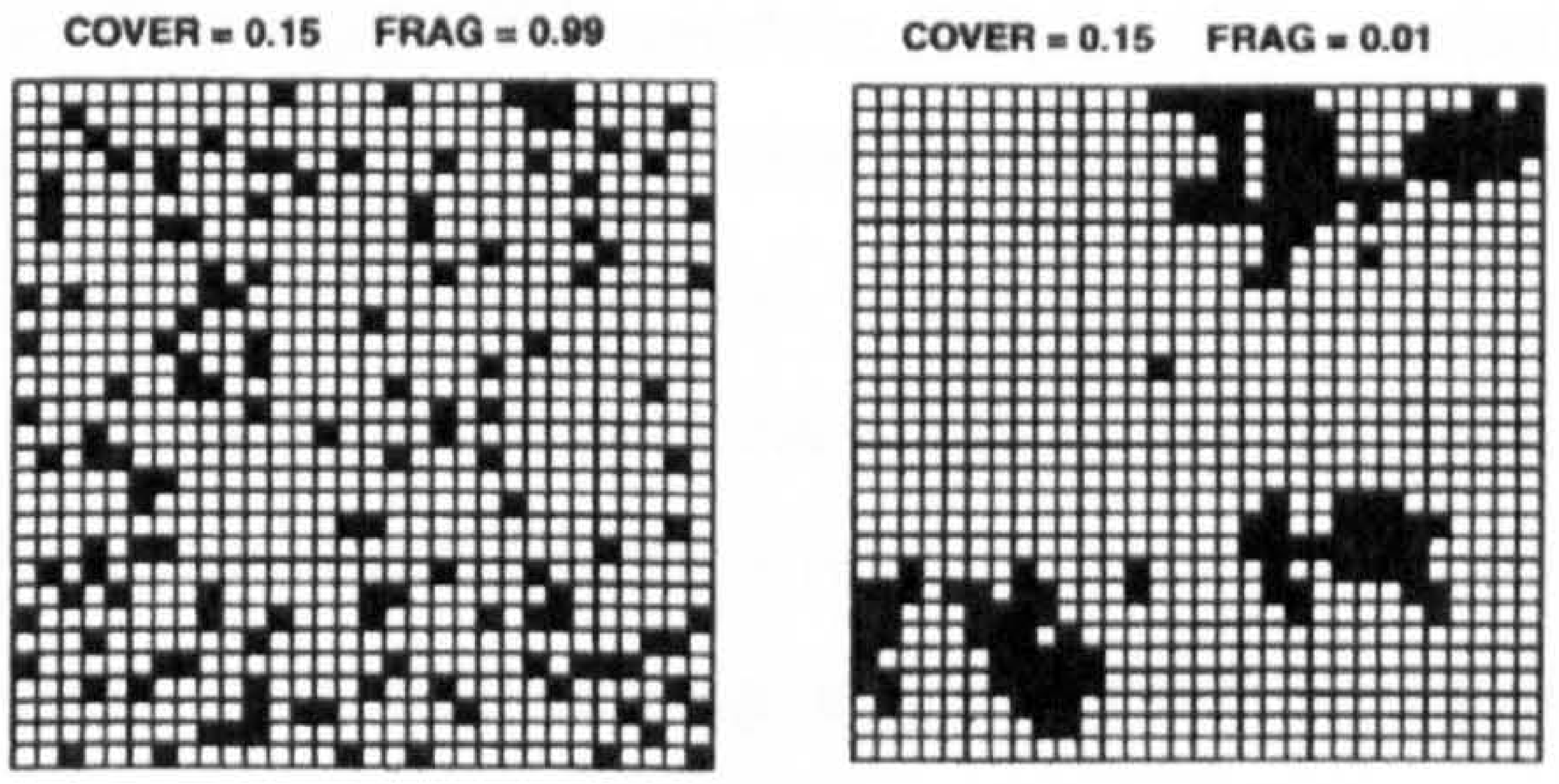
The largest body of work comprises observational studies, also referred to as empirical, or correlational studies, conducted within real landscapes where observation of species or habitat presence are linked statistically to landscape characteristics (Peterken and Game, 1984, Woolhouse, 1983, Opdam et al., 1985, Opdam et al., 1984, Woolhouse, 1987, Bellamy et al., 1996a, Trzcinski et al., 1999, Pharo et al., 2004, Andren, 1992, Radford et al., 2005). These may describe correlational relationships between landscape structure and species occurrence, from which deductions are made on potential causal mechanisms affecting species presence, and the effects that further habitat fragmentation may have. In order to compensate for the lack of experimental options, two approaches have been taken to isolate the effects of single variables. Research may compare a range of landscapes, holding similar values for one landscape feature, such as proportion of habitat cover, but differ across a range of values in other features. Alternatively statistical analysis may be used to control or eliminate one landscape element, such that other variables can be examined. Although observational studies provide an accurate description of species landscape use patterns, limitations include the time and cost of undertaking sufficient sampling to enable species occurrence / distribution to be linked to environmental /landscape variables, the difficulty of defining which measures of landscape form and structure are most likely to be drivers of species distributions, and the potential limitation of the results to the individual study landscape and species. Such limitations may be overcome by utilising species data collected for other purposes such as atlas studies, although these may have unknown accuracy levels and have scale limitations. Some observational research has resulted in low levels of explained variation. In such cases the variables recorded are assumed to have been inadequate to capture sufficient, relevant, ecological information defining the way the species perceives landscape structure, possibly due to inappropriate scales (Titeux et al., 2004), or may have ignored important effects such as variation in patch quality, predation or competition (Woolhouse, 1987).

4.2.2.2 Experimental studies

Several authors have pursued an experimental approach to more clearly investigate aspects of habitat fragmentation. These works often utilised "micro-landscapes" (Wiens and Milne, 1989, Wiens et al., 1993, Wiens et al., 1997, With, 1994) on which aspects of landscape structure such as patch size and isolation can be modified. These provide valuable insight into fragmentation, allowing theory to be directly examined without confounding variables. The results however may be limited by the small size (Hamazaki, 1996) and simplicity of many of the experimental species and landscapes examined, which are, in part, limited by the cost and practicality of undertaking such experiments at larger scales. The high degree of study flexibility in micro-landscape studies, in comparison to typical observational studies has been considered likely to accelerate more rigorous techniques in examining landscapes at other scales in landscape ecology (Wiens and Milne, 1989)

4.2.2.3 Modelling studies

The third key area of research involves modelling populations within artificial, computer generated, "neutral" landscapes (Saura and Martinez-Millan, 2000, Gustafson and Parker, 1992, Gardner and O'Neil, 1991, Gardner et al., 1987, Gardner et al., 1989, Fahrig, 1992, Doak et al., 1992, Fahrig, 1997, With et al., 1997, With, 1997, With and Crist, 1995). Neutral landscapes derive from percolation theory and allow investigations under conditions unaffected by ecological factors (e.g. competition, predation) or environmental variables (e.g. topography) (Gustafson and Parker, 1992, Gardner et al., 1987, With and King, 1997). Artificial landscapes are created using random landscape patterns (Andren, 1994) or may produce landscapes of varying degree of realism by being able to vary patch aggregation characteristics (With and King, 1999, Saura and Martinez-Millan, 2000). Landscapes are created within a grid, where adjacent squares of a habitat are considered to form a habitat patch (**Fig 4.3**). These are similar to "raster" data within GIS software and allow rapid generation of landscapes. Ecological modelling is used to examine species movement, dispersal, mortality and breeding success (**Fig 4.4**). Species dispersal characteristics, behaviour and landscape structure can be altered, and the effects examined on species distribution, abundance or dispersal success. Models may investigate single processes, such as dispersal, or more complex spatially explicit population models (SEPM) may examine features over time such as population growth and breeding success.



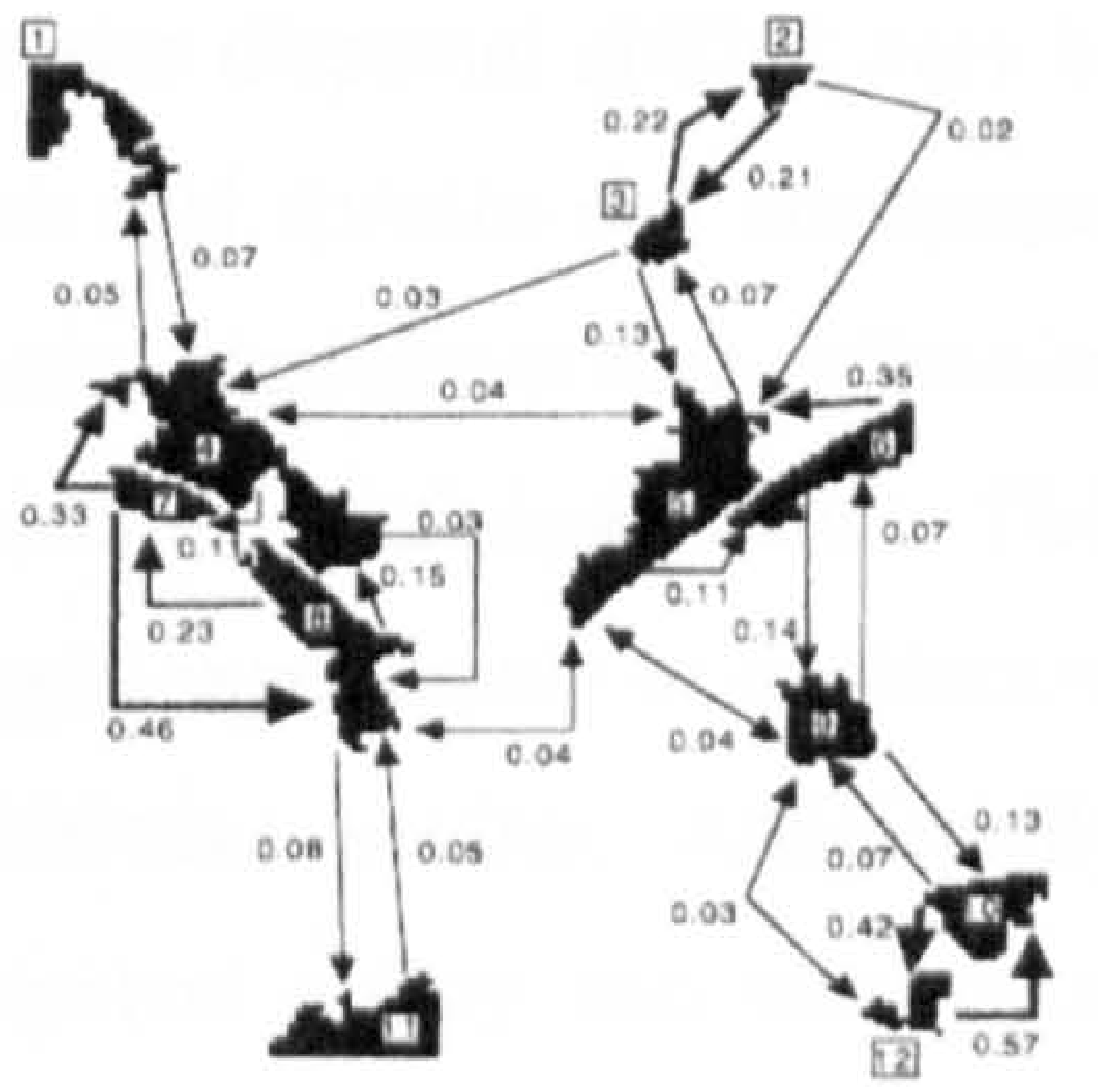


Figure 4.3
Artificial landscape modelling. Cover represents the proportion of potential breeding habitat (15%) (black cells), while FRAG indicates the extent of habitat fragmentation. Reproduced from (Fahrig, 1998)

Figure 4.4
Modelled landscape dispersal calculation of dispersal success between deciduous forest patches. Reproduced from (Gustafson and Gardner, 1996)

Modelling studies have examined various areas: the effects that proportion of suitable habitat (amount of remaining habitat following fragmentation) has on patch size and isolation by examining landscapes with different levels of cover in relation to predictions from percolation theory (Andren, 1994, Gardner and O'Neil, 1991, Gardner et al., 1987, Gustafson and Parker, 1992), the effects of landscape heterogeneity and landscape structure on species distribution patterns (With et al., 1997, With and Crist, 1995), the effects of different levels of habitat and matrix heterogeneity (Gustafson and Gardner, 1996) or habitat aggregation (With and King, 1999) on the successful dispersal of organisms between habitat patches, the effects of different landscape configurations (Fahrig, 1998) or patch isolation levels (Fahrig and Merriam, 1985) on survival of species / breeding populations, the effects of patch spatial arrangement / configuration on local population size (Fahrig and Paloheimo, 1988b) or population growth and spread (Sondgerath and Schroder, 2001), the effects of patch size and patch temporal lifespan on population size (Fahrig, 1992), the effects of different levels of patch aggregation on species distribution (With et al., 1997) and examination of the relative effects of habitat cover/loss or spatial fragmentation on population extinction (Fahrig, 1997, With and King, 1999). Use of neutral models has been particularly useful in the prediction of landscape response thresholds, identification of landscape connectivity levels, examination of species ability to perceive habitat patches, and examining the consequences of different levels of spatial heterogeneity or population subdivision for species populations (With and King, 1997).

Modelling has a number of strengths and limitations, and must strike a balance between ecological realism and model parsimony. The incorporation of aspects of species ecology and real landscape structure, although possible, may make results difficult to interpret, limit the insights to a certain species or landscape or make the functionality of the model complex. Conversely models that make too many assumptions and simplifications become unrealistic, again limiting the applicability of the results and their comparison to theoretical frameworks. The variables used within models, (habitat characteristics and species dispersal ability) may be hypothetical (Gardner et al., 1989) or based on observations from real species and landscapes e.g. (Fahrig and Merriam, 1985). Modelling research does not require significant data collection and can be varied and replicated many times to ensure reliable estimates of significance. This compares to the lower levels of replicates possible within real-world experiments or the limited observations and confounding effects found in landscape correlative studies. Additionally models may examine ranges of species behaviour, such as dispersal ability and perception range, that are not easily altered in experimental situations (Fahrig and Paloheimo, 1988a). However modelling studies have suffered from criticisms and limitations. Early application of random modelled landscapes have been criticized as not accurately representing real-world landscapes, where patches tend not to be truly randomly distributed, occurring more or less aggregated than expected (Gardner et al., 1987, Gustafson and Parker, 1992, Kirby and Thomas, 1994). Early analysis also tended to assume simplistic binary landscape representations: habitat patches within a hostile matrix e.g. (Fahrig, 1992, Gardner et al., 1989), and also utilised small grid landscapes. Many of these limitations have been addressed in recent models. Current models may focus not on single process such as dispersal success, but examine population dynamics in relation to landscape attributes, such as the effects of patch spatial arrangement on species population growth and distribution (Sondgerath and Schroder, 2001). New approaches have produced the creation of more realistic neutral landscapes (Saura and Martinez-Millan, 2000). Modelling studies may incorporate realistic landscape patterns derived from remote sensing data (Gustafson and Gardner, 1996) or model landscapes that are more complex than simple binary landscapes, mapping levels of relative patch habitat quality (Sondgerath and Schroder, 2001), or incorporating varying levels of multi-habitat and matrix use by modelled species (Wiegand et al., 2005). The results of these different approaches to the study of landscape usage and habitat fragmentation have been used to compare to predictions from landscape ecology theory and may be compared against a range of possible structure effects, as exemplified by the range of possible options for patch area and isolation effects (Fig 4.5).

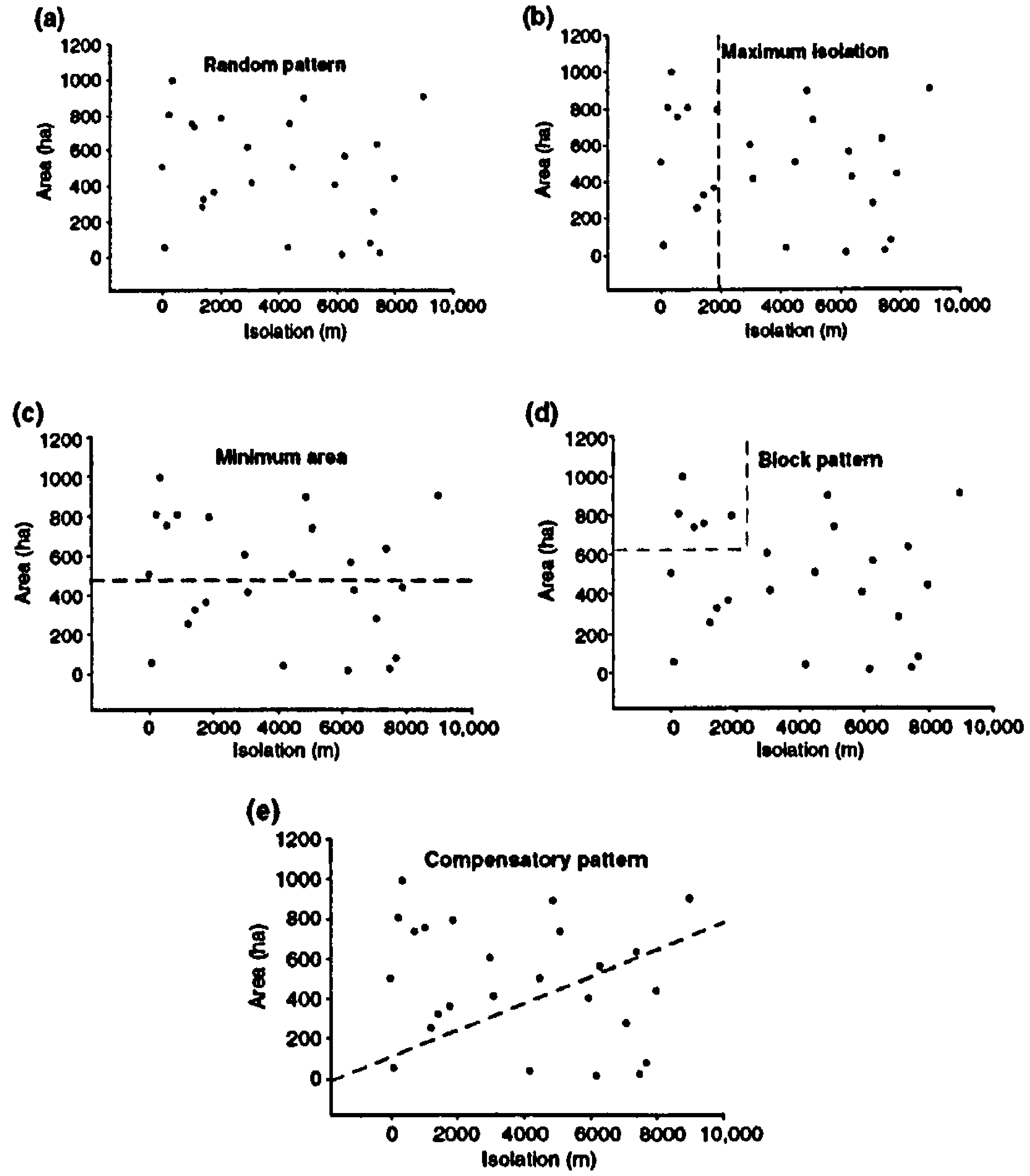


Figure 4.5
Possible effects of patch size and isolation variation on a species distribution pattern. Closed circles= occupied patches, Open circles=unoccupied patches. (a) random species distribution, unaffected by area or isolation, (b) dispersal limited species occurring only in patches below the threshold of isolation effects, (c) area sensitive species occurring only in patches above the area threshold, (d) block pattern caused by species that are both area and dispersal limited, (e) compensation pattern where species occur where patches may be occupied when isolated if they are sufficiently large, and small patches are occupied sufficiently close to occupied larger patches. Reproduced from (Watson et al., 2005) (after Lomolino, 1986:Lawlor, 1998).

4.2.2.4 Landscape size and scale

Scale must be explicitly considered with habitat fragmentation and related conservation issues (Doak et al., 1992): at one scale fragments may be of sufficient size for one species, but insufficient for others (Terborgh, 1992). Observational and experimental studies attempt to capture the appropriate scale to study species landscape use and deduce the effects of habitat fragmentation. Researchers have suggested that landscape definition should thus be species-centred (With, 1994), and have highlighted the importance of this in "multi-scale" approaches to examining landscape pattern and dynamics, avoiding the focus on human orientated scales and that allows landscapes structure to be examined in a way that is relevant to the organism under investigation (Wiens and Milne, 1989). Scale and grain effect data capture in compositional map; measurements made at different spatial scales, of the same area, may not be comparable (Turner et al., 1989). However fragmentation studies have been undertaken in landscapes of widely varying extent and grain and choosing the correct study scale may not be straightforward. Occasionally therefore, especially in multi-species studies, "landscapes" may not accurately reflect species perception. Modelling studies, comparing methods to define appropriate landscape study scale, have shown the difficulty of accurate definition of appropriate study scale (Cullinan and Thomas, 1992). Methods have been devised to study species movement behaviour at varying scales, for example using fractal dimension in order to deduce the appropriate scale of study (With, 1994) and measures of landscape structure can be integrated across scales (Wiens and Milne, 1989). The lack of explanatory power achieved within some landscape research has been attributed to studies potentially having been carried out at inappropriate scales to the relevant landscape process (Titeux et al., 2004). The wide range of scales examined within landscape studies means that a key goal in allowing increased comparison between different studies and regions and scales is the prediction of how ecological variables alter with changes in scale (Turner et al., 1989). Modelling studies indicate that pattern–process relationships can show varying reactions to changes in study scale, from being unaffected to showing significantly different effects (Gardner et al., 1989). Past research has recommended that work should be carried out in two stages, with initial preliminary studies being conducted to define the appropriate study scale (Cullinan and Thomas, 1992). However this may not always be practical, and exact definitions of appropriate scales may not be possible when many species are being investigated. An alternative to examining systems at a variety of scales, is to thresholds at which scale changes occur; within these ranges extrapolations across scales will be acceptable, but not beyond (Gardner et al., 1989). By carrying out studies across a range of scales these "domains" of scale which apply to certain processes or patterns may become apparent, and could be missed by analysis at a single scale (Wiens and Milne, 1989). A critical aspect to the interpretation of landscape ecological studies is therefore a statement of the grain and extent of data used (Turner et al., 1989).

In considering these issues it is useful to distinguish between "patch-level" and "landscape-level" studies. "Patch-level" studies examine patches within a single "landscape" where the occurrence or abundance of species are related to features of that patch such as its size, or its isolation from neighbouring patches (Hinsley et al., 1996, van Dorp and Opdam, 1987). Individual patches are the basic unit of study within the broader study area or "landscape". The study landscape is defined from broad geo-botanical factors and therefore landscapes tend to be defined at human-centred scales. Species perception and dispersal ability is likely to extend across several patches, not the entire study landscape, and this may be measured within local ecological neighbourhoods (Bellamy et al., 1996a, Hinsley et al., 1994). "Landscape-level" studies take the landscape as the unit of study and aim to compare processes, species distributions or abundances between "landscapes", relating species / populations to overall landscape structure rather than patch features. Data is collated for each landscape, e.g. mean patch size, shape, number of patches and habitat cover and are related to processes or species occurrence. These "landscapes" may occur in a continuous grid within a broader study area or region, or they may occur in separate discontinuous areas selected in order to represent landscape types. The extent of these landscapes may reflect typical dispersal or movement distance of mobile species (Villard et al., 1999) in which cases these are species-orientated landscapes. Or sizes may be determined by the past collation of data on a regular grid within distribution atlas schemes e.g. (Trzcinski et al., 1999, Titeux et al., 2004, Mouflis and Buckley, 2004).

While many studies show a clear distinction between patch and landscape studies this is blurred where complex data on habitat connectivity or isolation, beyond simple nearest neighbour value are collected. Data are frequently recorded within "ecological neighbourhoods" or "search distances" around focal patches in an attempt to define attributes of the local environment that may affect processes / species within the focal patch. Such measures may give an estimate of local landscape connectivity around the focal patch, which is considered to be more important than the value for the whole landscape due to the whole study landscape occurring at an extent too large to apply to the study species (Bellamy et al., 1996a, Bennett and Radford, 2004, Hinsley et al., 1994). Within single-species research distances are related to typical species movement. However studies examining species richness have two options in the use of search distances. A wide range of distances may be utilised and subsequently examined statistically to define which "ecological neighbourhood" distance is most related to occurrence (a posterior selection) or previous evidence from the literature on species dispersal (e.g. woodland birds) may be used to justify the collection of data within a single search distance (a priori selection). Examples include the use of 0.5km and 1km distances in lowland England, noted as being typical of the likely movement range of woodland resident breeding bird species during the breeding season (Bellamy et al., 1996a). Such variables (distance to nearest woodland with

1km, number of woodlands within 1km, area of woodland within 1km) therefore define species-orientated "landscapes" similar to the data collected in some "landscape level" studies. However key differences in the use of search distances in patch studies is that separate zones are calculated around each patch and thus local landscape covers and configurations remain linked to individual patch occurrence. The neighbourhoods overlap when calculated for different, patches while in landscape studies the variables are recorded from completely distinct, although possibly contiguous landscapes. The use of focal habitat search distances provides an opportunity to carry out multi-scale studies, as recommended by Wiens and Milne (1989) to accurately reflect species perception of their environment. Different distances may then correspond to individual species or to different species groups.

4.2.3 Ecological effects of fragmentation: predictions from models and experiments

4.2.3.1 Theoretical predictions

Reviews of habitat fragmentation have noted the importance of landscape spatial structure in understanding the impact on population survival (Fahrig and Merriam, 1994). Principal theories cited as providing insight are those of island biogeography theory and metapopulation biology (MacArthur and Wilson, 1967, Hanski, 1999, Hanski and Simberloff, 1997) (Section 3.2.2). Island biogeography theory predicts that larger, less isolated and older patches will hold higher species richness. Metapopulation theory predicts populations occurring among patches that have a higher density, with low isolation will be more robust to extinctions and therefore show longer persistence times, larger numbers of patches may also favour population persistence (Hanski and Simberloff, 1997, Hanski, 1999). Key theoretical predictions are the positive effects of patch size and the negative effects of isolation on species richness and population persistence. Therefore dispersal ability is critical to the functioning of these theories (Wiens, 1997). Fragmentation also leads to detrimental genetic effects in populations, including loss of genetic variation and increased divergence between isolated populations, caused by genetic drift, inbreeding and reduced gene flow (Young et al., 1996).

Several authors however have noted limitations to these dominant theories of fragmentation, particularly with respect to the effect that variation in the quality of habitat patches and the favourability of the habitat matrix may play in effecting populations, leading to a number of alternative suggested population structures such as mainland-island, patchy populations and source-sinks which may affect the predictions of the core theories (Pulliam, 1988, Watkinson and Sutherland, 1995, Harrison and Taylor, 1997). In particular recent discussions have contested the equilibrium status of patches, highlighting dynamic, and stochastic events and noting the non-equilibrium status of community structure (Hubbell, 2000).

Theoretical predictions approaches have been assessed by a number of experimental and modelling studies. Principal areas investigated have been the effects of patch size and isolation, matrix composition and landscape thresholds. Additional species variables examined include different life history attributes such as dispersal success, longevity and relative habitat specialism.

4.2.3.2 Experiments and ecological models

Patch size and isolation Modelling work confirmed the positive effects of patch size and negative effects of patch isolation on population size and also that patch temporal existence may be highly important in comparison to such spatial arrangements when length of habitat occurrence varies between patches or landscapes (Fahrig, 1992). A modelling study in forest patches in America found that isolated populations of white-footed mice were more likely to become extinct and took longer to be re-colonised than less isolated patches, predictions confirmed by subsequent field work (Fahrig and Merriam, 1985). Modelling has also shown that clustering of patches, particularly the scale at which clustering occurs, are important for dispersal success (Doak et al., 1992) and that habitat aggregation can strongly affect species landscapes connectivity (With et al., 1997), with aggregated patches enhancing dispersal success (With and King, 1999), patch occupancy (Verboom, 1991) and population persistence (Adler and Nurnberger, 1994).

Modelling has revealed that key factors in determining patch population size are the probability of dispersing individuals being able to detect patches in the landscape, the levels of emigration from patches, and principally that the dispersal distance of organisms affects whether spatial arrangement of patches has any effect on local population size (Fahrig and Paloheimo, 1988a, Fahrig and Paloheimo, 1988b). Habitat connectivity is thus affected by organism dispersal ability (Sondgerath and Schroder, 2001), if the average dispersal distance is low compared to the average inter-patch distance, then the spatial arrangement of patches may be particularly important (Fahrig and Paloheimo, 1988a).

Studies have shown the importance of patch size increasing population persistence in individual species e.g. in *Sitta europea* (nuthatch) (Verboom, 1991), patch size being considered to affect patch quality and increase potential colonisation / dispersal events. In their review of habitat fragmentation Fahrig and Merriam reported extinction events from several species populations, often at levels of 10-20% of the population (Fahrig and Merriam, 1994), highlighting the potential for meta-population effects to apply and for extinction and re-colonisation events to be linked to patch size and isolation, determining species distributions (Fig 4.6).

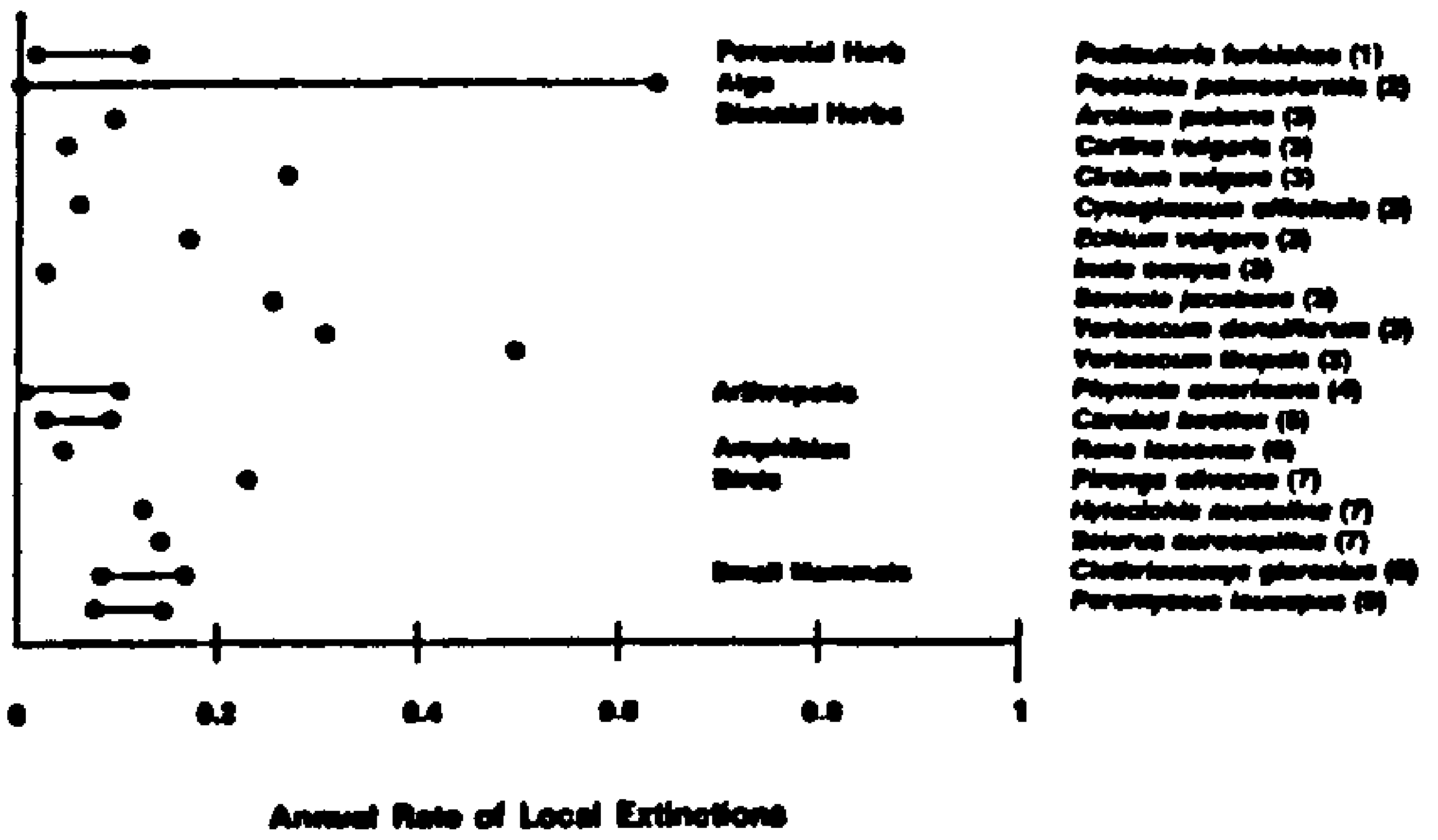


Figure 4.6
Rate of local extinctions per year. Reproduced from (Fahrig and Merriam, 1994), in which individual numbered references are listed.

Patch shape and edge-effects Rather fewer experiments or models have examined patch shape or edge-effects. An experiment using invertebrates supported theoretical predictions that patch shape will influence species mobility, showing that elongated patches support higher numbers of mobile species, providing larger contact areas to dispersing individuals, but also that the elongated patches showed higher temporal and spatial variation among individuals and patches than less elongated patches, suggesting they perhaps experienced more detrimental edge-effects in such patches (Hamazaki, 1996).

Habitat cover and thresholds Examination of patches within neutral landscape models have indicated that important threshold changes occur in patch number and isolation near the percolation threshold (Gardner et al., 1987, Andren, 1994) (Fig 4.7). Andren compiled model landscapes showing different proportions of habitat fragmentation, and found the dominant habitat remained as one continuous patch until 60% of original habitat remained, as predicted by percolation theory (Andren, 1994). It was not until less than 20% of the original habitat remained that increased effects of isolation began to occur, where further habitat reduction caused an exponential increase in the distance between remaining habitat patches (Andren, 1994). There is no simple linear relationship between cover of original habitat and patch size and isolation (Andren, 1994). In landscapes with high levels of habitat, detrimental fragmentation effects will be linked principally to loss of habitat area, while in landscapes with low covers of habitat the effects of patch size and isolation on species landscape use will become increasingly dominant. Examination of real landscape patterns showed these were more clustered in comparison with model landscapes and therefore showed a lower percolation threshold than expected from random patch distribution, patch size decreasing and patch number increasing exponentially at lower covers of overall habitat than expected from percolation theory (Gardner et al., 1987). A microlandscape experimental study examining grassland beetles found evidence for habitat cover thresholds affecting species behaviour, noting rapid changes in dispersal behaviour when habitat cover dropped below 20%, noting this was lower than expected from percolation models due to the way the beetles were able to easily move between both matrix and habitat patches in contrast to model assumptions (Wiens et al., 1997). "Critical thresholds" values therefore are not determined exclusively from landscape structure but result from the interaction of species perception and landscape structure (With and Crist, 1995, With, 1997).

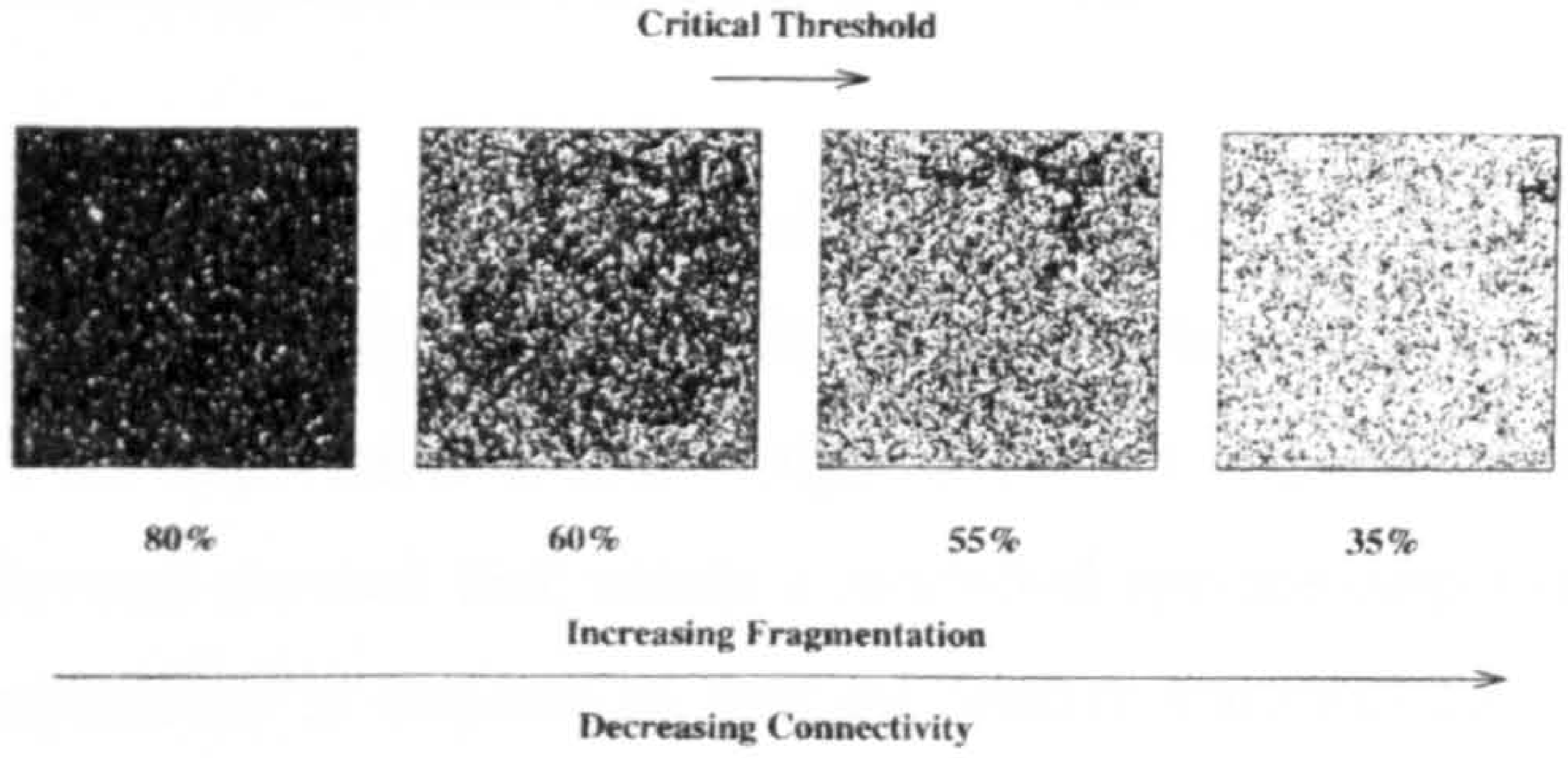


Figure 4.7
As the proportion of suitable habitat reduces in cover, connectivity reduces and patches become increasingly isolated. With increased isolation critical thresholds may occur in either landscape structure in neutral landscapes (as patches become increasingly isolated, patch size reduces and number of patches increases), or in the responses of species to landscape structure. Reproduced from (With, 1997).

Life history and population demography A study by Sondgerath and Schroder showed the importance of landscape structure varied depending on population demographic characteristics; the spatial scale at which fragmentation was perceived depended on population demography (Sondgerath and Schroder, 2001). Modelling research comparing factors including species reproductive traits showed that species with low reproductive potential may be particularly badly effected by habitat fragmentation due to effects on dispersal success (With and King, 1999). Models investigating life history effects have indicated that specialist species with relatively long distance dispersal will show aggregated distributions in landscapes where their habitat forms a minor component of the area in comparison to generalist species able to utilise additional habitats (With and Crist, 1995). Modelling in dynamic landscapes has shown that increased dispersal distance increased metapopulation persistence, although this was shown to be less of a clear advantage in landscapes with clumped patch distributions (Johst et al., 2002). Modelling studies have also shown the ability to utilise habitats within the landscape matrix can increase population size (Wiegand et al., 2005).

Relative importance of habitat fragmentation and application of predictions Several modelling and experimental studies have examined the relative effects of different aspects of habitat fragmentation in order to deduce important factors. When a range of landscape covers are examined several studies have indicated that the dominant effects on species are related to overall habitat cover or habitat loss rather than spatial fragmentation effects (Fahrig, 2001, Fahrig, 1997, Wiegand et al., 2005). Authors have therefore noted that habitat "patchiness" i.e. fragmentation does not always affect a species (With, 1997). Examining the effects of differing proportions of three habitats of differing quality, for species with nine ecological profiles, habitat cover accounted for 68% of the variation in population size with only 13% accounted for by fragmentation and 13% by species profile (Wiegand et al., 2005). However Wiegand et al (2005) noted this apparent dominance of habitat cover was significantly reduced when a range of more realistic landscapes were examined – i.e. lower amounts of primary habitat. Previous modelling has shown that the effects of habitat configuration were most important in landscapes with low habitat covers (Sondgerath and Schroder, 2001, With et al., 1997).

The relevant importance of habitat fragmentation has been linked to species dispersal ability and landscape connectivity. Comparisons of modelled landscape effects to direct experiments of the same species indicated the importance of knowledge of species dispersal behaviour (Fahrig and Paloheimo, 1988b). Several showed that where a modelled species dispersal ability was high and there was sufficient habitat to disperse to, that the relative fragmentation of the habitat may have no effect, all patches were effectively "functionally connected" (Wiegand et al., 2005). In modelled organisms the majority (89%) of variability in dispersal success in landscapes with different levels of habitat heterogeneity in the matrix, and thus different levels of patch boundary crossing behaviour, could still be accounted for by patch size and isolation (Gustafson and Gardner, 1996). Situations may clearly exist where the relative fragmentation of habitat is unimportant so long as habitat is present in sufficient amounts.

The extent to which these results of modelling and experimental studies are able to be generalised is unclear. Modelling has shown that the relative importance of landscape structure and the spatial scale at which species perceive habitat fragmentation varied with species population characteristics (Sondgerath and Schroder, 2001). Modelling also noted that to accurately predict the effects of habitat fragmentation requires sufficient understanding of species population biology, dispersal and habitat use characteristics (Wiegand et al., 2005). Ecological models will only be able to accurately reflect the impact of habitat cover and fragmentation on population dynamics by incorporating more "biological realism" within models (Wiegand et al., 2005). To account for these limitations models have recently broken away from the binary concept of structure and have examined landscapes with multiple habitat types e.g. (Wiegand et al., 2005) and varying levels of habitat quality (Sondgerath and Schroder, 2001, Johst et al., 2002), rather than earlier studies that assumed all patches were of similar habitat quality (Adler and Nurnberger, 1994). This has enabled the effects of multiple habitats and "poor quality" habitat to be modelled. In addition to the primary habitat (comprising the breeding or key habitat of the species), poor quality habitat may comprise sink patches or may act as "dispersal habitat" aiding species movement but not representing breeding habitat (Wiegand et al., 2005). The incorporation of such features makes models more realistic. In meta-populations or spatially structured populations, where species with moderate dispersal abilities occur in fragmented patches, the occurrence of "dispersal habitat" can be important in aiding periodic exchange of individuals between patches (Wiegand et al., 2005). Wiegand et al therefore recommend that the occurrence of dispersal habitat is incorporated in future metapopulation models.

4.3 Landscape structure, fragmentation and habitat quality: A review with an emphasis on woodland habitats

4.3.1 Introduction

The literature examining landscape scale species-habitat relationships and the effects of habitat fragmentation on species ecology is extensive. This section reviews observational studies that have investigated fragmentation effects on landscape and habitat patch quality or the occurrence of species within fragmented landscapes, with particular reference to woodland. In considering research several criteria allow the relevance of different effects to be assessed: a) the period over which fragmentation has occurred: are species still responding to recent fragmentation or has a habitat been fragmented within a landscape for a long period of time? b) the extent to which species colonisations and extinctions have actually been observed in fragmented landscapes? c) in what form plant and animal populations occur, are they present as remnant populations, in metapopulations or as source and sinks?

Literature exists from a range of habitats, from grasslands: (Cook et al., 2004, Fischer and Stocklin, 1997, Collinge, 2000), wetlands: (Knutson et al., 1999, Wettstein and Schmid, 1999) to woodlands (Peterken and Game, 1984, Peterken and Francis, 1999, Usher and al, 1992, Trzcinski et al., 1999, Jacquemyn et al., 2001, Radford et al., 2005, Jacquemyn et al., 2003). Woodland fragmentation / landscape-use studies exist from many geographical areas including the tropics: (Turner and Corlett, 1996, Terborgh, 1992, Gascon et al., 1999), temperate woods: (Wilcove et al., 1986, Freemark and Merriam, 1986, Opdam et al., 1985, Woolhouse, 1987, Bellamy et al., 1996a, Helliwell, 1976, Jacquemyn et al., 2001, Bennett et al., 2004, Jacquemyn et al., 2002, Jacquemyn et al., 2003) and boreal woodland: (Virkkala et al., 2004, Gu et al., 2002, Norton et al., 2000, Moen and Jonsson, 2003, Lobel et al., 2006, Enoksson et al., 1995, Lescourret and Genard, 1994). Woodland studies have examined fragmentation or landscape structure effects on a variety of species groups including birds: (Redpath, 1995, Andren, 1994, Gaston and Blackburn, 2002, Sisk et al., 1997, Hinsley et al., 1994, Opdam et al., 1985, Opdam et al., 1984, Woolhouse, 1987, Bellamy et al., 1996a, Helliwell, 1976, Trzcinski et al., 1999, Norton et al., 2000, Andren, 1992, Enoksson et al., 1995, Bennett et al., 2004, Radford et al., 2005, Lescourret and Genard, 1994, Hinsley et al., 1996, McIntyre, 1995), mammals: (Lidicker, 1999, Cook et al., 2004, Walsh and Harris, 1996, Fitzgibbon, 1997, Pardini et al., 2005), invertebrates: (Collinge, 2000, Didham and al, 1996, Webb, 1989, Allen et al., 2004, Komonen et al., 2000), vascular plants: (Helliwell, 1976, Jacquemyn et al., 2001, Kollmann and Schneider, 1999, Jacquemyn et al., 2002, Jacquemyn et al., 2003, Bastin and Thomas, 1999), and bryophytes: (Pharo et al., 2004, Lobel et al., Moen and Jonsson, 2003). Studies have examined wooded landscapes that have undergone fragmentation over different time periods from recent centuries / decades: (Laurance, 1994, Terborgh, 1992, Watson et al., 2005, Pharo et al., 2004, Radford et al., 2005, Gascon et al., 1999) to long fragmented European landscapes: (Peterken, 1981, Peterken and Francis, 1999, Peterken and Game, 1984, Bellamy et al., 1996a, Moen and Jonsson, 2003). Despite this extensive spread of research a significant proportion of work examined birds within broadleaved woodland patches. This is due to the high contrast and ease of distinction between woodland patches and matrix when examining habitat and remote sensing data, and the high levels of knowledge and data on bird ecology / distribution. A secondary, but significant, wealth of studies examined associations between woodland landscape characteristics and the diversity and composition of the ground-flora layer.

While some experimental fragmentation studies have been conducted within real landscapes (Summerville and Crist, 2001, Cook et al., 2005, Yao et al., 1999), rather than on artificial model landscapes, research typically relies on interpretation of "natural" or unplanned field experiments when opportunities arise to study landscape change following anthropogenic activities (such as major logging or construction projects), or through interpretation of current static relationships between current landscape structure and species distributions. Work on progressive habitat destruction may monitor habitat islands within which species changes are observed due to species extinctions and community changes related to individual patch size or isolation, many such examples coming from the recent fragmentation of tropical rainforests (Burkey, 1993, Laurance, 1994, Turner and Corlett, 1996, Gascon et al., 1999, Terborgh, 1992, Laurance et al., 2002). In contrast some studies have provided insight into fragmentation effects by examining landscapes undergoing colonisation, where the date of creation of woodland patches are known and developing species presence and community richness in patches of known age can be related to colonisation events that are, in turn, related to patch size and isolation. Woodlands in such work may arise through planting or natural colonisation and expansion following old-field abandonment, connected with agricultural decline in America and mainland Europe. Several studies have highlighted such effects on patch colonisation by woodland flora species in the UK (Usher and al, 1992, Peterken and Game, 1984, Peterken and Francis, 1999), North America (Matlack, 1994, Bellemare et al., 2002) and in Europe (Bossuyt et al., 1999, Grashof-Bokdam, 1997, Verheyen et al., 2003, Jacquemyn et al., 2001, Honnay et al., 1999d). Other research has assumed more that as equilibrium exists between current patterns of species occurrence and underlying landscape structure, particularly when examining more rapidly mobile species. This is an approach taken by many of the studies examining the relationship between woodland landscape patch characteristics, configuration and bird species presence where patch age, beyond measures of woodland canopy maturity, may not be considered as relevant, e.g. in UK: (Hinsley et al., 1994, Bellamy and Hinsley, 2004, McCollin, 1993, Woolhouse, 1987, Bellamy et al., 1996a, Helliwell, 1976, Redpath, 1995, Bennett et al., 2004), Europe: (Geerstema et al., 2004, van Dorp and Opdam, 1987, Enoksson et al., 1995, Opdam et al., 1985, Opdam et al., 1984), North America: (Villard et al., 1999, Boecklen, 1986, Freemark and Merriam, 1986, Trzcinski et al., 1999, McIntyre, 1995) and Australia: (Bennett and Radford, 2004, Watson et al., 2005).

The majority of works have examined fragmentation on woodland fragments within an agricultural matrix. Additionally several papers have examined effects on particular woodland types or growth stages within a forested matrix (Enoksson et al., 1995, McGarigal and McComb, 1995, Norton et al., 2000, Wickham and al, 1999) and in upland montane forest habitats (Lescourret and Genard, 1994). Woodland fragmentation effects have also been found in urban (Godefroid and Koedam, 2003, Bastin and Thomas, 1999) or peri-urban (Watson et al., 2005) landscapes, or have assessed the impact of different matrix types on fragmentation effects (Watson et al., 2005, Sisk et al., 1997). At a larger scale several studies have examined the relative effects of fragmentation when studied across climatic and environmental gradients across local regions (Bennett et al., 2004) and the influence of geographical location on effects of woodland fragmentation (Telleria et al., 2003).

This extensive literature on the effects woodland fragmentation has examined an array of effects beyond those from simple patch size and isolation. These studies are reviewed within the following sub-sections: landscape scale and extent, area of suitable habitat, patch size, core area, edge-effects and patch shape complexity, patch frequency, patch isolation, connectedness and connectivity, patch contrast, landscape matrix and habitat and landscape diversity and evenness measures and habitat corridors.

4.3.2 Scale and extent

Research on woodland fragmentation and species landscape use has been conducted across a range of scales, extents and with differing data resolution and intensity. Study areas and landscapes within patch-levels studies range from 20-50km^2 (Helliwell, 1976, Jacquemyn et al., 2001, Jacquemyn et al., 2002), to 200-500km^2 (Opdam et al., 1985, Redpath, 1995) and 1,000-3,000 km^2 (Bellamy et al., 1996a, Freemark and Merriam, 1986, Enoksson et al., 1995). Patch-level studies have ranged from limited data examining only 10-21 patches (Freemark and Merriam, 1986, McCollin, 1993, McIntyre, 1995, Woolhouse, 1987) to studies examining over 150-250 patches, in order to confirm patch and landscape effects (van Dorp and Opdam, 1987, Bellamy et al., 1996a, Hinsley et al., 1994, Jacquemyn et al., 2001, Jacquemyn et al., 2002, Peterken and Francis, 1999). Landscape-scale study resolutions range from 1km^2 grids (Titeux et al., 2004) to 100km^2 grid resolution (Trzcinski et al., 1999) and include studies comparing only 20-30 different landscapes (Radford et al., 2005, Villard et al., 1999), to work examining several hundred (Titeux et al., 2004) or over a thousand comparable landscapes (Mouflis and Buckley, 2004) in order to examine species landscape relationships.

A range of patch-scale research have also recorded a range of data within focal or ecological neighbourhoods around patches, effectively capturing measures of the local species "landscape" around focal patches. These studies allow information calculated at these local landscape scales to be matched to species occurrence. The extent of these landscapes around patches in woodland studies ranges from 0.5km to 5km radius, resulting in 78-7,850ha being measured (Table 4.2). Within studies examining bird species occurrence distance of 1km have been most typical. This is considered representative of short-distance dispersal / foraging, marking the extent of landscape use around a focal patch, or representing a territory around which foraging journeys occur.

Table 4.2
"Patch level" studies search distances used within studies of woodland birds, to calculate the number, area or % cover of woods occurring within the search distance.

0 5km	1km	2km	3km	4km	5km
0.78km^2 / 78ha	3.1km^2 / 310ha	12.5km^2 / 1,250ha	28.3km^2 / 2,830ha	50.2km^2 / 5,020ha	78.5km^2 / 7,850ha
(Bellamy et al., 1996a) (Hinsley et al., 1994) (Redpath, 1995)	(Bellamy et al., 1996a) (Bellamy et al., 2003) (Helliwell, 1976) (Hinsley et al., 1994) (Enoksson et al., 1995) (Bennett et al , 2004)	(Enoksson et al., 1995) (McCollin, 1993)	(Opdam et al., 1985) (Opdam et al., 1984) (Enoksson et al., 1995)	(Helliwell, 1976) (Enoksson et al., 1995)	(Opdam et al., 1984)

4.3.3 Landscape thresholds and proportion of woodland cover: when is fragmentation important in considering species distributions and abundances?

One quantification of fragmentation is the extent to which a particular habitat remains as a feature of the landscape, defined by proportion of cover. Total area of suitable habitat can have significant effects on the abundance, distribution and dispersal success of species and habitat conservation strategies can be expected to differ between regions where the landscape contains a high cover of a habitat to regions where it is scarce. Cover tends to be associated with other features, such as size and isolation. Landscape theories suggest there will be a clear and often linear link between habitat area (or habitat loss) and species population size: the random sample theory (Conor and McCoy, 1979). Thresholds may apply, beyond which habitat loss is associated with isolation and patch size reduction, where the total area of habitat may be insufficient to support a metapopulation or population. Threshold effects have been observed in landscapes with low covers of habitat (Andren, 1994). Andren reviewed a range of studies from real landscapes and found the results confirmed his model predictions. In landscapes with a high proportion of "total suitable habitat" species abundance/density could mostly be explained by the total area of habitat, while in landscapes with highly dispersed / fragmented habitats, with a low proportion of total suitable habitat, most studies found an effect of patch area and/or isolation (Andren, 1994). It was suggested the threshold below which patch size and isolation became increasingly important was between 10% - 30% of remaining habitat, for mammals and birds (Andren, 1994).

A range of woodland studies have examined habitat cover and found relationships with species richness. Positive associations are typically observed between proportion of forest and bird species presence or richness; in North America: (Villard et al., 1999, Trzcinski et al., 1999), Europe: (Geerstema et al., 2004) and Australia: (Bennett and Radford, 2004, Radford et al., 2005). Studies in England have also shown correlations between the richness of ancient woodland indicator flora and the proportion of ancient woodland habitat in the landscape (Mouflis and Buckley, 2004). Landscapes with higher covers of habitat hold more diverse communities. Several papers also report observed landscape threshold effects. Within $100km^2$ and $270km^2$ "landscapes" in Australia showing a wide range of woodland cover (<2% to 60%), analysis found a steep decline in woodland bird species richness where woodland cover reduced below 10% (Bennett and Radford, 2004, Radford et al., 2005). Where landscapes hold higher covers of woodland habitat, and especially where species are mobile patch shape or habitat aggregation are not important to species richness levels (Radford et al., 2005). Observational research thus confirms modelling predictions that habitat fragmentation will have increasingly detrimental effects when habitat cover in the landscape is low. However landscape size and scale, and species dispersal behaviour in such studies, affects their interpretation. Typically within landscape-level comparison studies habitat spatial occurrence data and species occurrence or abundance data are aggregated to the landscape unit as average measures, rather than collected in representative samples. This may affect results in that diversity and variation at lower scales within these landscapes may be lost when amalgamated to the broader landscape scale. Large landscape units of $100km^2$ are above the typical movement distances of woodland bird species and may mask the way species react at finer scales. However work examining landscapes from $6km^2$ to $100km^2$ found effects attributable to proportion of woodland cover, although one landscape study undertaken at a $1km^2$ contained a high degree (66%) of unexplained variation, potentially because the variables recorded within these landscapes did not relate well to the scale of use of bird species (Titeux et al., 2004).

Original predictions for habitat cover threshold effects came from random neutral models of landscape cover (Andren, 1994, Gardner et al., 1987). Real-world landscapes hold habitats that are more clustered, as seen with English ancient woods (Kirby and Thomas, 1994). Therefore real landscapes are expected to show a lower percolation threshold (Gardner et al., 1989). The woodland studies discussed and the general studies reviewed by Andren (1994) support the view that species richness will decline, and habitat fragmentation configuration variables will increase in importance in landscapes with less than 10-20% habitat cover (Bennett and Radford, 2004, Radford et al., 2005). In real landscapes, where patches are clustered, effects below 10%

may be more typical. Local populations may be sustained where aggregations of patches occur. However the constraints of study scale, in particular the scale at which landscape cover is calculated, and variation in species responses, limit generalisations.

The landscapes in which patch-level studies have confirmed important impacts of fragmentation, such as patch area and isolation on species presence, typically hold very low proportions of woodland cover. Covers range from 2-6% in lowland England (Bellamy et al., 1996a, Bennett et al., 2004, McCollin, 1993, Peterken and Francis, 1999, Peterken and Game, 1984) and from 6-8% (van Dorp and Opdam, 1987, Jacquemyn et al., 2002) to 20% in Europe (Jacquemyn et al., 2003). This research has been conducted in areas where fragmentation effects are most likely to be apparent, due to the low covers of woodland remaining, while studies in more wooded, or in upland areas, are rarer. Woodland species within these would be expected to have been detrimentally affected by woodland fragmentation, while many species most susceptible would be expected to have become extinct.

4.3.4 Patch size, species occurrence and habitat conservation quality

The patch size distribution within fragmented landscapes has ecological effects in addition to total habitat area and patch number. Patch size may often appear dominant in landscape research due to the association of patch area with other landscape variables. Associations have been recorded with species diversity, population size and population dynamics. Several mechanisms have been cited as causing these observed species-area effects. Within Island Biogeography the "equilibrium theory" suggested smaller islands / habitat patches held lower numbers of species due to the influence of long-term stochastic events where species on small islands were more likely to go extinct and less likely to be re-colonised, therefore on reaching "equilibrium" between such re-colonisations and extinctions, smaller islands on average will hold fewer species (MacArthur and Wilson, 1967). Metapopulation theory suggested that larger patches hold larger species populations, making species less prone to extinction and communities more diverse (Hanski and Simberloff, 1997). In contrast the "passive sampling" theory assumes that the species occurring in small patches are a random selection of those occurring in larger areas of habitat. Occurrence is related to the different abundance of species found in the larger area, species that occur at low abundances in large habitat areas being less likely to be found in a small area selection by chance (Connor and McCoy, 1979). Additional mechanisms include the existence of "minimum area requirements" for different species, which collectively lead to communities being richer in larger areas and the "habitat heterogeneity" hypothesis which states that larger stands of habitat are more likely to encompass a higher variation in environmental and habitat diversity due to variation in factors such as soils type, slope, aspect and resulting vegetation diversity which collectively leads to higher species richness (Rafe et al., 1985, van Dorp and Opdam, 1987, Peterken and Game, 1984). Smaller woods are therefore expected to be species-poor and to contain proportionately more generalist than specialist species, due to their less exacting habitat requirements (Farina, 1998). Patch size may also be linked to the occurrence of core woodland conditions where smaller patches comprise proportionally more edge than larger patches, affecting species usage (Section 4.3.3).

The relationship between patch area and species richness has been extensively confirmed: for example higher botanical ground-flora species richness in larger English Nature Reserves (Usher, 1979), English farm woodlands (Usher and al, 1992), within both ancient and secondary woodlands in Lincolnshire (Peterken and Francis, 1999, Peterken and Game, 1984, Verheyen et al., 2004), in both rural (Jacquemyn et al., 2001) and urban (Godefroid and Koedam, 2003) woods in Belgium and with richness of tree species (van Dorp and Opdam, 1987) and shrub species (Opdam et al., 1985) in Dutch woods. Another study from lowland England revealed an association between the combined cover of ancient woodland indicator species in the ground-flora community, and woodland ancient size, appearing to indicate a correlation between ground-flora species diversity and woodland size (Willi et al., 2005). Associations have also been shown with bryophytes richness in remnant deciduous patches in Swedish (Lobel et al., 2006) and Australian woods (Pharo et al., 2004). Studies have confirmed positive associations with avian species richness in woodlands in the UK: (McCollin, 1993, Bellamy et al., 1996a, Bennett et al., 2004, Mason, 2001), Europe: (Opdam et al., 1984, Hinsley et al., 1998, Opdam et al., 1985), North America (McIntyre, 1995) and Australia (Watson et al., 2005). However, several authors have reported a lack of associations with patch size. One study of UK woodland birds failed to observe associations between species richness and woodland size, although this study did not control for woodland type or structural features which may have swamped the importance of area (Helliwell, 1976). In examining the ground beetle fauna in both ancient and recent German woodlands Assman found no relationship between woodland size and species richness (Assman, 1999), while a study of resident birds within Swedish deciduous patches in a coniferous matrix failed to find associations between species presence and patch size (Enoksson et al., 1995), noting that where species may also use resources in the adjacent coniferous forest matrix that patch size will be less important, indicating the importance of landscape matrix context in fragmentation studies (Section 4.3.6).

In addition to measures of community species richness studies have examined species occurrence or abundance in relation to patch size. A wide range of studies have shown a positive relationship between woodland size and the probability of individual woodland bird species presence, or breeding, in lowland England: (Hinsley et al., 1994, Bellamy and Hinsley, 2004, Bellamy and al, 2000, Redpath, 1995, Hinsley et al., 1996), Europe: (Hinsley et al., 1998, van Dorp and Opdam, 1987), North America: (Boecklen, 1986, Freemark and Merriam, 1986) and Australia: (Watson et al., 2005), typically leading to higher observed species richness values being recorded in larger woodland patches. Studies have also confirmed that enlarging small woodland patches (by planting) led to increased bird abundance and richness in a study of farm woodlands in southern England (Bellamy and Hinsley, 2004). Additional work has recorded higher abundance of small mammals in larger woods in Cambridgeshire (Redpath, 1995) and higher frequency of bryophytes and lichens in boreal Swedish woods (Moen and Jonsson, 2003). Patch area has also been shown to be the most important factor explaining population size in a study of a woodland ground-flora species (Jacquemyn et al., 2002). Studies of ground-flora species have found that presence of species able to be related to patch area ranged from 57% of species in Lincolnshire woods (Verheyen et al., 2004) to 19% in herbaceous species in a Swedish mixed deciduous woodland (Dupre and Ehrlen, 2002).

Research has also examined species dynamics in relation to patch size. In Swedish woods *Parus major* (great tit) breeding success was shown to be poorer in smaller patches (Loman, 2003). Studies of woodland ground-flora indicated that smaller populations, in small woods may support reduced reproductive fitness, thereby increasing potential extinction events (Jacquemyn et al., 2002). In lowland England, local extinction of bird species from small patches was more likely to occur for woodland specialists, than woodland generalists, following bad weather (Hinsley et al., 1994). Woodland "specialists", "core" or "interior" species typically show stronger associations than "generalist" or "edge" species. Where species are able to utilise resources and habitats beyond the patch the impacts of patch area may be reduced (Woolhouse, 1987, Enoksson et al., 1995).

Of the range of studies confirming species-area effects authors have cited support for a number of theoretical causes. Bellamy et al indicated that the number of bird species found in small woods was similar to the number found in samples of larger woods and could be accounted for by the random sample effect, for woods up to 2ha in size. However as size increased proportionally less species occurred in smaller woods than would be expected from similar sized samples of extensive woods, indicating additional effects were occurring (Bellamy et al., 1996a). Several studies have indicated that immigration and extinction events vary both with woodland size and isolation and are important processes in structuring woodland communities, noting that for some species woodland can be regarded as habitat islands in the sense of island biogeography (Opdam et al., 1985). However researchers have also noted that the equilibrium theory is unlikely to apply to all woodland species due to species-specific effects of isolation and area, due to matrix effects, therefore total woodland communities are unlikely to be in equilibrium as predicted by island biogeography effects (Opdam et al., 1985). Bellamy et al suggested the equilibrium theory did not apply in their study of woodland birds, failing to link patch features to species extinction or colonisation events, although the study was based only on three years of data (Bellamy et al., 1996a).

Much research has indicated that the causes of species-area relationships in woodlands principally relate to the effects of increased habitat and environmental heterogeneity in larger patches. In plant studies authors have noted that the species-area relationship may be due to the correlation between habitat heterogeneity and patch size (Peterken and Game, 1984) or management practices associated with different sized woods (Peterken and Francis, 1999). Structural and topographical features (soils types, ride occurrence and proportion of open space habitats) have been shown to influence botanical diversity levels and occurred more frequently in larger woods (Peterken and Francis, 1999, Peterken and Game, 1984).

Evidence in support of patch-size thresholds is more limited. When examined individually a variety of both plant (McCollin et al., 2000) and bird species (Hinsley et al., 1994) are known to occur in small woodland patches, leading some authors to refute the existence of strict minimum-area requirements for English woodland flora (Peterken, 2000b) and avian fauna (Hinsley et al., 1994). As a broad generalisation it is only when communities are examined that the higher probabilities of species occurrence for a range of species in larger patches leads to an effect of higher richness in larger woods. Therefore exceptions are likely to exist for individual woodland stands but the trend will be more robust when examining a number of sites within a landscape.

The species-area effects seen in woodlands may be most pronounced when examining many sites across heterogeneous areas, and may have several, combined causes with strong effects from environmental heterogeneity, site history, matrix contrast and woodland management. Research has indicated these effects may be less reliable when examining woods over selected ranges of area (e.g. parts of the whole range in patch sizes) (Kirby, 2004), or when examining woods within restricted homogeneous landscapes, or where there is less of a contrast to the surrounding habitat matrix (Enoksson et al., 1995). Studies have also indicated that the relationship may be lost or weakened when data are amalgamated to the landscape scale, a study of 10km^2 landscapes in England found the mean semi-natural ancient woodland patch size showed only small correlations with richness of botanical indicator species between landscapes (Mouflis and Buckley, 2004), suggesting that in combining the regional richness across landscapes the effects of individual patch size on ancient woodland indicator flora is partly lost.

The positive effects of patch size on species presence and community richness may depend on species mobility and landscape connectivity. Immobile species are restricted in occurrence to a habitat patch, with movement opportunities limited to events of dispersal and colonisation while mobile species may be able to use several separate habitat patches as a single cluster or "resource patch"(Farina, 1998). Where species are able to utilise several associated patches

(patches are functionally connected) they may be less likely to be effected by single patch size and may show combined effects of patch size and patches number, leading several studies to include measures of habitat area within a search distance around the focal patch as a variable (Bellamy et al., 1996a, Hinsley et al., 1994). However evidence exists where although bird species were shown to use several patches presence or breeding success was still related to individual patch size (Hinsley et al., 1994, Loman, 2003), suggestions being that additional travelling and foraging costs may occur in birds using several patches compared to a single larger habitat patch (Loman, 2003). Within mobile fauna such work has noted the conditions necessary for metapopulation existence may therefore occur, where the current presence of populations in small patches may not necessarily represent long-term viable populations (Hinsley et al., 1994) and such patches may represent population sinks (Loman, 2003). While such general trends with regard to species richness and woodland size may emerge, the implications of patch size for individual species may not always be clear. A detailed study of *Strix aluco* (tawny owl) in Cambridgeshire indicated that although patch size was positively associated with population distribution, that due to patterns of territory usage where more frequent territorial disputes occurred as territories increasingly overlapped in larger woods, that breeding success of the species was actually highest in intermediate sized woods due to a trade-off between these conflicting biological effects (Redpath, 1995).

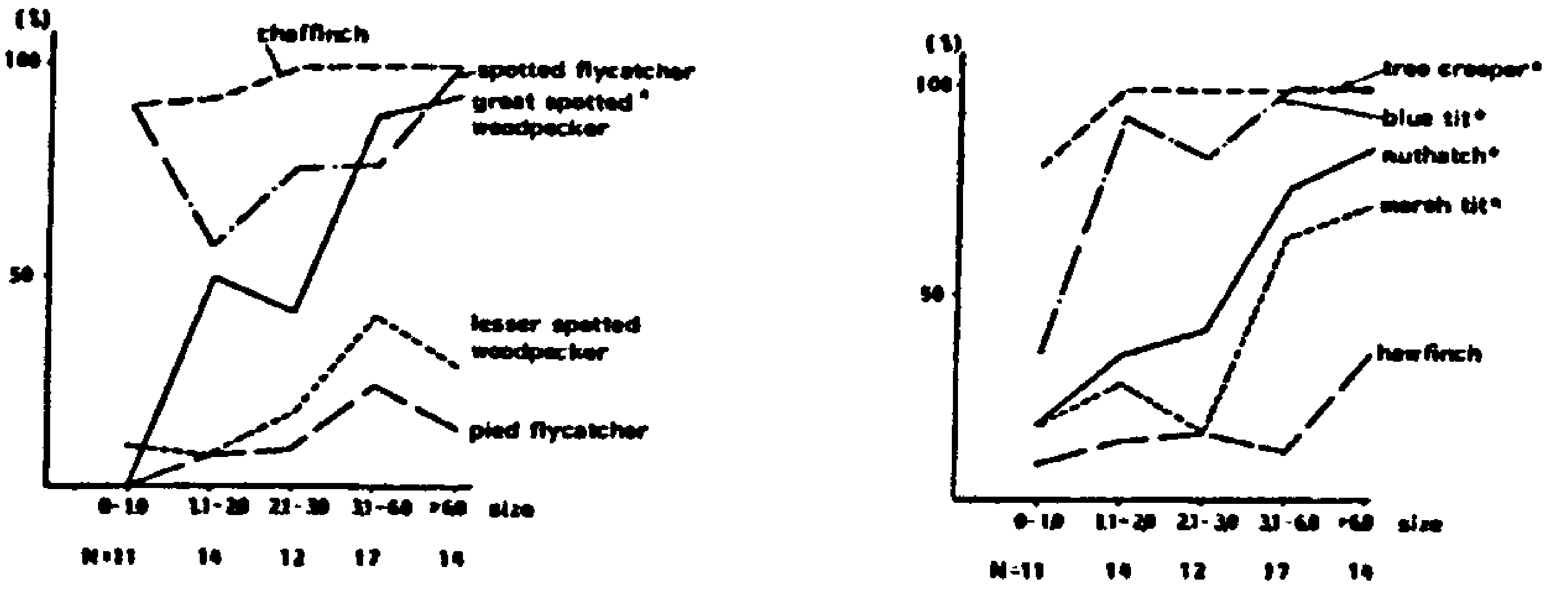


Figure 4.8
Frequency of occurrence of species restricted to mature deciduous woods in Holland, Reproduced from: (Opdam et al., 1985)

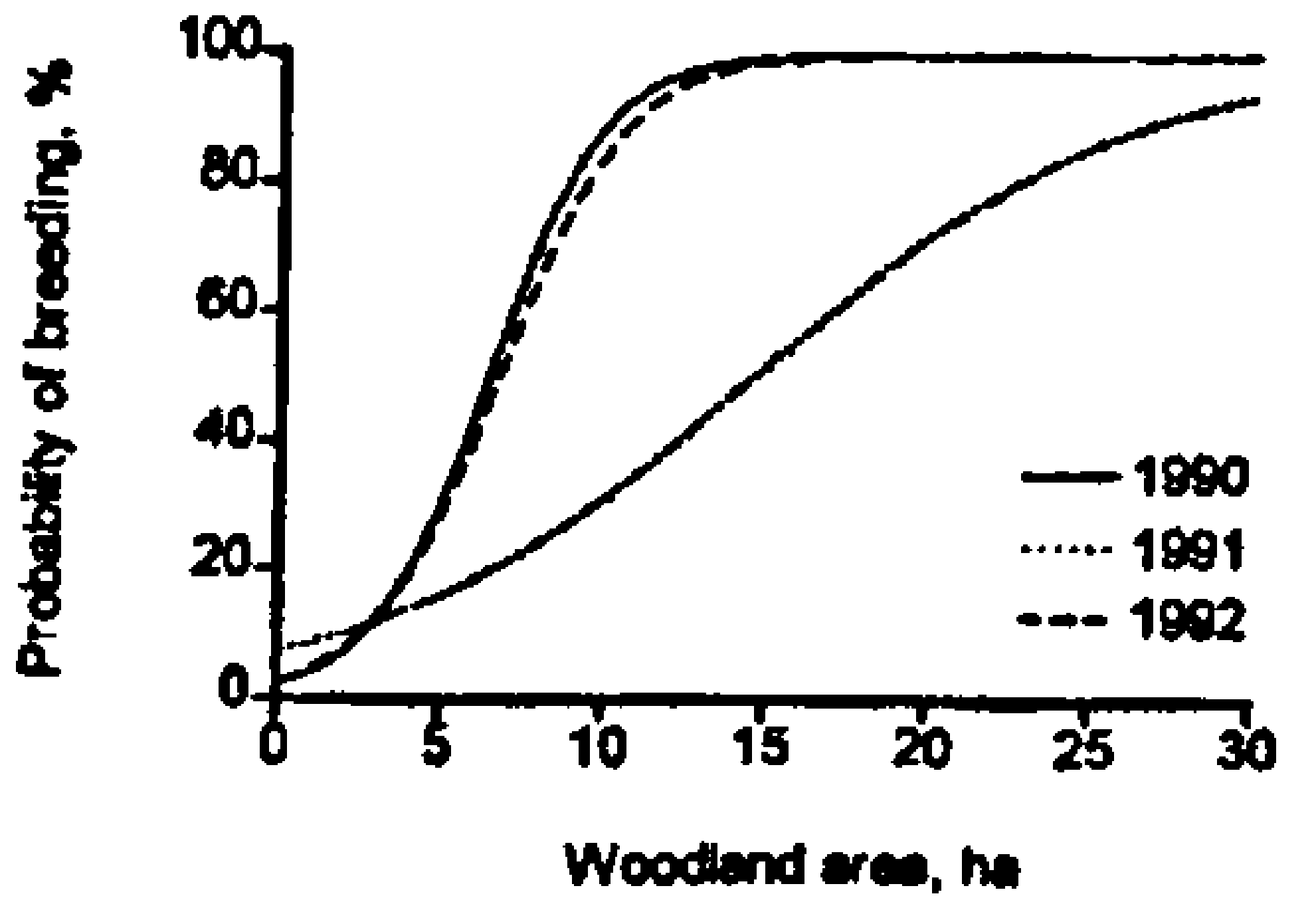


Figure 4.9
Incidence function for presence of *Dendrocopos major* (great-spotted woodpecker)in Cambridgeshire, reproduced from (Hinsley et al., 1996).

Although several studies on patch size effects have refuted the existence of strict thresholds for individual species, others have highlighted patch size thresholds that are considered useful for interpretation of diversity levels, or were noted as useful to guide woodland conservation / restoration. For fauna, studies in lowland England have noted bird species richness increased most rapidly, up to 2ha. At this size most of the common bird species were typically present, although not necessarily breeding, or in large numbers (Bellamy et al., 1996a), while in Holland a 3ha patch size was noted of importance for presence of forest interior birds (van Dorp and Opdam, 1987). A study of tawny owls in southern England noted that occurrence did not reach 100% until size reached 4ha (Redpath, 1995), while Hinlsey's study in southern England revealed the probability of the majority of bird species breeding in a wood did not reach 100% until woods reached 10ha (Hinsley et al., 1994, Hinsley et al., 1996). Mason (2001) supported the benefits of increasing small woodland to a size of 10ha to benefit avian diversity. However one study found that some bird species populations were too small even in woodland patches of 10ha to avoid periodic extinctions (Hinsley et al., 1995a).

Avian studies in the UK have examined woods in the range 0.02–30ha (Bellamy et al., 1996a, Bennett et al., 2004), 0.7–14.5ha (McCollin, 1993), 0.12–62ha (Helliwell, 1976), with European studies holding similar values and with the majority of patches in such studies lying to the lower end of the range. In such landscapes woods above 10-20ha in size are large while woods beyond 50ha are atypical. Several avian studies have considered that woods beyond thresholds of either 2ha (Bellamy et al., 1996a), 8ha (Helliwell, 1976) 10ha (Bellamy et al., 1996a, Bennett et al., 2004) 20ha (Opdam et al., 1985) or of truly extensive woods (100's ha) (Opdam et al., 1984, Opdam et al., 1985) can be regarded as *source* patches for woodland species, due to their size ensuring they will support a high species richness and presence of breeding populations of many avian species.

Botanical studies have also recommended a minimum of 1.5ha and preferably 5ha to achieve the most potential flora diversity in new farm woodlands in England (Usher and al, 1992) while studies by Peterken found that overall woods above 3-5ha were likely to support enhanced richness due to presence of open space features relating to management, with >20ha in size likely to ensure such spaces remain permanent features (Peterken and Francis, 1999).

4.3.5 Core area, edge-effects and patch shape complexity

The relationship between area and perimeter varies with patch shape and size and this perimeter/area ratio is a frequently measured landscape metric (McGarigal et al., 2002). Fluctuations in patch size, shape and perimeter/area ratio give rise to variation in "core area" calculated following the definition of a distance from the patch boundary that is relevant to the study habitat or species within which observable "edge-effects" may occur. The relationship between patch size, shape, core area and edge-effects may be complex. A larger proportion of perimeter to area occurs at smaller patch sizes, while the more a patch shape deviates from a circle and becomes linear, the greater the ratio of perimeter to area will be. Therefore core area will be lowest and edge-effects highest in patches that are either small, have complexly shaped or linear.

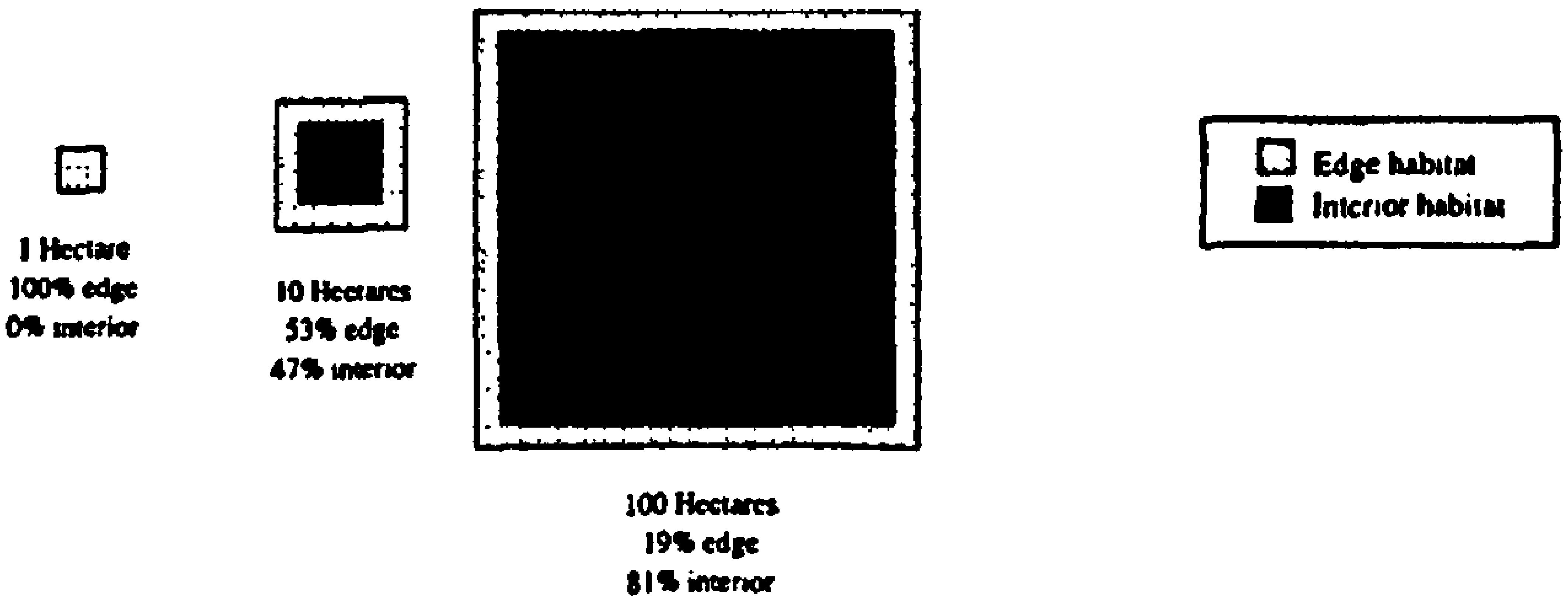


Figure 4.10
Edge-effects, core area and patch size. Core area proportion increases with patch size. Based on 50m edge width. Reproduced from (Collinge, 1996).

"Edge-effects" in woodland habitats are typically considered to be detrimental (Murcia, 1995), while "core area" has more positive associations, representing an area of stable conditions within the patch, where edge-effects are absent. However depending on the depth of edge-effects and patch shape some patches may not contain any "core area" (Fig 4.10). Negative edge-effects within woodlands have been shown to be associated with a diverse range of factors. Increased avian nest predation has been recorded in small and complexly shaped woods in Sweden (Andren, 1992), while a study of woodland birds restricted to mature deciduous woods found lower species richness in oblong shaped woods with more "edge" (Opdam et al., 1985). A study of ancient woodland ground-flora in Cambridgeshire examining agricultural pollution found longer thinner woods had higher covers of nitrophilous species and low covers of woodland interior "stress-tolerator" species (Willi et al., 2005). Forest edges have been shown to concentrate and collect nutrient pollution (Weathers et al., 2001). Agricultural pollution has been shown to detrimentally effect ground-flora species in ancient woodlands in Kent (Gove et al., 2004b, Gove et al., 2004a) (Fig 4.11), Cambridgeshire (Willi et al., 2005) and in Lincolnshire and Derbyshire (Bateman et al., 2004). Additionally competition from weed species along edges of ancient woods in Belgian has been shown to affect ground-flora (Honnay et al., 2002b). Various works have recorded alterations in environmental conditions along woodland edges compared to interiors, including altered light levels, air temperature and soil characteristics, shown for ancient woods in an arable matrix in Belgium (Honnay et al., 2002b), North American temperate woods (Collinge, 1996, Matlack, 1993), and detrimentally affecting presence of bryophytes in European boreal woods (Moen and Jonsson, 2003, Lobel et al., 2006). Particular forms of edge will have different effects depending on the "contrast" between patch types. Effects are expected to be lower when habitats show less contrast, as between adjacent stands of different woodland types than between highly contrasting habitats such as woodland and arable or improved grassland. Edge-effects have also been shown along roads where multiple detrimental affects causing mortalities or reduction in breeding success of birds and mammals occur (Spellerberg, 1998).

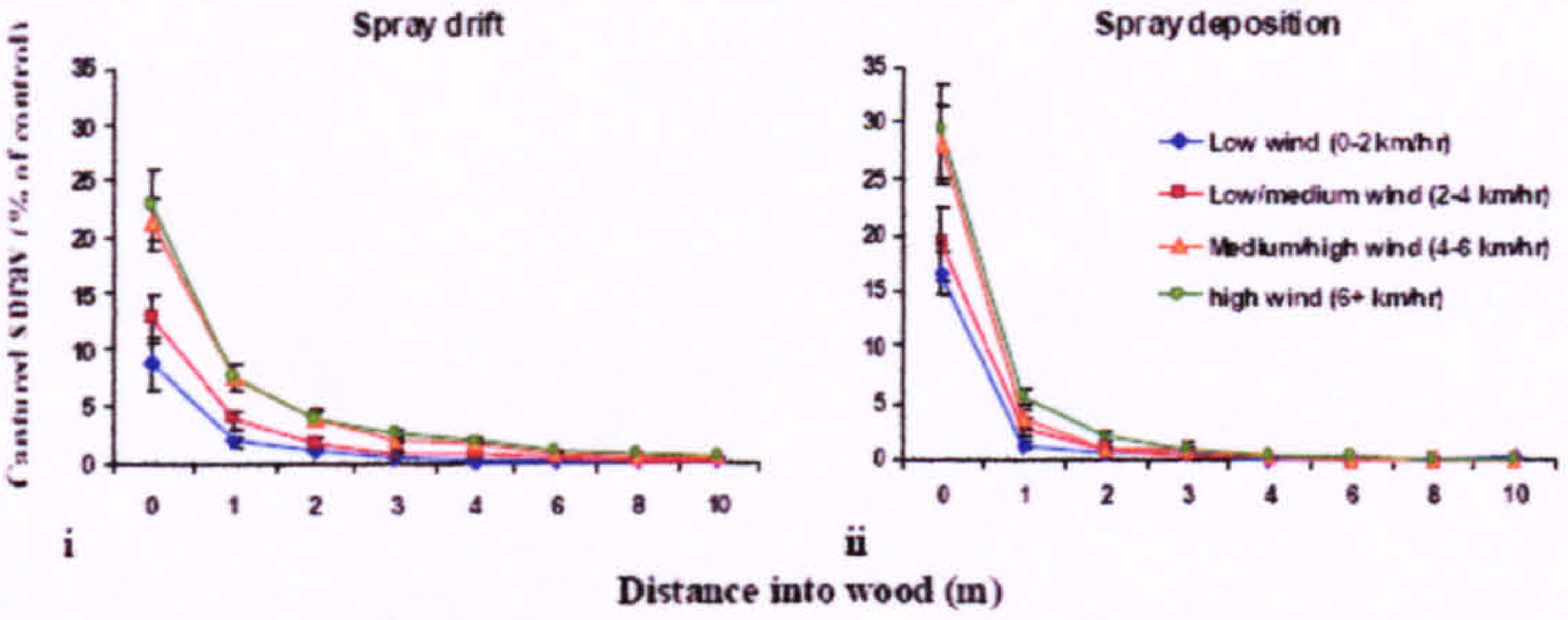


Figure 4.11
Edge-effects of pesticide spray within ancient woodland margin in Kent. Reproduced from (Gove et al., 2004a)

In contrast to detrimental edge-effects research has also indicated that edges may produce beneficial conditions. Fuller et al summarised a number of positive factors influencing bird diversity in relation to edges in British lowland woods including a higher complexity of vegetation, variety of tree and shrub species, more flowering trees and shrubs, high levels of insects, and adjacency to potentially beneficial farmland habitats all of which may encourage increased use and occurrence of certain bird species along woodland edges (Fuller et al., 1995). In lowland England certain species may be associated with edge conditions (Hinsley et al., 1994), in particular species classified as "edge" species which use woodland habitats in addition to farmland, non-woodland habitats (generalists) have been shown to be positively associated with the length of patch perimeter (lowland England) (Bellamy et al., 1996a). Similarly management that creates edges or "rides" within woods has been shown to increase botanical ground-flora species richness in lowland English woods (Buckley et al., 1997), although such species may not be strict woodland interior specialists. Additionally in a study of carabid species within ancient woodland sites in Manchester the number of species occurring within the edge zone was found to be associated with patch mean shape index, increasing with patch shape, while the study also indicated the presence of houses adjacent to woods may be beneficial in maintaining woodland climate conditions and may reduce edge-effects in comparison to open habitats (Allen et al., 2004). At a landscape scales analysis has had some success in linking patch shape complexity measures to positive species richness in flora and bryophytes (Moser et al., 2002).

These associations of species to edges have led to a distinction in some studies between woodland "interior" or "specialist" species and woodland "edge" or "generalist" species (Section 4.3.11). Woodland interior / specialists are detrimentally affected by edge-effects. Such species will show occupancy effects in relation to patch size and may be more closely associated with measures of patch core area or shape, whereby patches of apparently sufficient size may not be occupied by the species where the patch does not contain sufficient core area. Woodland edge or generalist species in contrast may be positively associated with the conditions found at woodland edges or may be able to use woodland interior and edge habitats equally well. Where studies do not distinguish between these species groups the effects of patch shape and core area may be less apparent due to the confounding processes acting on each species group. Therefore one effect of increased forest fragmentation may be the replacement of forest interior "specialists" with edge "generalists", which may occur in smaller or more linear and complexly shaped patches while utilising the habitats within the surrounding matrix (McCollin, 1998).

Several researchers have examined patch shape or core area and have not found significant effects. A study of small farm woodlands in England failed to show any association between patch shape and botanical richness (Usher and al, 1992) while a study examining ancient woodland indicator species flora in different landscapes (120km^2) in England failed to show any significant correlation between mean shape index and indicator species richness (Mouflis and Buckley, 2004), although the individual patch effects may easily be obscured by such a study amalgamating patch shape data to the landscape level where the relationship may not be well captured when aggregated across landscapes using mean shape values.

Where edge-effects have been found within studies the extent to which these have penetrated into woodland patches is variable. Distances range from 3m for competitive effects in ground-flora in woods in Belgium (Honnay et al., 2002b), agricultural pollution extending 5-10m in Kent ancient woods (Gove et al., 2004b), to 30-40m in Lincolnshire and Derbyshire woods (Bateman et al., 2004), c.50m affecting bryophytes and lichens in boreal woods in Europe (Moen and Jonsson, 2003), while one study in ancient woods in Cambridgeshire reported that ground-flora diversity and composition was highly altered near arable boundaries but that elevated nitrogen levels occurred up to 100m (Willi et al., 2005). Typically larger distances have been recorded where environmental effects are measured (levels of exposure, light, humidity and air-borne pollution) than studies showing actual differences in plant or faunal communities. This is illustrated by the study by Honnay et al which confirmed differences in plant community composition between edge and core forest zones but concluded the woods were relatively impermeable to invasion by weedy competitive plant species beyond an initial 3m edge zone even though environmental effects could be detected much further (Honnay et al., 2002b). A review examining microclimate in temperate forests found maximum effects ranging from 10m to 60m, with an average of the studies being approx 46m (McCollin, 1998). However even these environmental edge-effects although probably typical of all open-ground to forest edges, will vary with aspect (Honnay et al., 2002b), topography and edge contrast, while some, such as the increased predation rates, shown by Andren (1992), are also highly matrix context dependent.

Edge-effects due to patch size and shape are widely evident in the literature and have been shown to extend for considerable distance from climatic and pollution effects. However the impacts of such edge-effects may be highly species-specific and may be affected by many variables. Due to such variation, patch shape and core-area effects are most usefully examined using landscape-mosaic models, where edge-effects may be quantified based upon the nature of adjacent patch boundaries and levels of patch contrast, as exemplified by the careful clarification of edge-effects due to patch adjacency type by (McGarigal and McComb, 1995).

4.3.6 Patch frequency and woodland conservation value

The number of habitat patches may have important effects in addition to the area of suitable habitat present, although such affects may be difficult to distinguish from those of isolation. Theories of the value or contribution of multiple patches stem from island biogeography and metapopulation theory (MacArthur and Wilson, 1967, Hanski and Simberloff, 1997) and from the positive effects of habitat heterogeneity (Section 3.2.2). Principally the effects relate to the fact that multiple nearby patches can represent a larger pool of potential colonists. Multiple patches may increase the persistence of a metapopulation by enhancing patch colonisation rate in comparison to patches that are located close to fewer patches. Additionally in examining species richness across multiple patches several habitat patches may hold more species than fewer patches of the same area. The occurrence of multiple patches adjacent to a focal or newly created patch may thus have two effects. At one level the occurrence of multiple patches may be a correlate of source patch area, more patches representing more area, with consequent associations with increased population sizes and species richness. However even when the area of nearby patches remain constant their may be benefits from multiple patches than the same area in fewer patches. Critically such effects are dependent on study species dispersal / movement distances relative to inter-patch distances. Mobile species are continually affected by patch number in the way they utilise habitat and resources, while immobile species are only affected at events of dispersal and colonisation.

Several studies have confirmed the positive effects of habitat patch number on species presence, abundance or diversity. The number of ancient woodland indicator species flora showed a large correlation with number of ancient woodland patches in 10km^2 landscapes across England (Mouflis and Buckley, 2004), a relationship that was shown to be important even after controlling for overall area of ancient woodland in the landscape. Woodland bird species presence was also positively correlated with number of woodland patches in landscapes in North America (Villard et al., 1999). At a finer, patch-scale, studies have identified that multiple patches can hold higher levels of biodiversity than the same area of habitat in fewer patches, indicating the number of patches could be used as a surrogate for woodland biodiversity (Woolhouse, 1987, Peterken and Francis, 1999). In Lincolnshire Peterken and Francis (1999) found that collectively several small woods held a richer ground-flora than fewer large woods of the same total area, while for UK woods Woolhouse showed that several small woods held a higher number of bird species than fewer woods of the same total area (Woolhouse, 1987). Local woodland species richness is not related just to the area of woodland in the local landscape but also to the number of patches present.

In addition to biological explanations for such effects, resulting from colonisation and re-colonisation, and metapopulation theory, multiple sites may enhance local biodiversity due to the variation in conditions occurring at multiple patches compared to fewer, more homogenous patches. Separate woods may receive different management, favouring different species, at different times. These observations also suggest that several separate woodlands that occur in a diffuse group over a wide land area hold more species than woodlands that are separate but occur close together. This is because nearby woods are more likely to hold similar environmental conditions. Therefore once again the scale of study affects the relationships observed. In many woodland studies that record multiple patches, patch number is measured by within an envelope or ecological neighbourhood. This is typically based on species dispersal or movement ability, e.g. woods recorded within a 1km or 2km radius of a woodland in the study of mobile woodland bird populations (Hinsley et al., 1994, McCollin, 1993).

In summary, regional or multi-patch floral and avian diversity may be related to the number of patches present in addition to area. The scale at which this trend occurs will be related to species movement and dispersal abilities. When distance between patches is much greater than typical species movement or dispersal distances then no relationship would be expected between patch number and species occurrence or diversity. These relationships generally apply to historically fragmented landscapes with remnant vegetation patches, such relationships are likely to reduce as the proportion of the landscape remaining intact increases. For example in landscapes where the majority of patches occur close to other larger patches the number of patches is unlikely to be significant beyond the affects of adjacent patch area in the local landscape.

4.3.7 Isolation, connectedness and connectivity: effects on species occurrence, habitat quality and conservation interest

A critical variable in defining spatial configuration is isolation. Patch isolation and its inverse, landscape connectivity, affect species abundance, community assembly and diversity. These may be investigated by examining structural connectivity or species-ecology based functional connectivity (Fig 4.12, 4.13). Structural metrics examine the landscape pattern that can be measured purely from structural features of patch occurrence while functional measures interpret the landscape in relation to species attributes such as dispersal distances or accounting for differences in landscape resistance or hostility. The simplest structural isolation is the physical isolation of a patch from nearby patches, measured by Euclidean or "nearest-neighbour" (NN) distance. Although structural measures may be relevant to mobile species such as birds, more functional approaches are required for less mobile species, or species dispersing across habitats on the ground. Connectivity measures the result of patch spatial structure on species populations, by examining patch isolation in relation to species movement ability, to define functionally connected habitat (With et al., 1997). Habitat patches may be functionally "connected" even though they are physically isolated, to the extent that analysis may reveal "virtual" functional movement corridors to exist (Vuilleumier and Prelaz-Droux, 2002). Connectivity explicitly considers species ecology and is not related solely to the study landscape but also to the study species (Wiens et al., 1997). Merriam (1984) considered connectivity to measure the processes "by which the subpopulations of a landscape are interconnected into a demographic functional unit" (Merriam 1984 in Farina 1998). This highlights the importance of species dispersal and landscape resistance, and connectivity can also be viewed as the inverse of a measure of hostility of inter-patch habitat (Farina, 1998). Where landscapes are hostile to species movement functional connectivity may be low, where landscape are less hostile and more easily traversed, connectivity may be high.

Ideally the variables used within studies to characterise isolation or connectivity will accurately reflect the way a species utilises the landscape, however this is not always possible, given limited datasets and lack of species knowledge. Researchers have noted that it is unreasonable to assume that dispersal routes will simply be the shortest routes between two patches (Gustafson and Gardner, 1996). However the use of simple edge to edge patch isolation may be appropriate in examining landscape use by freely dispersing bird species (van Dorp and Opdam, 1987, Opdam et al., 1984) and has been utilised to reflect botanical colonisation and isolation (Jacquemyn et al., 2001, Jacquemyn et al., 2002, Dupre and Ehrlen, 2002).

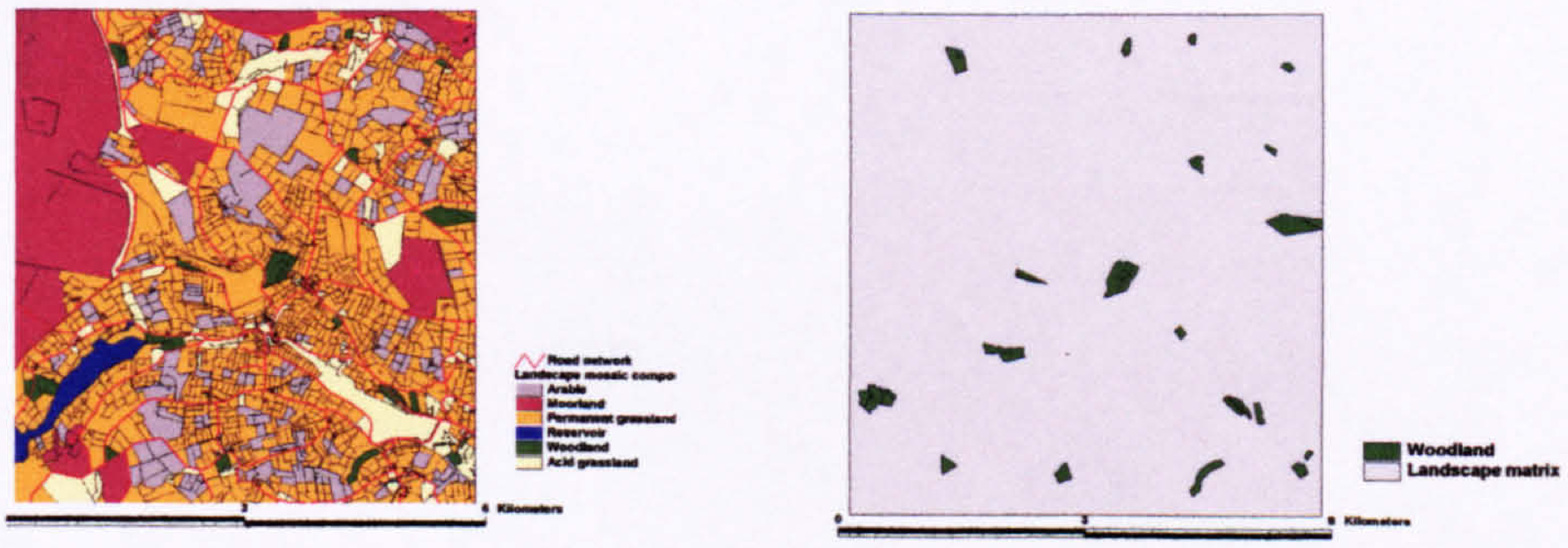


Figure 4.12
Examples of landscape mosaic (left) and binary (right) landscape representations

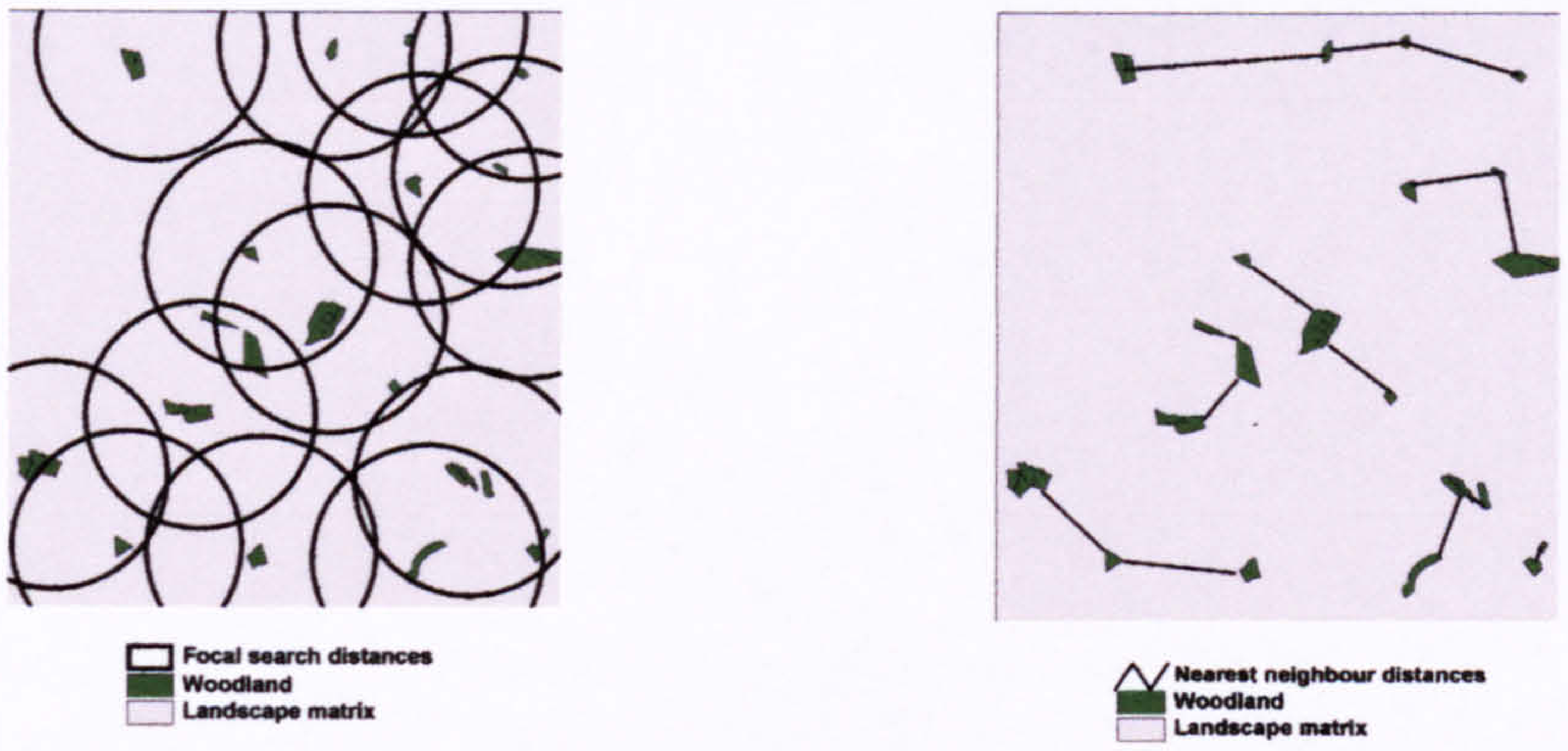


Figure 4.13
Focal search distance (functional) and nearest neighbour (structural) measures of woodland isolation (Produced by the author).

Several studies have used proximity measures combining physical isolation and patch area within a focal neighbourhood (Usher and al, 1992) (**Fig 4.13, 4.14**), although some studies incorporating such measures found their inclusion provided no additional information or explanatory power beyond that provided by simple nearest neighbour index (Dupre and Ehrlen, 2002, Bastin and Thomas, 1999). Other studies have taken inspiration from source-sink models and examined isolation from species "source" habitat; the nearest large woodland above a threshold indicative of high biodiversity levels and large population size, e.g. 10ha (Bennett et al., 2004), 20ha: (Opdam et al., 1985), 25ha: (Opdam et al., 1984) and distance to "extensive" forest of many 100's of hectares (Opdam et al., 1985, Opdam et al., 1984). Many of these approaches are based upon simple "binary" landscape interpretations, following island biogeography predictions (**Fig 4.12**). However additional approaches based upon the landscape mosaic model are appropriate for certain plant and bird species or species such as bats that utilise the landscape in more complex ways, following structural corridors such as hedges and lines of trees when foraging in English landscapes (Entwistle et al., 2001, Greenaway, 2004, Greenaway and Hill, 2004, Walsh and Harris, 1996) (**Fig 4.14**). For these species the use of simple edge to edge woodland distance would not accurately reflect isolation, the connectivity of the landscape being determined by the configuration and structure of multiple habitat types, potential dispersal distances being altered based upon resistance within different habitat types and differing boundary crossing behaviour at habitat/matrix patch boundaries. Such studies therefore may examine flight lines along networks of connected trees and hedges around patches (Bellamy et al., 1996a).

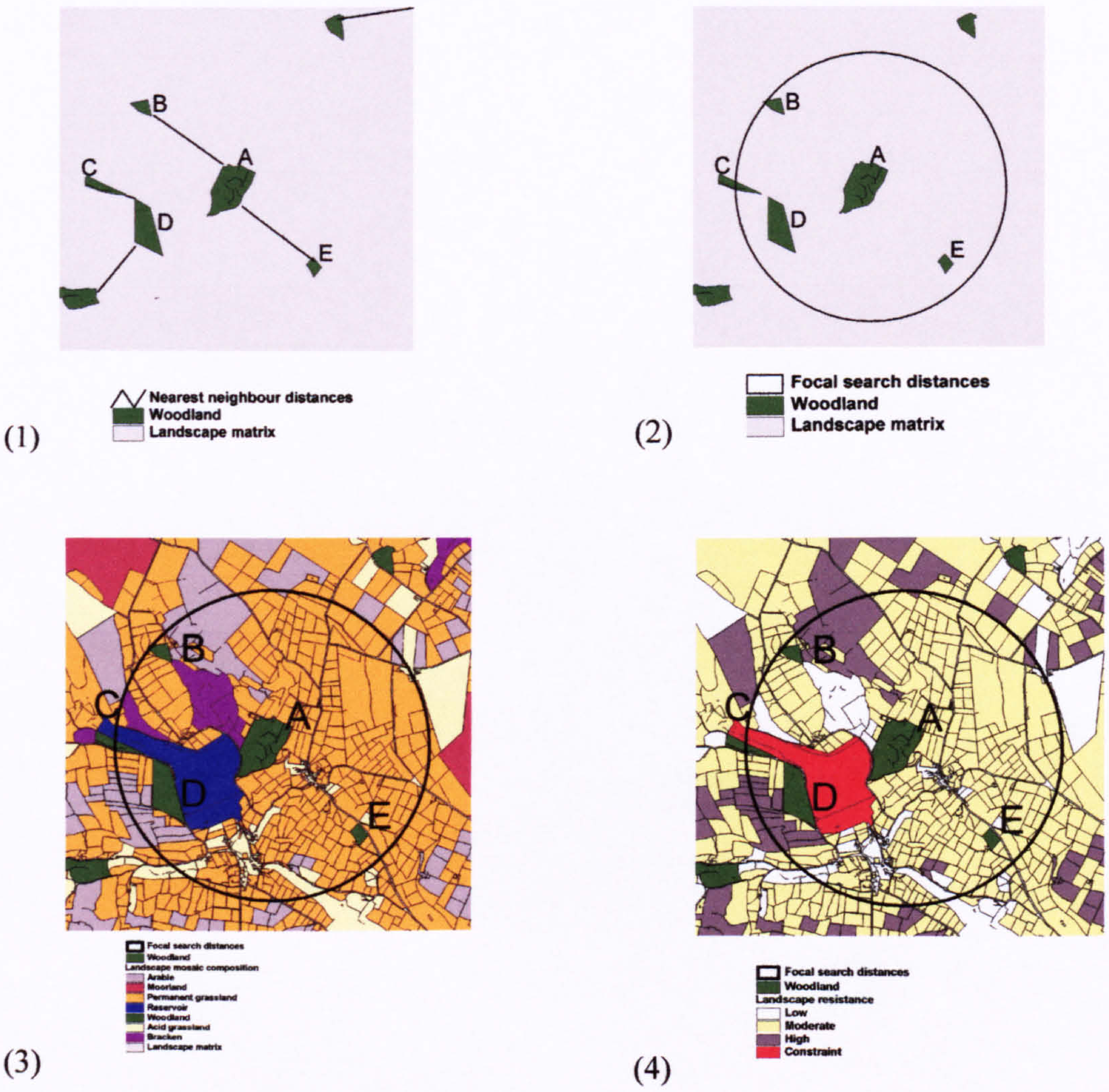


Figure 4.14

Varying representations of connectivity between patches A,B,C,D and E based upon consideration of dispersal from patch A and varying with study methodology. (1) and (2) are based upon binary landscape representation while (3) and (4) are based upon the landscape mosaic perspective. (1) Nearest neighbour distances may be used to indicate potential connectivity between patches, based purely upon the shorted distance, assuming this may be linked to probability of dispersal. (2) Utilising a focal search distance based upon species dispersal ability indicates all patches (A,B,C,D,E) are functionally connected. (3) Taking a landscape mosaic approach it can be seen that while patches A,B,D and E remain functionally connected patch C can no longer be considered functionally connected for ground dispersing species as it is not able to be reached within the set dispersal distance, assuming that open water represents a complete barrier to species dispersal. (4) Taking into account landscape resistance values reveals that only patches A and B may be considered functionally connected for ground dispersing species, as al other patches are separated by patches that represent barriers to dispersal or that have high landscape resistance meaning species could not disperse long distance to reach patches at the edge of there potential dispersal range (which is based on dispersal across favourable, low resistance habitats) (Produced by the author).

A variety of studies have confirmed associations between woodland isolation and species presence, breeding, reproductive success or overall patch diversity levels. Empirical studies have shown that isolation negatively effects botanical species richness in England: (Peterken and Game, 1984, Mouflis and Buckley, 2004), America (Matlack, 1994) and Europe: (Jacquemyn et al., 2003) (**Fig 4.15**). Studies have shown species richness in secondary woods is related to their isolation distance from ancient woods, (Peterken and Game, 1984, Dzwonko and Loster, 1992, Dzwonko, 1993, McGarigal et al., 2002, Matlack, 1994). Studies examining ground-flora have shown that distance to established woodland areas negatively affects the richness of developing young woods (Dzwonko, 1993, Jacquemyn et al., 2003, Butaye et al., 2001). European studies examining herbaceous ground-flora species have also shown that particular species can be affected by isolation, their distribution being less frequent in isolated woodland patches, more isolated areas within single woods, or isolation producing distinct aggregations of species distributions (Verheyen and Hermy, 2001, Dupre and Ehrlen, 2002, Jacquemyn et al., 2001). However exceptions occur, one study from lowland England suggesting that isolated woods were more valuable to conservation as they were richer in plant species than less isolated woods close to nearby large woods, although the authors considered this due to the more intensive management of the larger woods effecting species presence (Helliwell, 1976). Conflicting results have been reported for studies of bryophytes in isolated woods. One boreal study indicated distances of more than a few hundred metres were sufficient to prevent colonisation (Lobel et al., 2006), while in Australian woods bryophyte presence was found to be more responsive to internal patch features (structure, substrate) than landscape context, leading the authors to suggest the species had high colonisation ability (Pharo et al., 2004). Isolation effects have also been reported for woodland fungi. Studies showing that bracket fungi occurred with reduced frequency, and the invertebrates communities they supported were impoverished in woodland fragments separated by 1.2 to 1.7km (Komonen et al., 2000).

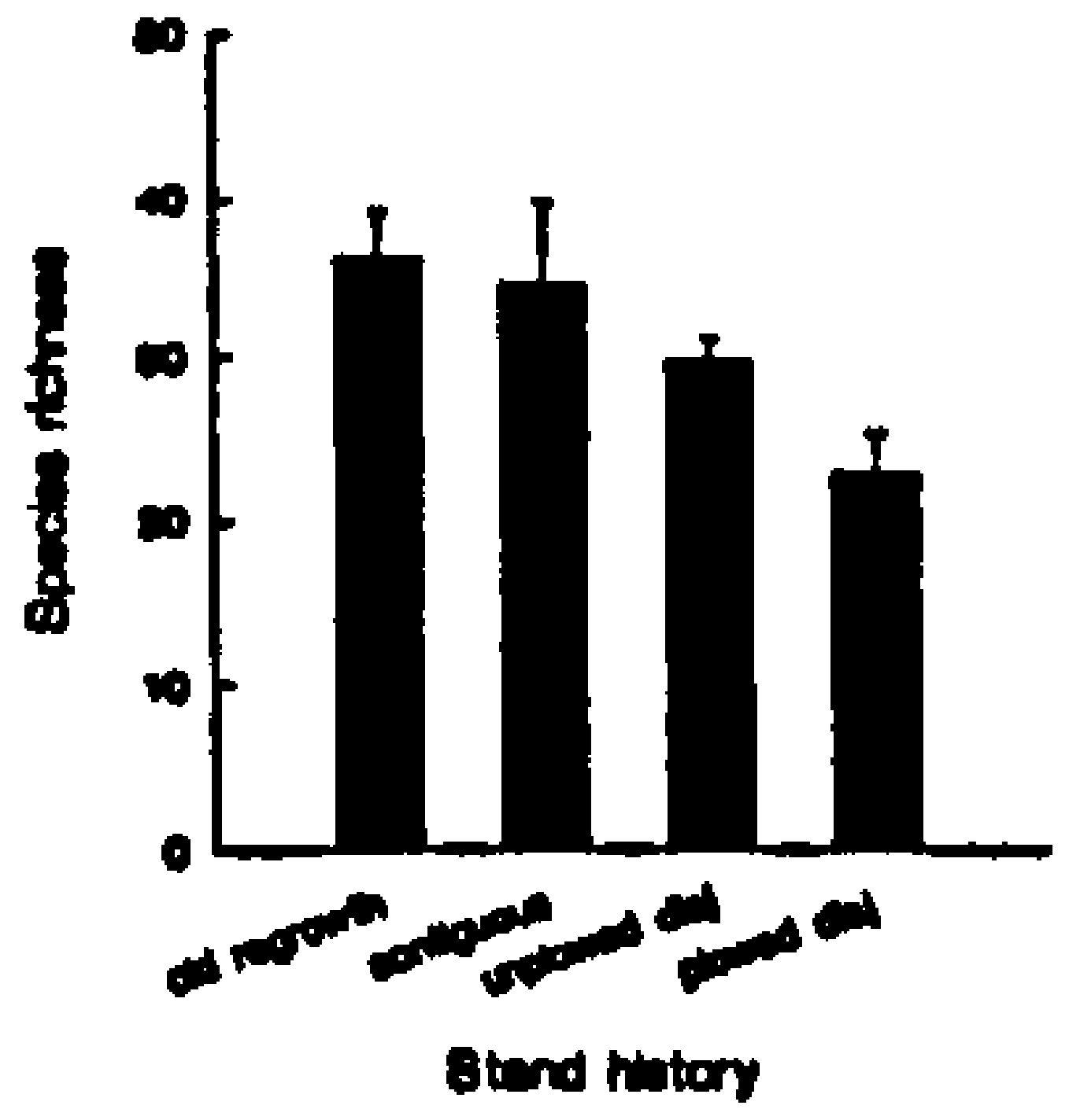


Figure 4.15
Species richness in contiguous and disjunct (disj) woodland patches in North America with different management history. Error bars indicate +-1 SE. Reproduced from (Matlack, 1994)

Several of these botanical studies highlight the poor dispersal ability of woodland flora, in particular the group known as "ancient woodland indicator species" and the differences between ancient woods and secondary woods. A variety of studies have shown an effect between isolation and the presence of herbaceous ground-flora. In Belgian deciduous woods Verheyen and Hermy found both isolation and dispersal mechanisms affected forest plant distribution, with dispersal limited species being less likely to occur in isolated woodland (Verheyen and Hermy, 2001). However the authors noted that the "low colonizing capacity of ancient forest plants cannot be attributed to a single cause, rather both dispersal and recruitment are limiting but the relative importance varies" (Verheyen and Hermy, 2001). The authors were able to define plants into different species groups susceptible to different limitations. A study in Swedish deciduous woods also found a relationship between isolation and plant species occurrence and dispersal ability (Dupre and Ehrlen, 2002). Plants with low seed production or showing vegetative spread were less likely to occur in isolated woods, suggesting dispersal limitation (Dupre and Ehrlen, 2002), confirming predictions made by modelling studies that isolated patches contained more "far dispersing species" and less "short dispersing species" (Baillie et al., 2000). Additionally forest specialists were less likely to be found in isolated patches than woodland generalists (Dupre and Ehrlen, 2002). However although such correlations were noted, when the distributions of 57 species were modelled from field data only 4 species held distributions that were considered to result from edge to edge isolation (although the authors note the possible limitation of the study in defining connectivity measures) (Dupre and Ehrlen, 2002). A study examining ground-flora colonisation of recent woods in a well wooded area of Belgium (20%), found that species richness was significantly lower in isolated woodland patches than non-isolated patches, where isolation was classed as woods occurring more than 100m from nearby woods (Jacquemyn et al., 2003). Even when secondary and ancient woods are touching studies have found that successful colonisation of new woodland by ancient woodland ground-flora declines with distance into the new woodland patch (Bossuyt et al., 1999, Brunet and von Oheimb, 1998, Matlack, 1994). Within Lincolnshire Peterken and Game suggested that isolation effects were linked to dispersal, some flora not being able to colonise distant sites, even over considerable periods of time (Peterken and Game, 1984). Peterken and Game (1984) showed how the richness of secondary woods developing in isolation from ancient woods was lower than secondary woods developing adjacent to ancient woods, in a lowland arable landscape. Interestingly one study failed to show any association between botanical richness and woodland proximity although the study examined secondary woodlands of limited age (40-90yrs old), within a landscape lacking ancient woodland sites, possibly meaning that the short timescale and lack of ancient woodland species meant isolation had less of an effect (Usher and al, 1992). At a larger scale, examining 10km^2 "landscapes" across England researchers showed a moderate negative correlation between ancient woodland isolation (mean nearest-neighbour and mean 2nd nearest neighbour) and landscape richness of botanical ancient woodland indicators species suggesting landscapes with more isolated ancient woods tended to hold lower overall richness of AWI species (Mouflis and Buckley, 2004). The isolation of ancient woodland patches from secondary woods (measured as mean Euclidean distance) was not significantly associated with occurrence of ancient woodland indicator species. Importantly these relationships remained even when accounting for the relationship with overall area of ancient woodland between landscapes with different isolation (Mouflis and Buckley, 2004). This implies that decreased isolation of ancient woods from secondary woods at this landscape scale does not promote landscape richness of ancient woodland indicator species. These recent findings, indicating an isolation effect on ancient woodland sites contrast with earlier studies by Peterken and Game (1984) who considered that, within their study area, isolation did not affect richness within existing ancient woodland sites, where occurrence was determined primarily by within-patch features (vegetation structure, soils). This indicates that ancient woodland sites, having acquired woodland species at a time when land management was less intensive, and having existed over long periods of time, were not affected by becoming subsequently more isolated. Instead the plant species showed remnant population dynamics in the sites in which they have perhaps always occurred. This indicates that ancient woodland ground-flora would be unlikely to exist as metapopulations. Interestingly, the national study by Mouflis and Buckley (2004) supports the view that such ancient woodland flora could exist as metapopulations, aided by decreased isolation. Both studies suffer limitations; Peterken and Game being limited to a single landscape where the matrix is truly hostile (arable), and Mouflis and Buckley relying on atlas data collated for different purposes, with potential bias to selected sites and occurring at a large scale – 10km grids relative to plant mobility, reflected by the authors' call for the study methodology to be repeated at finer scales.

These observations between ancient and secondary woods can be linked to the different predictions made by island biogeography for mainland-island and oceanic islands. The theory predicts that in newly created oceanic islands species diversity will increase over time to reach a level influenced by the number, size and distance to colonisation sources. In islands formed from the mainland by reduction in area, analogous to ancient woodland becoming fragmented in an agricultural matrix, the theory predicts that relaxation will lead to lower species number over time as species become extinct in relation to island size and distance to colonisation sources. The observations with woodland ground-flora indicate that such extinctions do not occur in ancient woodland flora perhaps counteracted by the long history of active management and the remnant population capability of the plant species examined.

A variety of studies have also examined the avian fauna of isolated woods. Researchers have reported negative effects on the number of woodland dependant bird species occurring within isolated woods in lowland England (Bennett et al., 2004), Europe: (Opdam et al., 1984) and North America (McIntyre, 1995). Other studies have examined effects on individual species and found presence to be negatively associated with: mean nearest neighbour distances in woods in England (McCollin, 1993, Redpath, 1995, Hinsley et al., 1994), Europe (van Dorp and Opdam, 1987), North America (Villard et al., 1999), isolated as opposed to aggregated woods in Sweden (Enoksson et al., 1995) or by distance from the nearest extensive large forest area in Holland (Opdam et al., 1985, Opdam et al., 1984). Other faunal studies have examined species dynamics and found isolation was associated with increased risks of patch population extinction for woodland mice in America (Fahrig and Merriam, 1985).

In contrast to many studies, several researchers have examined isolation utilising the landscape mosaic model, incorporating multiple habitat types and examining the effects of patch structure within the matrix. One study in lowland England showed the presence of hedgerows in the immediate landscape around woods increased species richness of woodland resident birds (Bellamy et al., 1996a).

The relevance of patch isolation differs between landscapes and species depending on the relationship between species mobility, dispersal, patch size and isolation. Where species frequently move between habitats or use several patches in a landscape the use of connectivity measures may be more appropriate, patches being considered connected where they lie within a distance defined by species mobility. Patches lying beyond typical species movement distances can then be considered to be functionally isolated. Very mobile species therefore may only be affected in extreme cases of isolation. However even for very mobile species movement and dispersal events carry some form of cost, either in terms of energy expenditure or through increased chance of predation, and therefore isolation is likely to remain a relevant factor. A central problem in studies examining connectivity is the categorisation of dispersal distances. Often research attempts to model relevant isolation levels or define functional connectivity in relation to typical dispersal distances. Dolman and Fuller (2003) noted that dispersal generally has a "leptokurtic" distribution (the majority of individuals disperse short distances but a small number show very long-distance dispersal), highlighting the problem this causes studies, which generally do not know the extent of this leptokurtic tail due to lack of data. Distances may therefore be estimated from observations or short term data, but in such cases the effect of any modelling or effects should also be related to these short timescales and the effects of longer term, gradual colonisation may remain unknown. Notably several studies of plant migration have been limited by the low age range of patches e.g. (Brunet and von Oheimb, 1998), the potential for occasional long-distance dispersal may aid colonisation of some species at distant sites, given longer timescales. Dispersal rates of ancient woodland ground-flora into adjacent secondary woods would take 100yrs to colonise a 0.25ha created adjacent site and 200yrs to colonise a 1ha site assuming a compact shape (Brunet and von Oheimb, 1998).

Ultimately the distances over which woodland may be functionally isolated will depend on the combined effects of species dispersal ecology, patch configuration and relative matrix permeability. Studies indicate that freely dispersing birds may be affected by isolation distances of 1-2km even though they are able to disperse further, while woodland flora and immobile invertebrates may be negatively affected by distances of as low as 100-500m although such affects will be limited where much longer timescales are available for patch colonisation.

4.3.8 Patch contrast and the landscape matrix: effects on species occurrence, habitat quality and conservation potential

In addition to patches spatial arrangement, patch populations are also affected by the landscape matrix and the contrast between adjacent patch types. These affect boundary crossing behaviour, landscape hostility and hence species dispersal, movement and the relative importance of edge-effects. Matrix effects are therefore closely related to patch shape, functional connectivity, the number of habitat patches around focal patches and the total proportion of habitat within the landscape. The theoretical basis for these effects and the corresponding metric usage are less related to strict island biogeography and metapopulation biology theory than other fragmentation metrics due to the frequent treatment of matrix in a simplistic way as a hostile "sea", although recent examples of metapopulation modelling studies have attempted to include more realistic treatment of matrix variation. Studies examining forest remnants have noted the importance of matrix in determining dynamics and have called for a "whole-landscape" approach to fragmented population conservation, rather than the past emphasis on patch binary approach (Gascon et al., 1999). Quantification of the landscape matrix is the basis behind the landscape mosaic approaches to landscape study (McGarigal et al., 2002), while aspects of gradient theory may also apply where patches are frequent in relation to the factor gradients examined within studies (McGarigal and Cushman, 2005).

A variety of woodland studies have examined patch contrast and landscape matrix effects, noting that the form and composition of the matrix can influence avian species occurrence, community composition, dynamics and relative predation within woods in North America (Sisk et al., 1997, McGarigal and McComb, 1995), Australia (Watson et al., 2005), Europe (Andren, 1992), and England (Bellamy et al., 2003), woodland ground-flora communities in England (Gove et al., 2004b, Gove et al., 2004a) and in determining the composition and dynamics of vertebrate communities within tropical forest remnants (Gascon et al., 1999).

The principal effects of the landscape matrix relate to two main areas, effects associated with patch contrast and edge-effects and effects associated with the proportion of habitat types occurring within a landscape, and the way species utilise "matrix" habitats in addition to the primary woodland habitats. Patch contrast relates to the degree of difference between the woodland and neighbouring patches. Where levels of contrast are high, as between structurally diverse woodland and a uniform arable field, edge-effects relating to microclimate are highest. Additionally contrast may incorporate measures of naturalness. Studies in Kent showed botanical species richness within ancient woodland margins was related to the naturalness and intensity of management of adjacent grasslands. Richness was higher where ancient woodland sites were adjacent to semi-natural, unimproved, grassland than to intensive arable land, higher contrast leading to more intense edge-effects (Gove et al., 2004a). Patch contrast also affects species movement and thus connectivity between patches, where contrast is low species are considered to be more likely to disperse between patches or to be able to persist or survive within such patches while dispersing, in comparison to species attempting to cross boundaries with high contrast. In tropical rainforest remnants the contrast between the matrix developing around rainforest remnants affected species use, species sensitive to fragmentation being more likely to use the matrix where it more closely resembled the original primary forest type (Laurance et al., 2002). Several tropical forest studies have indicated the beneficial effects of holding ecologically similar matrix adjacent to patches producing e.g. favourable microclimate or increased re-colonisation (Turner and Corlett, 1996).

The second principal area relating to landscape matrix is the ability of species to utilise matrix habitats in addition to woodland. At extremes species may either be completely confined to using woodland (specialists) or may use woodland in combination to matrix habitats (generalists) (Section 4.3.9). The composition of the matrix will therefore affect the way species utilise the landscape. As woodland fragmentation occurs and the proportion of matrix increases, the effects on specialist and generalist species will differ, and these effects will depend on the contrast between landscape matrix and woodland. In a study of tropical forest remnants, vertebrate species that utilised the matrix in addition to patch habitats, persisted or increased after fragmentation, while species that avoided the matrix reduced or disappeared (Gascon et al., 1999). In North American oakwoods avian species composition and abundance was found to differ within otherwise similar woodland patches located within different matrix habitats, due to the differing levels of matrix usage; from complete avoidance, to equal usage (Sisk et al., 1997).

Effects relating to the matrix affect the relationships between other landscape variables and species usage. One study, examining resident avian species within deciduous patches in a European coniferous forest, failed to observe an association between patch size and species presence and noted this may be because species were able to utilise the surrounding forest matrix, which only showed a relatively low contrast from deciduous forest, the importance of individual patch size being less where resources were also accessible within matrix habitats (Enoksson et al., 1995). Additionally an American study found low effects of isolation and habitat configuration on avian species within forested landscapes with lower levels of contrast between patches (McGarigal and McComb, 1995). An Australian study found differing woodland isolation effects on bird species richness between different matrix types (agricultural, urban and peri-urban) (Watson et al., 2005). In Andren's study of nest predation by corvids, predation rates in edges increased where the proportion of agricultural land increased, due to the agricultural matrix supporting populations of the corvid predator species (Andren, 1992), explaining why predator effects on nests are often most common in landscapes fragmented by agriculture rather than other land-uses (Chalfoun et al., 2002) and illustrating the complex interactions that may occur between patch shape and matrix composition. Other avian studies have reported matrix affects on population synchrony between patches, with implications for patch-level extinctions (Bellamy et al., 2003). Botanical reviews have noted that where woods are surrounded by highly contrasting habitats they may be detrimentally affected by generalist /

invasive species, which may colonise such patches (Murphy and Lovett-Doust, 2004). However studies have noted woodland edges may be relatively impermeable to such invasion by weed plant species from the agricultural matrix, beyond a narrow edge zone (Honnay et al., 2002b).

The majority of studies examining woodland fragmentation, especially temperate studies, have examined woods within lowland, agricultural and often arable, landscapes (Bellamy et al., 1996a, Hinsley et al., 1994, Peterken and Francis, 1999, Peterken and Game, 1984, Freemark and Merriam, 1986, Opdam et al., 1984). Many of these studies specifically selected such highly contrasting matrix habitats such that the approximation to the hostile "sea" of island biogeography theory would be approximated, hypothesising that under such situations the effects of patch area and isolation would be most apparent. A range of more recent studies however have highlighted the importance of matrix habitat in affecting potential species-landscape relationships, possibly negating such patch and isolation effects in certain landscapes (Enoksson et al., 1995, McGarigal and McComb, 1995) and highlighting the importance in clarifying the definition of species matrix usage ability (specialist and generalists), in predicting potential landscape patch effects. The results of these studies have shown that patch contrast and matrix composition are highly important for patch species presence and richness. The form of the matrix may heavily influence species dispersal and the accessibility of resources within habitat patches, altering patch occupancy in hostile matrix landscapes compared to more permeable landscapes. Such ideas are therefore linked to the concepts of functional isolation and connectivity, core area and edge-effects, key concepts being landscape hostility and contrast. While matrix effects may appear more important for highly mobile species such as birds and larger mammals, where species able to use multiple habitats can access different patches, the matrix also has implications for botanical species. Woodland flora may be able to survive and reproduce in certain matrix habitats, such as streamside, flushes or hedgerows holding the potential for landscapes with these habitats to show higher functional connectivity, aiding colonisation of new or recovered patches, while the variety of dispersal methods may also be affected by matrix type. Flora dispersal methods reliant on animals (ingestion, adhesion) will be directly affected by matrix effects on animal behaviour while other methods such as water and wind dispersion may be affected indirectly by the form of matrix encouraging or preventing such dispersal, habitats with differing levels of contrast being more or less likely to favour dispersal types. For example, highly managed watercourses within arable landscapes being less likely to provide suitable dispersal routes for woodland flora than natural streams while differing patches may affect air flows and the distances over which seeds disperse. In summary the relative favourability or hostility of the surrounding matrix may affect the patch quality. An identical patch may support richer, higher "quality" communities when occurring within a "favourable" matrix than within a "hostile" matrix, all other factors being equal but in such cases, where patch contrast is particularly low, the relative importance of patch isolation and size may be reduced in comparison to patches occurring in more hostile matrix types.

4.3.9 Habitat corridors, patch habitat quality and conservation potential

Habitat corridors have been defined as narrow strips surrounded by habitats of other types (Farina, 1998, Forman and Gordon, 1986) and are thought to influence species mobility and dispersal. Corridors are typically not of sufficient size or quality to be utilised to the same extent as habitat patches, but may be used by species for dispersal or may be occupied and utilised in a sub-optimal way, compared to habitat patches. Corridors also occur where functional, rather than structural, connectivity occurs. Modelling studies have indicated that corridors may be "virtual", not readily visible from landscape structure, being the result of various interacting landscape features funnelling species movement (Gustafson and Gardner, 1996, Vuilleumier and Prelaz-Droux, 2002, Pe'er et al., 2005). The concept of habitat corridors collectively spans issues of patch size, shape, contrast, connectivity and connectedness and are believed to be beneficial to aid species mobility and dispersal, although the concept is controversial, with mixed evidence and acceptance in the literature (Dawson, 1994, Dolman and Fuller, 2003, Dover, 2000, Mann and Plummer, 1995, Hobbs, 1992). Depending on the definition of habitat corridors and the method by which landscape data is collected during landscape studies, habitat corridors can be viewed simply as linear patches where the main effects on species presence are due to patch size, shape or core area. Additionally the confirmed presence of corridors in studies may depend on whether a binary island biogeography approach or a more mixed landscape mosaic approach has been taken.

A variety of studies have examined habitat corridors within woodland landscapes: (Haas, 1995, Tischendorf et al., 1998), as critical issues in conservation and planning: (Kirby, 1995, Dover, 2000, Dawson, 1994, Mann and Plummer, 1995, Hill and al, 1992), and as factors effecting species presence and diversity (Shirley and Smith, 2005). Dover (2000) notes the wide variety of definitions and intended uses of terms relating to habitat or wildlife corridors included in development plans in the UK and that these were typically focused on urban rather than rural areas (Dover, 2000). In examining the possible beneficial effects of corridors on woodland colonisation Dolman and Fuller (2003) note the literature contains "much confusion and apparently opposing views". They identify three areas where ambiguity can affect the results of a study by a lack of clear distinction between; (1) corridors that essentially allow movement of individuals dispersing from a population source within a "core" area to other suitable patches of habitat, (2) resident populations occupying linear habitats, that may allow range expansion by "percolation" and (3) the periodic use of linear habitats during foraging movements within an individuals home-range (Dolman and Fuller, 2003). The effect of corridors may therefore be highly species dependant and indeed the creation of habitat corridors to aid dispersal may actually act as a barrier to the dispersal of different species, and must be carefully considered (Kirby, 1995). In conclusion of their review Dolman and Fuller (2003) noted that "corridor networks of hedgerows and other linear features will help rapid colonisation of new woodlands by widespread and common species that would get there anyway, but will be of little help in the dispersal and colonisation of specialist and "core" species that are of greatest conservation concern". A recent study in England also concluded that dispersal of specialist flora would not be aided by hedgerows networks (McCollin et al., 2000). In reviewing flora dispersal and connectivity Murphy and Lovett-Doust noted several studies exist highlighting that continuous "corridor" features are not required to ensure connectivity, which may exist through other elements such as "stepping stone" habitat patches (Murphy and Lovett-Doust, 2004).

The concept of habitat corridors remains controversial. Their application may be different when examining mobile species compared to immobile flora. Over short timescales fauna may utilise corridors to move between patches, thus increasing connectivity, but the relationship may be complicated where species also utilise such corridors as resource habitat, and therefore their existence is less likely to be important for woodland specialist species (Section 4.3.11). The use of corridors by specialist woodland plants appears less likely. However, over longer timescales, habitats considered as corridors such as hedgerows or green lanes may serve as remnant patches supporting woodland flora and allowing future colonisation of any new woodland sites created adjacent to such features, rather than encouraging species movement over shorter timescales. In summary the concept has proved useful in promoting consideration of landscape connectivity, but when following a full landscape mosaic approach and incorporating assessment of matrix habitat usage and hostility the separate definition of habitat corridors may be unnecessary, already being incorporated in assessment of matrix favourability. However the use of the concept may remain useful in disseminating the results of such research or landscape planning to planners and the public.

4.3.10 Fragmentation effects and patch habitat quality: structure, heterogeneity and age

Theoretical predictions of habitat fragmentation effects differ relating to relative patch habitat quality. Predictions based upon island biogeography theories note the increased species richness on larger habitat patches may relate in part to increased habitat heterogeneity, size being indicative of enhanced patch habitat "quality". In contrast many early metapopulation studies, defining species population effects assumed all habitat patches to be of equal quality, but that patch size was linked to population size, in order to model patch size and isolation effects, a factor followed in many recent modelling studies. In woodland species-landscape observational work, many have incorporated measures of relative habitat "quality" at the patch scale, indicated by factors such as woodland structure and topography in order to compare the importance of "within-patch" quality against abiotic variables (patch size, shape) and landscape-level variables, in affecting species occurrence and richness, especially in studies of avian species richness or occurrence (McCollin, 1993, Enoksson et al., 1995, Watson et al., 2005, van Dorp and Opdam, 1987, Boecklen, 1986, Freemark and Merriam, 1986, Opdam et al., 1985, Bellamy et al., 1996a, Bennett et al., 2004, Hinsley et al., 1994). Rather fewer studies have specifically examined the association between within-patch quality and botanical species presence, but see: (Helliwell, 1976). Some relationships, however, are reported in studies primarily examining avian patch relationships e.g. (Bellamy et al., 1996a, van Dorp and Opdam, 1987, Opdam et al., 1985).

The potential for a wide variation in the internal characteristics, structure and hence ecological quality of woodland patches is a problem for much landscape research. Analysis may ignore within-patch variation, assuming spatial structure or abiotic patch factors will still become apparent (Opdam et al., 1984, Woolhouse, 1987). However this risks spatial configuration and landscape effects being swamped by within-patch variation. Studies may therefore attempt to control within-patch variation when selecting study sites, for example examining only patches of a similar woodland type or age, assuming these will be similar structurally, compositionally and in perceived habitat quality, for example choosing only mature broadleaved woodlands e.g. (Opdam et al., 1985, van Dorp and Opdam, 1987).

Several studies have recorded effects of within-patch variables on avian species richness and occurrence. In Holland research found that structural heterogeneity, variation in trunk diameter (linked to the number of different tree age stages present) and cover of shrub layers significantly contributed to the explanation of variation in bird species numbers between patches (van Dorp and Opdam, 1987). Significant effects have also been recorded of within-patch variables being incorporated in models explaining presence of individual avian species such as shrub density affecting *Pyrrhula pyrrhula* (bullfinch) and *Muscicapa striata* (spotted flycatcher) presence, and within woodland habitat diversity affecting *Prunella modularis* (dunnock) and *Carduelis chloris* (greenfinch) in lowland English woods (Hinsley et al., 1994).

Where work has identified the influence of habitat heterogeneity on species richness; levels of variation explained have varied from 12%-34% in bird studies (Bellamy et al., 1996a, Boecklen, 1986, Freemark and Merriam, 1986), and 29% and 34% in plant studies (Bastin and Thomas, 1999, Peterken and Game, 1984).

Several studies have also recorded associations between within-patch variables and botanical species presence or diversity. Woodland age, successional stage, management, topography and soil type affect botanical species colonisation by affecting recruitment arriving at the site. Additionally such factors will affect the growth and abundance of species within existing woodlands. Verheyen and Hermy noted colonisation of isolated new woodlands by ancient woodland flora species in Belgium was related to both the dispersal ability of plants and the developing habitat quality of the newly created woodland, older stands being more suitable for colonisation (Verheyen and Hermy, 2001). Another study in Belgium, of woodland ground-flora richness, found a significant interaction between patch area and patch age on species richness, the authors noting the effects could not be considered in isolation (Jacquemyn et al., 2001). Species richness was not related to area for woods younger than 51 years, following which there was a significant effect of increased area on enhanced species richness (Jacquemyn et al., 2001). The authors suggested the relationship was caused both by gradual colonisation of species over time and to the effect of increased structural and micro-environment diversity such that dispersal events into older or more established woods more successfully resulted in recruitment (Jacquemyn et al., 2001) (Fig 4.16). Interestingly however, in Lincolnshire woods Peterken and Game noted that ground-flora richness did not increase with age in secondary woods (Peterken and Game, 1984), perhaps because of the dominant effects of isolation in this hostile arable landscape. In finding no patch area effects on botanical richness in North American woods one study noted the effects of intense cultivation, in reducing micro-topographic diversity, may limit affects of area in secondary woods, meaning a mix of previous management and landscape effects may dictate relative importance (Matlack, 1994).

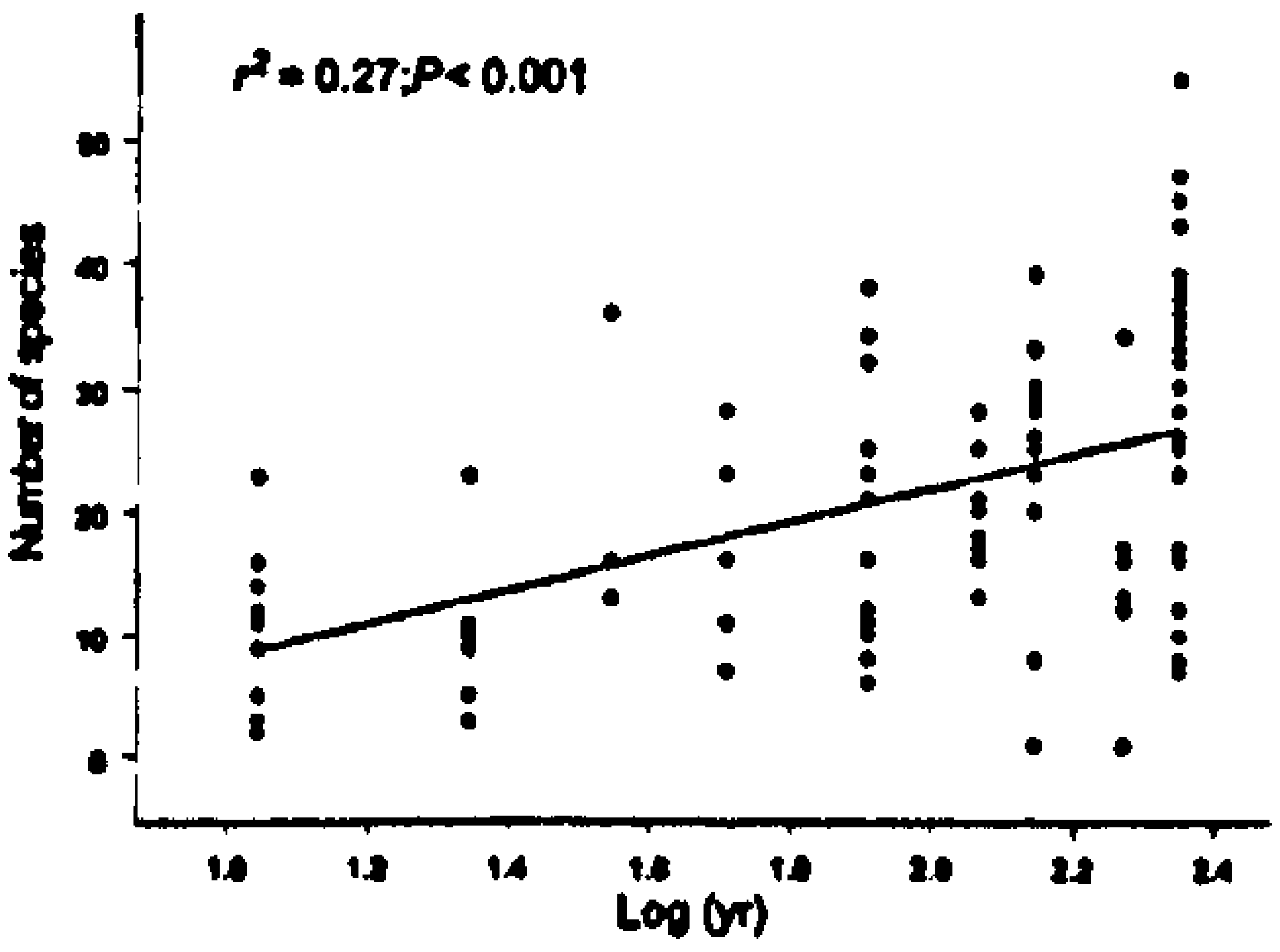


Figure 4.16
The relationship between woodland patch botanical species richness and patch age, for non-alluvial forest stands. Reproduced from (Jacquemyn et al., 2001).

Within established woods flora diversity and abundance may be related to within-patch features that are indicative of habitat quality or management. In Shropshire woods, a number of significant correlations were found between ground-flora richness and structural variables. Species richness was negatively correlated with the percentage tree cover (-0.27 to -0.44) and altitude range (-0.21 to -0.32) and positively correlated to the percentage cover of shrubs (0.22) and native trees (0.27 to 0.36) (Helliwell, 1976). In Lincolnshire Peterken and Game concluded that botanical species richness within ancient, long-established, woods was "determined by the range of soils and variety of vegetation structure" rather than affected by factors such as isolation, in contrast to developing secondary woods (Peterken and Game, 1984). In their study the number of soil types and ride length explained 42% of the variance in species richness (Peterken and Game, 1984). Surprisingly flora richness was not significantly affected by the proportion of replanted conifers within ancient woods suggesting diversity was retained after replanting (Peterken and Game, 1984). Such statements assume that these long established ancient woods developed or were colonised when sources of colonists was not a limiting factor – i.e. the proportion of suitable habitat in the surrounding landscape was high (Section 4.3.3) or the surrounding matrix was of low contrast (Section 4.3.8), and therefore internal features dominate the composition of these woods rather than landscape scale inter-patch variables. However while this may apply to immobile and long persisting flora similar relationships are unlikely for mobile fauna.

A variety of studies have recorded associations between within-patch abiotic variables such as woodland size and shape and woodland structural or management features. Several studies have found woodland size to be correlated to features of vegetation structure or habitat quality, noting larger woods are more heterogeneous and diverse (Bellamy et al., 1996a, van Dorp and Opdam, 1987). In lowland England wood area was found to be correlated with canopy density, the number of features corresponding to structural heterogeneity and the number of tracks (Bellamy et al., 1996a). Peterken and Game noted measures of habitat diversity including ride length, soil type and soil pH range were correlated with patch size (Peterken and Game, 1984), while Peterken and Francis found that structural features such as rides and evidence of management was more frequent in larger woods and significantly affected the diversity of open space species (Peterken and Francis, 1999). In Holland of 12 recorded structure variables many were correlated with woodland size, being presence of water (streams, ditch or pond), presence of conifers, forest structure layers, number of forest communities, tree size DBH, number of tree species, tree density, and presence of saplings (van Dorp and Opdam, 1987). Also in Holland wood size was positively related to structural heterogeneity and shrub species diversity (Opdam et al., 1985). A boreal woodland study found a strong effect of the number of deciduous host trees per stand on explaining bryophyte species presence (Lobel et al., 2006) implying potential use of cover of deciduous trees as a substitute for bryophyte diversity. However, some studies have failed to find such associations. One English study examined correlations between patch area and 28 measures of woodland habitat structure, but found no significant correlations, although the study examined only a limited number of patches within a relatively small geographic area (McCollin, 1993).

These relationships support the frequently observed species-area relationship of enhanced richness in larger woodland stands, but also note that where woods are particularly structurally diverse or variable they may deviate from such general species-area trends. The structural diversity typical of mature broadleaved woodland will be affected by management history and site conditions but can take significant periods of time to develop, perhaps taking 50-100 years to provide the necessary structure and composition for specialist woodland bird species

(Bellamy and Hinsley, 2004), while Jacquemyn noted that forest succession to saturated woodland patches may take at least 200yrs, especially for less diverse woodlands on non-alluvial soils (Jacquemyn et al., 2001). Therefore issues of woodland habitat fragmentation and quality must incorporate relative patch quality and age in considering potential landscape effects.

4.3.11 Fragmentation effects, population form and species life-history traits

Not all species occurring within woodland habitat exhibit similar forms of population structure or life-history traits. Theoretical studies indicate that different forms of population structure may respond differently to fragmentation and landscape spatial structure. However population form is not easily measured, and may often be inferred from observations on the association between species occurrence and landscape structure leading to circular arguments. Populations are principally defined by their potential to interbreed and therefore critical factors to define population boundaries are mixing of individuals, the colonisation or re-colonisation of patches and the relative dispersal distances of species compared to local distributions.

Mixed evidence exists for the identification of specific populations such as metapopulations within real landscapes; evidence from woodland studies is also variable. Several recent reviews have concluded that typically plants do not occur as metapopulations (Freckleton and Watkinson, 2002, Bullock et al., 2002, Husband and Barrett, 1996). In particular plants are considered likely to occur in patchy remnant populations where they may persist due to their long lifespans and vegetative reproduction, meaning metapopulation dynamics may not be observed in woodland specialists, especially over shorter timescales. With regard to plants therefore the gradual accruement of species may be considered more similar to the classical island biogeography theory than metapopulation dynamics, impacting on the colonisation effect of newly created or restored habitat patches, while spatial patch and landscape considerations may be less important for regular extinction dynamics.

Several studies have observed or cited evidence suggesting metapopulation structure within populations for botanical species (Mouflis and Buckley, 2004), birds (Opdam, 1991) and bryophytes (Lobel et al., 2006). In England, negative correlations between ancient woodland ground-flora richness and ancient woodland nearest-neighbour isolation were interpreted as indicating low ancient woodland isolation levels promoted retention of flora richness, possibly due to allowing migration between sites within metapopulations (Mouflis and Buckley, 2004). Such relationships would support potential metapopulation structure in ancient woodland plant populations. Interestingly the study did not find that landscape ancient woodland indicator species richness was associated with low isolation between ancient woodland sites and secondary woodland sites (Mouflis and Buckley, 2004). This could indicate the presence of secondary woods does not aid in the long term retention of species within ancient woodland sites, by a lack of migration between such sites, or that other factors are occurring in landscapes where ancient woodland sites show low isolation and higher ancient woodland flora richness. Secondary woods may not represent sources of ancient woodland colonists due to a lack of suitable structure or habitat quality, or due to the relatively high turnover (woodland destruction and creation) in comparison to ancient woodland sites, meaning that populations do not build up. These results contrast interestingly with those of individual woods in Lincolnshire where the isolation of ancient woodland sites was not considered to be a significant factor in affecting ground-flora richness (Peterken and Game, 1984). Peterken and Game indicated that flora richness and diversity was determined by patch factors primarily, presumably acting upon the species as they originally colonised the site within historic, richer landscapes and indicating such populations currently occur as remnant, isolated populations with very little or no mixing between patches. Studies have provided evidence that patch area and isolation may affect persistence of plant populations by affecting colonization and the relative reproductive success of small populations in small patches (Jacquemyn et al., 2002)

A variety of evidence has emerged for more mobile fauna groups. Researchers typically interpret strong associations between woodland dependent bird species and the isolation or area of woodland occurring around focal woods patches as evidence for potential metapopulation structures e.g. (Bennett et al., 2004). Opdam reviewed a range of avian-landscape studies and concluded that metapopulations occurred; extinction probability of patches was related to area, which was also proportional to species patch population size (Opdam, 1991). McCollin, in his study in England took the strong relationship between landscape isolation variables and bird species richness as evidence of the existence of a metapopulation form between the core-satellite and patchy population models, noting that a larger areas of woodland outside his study area was likely to represent a source of species (McCollin, 1993). One study notably illustrated a link between patch size and patch-level species extinction, suggesting larger populations in bigger woods were less likely to go extinct, but did not observe correlations between patch colonisation and patch variables, although interpretations of such dynamics are limited by the 3yr study duration (Bennett et al., 2004).

Where population events have not been able to be observed, differing fragmentation effects may still be seen in species with different life history traits. Modelling and theoretical studies predict that the relationship between species dispersal and typical inter-patch distances is critical to potential fragmentation effects. A study of ground-flora in Belgium found species with short distance dispersal (autochores and myrmechores) had more aggregated distributions among woods than species with long distance dispersal (endozoochores and anemochores) (Jacquemyn et al., 2001). One study of remnant vegetation fragments in an urban area noted how "late successional" woodland ground-flora species were most successfully modelled by factors such as patch area and isolation, compared to early or mid-successional species with different life history traits (Bastin and Thomas, 1999).

Recent analysis of fragmentation effects has noted that effects are predicted to differ between generalist species, able to use many features in a landscape matrix as habitat, than specialists which are fully restricted to single habitats. Several woodland studies have confirmed the differing effects of fragmentation when such species are compared. A European study found that when all woodland bird species were examined woodland isolation did not significantly affect species, but when specialists, restricted to mature deciduous woods were examined, isolation effects were observed (Opdam et al., 1985). One study of bird occurrence in British woods found no association with isolation (Helliwell, 1976), although this was criticised, noting too many species were examined in relation to presence within the woodland patches, without discriminating woodland and generalist species (Opdam et al., 1985), while additionally the heterogeneous woods examined may have caused variation in structural habitat features to have overwhelmed any landscape factors (Opdam et al., 1984). A study of birds in lowland English woods noted that patch and landscape effects were stronger predictors of species richness in woodland dependent birds than of the avifauna as a whole (Bennett et al., 2004). The study also showed the differences in landscape use by groups of woodland dependent resident, as opposed to migratory, birds. Migrants responded to aspects of regional environmental gradients in 5x5km landscapes, in addition to patch features, across an area of lowland England, in contrast to residents which responded primarily to patch features, indicating the wider ranging behaviours of migrant species (Bennett et al., 2004).

In summary, direct evidence for the effects of species population form and species life history traits on potential landscape-scale fragmentation effects remains varied. Studies confirm that life history traits such as short distance dispersal will lead to increased potential fragmentation affects and affect species distributions within networks and that specialist species will be more affected by fragmentation than generalist species. However evidence for the existence of different population forms remains varied. Debate remains about typical flora population type. Studies exist confirming or indicating potential metapopulations (Harrison and Taylor, 1997, Lobel et al., 2006, Valverde and Silvertown, 1997), while much evidence remains that flora may exist in truly isolated remnant population forms or as patchy population (Freckleton and Watkinson, 2003, Eriksson, 1996). In mobile fauna many authors have noted potential for metapopulations to occur, although longer-term studies are required. It has been noted that island biogeography theory remains a useful system with which to study habitat fragmentation effects if specialist species rather than generalists are examined (Cook et al., 2002).

4.4 Discussion: Patch character, landscape configuration and context: evidence, trends and application in the study and conservation of woodland habitats

A wide variety of factors have been shown to influence woodland patch habitat quality and patches use by species. These include patch area, shape and its relation to the surrounding matrix, while studies have also shown the importance of within-patch measures of quality including time since creation / disturbance, management and that the potential effects may be affected by study species life history traits such as dispersal ability and relative habitat specialism. The majority of observational fragmentation studies have emphasised the effects of the principal landscape characteristics of patch size and isolation, following from classic metapopulation and island biogeography theory, many studies confirming the predictions of neutral landscape modelling that these metrics would account for most of the variation in species landscape processes such as dispersal success e.g. (Bennett et al., 2004). Small, isolated woodland fragments will retain less species than larger, more well-connected fragments and the species that remain will not be a random selection of the original potential species (Dolman and Fuller, 2003). However these effects may depend on the species examined and critically on the way in which landscape elements are quantified. Generalisation of reactions to landscapes structure and habitat fragmentation are limited due to the effects of scale on the definition of patch and landscape boundaries, while observed effects may be species specific, or related to a number of life history traits. However with these reservations acknowledged the body of research now conducted allows a number of trends to be observed. A sufficient variety of observational and experimental studies, have been conducted to allow the predictions from landscape ecology and modelling studies to be examined. Evidence suggests effects will be strongest in landscapes with lower covers of habitats and where species hold limited dispersal characteristics in relation to inter-patch distances. Strong effects have been attributed to study scale in relation to species perception and the accuracy with which landscape variables capture landscape structure from a species perspective (With, 1994, Wiens and Milne, 1989). Many of the studies examined were able to report strong relationships between individual elements of landscape structure and species richness or individual species presence and abundance (Bellamy et al., 1996a, Freemark and Merriam, 1986, Peterken and Game, 1984, Usher and al, 1992). Additionally many studies developed mixed models where combined factors of different landscape structure elements were related to species variables (Jacquemyn et al., 2002, McCollin, 1993). While older studies examined effects due to selected fragmentation parameters such as size and isolation, recent studies are highlighting the complex interactions that may occur at different scales affecting species, attempting to extract the effects attributable to structure at plot, patch and landscape scale (Cushman and McGarigal, 2002). An approach recently followed in variance partioning of ground-flora data within Britain (Corney et al.,

2006) and in examining patch, local and regional effects on avian composition in England (Bennett et al., 2004).

The success of predictive variables in explaining species richness or presence will largely depend to the degree to which they capture aspects of the landscape that are directly relevant to the species examined. Because many of these effects vary in strength and relative importance between landscapes elements and between study species, some of the most comparable results are in interpreting the differing effects of landscape structure on species richness.

Avian Studies Research examining the effects of habitat fragmentation / spatial structure in determining avian species richness frequently identifies combined effects of several patch or landscape variables. Analysis may highlight the importance of within-patch variables, woodland structure or topography, abiotic patch variables such as patch area and shape, between patch variable such as patch isolation and landscape level variables such as proportion of woodland cover. Studies have attempted to define the relative importance of these separate variables or may present models where combinations of variables collectively explain species occurrences. Early work examining avian communities within different landscapes differing in fragmentation and patch configuration found that woodland size was the best predictor of bird species richness and occurrence, although the study attempted to select relatively similar woods to control excessive variation in within-patch features (van Dorp and Opdam, 1987). A study in North America found that area alone explained 50% of the variation in patch bird species richness with a further 17% of variation explained by habitat heterogeneity factors (Freemark and Merriam, 1986). An American study indicated than once patch size was accounted for that within-patch habitat heterogeneity / complexity was an important factor in determining patch species richness (Boecklen, 1986). A study of woodland bird species occurrence in North Humberside found the presence and number of species was principally related to patch level factors, rather than "within-patch" variables of vegetation structure (McCollin, 1993). 84% of the variation in species richness of woodland birds was explained by a model based upon isolation, area and shape (McCollin, 1993). In lowland England a study found variables relating directly to the patch were more important in explaining the numbers of resident woodland bird species present than variables relating to the surrounding landscape, with 68%-74% of species variation explained by patch features (Bellamy et al., 1996a). In designing their study methodology and selecting landscape variables to study common resident woodland bird species in lowland England Bellamy et al (2003) noted that in addition to patch area the area of woodland and the length of hedgerow within 1km of woodland had shown the strongest effects on bird species from a number of available measures used in previous studies e.g. (Hinsley et al., 1995a, Hinsley et al., 1995b, Bellamy et al., 1996a, Bellamy et al., 1996b). A recent study in lowland England explained 70% of the variation in woodland dependant birds by a regression model incorporating patch area and the length of hedges and area of woodland within 1km of the wood patch (Bennett et al., 2004). Opdam reviewed a number of landscape studies of bird species presence and richness studies and noted that in most studies the effect of patch size was typically more important than isolation in explaining presence or richness but noted that this may be affected by scale and the particular species being studied (Opdam, 1991). Research has therefore typically been successful in explaining variation in species numbers due to landscape composition and patch features. These results are affected by study scale. The importance of patch level effects has clearly been shown, and that composition in the local landscape can have strong impacts on species presence. Research has successfully shown effects of spatial structure at larger spatial scales in 10km grids (100km^2) (Bailey et al., 2002, Radford et al., 2005). However, while the importance of habitat cover has been shown in these larger landscapes, e.g. explaining 55%-60% of variation in woodland birds in 100km^2 landscapes in Australia, mean patch shape complexity and woodland aggregation only explained 10%, while bio-geographic factors, elevation range and geographic position explained 14% (Radford et al., 2005). Additionally a study aggregating values at a 1km^2 scale also returned relatively low explanation of richness by spatial variables and 66% of variation remained unexplained at this scale (Titeux et al., 2004). These studies suggest that information relevant to the way that bird species perceive their environment are lost when landscape composition values are aggregated and simplified to the 100km or 1km scale. More successful models have been created when examining both individual patch variables and the surrounding landscape at smaller scales of 3.1km^2 (Bellamy et al., 1996a, Bennett et al., 2004, Hinsley et al., 1994) 6.25km^2 (Villard et al., 1999)and 12.5km^2 (McCollin, 1993), believed to reflect distances typically covered by woodland bird species (Fig 4.17). This importance of study scale was reflected within a review examining the relevance of including landscape factors in species studies. The review found that landscape factors, such as area of suitable habitat within the vicinity of patches, was significant in predicting species presence and abundance for vertebrates, but not for most invertebrates, and that these effects were likely to due to the difficulties in attempting to catch "landscape" variables for small immobile invertebrate species (Mazerolle and Villard, 1999). Therefore the commonly used scale distances within patch studies measuring focal search distance of 1km, appear to closely match the range at which bird species will typically react to landscape structure, being principally affected by individual patch characteristics and the form and structure of patches within the surrounding 1-2km (310ha-1,250ha).

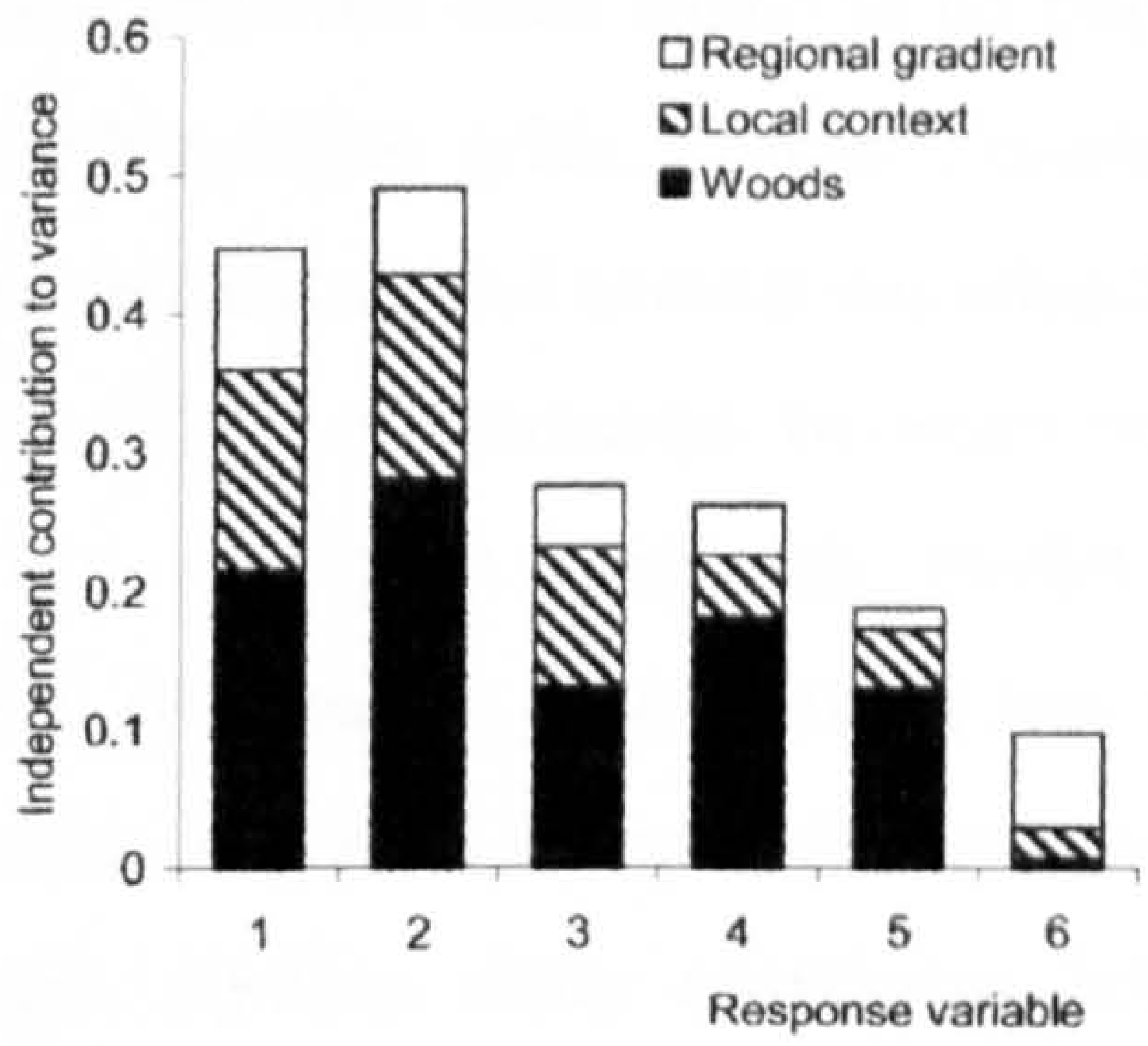


Figure 4.17
A study examining the extent to which within-patch / woods, local landscape context and regional gradients in environmental values affect avian species richness in lowland England. The total independent variance attributed to each variable is plotted for (1) Species richness of woodland migrants (2) species richness of woodland dependants (3) species richness of edge species (4) turnover of woodland migrants (5) turnover of woodland dependants (6) turnover of edge species. Reproduced from (Bennett et al., 2004)

Botanical Studies Research of woodland flora has also confirmed significant effects of patch and landscape variables on species occurrence and richness. Plant distributions have been explained by combined effects of patch area, isolation and within-patch habitat factors (Jacquemyn et al., 2002). Research examining species richness has highlighted the overall importance of patch area, with additional significant contributions from within-patch measures of patch diversity (Bastin and Thomas, 1999, Jacquemyn et al., 2001, Peterken and Game, 1984, Usher and al, 1992). Studies have also typically shown negative effects of patch isolation, although this is often of lower explanatory power than these pervious variables, e.g. (Peterken and Game, 1984, Bastin and Thomas, 1999, Jacquemyn et al., 2003). However some work have shown isolation to be highly important, occasionally more so than patch area (Matlack, 1994). This study noted, that in examining abandoned fields, that previous agricultural activity may have homogenised these sites, potentially removing some of the beneficial associations between patches size and micro-topographic diversity, leading to enhanced effects of isolation.

In one English study, correlations between ancient woodland indicator flora richness and woodland presence, found that overall species richness in landscapes was most affected by the number of ancient woodland patches, the area of ancient woodland, the area of broadleaved woodland and was negatively affected by ancient woodland isolation (Mouflis and Buckley, 2004). The number of patches was found to remain important even after accounting for its association with the area of ancient woodland present. Landscapes with more frequent ancient woods contained higher indicator species richness (Mouflis and Buckley, 2004). This research points to a clear benefit of multiple patches to woodland conservation. Potential causes could derive from metapopulation theory or the association of differing habitat quality and environmental conditions in multiple patches driving enhanced diversity. However, where such work relies on atlas data there is always the danger that richer wooded landscapes were visited preferentially resulting in higher species records, compared to less wooded landscapes.

The scale at which patch fragmentation has been found to be important to botanical species has varied. One study conducted within 10km grids, 100km^2, examined the relationship between woodland dependent species and woodland and ancient woodland fragmentation across England (Bailey et al., 2002). The study found relationships between modelled bird distributions and woodland fragmentation but was unable to accurately model invertebrate or plant species distributions (Bailey et al., 2002), presumably because these less mobile species did not respond to landscape structure at this scale, or at least not to variables summarised to the coarse scale of 10km grid squares. Interestingly, another study examining 10km landscape grids observed relationships between ancient woodland isolation values and ground-flora richness and suggested this may be due to the positive effects of patch aggregation maintaining ground-flora diversity at this scale (Mouflis and Buckley, 2004). Additional insight into applicable scales for botanical studies can be gained by examining the range of inter-patch distances examined within botanical studies reporting significant patch size or isolation effects. These ranged from 10-480m within an American study (Matlack, 1994). Mean nearest neighbour distances for English ancient woods are 739m, and this dropped to a mean of 350m in local landscapes that held high richness of ancient woodland indicator species (Mouflis and Buckley, 2004). Other authors have not considered actual distances but have classified woods as isolated when more than 10m (Peterken and Game, 1984) or 100m (Jacquemyn et al., 2003) from potential source woods. Therefore, within the range of woodland landscapes sizes, with differing woodland isolation and patch size levels a range of patch and landscape habitat fragmentation effects have been recorded on botanical richness, which are of use in considering conservation planning.

4.5 Chapter Summary

Fragmentation effects and conservation

- Many theoretical predictions have been confirmed by observational and experimental studies
- Research has confirmed fragmentation effects occur in English wooded landscapes
- Several features of woodland spatial configuration and landscape structure can reliably be associated with species richness, and potential conservation interest, in English woods.
- Within typical English landscapes where woodland cover is low (<10%) the landscape is likely to have suffered losses of woodland dependent species and to hold reduced species compared to more wooded landscapes, additionally patch size, isolation and configuration effects are likely to be important to species distribution and abundance.
- Features promoting higher species richness and higher populations of potential woodland colonists to move to additional new woodland sites are: larger woods, compact shaped woods, and local landscapes with more frequent woods with higher woodland connectivity (functional or structural).
- Studies have indicated the importance of within-patch habitat quality factors on biodiversity levels., these measures may be related to patch abiotic and landscape factors, and interact with spatial configuration effects in complex ways
- Patch quality factors including woodland maturity, and woodland structural and microhabitat diversity favour enhanced richness
- Effects of fragmentation and potential edge-effects are likely to be most extreme where the contrast to the landscape matrix habitat is high
- Thresholds have been suggested whereby woods will have reached levels of diversity if of a certain size, or will be connected if below a certain isolation level.
- Reported fragmentation effects are likely to occur across a number of species groups, with landscape configuration impacting on populations even if true metapopulations do not occur, however exact effects and their intensity will vary with study scale and species, critical features being species area requirements and dispersal ability
- At scales where landscape composition changes at patterns beyond the scale of species dispersal ability or area requirements, fragmentation will effectively be too high and species will respond only to immediate patch features, conversely where landscapes are highly connected species will respond to overall landscapes features rather than individual patch features
- Although habitat fragmentation does not always lead to populations holding a metapopulation structure, it will still have important effects on the spatial structuring of populations

Knowledge limitations and future research

- Despite limitations to research in this field a range of factors can be extracted that are relevant to future fragmentation work and to research aiming to address the detrimental effects of ongoing woodland fragmentation through targeted, landscape-scale conservation action.
- Research has suggested the use of generalisations based on species guild structure and life history types among species groups to link landscape structure to biodiversity effects.
- Practical applications are limited by the range of study organisms currently examined, mainly woodland ground-flora and avian communities.

- Studies have confirmed expectations from theoretical modelling of hypothetical species increasing confidence in the generality of such relationships
- Research has typically been conducted in lowland agricultural landscapes with high contrast between woodland and matrix, and low levels of woodland cover
- Only rarely were studies concerned with deducing the form of species population structure being examined
- Observational studies are limited by the range of study landscapes currently examined, in particular a lack of fragmentation studies in upland habitats within the UK or from landscapes with higher local proportions of woodland cover
- Theory and suggestions from individual studies suggest that future examination and application at multiple scales to model species landscape / patch use would be fruitful
- In considering the broader effects of habitat fragmentation from a conservation perspective limitations are apparent from many current observational studies, while observational research of current landscape form can deduce current species landscape relationships, due to fragmentation already having occurred, the species most sensitive to habitat fragmentation will already have been lost or will be very rare in a landscape - deriving predictions from current distributions will be biased
- Ideally using underlying driving patterns of species diversity will be preferable to exmaine, ideally applicable at multiple scales
- The relationship between within-patch habitat quality and patch and landscape abiotic / structure factors warrants further research
- Patch area and isolation factors may be less relevant and within-patch habitat quality and management more relevant to long lived sessile organisms in contrast to relatively short-lived and mobile species such as birds
- Certain species can persist in landscapes even after isolation simply due to their longevity, and may be surviving in areas within which they can no longer successfully reproduce, for example due to a lack of pollinators, dispersals of seeds or climate effects

Chapter 5
Woodland conservation ecology

A literature review of ancient woodland, Upland Oakwoods, woodland ecology and conservation

5.1 Introduction

This chapter reviews woodland ecology, with an emphasis on Ancient Woodlands and Upland Oakwoods. The importance of ancient woodland sites is discussed and aspects of woodland ecology relating to colonisation and succession are reviewed. The ecology of species groups characteristic of, or dependent on, Upland Oakwoods are discussed and important features for these species are highlighted. Finally current research relating to the creation, restoration and enhancement of woodlands, with emphasis on Upland Oakwoods and ancient woodland sites, are discussed. Features that are of potential use in the prioritisation of sites within spatial conservation strategies are highlighted.

5.2 Woodland classification and ancient woodland site status

5.2.1 Woodland site classification

The UK has a long tradition of descriptive woodland classification (Moss et al., 1910, Tansley, 1939). In recent decades woodland communities have been described and classified under various systems (Peterken, 1993, JNCC, 1993, Rodwell, 1991, Peterken, 1996). These focus on defining naturalness, and categorise relationships between vegetation and underlying edaphic, topographic or management features (Table 5.1, Fig 5.1).

Table 5.1
Descriptive features used in the classification of woodland sites

Feature	Description
Canopy	The dominant canopy type is frequently used as the main feature to classify woodland. This can be identified to varying levels of complexity from broad classes such as broadleaved, coniferous or mixed, to more detailed descriptions based on dominant species at the stand level or NVC communities
Ground-flora	The range of species dominant in the ground-flora of the woodland
Naturalness (Origin)	A woodland site may be classified by its perceived naturalness or from evidence of its origin – for example as a result of natural woodland colonisation processes or as a direct result of human management i.e. planting
Longevity / history	The inferred history of a site may be used to define woodland categories such as recent, secondary, long-established and ancient woodland

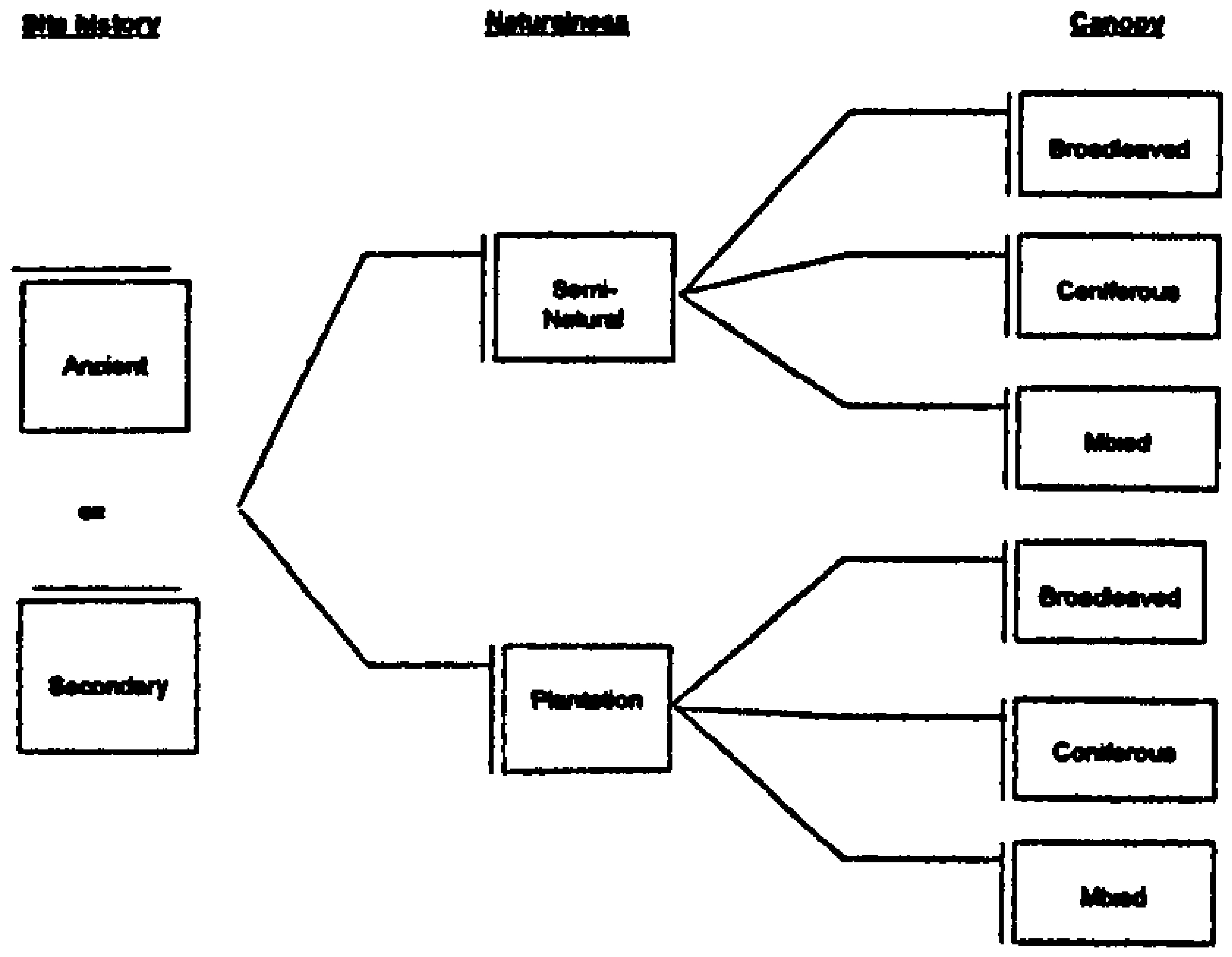


Figure 5.1
Classification of woodland sites by history, naturalness and canopy dominance. Ancient = Woodland sites known or inferred to have existed since at least 1600AD, Secondary = woodland known or inferred to have developed in recent times (after 1600AD), Semi-natural = woodland dominated by semi-natural process such as colonisation and regeneration, Plantation = derived from planting of tree species, Broadleaved = dominated by broadleaved, deciduous tree species, Coniferous = dominated by conifer tree species, Mixed = dominated by a mixture of coniferous and deciduous trees.

Classifications are often utilised to summarise site ecological interest (Fig 5.1). Higher naturalness levels and increased longevity are ecologically valuable, and worthy of conservation (Peterken, 1977a). During the 1970's there was increased awareness of the conservation importance of long-established woods that had remained following the decline in importance of local woodland products, and recent intensification of commercial, conifer-based forestry. The term "Ancient Woodland" was used for sites where it was known, or inferred, there had been continuous woodland cover since AD 1600 (Spencer and Kirby, 1992, Peterken, 1977a, Kirby and Goldberg, 2005). In Europe however ancient woods may be considered to be from 200 to 400 years old depending on data sources (Wulf, 2004, Bossuyt et al., 2002). An Ancient Woodland Inventory (AWInv) was produced for each county in England and Wales detailing the locations of these sites (Spencer and Kirby, 1992). Sites were located by combining a range of information including historical maps and archaeological evidence. In many areas early historic maps were unavailable and a combination of the earliest available maps and additional information from place names or site surveys were utilised (Kirby and Goldberg, 2005). In England Ordnance survey (OS) maps and aerial photographs were used to plot boundaries and the extent of areas considered Ancient Semi-Natural Woodland (ASNW) or Plantation on Ancient Woodland Site (PAWS). Pasture woodland and woods less than 2ha were not included (Kirby and Goldberg, 2005). During the late 1990's the inventory was digitised and became available in Geographic Information System (GIS) format (Reid et al., 1999). Great Britain is currently the only European state to have completed a review of ancient woodland sites (Wulf, 2003).

The creation of the inventory allowed the relative conservation importance of woods to be assessed on a county or national level. The classifications however are a simplification of the status by which individual woodland sites could have been classified at various points in their history (Table 5.2). There is therefore variation in the longevity of site conditions that may have existed within sites identified within the broad categories. The implications are that while current site classification provides an accurate assessment of current ecological conditions, and ancient woodland status confers knowledge that forms of woodland have existed at the site for long periods, the exact ecological conditions present at the site over time cannot be entirely inferred from current woodland cover. The current site classification of woodland will impact greatly on its current ecological value but may not always reflect its long-term management history.

5.2.2 Ancient woodland and conservation

5.2.2.1 Woodland cover and Ancient Woodland extent

England is not well wooded in comparison to other European countries, with current woodland cover standing at 8.6% (1.1 million ha) (Anon, 2005a). Ancient woodland covers 2.6% of England (Pryor, 2003, Peterken, 2000b, Thomas et al., 1997). Between 39-42% of the Ancient Woodland resource has been converted to PAWS in England (Pryor, 2003, Pryor and Smith, 2002) (Table 5.3). Larger ancient woodland sites are more likely to have been replanted (Spencer and Kirby, 1992).

Table 5.3
Extent of Semi-Natural (SN) and Ancient Woodland (AW) site types. ASNW = Ancient semi-natural woodland, PAWS = plantation on ancient woodland site, OSNW = other semi-natural woodland, AW = Ancient woodland. ASNW-NIWT, PAWS-NIWT and total AW-NIWT from combined analysis of ancient woodland and National Inventory of Woodland and Trees (NIWT) data. Source; (Thomas Kirby and Reid 1997, Pryor 2003.; Pryor and Smith 2002).

Category	England	Wales	Scotland	Britain
ASNW	206 000	30 700	89 100	325 800
ASNW-NIWT	193 500	27 000	64 500	285 000
PAWS	135 100	29 900	59 100	224 100
PAWS-NIWT	140 100	24 700	54 700	219 600
OSNW	209 800	51 700	44 000	305 500
Total AW	341 100	60 600	148 200	549 900
Total AW-NIWT	333 600	51 700	119 300	504 600
Total SN	415 800	82 400	133 100	631 300

5.2.2.2 The ecological interest of Ancient Woodland Sites

Ancient Woodland sites are of considerable importance for UK woodland conservation (Marren, 1992, Spencer and Kirby, 1992, Thomas et al., 1997, Peterken, 1996, Rackham, 2003, Peterken, 1977a, Kirby and Goldberg, 2005). The continuity of woodland cover and longevity of management has resulted in high levels of ecological interest, compared to more recently developed woodlands. Ground-flora species differ between ancient woodlands and recent woodlands (Dzwonko, 2001) while studies have shown Ancient sites are richer in ground-flora and beetle fauna than recently developed woods (Peterken and Game, 1984, Assman, 1999, Lawesson et al., 1998). Ancient sites hold a higher richness of specialist ancient forest ground-flora, than recent forest sites (Bossuyt and Hermy, 2001), and contain rare flora, less likely to be found in recent woodland sites (Brunet, 1993).

Table 5.2
Potential variations in the range of woodland sites considered to be "Ancient Woodland". Natural = dominated entirely by natural process of colonisation and regeneration without the effects of human management, Semi-natural = dominated by natural process and native species but may show limited influence of management such as felling and coppicing, Plantation = dominated by the influence of regular planting and felling or tree crops

Woodland history periods				
Post glaciation	c. 1600AD	1600AD to 1800AD	1800AD to Present	Summary of woodland site history and conditions
Natural	Natural	Natural	Natural	True, undisturbed original woodland cover ancient "wildwood" –unlikely to exist to any large extent in the UK
Natural	Natural / Semi-Natural	Semi-Natural	Semi-Natural	Site may have existed as true semi-natural wildwood in 1600 or already been influenced by traditional woodland management, but has retained general semi-natural conditions.
Natural	Natural / Semi-Natural	Semi-Natural	Plantation	Woodland may have remained semi-natural with or without management until relatively recent times when converted to plantation cover.
Natural	Natural / Semi-Natural	Plantation	Plantation	Woodland may have been natural or semi-natural but managed in 1600 but subsequently converted to plantations, such that sites hold a relatively long history of such plantation use, which is still reflected in current use.
Natural	Natural / Semi-Natural	Plantation	Semi-Natural	Woodland cover may have semi-natural origin, but have been converted to plantation in historic times and more recently reverted to semi-natural cover conditions.
Open ground	Natural	Natural	Natural	Originally open ground habitats but developed as natural woodland cover since c.1600AD
Open ground	Natural / Semi-Natural	Semi-Natural	Semi-Natural	Originally open ground habitats but developed as true natural or managed semi-natural woodland cover since c.1600AD
Open ground	Natural / Semi-Natural	Semi-Natural	Plantation	Originally open ground habitats but developed as true natural or managed semi-natural woodland semi-natural woodland cover since c.1600AD and converted into plantation cover in recent history.
Open ground	Natural / Semi-Natural	Plantation	Plantation	Originally open ground habitats, developed as natural or semi-natural woodland by 1600 but subsequently converted to plantations, such that sites hold a relatively long history of such plantation use, which is still reflected in current use.
Open ground	Natural / Semi-Natural	Plantation	Semi-Natural	Originally open ground habitats, developed as natural or semi-natural woodland by 1600, but have been converted to plantation in historic times and more recently reverted to semi-natural cover conditions.

Sites may also have soil profiles typical of undisturbed forests (Spencer and Kirby, 1992). The distinction between ancient and recent woodland flora and the acknowledged importance of Ancient woodland sites may be less clear however where woodlands have been grazed over long periods (as may occur in the uplands), where the majority of secondary woods occur close to ancient woodland sites (allowing their colonisation), where refuges exist for woodland species beyond woodland sites (where species may persist through periods of woodland absence before colonising secondary woods), and where an oceanic climate or lower intensity land uses allows species to occur beyond woodland sites, within the landscape matrix (Peterken, 1974, Spencer and Kirby, 1992, Peterken, 1977a).

5.2.2.3 Ancient Woodland site and landscape characteristics

Ancient woodland size and frequency varies across the UK (Spencer and Kirby, 1992), influenced by landscape form and local history and intensity of woodland use. Important factors include: density of human populations, importance and value of land and ease of transport. Additionally presence, by definition, is affected by levels of map and archaeological data. In the lowlands ancient woods may occur as compact sites at parish boundaries, but in the uplands often follow topographic features such as steep slopes, resulting in smaller linear sites (Spencer and Kirby, 1992). The association between inaccessible topography such as cliffs and rock exposures and long-established woodland or ancient tree cover is a widespread phenomenon (Larson et al., 2000). Sites out of the influence of man and domestic stock allow growth of ancient trees, spanning centuries and perhaps across periods when woodland at more accessible sites has been lost (Larson et al., 2000). This shows the importance of features that limit access to livestock as refuge areas of woodland during times of increased intensity of land-use (Larson et al., 2000). In America associations between topography and remnant ancient woods have been examined using topographical factors to map the location of remnant ancient forests, logic being they are restricted to non-commercial steep slopes and poor soils (Therrell and Stahle, 1998, Stahle and Chaney, 1994).

The UK ASNW resource is fragmented and typically consists of small scattered sites. 80% of sites contain less than 20ha of ASNW, while nationally only 500 sites are larger than 100ha (Thomas et al., 1997, Peterken, 2000b). A study of 5 English Natural Areas found wide differences in the frequency and cover of sites, ranging from 15% at 100 sites (High Weald) to only 0.2% cover at 3 sites (Lancashire plain) (Kirby and Thomas, 1994). Sites frequently occurred in clusters and along linear features such as slopes or valleys which resulted in reduced distances between woods in the direction of these features (Kirby and Thomas, 1994). In the Cumbrian uplands analysis revealed ancient woodland inter-patch distances (350m +-160m and 138m +- 33m) were higher than distances between ancient woodland sites and non-ancient woodland, indicating ancient woods were more isolated from their nearest ancient neighbours, than from nearest non-ancient woods (Baalman and Kirby, 1995), a trend confirmed by a recent study that examined all ancient woods within England (Table 5.4) (Mouflis and Buckley, 2004).

Table 5.4
Summary statistics recorded for Ancient Woodlands within 10km grid covering England. ASNW = ancient semi-natural woodlands, PAWS = plantation on ancient woodland site, MNN = mean nearest neighbour distance, MNN2 = mean distance to second nearest neighbour, MNN AW-SW = mean nearest neighbour distance between ancient woodland and secondary woodland. Source: (Mouflis and Buckley, 2004).

Ancient woodland variable	Mean	SE
Total Area (ha)	289.2	11.6
Frequency	19.69	0.59
ASNW area (ha)	184.2	8.0
PAWS area (ha)	113.7	4.6
Mean patch size (ha)	14.29	0.4
MNN (m)	739	31.4
MNN2 (m)	1384	44
MNN AW – SW (m)	511.4	10.3

5.3 Woodland species ecology and dispersal

5.3.1 Introduction

The following section details a number of studies relating to woodland ecology with relevance to Upland Oakwoods, or their component species. Species groups characteristic of, or dependant upon Upland Oakwoods are examined in relation to the development of new Upland Oakwoods: colonisation ability, succession and movement of species between existing woodland sites. Species have preferences for different sets of ecological and environmental conditions. The NVC communities comprising Upland Oakwoods occupy distinct site types and woodland composition and diversity will in part reflect the conditions leading to the development of these communities (Rodwell, 1991).

5.3.2 Woodland species assemblages

Several groups of species are characteristic of upland Oakwoods habitats, the most apparent comprising the dominant trees and shrubs in the canopy and the various ground-flora communities. Important avian fauna, invertebrates and mammals also occur. The NVC woodland descriptions detail the frequencies of tree, shrub and ground-flora that comprise Upland Oakwood habitats (Rodwell, 1991) (Table 5.5). The composition of these communities arises from interaction with the surrounding biotic and abiotic environment. Species have different tolerances to environmental conditions: hydrology, fertility, light levels and to competition from other species. Species may be grouped by life history "strategies" termed, competitive, stress-tolerant or ruderal (Grime, 2001, Grime et al., 1992). The range of conditions existing within a wood can therefore determine the species present. The conditions prevalent at a certain point in time will tend to favour a subset of the species present, and when conditions change, for example through felling or drainage then a different subset of species will be favoured. A classic example being when part of a wood is felled or coppiced (Peterken, 1993, Mason and Macdonald, 2002).

Woodland fauna also have tolerances for combinations of biotic and abiotic environmental conditions, but are more readily able to move between sites when conditions change. A critical factor in relation to such movement is scale. Larger and more mobile species such as birds and the winged invertebrates may move over landscapes of many kilometres and therefore these species regional distributions tend to reflect the distribution of those conditions across the landscape. Less mobile species such as un-winged invertebrates are restricted in their movement ability. Such differences are particularly important for the conservation of these species and potential spatial strategies to address their conservation.

The range of bird species listed within the Peak District Upland Oakwood Action Plan include several nationally significant species and several long list BAP species (Table 5.6) (Peak District National Park Authority, 2002). These species, for which Oakwood conservation is important, do not share exact preferences in woodland conditions. While generalists may be favoured by initiatives that address the conservation of a number of habitats across the landscape, the conservation of specialist species are more closely tied to the conservation and condition of their principal habitat. While certain avian fauna may have relatively detailed habitat preferences (Stowe, 1987), for many birds species woodland habitat structure and diversity are important, with sites richer in woodland flora, and structurally diverse providing increased opportunities for these species feeding and nesting.

In addition to avian fauna scarce invertebrates are listed within the Upland Oakwood BAP. A review noted that of Dark Peak habitats semi-natural woodland held the most associated invertebrate species, including Northern Wood Ant (*Formica lugubris*) (BAP sp.) (Drake et al., 1998). Several habitat features were considered to be important for the occurrence of scarce invertebrates in Dark Peak semi-natural woodland and dead wood (Table 5.7) (Drake et al., 1998).

Table 5.7
Invertebrate species requirements within the Dark Peak Natural Area, for selected habitats. Source (Drake et al., 1998).

Habitat	Sp. groups	Species requirements
Semi-natural woodland	Lepidoptera	Wide variety of native trees and shrubs
	Coleoptera	Flowery open spaces
	Diptera and	Pollen and nectar sources, standing and fallen dead wood, Carr
	Hymenoptera	Fungal fruiting bodies on or associated with trees
	Mollusca	Well-developed ground-flora including Mercurialis
Dead wood	Coleoptera Diptera	Standing and fallen dead wood fungal fruiting bodies on or associated with trees, ancient hulks

Table 5.5
Species recorded at a frequency of II or more within the component Upland Oakwoods NVC communities. Data from (Rodwell, 1991).

Latin	W10e	W11a	W11b	W11c	W11d	W16b	W17a	W17b	W17c	W17d	W4a	W4b
Acer pseudoplatanus	e											
Alnus glutinosa												b
Betula pendula				c	d	b				d		
Betula pubescens		a	b	c	d	b	a	b	c	d	a	b
Fraxinus excelsior	e	a			d							
Quercus robur	e				d					d	a	
Quercus petraea	e	a	b		d	b	a	b	c			
Ulmus glabra	e											
Trees sp.	5	3	2	2	5	3	2	2	2	3	2	2
Corylus avellana	e	a	b	c	d		a	b	c			
Crataegus monogyna	e	a							c			
Ilex aquifolium	e					b		b				
Salix cinerea											a	b
Sambrucus nigra	e											
Sorbus aucuparia		a	b			b	a	b	c	d	a	
Shrubs sp.	4	3	2	1	1	2	2	3	3	1	2	1
Agrostis capillaris		a	b	c	d		a		c			
Agrostis canina montana		a	b	c			a		c	d		
Ajuga reptans				c	d							
Anemone nemorosa			b	c	d							
Angelica sylvestris					d							
Anthoxanthum odoratum		a	b	c	d		a	b	c	d		
Athyrium felix-femina			b									
Blechnum spicant			b	c			a	b		d		
Carex nigra												b
Cerastium fontanum					d							
Cirsium palustre												b
Conopodium majus			b	c	d							
Deschampsia cespitosa	e	a										b
Deschampsia flexuosa		a	b	c	d	b	a	b	c	d	a	
Digitalis pupurea		a							c			
Dryopteris affinis (borreri)		a										
Dryopteris dilatata	e	a				b		b			a	
Dryopteris filix-mas	e	a										
Epilobium angustifolium											a	
Festuca ovina			b	c			a		c	d		
Festuca rubra					d							
Galium saxatile		a	b	c	d		a	b	c	d	a	
Holcus lanatus					d							b

Latin	W10e	W11a	W11b	W11c	W11d	W16b	W17a	W17b	W17c	W17d	W4a	W4b
Holcus mollis	e	a	b	c	d		a		c			b
Hyacinthoides non-scripta	e	a	b		d							
Hydrocotyle vulgaris												b
Hypericum pulchrum			b	c	d							
Juncus effusus												b
Lathyrus montanus				c								
Lonicera periclymenum	e	a	b	c	d			b			a	
Lotus ulginosus												b
Luzula multiflora					d							
Luzula pilosa		a	b	c								
Melampryum pratense				c			a			d		
Molinia caerulea							a				a	b
Oreopteris limbosperma			b									
Oxalis acetosella	e	a	b	c	d		a		c	d		
Poa pratensis				c	d							
Potentialla erecta			b	c	d		a			d		b
Primula vulgaris		a	b		d							
Pteridium aquilinum	e	a	b	c	d	b	a	b	c	d	a	
Rubus idaeus				c								
Rubus fruticosus	e	a							c		a	
Rumex acetosa					d							
Stellaria holostea	e	a			d							
Succisa pratensis			b	c								
Teucrium scorodonia		a	b	c	d							
Trientalis euopea				c								
Vaccinium myrtillus			b	c		b	a	b		d		
Veronica chaemaedrys			b	c	d							
Veronica officinalis			b	c	d							
Viola palustris												b
Viola riviniana	e	a	b	c	d							
Trees sp.	5	3	2	2	5	3	2	2	2	3	2	2
Rank	1	2	3	3	1	2	3	3	3	2	3	3
Shrubs sp.	4	3	2	1	1	2	2	3	3	1	2	1
Rank	1	2	3	4	4	3	3	2	2	4	3	4
Ground-flora sp	11	20	26	28	27	4	14	7	11	11	8	11
Rank	6	4	3	1	2	9	5	8	6	6	7	6

Table 5.6
Compiled habitat preference information for bird species listed within Upland Oakwoods in the Peak District Local Biodiversity Action Plan (LBAP). Source (Kirby et al., 2000). These include target figures recommended by English Nature as enabling assessment of favourable condition for the species listed. **=nationally significant, *=long list species. AM = arithmetic mean, SD = standard deviation. Dispersal distances from (Paradis et al., 1998).

Species	Woodland composition conditions	Associated Diversity issues	Surrounding landscape issues	Dispersal distance (km)			
				Natal/ colonising		Breeding / local	
				AM	SD	AM	SD
Pyrrhula pyrrhula Bullfinch**	>25% cover of *Pyrus, Malus, Rumex, Rubus, Betula, Fraxinus, Acer pseudoplatanus, Crataegus, Ilex.* 3:1 ratio of dense to open woodland conditions	Abundant shrub layer invertebrates, and fruit and seed bearing plants	Semi-woodland habitat, scattered trees	4.6	9.8	2.5	0.4
Turdus pilaris Fieldfare**	>25% cover including target species *Crataegus, Ilex, Sorbus, Taxus* or *Juniperus.*	Abundance of fruit bearing plants, and ground surface invertebrates. Areas of locally dense scrub>5m tall					
Caprimulgus europaeus Nightjar**	Sparse woodland and scrub, patches of bare ground	Abundance of night flying insects	Associated open habitats				
Phoenicurus phoenicurus Redstart**	Abundance of deadwood for nesting opportunities. >20% standing trees with >10% dead wood per tree. Predominantly closed canopy with sparse ground-flora. >75% canopy cover, <25% ground cover.	Abundance of canopy invertebrates		12.2	11.9	-	-
Turdus philomelos Song Thrush**	Patches of mature scrub and areas of short grassland cover. >25% cover of ilex, *Crataegus* or *Prunus.*	Abundance of fruit bearing plants and ground surface invertebrates.	Semi-woodland habitats, scattered trees, parkland etc.	7	21.6	4	21.8
Muscicapa striata Spotted Flycatcher**	Large open glades or clearings >50m sq. Presence of nesting sites or cavities	Abundant flying insects	Semi-woodland habitats,	12.8	17.6	5.9	12.1
Passer montanus Tree Sparrow**	Abundant small tree cavities. >25% cover of *Stellaria, Chenopodium, Triticum*	Abundance of seed and fruit bearing plants	Semi-woodland habitats and farmland	8	17.5	5	23.3
Buteo buteo Buzzard*	Abundant trees >14m for nesting use. Several woodlands >1ha	Associated habitats with diverse and abundant prey sources (small birds, mammals, etc)	Open ground for feeding	24.2	24.1		
Tetrao tetrix Black Grouse*	Many small to large woodlands among open country. >50% cover of Larch, *Pinus, Juniperus, Sorbus, Crataegus.* Several woods >1ha and forming over 50% of total area. >75%of woods with tall shrub layer >1m tall		Mosaic of woodland and open ground habitats				
Sylvia atricapilla Blackcap*	Mix of tall trees and shrub/rich ground vegetation. >50% wood area with understorey and ground vegetation. >25% of *Rubus, Sambrucus, Crataegus,* or orchard sp.	Abundance of fruit bearing plants, canopy and shrub layer invertebrates	Hedges, parkland	41.2	37.9	27.5	32
Parus caeruleus Blue tit*	High frequency of mature tree with understorey. Abundant small tree cavities >30% *Quercus, Alnus* or *Betula,* >50% understorey and ground cover > 25% cover of *Fagus, Pinus, Castanea, Crataegus.*	Abundant fruit and seed bearing plants, canopy invertebrates	Semi-woodland habitats, parks, gardens etc.	5.3	15.2	2.3	10.2
Regulus regulus Goldcrest*	Several woodlands >10ha representing >50% of total area A diversity of tree species, but mainly conifers. >30% *Picea, Larix, Pinus*	Abundance of canopy and shrub-layer invertebrates	Hedges, parkland, reedbeds				
Carduelis carduelis Goldfinch*	Mix of dense scrub or hedges and shorter vegetation. 1:1 ration of short <5cm to tall1-3m vegetation for breeding. >25% cover *Alnus, Betula, Compositae, Dipsacus, Cirsium, Pinus.*	Abundance of ground-surface and shrub-layer invertebrates, seed bearing plants	Semi-woodland habitats, parks, orchards, garden, heathland	11.1	18.2	10.6	20.8

Species	Woodland composition conditions	Associated Diversity issues	Surrounding landscape issues	Dispersal distance (km) Natal/ colonising AM	SD	Breeding / local AM	SD
Dendrocopos major Great Spotted woodpecker*	High frequency of medium to large woodland in the landscape. Several woods >2ha and >50% of target area overall. High frequency of large diameter mature trees. Mature *Fraxinus, Quercus, Fagus* >20cm DBH. >25% cover *Fagus, Carpinus*, >20% of standing trees with >10% dead wood per tree, with locally frequent fallen deadwood.	Abundance of seed and fruit bearing plants, in situ, standing and fallen deadwood, canopy invertebrates and birds	Semi-woodland habitats, hedges, gardens	16.5	19.8	3.7	10.4
Parus major Great Tit*	High frequency mature trees, dense understorey. Abundant small tree cavities. >25% cover of *Fagus, Corylus, Quercus*. >30% cover of *Quercus, Alnus, Betula, Sorbus* and >50% understorey	Abundance of seed and fruit bearing plants, ground and canopy invertebrates	Farmland, gardens, parkland	5.3	17.9	2.5	12.3
Picus viridis Green Woodpecker*	Mix of mature trees + grasslands. High frequency large diameter mature trees: *Fraxinus, Fagus, Quercus* >20cm DBH. >20% standing trees with >10% dead wood per tree	Abundance of deadwood, ants and other ground-surface invertebrates	Pasture, parkland, gardens, heaths	7.6	10.1	1.7	2.5
Carduelis chloris Greenfinch*	>25% cover of *Polygonum, Sinapis, Rubus*	Abundance of ground-surface and shrub layer invertebrates, seed bearing plants	Farmland, scrub, trees, hedges, parkland, gardens	4.2	6.4	7.5	22.1
Coccothraustes coccothraustes Hawfinch*	>25% cover of *Quercus, Fraxinus, Fagus, Carpinus, Acer, Crataegus, Taxus*, and other fruiting trees.	Abundance of canopy invertebrates, fruit and seed bearing plants	Hedges, orchards				
Dendrocopos minor Lesser spotted woodpecker*	>20% standing trees holding >10% dead wood per tree	Abundance of dead-wood and canopy invertebrates, deadwood in situ and fallen	Parkland, orchards, avenues of trees				
Asio otus Long eared owl*	Open terrain with patches of trees and scrub. High frequency of dense scrub amongst open ground. Patches of dense scrub >100sq.m.	Abundance of live mammals and birds	Plantations, hedges, farmland, moors	47.8	56.9	4.8	4
Parus palustris Marsh tit*	High frequency of medium to large woodlands in the landscape. Several woods >3ha and forming >50% of the land area. High frequency of mature trees with dense understorey and bare ground. >30% cover *Quercus, Betula* at >20cm DBH and 50-75% understorey with >25% bare ground. >25% cover of *Fraxinus, Fagus, Quercus, Corylus, Sambrucus, Cirsium, Lonicera.*	Abundance of shrub-layer and canopy invertebrates, seed and fruit-bearing plants	Alder carr, riparian trees, orchards, gardens, parks				
Sitta europaea Nuthatch*	Many mature, large crowned trees. Mature *Quercus, Fagus, Castanea* at >20cm DBH. >25% cover of *Corylus, Fagus, Quercus, Carpinus,*	Abundant tree cavities, seed and fruit-bearing plants, bark-dwelling + canopy invertebrates	Parks, Gardens	6.5	8.8	0.8	1.6
Ficedula hypoleuca Pied Flycatcher*	High frequency of medium to large woodlands. Several woods >1ha forming >30% of total area. High frequency of mature trees, with open canopy and sparse ground-flora. Trees>10m abundant with <75% canopy and <25% ground cover	Abundance of flying insects and canopy invertebrates, small tree cavities		20.6	16.5	20.6	17.7
Accipiter nisus Sparrowhawk*	Many small to large woodland amongst open ground Several woods >1ha representing 25%-75% of total area. Mature trees at medium density with areas of dense trees and scrub. Trees10-20m abundant, averaging 2-5m apart with <25% thicket stage conifer or areas of deciduous scrub.	Abundance of live birds and mammals	Farmland	15.6	22	17.3	36.2
Strix aluco Tawny Owl*	Many small to large woodland amongst open country. Several woods >1ha and representing >50% of total area	Abundant large tree-cavities, mammals, birds, amphibians and ground-surface invertebrates	Farmland, parks	8.3	13.5	3.1	7.4
Anthus trivialis Tree Pipit*	Mix of short to medium height swards, areas of taller trees + scrub. Vegetation <40cm, with patches of trees and scrub at >2m representing <25% of total cover	Abundance of ground-surface and shrub-layer invertebrates.	Grassland, heaths, hedges,				

Other notable fauna include bats, which may include *Plecotus auritus* (brown long-eared), *Nyctalus noctula* (noctule) and *Nyctalus leisleri* (leislers). These require suitable roosting trees with old woodpecker holes or rot holes (Entwistle et al., 2001). Species such as *Plecotus auritus* feed directly from tree leaves and branches and therefore require sufficient volumes and diversity of arboreal invertebrates beyond the range of moths and other flying invertebrates that may be caught in flight by other bats (Walsh and Harris, 1996). Bats are highly mobile moving different distances to forage around roost sites. Approximate distances include: 2km: *Pipistrellus*, 1.25-5.7km: *Eptesicus serotinus*, 2-10(20)km: *Nyctalus noctula*, 5-17km:*Nyctalus leisleri*, 1.5-3km: *Plecotus austriacus*, 7-8km: *Myotis daubentonii*, 0.7-1.2km: *Myotis mystacinus*, >10km: *Myotis brandtii*, 0.6-3km: *Myotis natter*ei, and rarer species such as *Myotis bechsteinii* may forage for only 0.5-1.5km around roosts. Movements for many species are strongly orientated along landscape structural features (Boye and Dietz, 2005).

Knowledge of species habitat preferences can allow management decisions to be tailored when sites are known to be important for certain species. However when species distribution information is lacking management may attempt to benefit broad species richness at a site, or to provide a sufficient spread of conditions between different sites that a wide variety of species may be favoured.

5.3.3 Woodland colonisation, regeneration and dispersal

5.3.3.1 Introduction

Of critical importance to the development of site diversity, and the relationship of factors promoting site biodiversity (Section 5.4), and their integration with spatial, landscape and fragmentation impacts, are the ability of species to colonise a site and issues of woodland succession.

Woodlands may originate through natural colonisation or planting while both natural and artificial woods acquire additional species through colonisation. Colonisation and succession events share a number of important features. Key aspects are the successful growth to maturity of colonists, production and dispersal of propagules and successful establishment and growth. Theories of succession note species initially able to colonise hold different dispersal strategies and environmental tolerances, "pioneer" species, in contrast to species that may arrive later during succession that cannot tolerate conditions in early successional sites before full woodland canopy and soil conditions have developed (Begon et al., 1996). Primary succession research suggests floral succession is often determined by stochastic events and by a sites landscape context equally as to the site characteristics (del Moral et al., 2005). Oakwoods often naturally develop on infertile sites with low nutrient availability and poor soils, e.g. from pre-cursor heathland vegetation or acid grassland, typically with *Betula sp.* colonising initially. A slow process of "facilitation" succession may occur, with the gradual development of conditions that become increasingly suitable for a wider range of species (Hester et al., 1991b, Hester et al., 1991a).

Suitable conditions are required for growth and establishment. Studies relating to tree and shrub germination highlight a complex mix between micro-site selection for suitable germination sites and the effects of loss of seeds or seedlings through predation (Ovington and MacRae, 1960, Shaw, 1968b, Shaw, 1968a, Watt, 1919, Kinnaird, 1968, Jarvis, 1964). A study of oak concluded recruitment was seed limited in open sites, where vertebrate herbivores were scarce, micro-site limited in dense forest canopy, and herbivore limited in other sites, for example where rabbits were abundant (Crawley and Long, 1995). Other studies examining tree species within UK oakwoods showed seed predation was generally attributable to small mammals rather than invertebrates or birds (Hulme and Borelli, 1999). Ultimately the range of woodland habitats that develop at a site following initial succession and the communities they support are dependant on the continued presence and regeneration of the woodland canopy, shrub layer and ground-flora.

5.3.3.2 Woodland flora colonisation and dispersal

Woodland flora show a variety of reproduction strategies, colonisation speed and distances differ by species (Table 5.8). Species that reproduce vegetatively are restricted to slow dispersal over short distances, while species producing seed use various strategies: mechanical, wind, water, birds, via the fur of passing animals, ants or alternatively seeds may attempt to disperse not in space, but in time, with the creation of long-term seedbanks which allow plants to colonise favourable conditions when they are occur again in the future.

Table 5.8
Colonisation distances for selected native tree species. Table compiled and adapted from Birks (1989), Grime et al (1992) and Harmer (1999).* = Colonisation rates are calculated from the estimated rates of spread of different species across the UK during the Holocene. ** = colonisation generation rates are expressed as the colonisation distance required to achieve Holocene dispersal rates, accounting for a species lifespan to maturity for each species.

Tree species	Colonisation distance (m)	Pre-historic UK rates of colonisation m/yr*	Pre-historic UK rates of colonisation km/gen**	Dispersal method
Quercus	20	350-500	7-10	Animal
Betula	100-200	250	2-5	Wind
Fraxinus	50-100	50-200	1-3	Wind
Alnus	20	500-600	5-6	Water
Salix	100-200	-	-	Wind, Water
Corylus	20	500	7.5	Animal
Populus tremula	100-200	-	-	Wind
Tilia cordata	50-100	450-500	10	Wind
Malus sylvestris	20	-	-	Animal
Ulmus	50-100	550	8	Wind
Pinus		100-700	1-7	Wind
Fagus		100-200	4-8	Animal

Footnote: current colonisation distances are those considered to allow sufficient dense stocking of sites in the short term, do not account for sporadic dispersal events over longer timescales. Holocene dispersal events would have occurred into open or patchy landscapes over very long periods and therefore may be higher than regular dispersal events possible now.

Tree regeneration is of prime importance to maintain woodland conditions. As a keystone species oaks are of particular interest. Acorn production varies with tree age and form (Crawley and Long, 1995) and are dispersed by gravity or animals. Jones (1959) noted distances of 180-270m from a mix of bird and small mammal burial, with *Garrulus glandarius L.* (jays*)* recorded as transporting acorns 4-5 km based on anecdotal evidence (Jones, 1959). Several of the major acorn predators are also dispersers: jay, *Sciurus carolinensis* (squirrels), *Apodemus sylvaticus L.* (wood mouse) and *Columba palumbus L.* (wood pigeon) (Crawley and Long, 1995). Studies of grassland colonisation by oak and hazel noted rodents may disperse seeds 10-20m and jays 400-500m (Kollmann and Schill, 1996).

Reviews of plant dispersal note many species do not have specialised dispersal mechanisms, and therefore only achieve short distances (Wilson, 1993, Portnoy and Willson, 1993). In secondary woods ground-flora occurrence in isolated fragmented woods was related to life-history and dispersal type (Dzwonko and Loster, 1992). Distances differ depending on life-history and dispersal mechanism, being greatest for wind-dispersed species, followed by animal-dispersed species, ballistic/mechanical dispersed and with species with no observed mechanism having shortest distances (Wilson, 1993, Portnoy and Willson, 1993). However these distinctions have not always been well observed in studies (Portnoy and Willson, 1993).

British trees and shrubs show a variety of dispersal mechanisms (Table 5.8). Species with fruit covered seeds e.g. holly (*Ilex aquifolium*) ensure dispersal by birds (Peterken and Lloyd, 1967). Additionally a variety of trees and shrubs are dispersed by wind (*Betula*, *Fraxinus*,) and water (*Alnus*) (Grime et al., 1992). Long-distance birch colonisation of up to 500m has been observed, although 90% fell within 60m of parent trees (Thompson, 2004).

Ground-flora species also show a range of dispersal. Certain species spread vegetatively e.g. *Lamiastrum galeobdolum* (yellow archangel) has been observed spreading 50-156cm per year (Salisbury, 1976 in (Packham, 1983) and *Hyacinthoides non-scripta* (bluebell) showing seed transport of 1m (Knight, 1964). Generally however evidence for dispersal distances of oakwood flora are rare. Such short-distance movement assumes suitable woodland or semi-woodland habitat exists along which a species may spread and is therefore not relevant to dispersal across non-woodland habitat. Animal, water and wind dispersal are required to disperse across matrix habitats. Examples include *Geum rivale* and *Geum urbanum* with 100-150 hooked achenes per seed head and experiments by Kiviniemi (1996) showing they may be dispersed on animal fur from metres to a few kilometres or further by such methods (Taylor, 1997a, Taylor, 1997b). Several ground-flora species are also dispersed following the action of grazing or browsing animals. A recent study recorded over 84 species dispersed by deer and lagomorphs in a lowland plantation, accounting for over 6,500 seeds per hectare (Eycott et al., 2004). Grazing red and fallow deer held a higher pellet seed content than browsing deer and, with their wide ranging behaviour, may act as dispersers of forest plants (Eycott et al., 2004). The presence of deer in new woodland may not be as detrimental to conservation as is commonly assumed (Eycott et al., 2004). Therefore while typical estimates of ancient woodland ground-flora dispersal reach 16-32m / century (Honnay et al., 1999c), rates would be higher where occasional animal dispersal was occurring.

A critical issue in woodland colonisation is the difference between local and long-distance events (Table 5.8). There are differences of opinions among authors as to the importance of long-distance dispersal to plants and whether local dispersal is sufficient to explain, predict or study plant dynamics. Cain et al consider long-distance dispersal to be important and note effort should be given to collecting data on the tail of dispersal distributions (Cain et al., 2000). However although the behaviour of the tail of seed distribution is critical in examining dispersal (Portnoy and Willson, 1993), it is rarely accurately measured. Critical factors are the timescales over which colonisation events are observed and the frequency of long-distance events. It is very difficult to measure maximum plant dispersal and therefore local events are more typically observed (Wilson, 1993). Studies have modelled both local and long-distance dispersal of tree species and have noted specialised models with fatter tails of spread may better predict dispersers (Clark et al., 1999). Typically models are often used in analysis of dispersal events at broader spatial scales (Clark et al., 1999). Some research has attempted to accurately examine the relative ability of different flora dispersal mechanisms within forest dynamics, rather than using generalised mean distances. The research found that modelled composition was more similar in plots within 150m of each other compared to more widespread plots (Hanson et al., 1990). Such complex models, separately modelling bird, mammal and gravity dispersal are rare, and most research utilises broad generalisations of dispersal by study group.

Typically seed rain reduces with distance from the source plant (Clark et al., 1999). A review of measured seed dispersal in woodland plants revealed very few confirmed events of long-distance dispersal, maximum distances often being less than 50m and most less than 100m (Cain et al., 1998). Due to lack of observation on the dispersal tail the review noted these were indeed likely to be underestimates of true maximum dispersal events. Real-world flora colonisation events show colonisation is affected by other factors apart from obvious dispersal mechanisms, due to the larger distances moved (Wilson, 1993). Therefore for a variety of species extremes dispersal events such as storms or occasional long-distance transport by fauna may allow flora dispersal up to 10km (Clark et al., 1999). The relative importance of local versus long-distance plant migration has been considered to be highly important in the conservation of diversity in current fragmented landscapes, determining whether distant fragments are likely to be able to exchange propagules or be colonised (Matlack, 2005). One study in North America found a high predictive power of modelling based on local dispersal events, compared to observed trends in a real landscape and therefore noted extreme long-distance dispersal events were probably rare on a human timescale, supporting the short-term importance of local dispersal (Matlack, 2005).

If occasional long-distance dispersal is important to woodland colonisation then models using nearest-neighbour (NN) distances to examine likely colonisation events may not reflect the actual likelihood of colonisation (Cain et al., 1998). However this view depends on which mechanism may be occurring in the long-distance transport of plants, and critically on the timescales in difference between typical local dispersal and occasional long-distance dispersal. In the shorter term NN distances may reflect dispersal if they are within potential local colonisation distances, and / or are related to typical between-patch movement by fauna, potentially transporting seeds. The area thus remains unclear. Research has noted that if plant populations occur in metapopulations they will have to be linked by long-distance dispersal events which may be random as distances of isolation in metapopulations are often beyond the short-distance abilities of plants (Cain et al., 2000). Alternatively in the shorter term such populations may be truly reproductively isolated and act as separate populations. This is illustrated by the paradox of historic long-distance plant dispersal and current observations of short-distance dispersal in plants. Plants have dispersal with leptokurtic distributions and therefore as time increases the probability of long distance dispersal increases (occasional events are long-distance much beyond the ranges expected from mean dispersal values) (Clark, 1998). Because such events are stochastic they may reduce some of the differences seen between life-history determined short-distance dispersal, but only over long timescales (Clark, 1998).

In relation to conservation and restoration work the range of research suggests that plant species occurrence in a landscape is the result of the effects of local dispersal events limited by the spatial arrangement of habitat (Matlack, 2005, Matlack, 1994). Extreme long-range dispersal may affect landscapes over long periods of time but in the short-term, up to centuries, local colonisation events dominate (Matlack, 2005, Malanson and Armstrong, 1996). Therefore broad species traits can be used to estimate potential colonisation distance and colonisation success can be expected to increase with the size / quality of a source population producing the seeds.

5.3.3.3 Woodland fauna colonisation and dispersal

Woodland fauna are more mobile, species such as birds and bats easily dispersing across distances of several kilometres. UK bird dispersal distances have been compiled from ringing records and give an indication of typical movement distances (Paradis et al., 1998) (Table 5.6). However several faunal groups are known to have relatively poor powers of dispersal, including

Coleoptera, Mollusca and dead wood insects (Peterken, 1993). These species, due to the absence of wings and their small size may be incapable of travelling large distances. Additionally their behaviour may mean they avoid crossing non-woodland habitats and so are restricted to continuous areas of woodland habitat. One author has suggested a saproxylic beetle species has been unable to migrate a distance of only 500m between host trees in over 400 years at an English parkland site (Alexander 2003b in (Alexander, 2004). Distances beyond 20m-200m may represent a barrier to many such species.

5.3.3.4 Woodland species dispersal and Ancient Woodland Indicator Species (AWIS)

Associated with the development of interest in the conservation of ancient woodland sites (Spencer and Kirby, 1992, Rackham, 2003, Thomas et al., 1997), has been a range of research into "ancient woodland indicator species" (AWIS), characteristic of long-established sites (Hermy et al., 1999, Peterken, 2000a, Rose, 1999, Spencer, 1990, Wulf, 1997, Peterken, 1974). Authors have noted that where indicators of long-term habitat continuity are selected they should have low dispersal ability and have a "perennial stayer" in life history types to be able to identify potential sites of long-term habitat persistence (Norden and Appelqvist, 2001).

Ancient woodland sites have a long history of continuous woodland cover and their flora and fauna developed in times when woodland was more common and land-use less intensive. Differences have been shown in the flora of ancient and secondary woods (Peterken, 1981, Peterken and Francis, 1999, Peterken and Game, 1984), and have examined the dispersal potential of different flora (Hermy et al., 1999, Verheyen and Hermy, 2001, Dzwonko, 1993, Dzwonko, 2001, Dzwonko and Loster, 1992, Brunet, 1993, Brunet and von Oheimb, 1998, Brunet et al., 2000). A range of species have been identified which, due to a combination of low powers of dispersal and / or exacting habitat requirements are restricted to ancient woodland sites. Lists of these have been produced for England (Peterken, 1974, Kirby and Goldberg, 2005). Where these species occur within a site they are taken as indicative of longevity of woodland conditions and may be used to infer that a site is ancient woodland (Miles and Miles, 1997, Peterken, 2000a, Rolstad et al., 2002, Rose, 1999, Wulf, 1997), although the identification and use of such indicators may not be straightforward or always reliable (Norden and Appelqvist, 2001). Due to the range of effects influencing individual species presence the predictive power of these indicators is highest when a number of species are present. Ancient forest species are poorly represented in seed banks and only a limited number have persistent seed banks, where they do occur in the seed bank the seed density is low (Bossuyt and Hermy, 2001). Previous land-use strongly affects seed-bank composition, it can take over 50 years for the decline of species typical of previous land-use in secondary woods (Bossuyt and Hermy, 2001). Therefore ancient forest species persist through ongoing occurrence in the ground-flora and when lost from the vegetation may only return following colonisation or spread, rather than re-establishment from seedbanks. Unfortunately however such colonisation is limited by generally poor powers of dispersal such that very close distances are required for successful colonisation to occur.

To a lesser extent faunal species have also been considered to be indicative of ancient woodlands, e.g. molluscs and invertebrates being proposed (Peterken, 1993). Dolman and Fuller note in their review of woodland colonisation and dispersal that "from the evidence available it seems reasonable to conclude that many *specialist* woodland species have severely limited dispersal ability, with the result that populations have only persisted at sites with a continuity of suitable conditions" (Dolman and Fuller, 2003). Current research has suggested that ancient woodland species can reliably be used to indicate woodland longevity at a site but that the particular species may differ between geographical areas (Rose, 1999, Kirby and Goldberg, 2005, Peterken, 1974). Additionally complex interactions may occur that allow species to persist beyond woodlands in certain habitats or locations, such that links with ancient woodland sites may be broken. Therefore the exact use of such indicator species must be qualified for particular sites and areas.

5.4 Woodland site diversity

5.4.1 Introduction

Factors within woodlands associated with high biodiversity are critical to conservation managers and relevant to woodland restoration and creation strategies. Woodlands and potential woodlands differ in size, composition and micro-climate among many additional factors, influencing their diversity and quality. It is of interest to note the factors most relevant to woodland richness and diversity with a view to Upland Oakwood conservation, restoration or creation. This section will concentrate on internal features promoting woodland diversity, assuming species are able to colonise sites. Principal features are therefore: regional location, climate, elevation, geology, longevity of woodland cover and longevity of agricultural land-use, canopy composition and management, maturity of woodland cover, deadwood and veteran trees, structural diversity, topography, aspect, type and diversity of soils, hydrology and the presence of watercourses.

5.4.2 Regional location, climate, elevation and geology

Climate and elevation have a strong influence on the development of woodland. These relate to the effect of altitude on temperature, growing season and soil development (Pyatt et al., 2001, Ray et al., 2003a). Oakwoods only occur over broadly acidic geology and are typical of upland and upland fringe regions (Rodwell, 1991). Within this range woodland diversity would be expected to decline toward the limits of occurrence as either only impoverished sites, or specialist community types survive along the tree-line habitats, or where occurrences grade into moorland scrub (Rodwell, 1991, Hale et al., 1998, Good et al., 1990). Therefore typical Oakwood diversity may be assumed to hold an optimum level within particular climatic and elevation bands, although diversity across woodland estates and complexes will also be affected by the occurrence of marginal or transitional NVC communities, for example where climate is ameliorated in steep humid ravines.

5.4.3 Longevity of woodland cover, agricultural land-use and pre-cursor habitats

Long-established and ancient woodlands hold higher species richness than more recent secondary woodlands (Peterken and Game, 1984, Bossuyt and Hermy, 2001), contain rare or uncommon species not found in recent secondary woods (Dzwonko, 2001, Brunet, 1993) and secondary woodlands arising in historic times hold biodiversity levels associated with the time since creation (Wulf, 2004). Woods may take considerable time to reach their potential diversity following succession or creation, even when occurring close or adjacent to colonisation sources (Peterken, 1996, Wulf, 2003, Peterken, 2000b). Where woods are known to be secondary, current biodiversity may also be related to the type and intensity of land-use previously undertaken at the site (Wulf, 2004, Honnay et al., 1999c, Dupouey et al., 2002). Previous agricultural land-use will mean populations of undesirable species occur in the seed bank and studies indicate it can take 50-100 years for these to decline and begin to resemble ancient woodland sites, even though by this time the vegetation cover of the sites will not yet have approached that of ancient woodlands sites (Bossuyt et al., 2002, Bossuyt and Hermy, 2001).

Longevity may be associated with the colonisation probability; longer established sites have experienced more colonisation events over time, gaining higher biodiversity. Additionally longer established woodlands will have been created with less intensive methods than recently created woodlands, and would have arisen at times when landscape management intensity was lower, more propagules were available and the matrix was more favourable to woodland species movement. Long-established woods may have preserved relatively unaltered soil structures (Spencer and Kirby, 1992) and therefore be less nutrient-rich than sites created on ex-agricultural land. A study in North America indicated the period at which secondary woods arose affected species richness. While ground-flora richness did not differ between primary woods and 19th century secondary woods, primary woods had a higher richness than 20th century secondary woods suggesting either time related dispersal limitation or differences in the matrix habitats allowing colonisation of new wood sites between these periods (Bellemare et al., 2002). A study examining ground-flora found woods developing on grassland rather than arable sites held higher richness of herbs and shrubs (Wulf, 2004). Woods developing on grasslands became richer and this may be due to the ability for woodland species to survive as small populations in grassland habitats (Wulf, 2004) or due to the lower levels of remnant nutrients compared to arable sites. The incorporation of existing habitats such as mires, crags, treeline scrub, rich grasslands, dwarf shrub heaths, remnant small woodlands, and riparian areas into new upland woodlands is considered to increase resulting woodland biodiversity (Ratcliffe and Peterken, 1995). An American study of early European agricultural area found sites re-colonised from nearby ancient woods were lower in floral diversity when they had been ploughed, than un-ploughed sites. The implication being that even historical ploughing, in the absence of high increases of artificial nutrient inputs, reduced the micro-topography reducing subsequent flora colonisation and establishment (Matlack, 1994). Richness and composition of ground-flora or ancient woodland species developing in secondary woods is also negatively associated with the length of agricultural use of pre-forest land due to build up of soil nutrients and associated factors (Dupouey et al., 2002, Honnay et al., 1999c). Such trends may persist even after considerable periods of time, and may be irreversible on a historical human timescale (Dupouey et al., 2002).

5.4.4 Canopy composition, diversity and woodland management operations

Intensive forestry management may have a negative impact on woodland species due to the use of non-native species, site preparation methods and intensive management regimes (Bengtsson et al., 2000, Hartley, 2002) while the levels of native vegetation remaining in plantations can significantly affect biodiversity levels in plantation forests (Hartley, 2002).

Woodland fauna Bird communities differ between broadleaved and coniferous stands (Donald et al., 1997, Donald et al., 1998), mixed stands hold intermediate communities (Donald et al., 1998), while broadleaved stands hold higher species richness (Donald et al., 1997). Broadleaved woodlands are important for UK bird populations, being a key habitat linked to species occurrence (Fuller et al., 2005). Deciduous trees are also preferred by bats for roosting, and woods with a high proportion of deciduous trees, especially oak and beech support more bat species than conifer woods (Boye and Dietz, 2005). The presence and cover of broadleaves within conifer woods impacts bird communities, with richer avifauna occurring in managed conifer woods if broadleaved trees are dispersed than concentrated (Bibby et al., 1989, Donald et al., 1998), suggesting that among managed plantations mixed species stands will hold richer bird communities than pure conifer stands (Peck, 1989). In addition to composition, bird species density and richness were correlated with tree species richness, due to preferences for different trees for feeding (Peck, 1989). All species need not be abundant to benefit from positive effects of multiple tree species in a forest stand (Peck, 1989). In particular presence of oak, birch and alder can significantly enhance woods as bird habitat (Peck, 1989). Studies have also indicated the number of insects associated with British tree species differ between species (Kenedy and Southwood, 1984), suggesting diverse stands will hold higher insect diversity.

Woodland flora Flora occurring in non-native stands associated with plantation forestry may be affected by operations in establishing woodlands, resulting in lower diversity where sites have been ploughed or intensively managed (Matlack, 1994). The heavy shade and dense nature of conifer stands has negative effects in comparison to broadleaved woodlands. Studies of PAWS found higher ground-flora richness and cover under less shaded oak compared to dense beech or conifers, mixed stands were intermediate (Kirby, 1988b). Other research found broadleaved stands held richer flora than conifers (Coroi et al., 2004), typically broadleaved woods are acknowledged as holding higher conservation value (Coroi et al., 2004). A study of fungi in conifer woods also found species richness to be correlated with diversity of ground vegetation diversity, emphasising occurrence of hotspots between such different communities, possibly related to hydrology or humidity / soil factors (Humphrey et al., 2000). In managed forests analysis found plant richness was negatively affected by the levels of conifers present in the neighbourhood surrounding a stand (Skov, 1997). Dominant canopy type can strongly influence ground-flora composition in managed woods. In semi-natural ancient woodland however composition is largely determined by topography, soils and edge effects, with only minor effects attributable to canopy type (Thomsen et al., 2005). In these natural woods the canopy effects observed were mostly attributable to availability of light and to a lesser degree the vertical structure of the canopy composition (Thomsen et al., 2005).

Management Management events initiating canopy composition and affecting structure strongly impact on woodland composition. Conservation should aim to mimic natural disturbance events likely to occur in an area to promote or retain woodland biodiversity (Roberts and Gilliam, 1995, Bengtsson et al., 2000, Attiwill, 1994, Hartley, 2002), typical events being tree-fall, storms, floods, lighting strikes, fire and presence of larger grazing herbivores and the potential consideration of these to re-stocking and felling regimes. Research of disturbance and management found that species diversity was highest where woods experienced lower disturbance levels or where regeneration had occurred for the longest periods (Onaindia et al., 2004). Due to the majority of species in forest seed banks not being typical of the vegetation cover, forest management should be careful not to cause excessive disturbance or encourage regeneration of invasive weed species from the seed bank (Bossuyt et al., 2002). Methods suggested to benefit biodiversity in plantation forests include retention of snags and deadwood, growth of multi-species stand and mixtures, use of native species and retention of deadwood (Hartley, 2002).

5.4.5 Maturity of woodland cover, deadwood and veteran trees

Stand maturity is associated with increased structural diversity, the development of closed and more humid woodland conditions, and also increased occurrence of deadwood, veteran trees and semi-natural conditions, all providing specialised habitats and niches that may be associated with species of conservation concern (Drake et al., 1998). Stable and humid woodland conditions can be important for lower plant occurrence and conservation (Hodgetts, 1996, Thompson, 2005). The composition of avifauna communities have also been found to be partly determined by woodland canopy age / maturity, and this is likely to reflect in part increased structural diversity and presence of veteran and deadwood features (Donald et al., 1998). Bird species use of conifer plantations are strongly affected by plantation age, very young and dense closed stands being used less than intermediate stages (Patterson et al., 1995). Bat roosts are also more likely in older trees with aged or veteran features (Forestry Commission, 2005), while deadwood invertebrates are associated with hollow, dead or veteran trees, being more common where such features are abundant or at sites with frequent high quality deadwood habitat (Ranius, 2002). Additionally the volume of deadwood has been found to be associated with saprotrophic fungi diversity in conifer plantations (Humphrey et al., 2000), although deadwood volume is typically less in managed than in natural stands (Hodge and Peterken, 1998). Research has noted the importance of deadwood and retained trees in providing hollows and cavities for use by wildlife species, they also note the problems of retaining such structures and ensuring their persistence within managed stands (Gibbons and Lindenmeyer, 1996). The importance of deadwood habitat to woodland biodiversity however is now recognised and management guidelines have been developed (Forestry Commission, 2002).

5.4.6 Structural diversity

Structural diversity refers to the layering of canopy, understorey and shrubs. Structure effects may be complex, with affects on woodland fauna, through the provision of niches, or woodland flora through impacts on shading, forest floor humidity and competition. Many different methods have been proposed for assessment of stand structure (Lahde et al., 1999). Broader assessment of diversity may also include presence of veteran / significant trees and deadwood (Lahde et al., 1999).

Woodland fauna Early works indicated positive associations between complex woodland structure and bird diversity (MacArthur and MacArthur, 1961), later studies showed more complex relationships involving structural diversity and stand age (James and Wamer, 1982). Woodland diversity and structure are important for woodland avifauna, namely shrub layer composition and structure (Diaz, 2006) and structural differences between stands affect bird species composition. Stand diversity (tree species and tree sizes) was associated with bird species richness and total abundance in a UK study (Donald et al., 1998). Woodlands that are structurally complex have been found to be rich in woodland bird species, therefore both young forests in succession and mature broadleaved woods may be equally rich, while dense uniform woods of limited tree diversity, or coniferous forests have the lowest bird species richness (James and Wamer, 1982). A study in the Forest of Dean found higher richness of birds occurred in broadleaved stands with more frequent undergrowth (Donald et al., 1997). Another study found bird species abundance was highest where there was high canopy height and high tree species richness (James and Wamer, 1982). Bird species use of conifer plantations have been shown to be strongly affected by the age of plantation crop and season, young and dense closed stands being used less than intermediate stages (Patterson et al., 1995). Interestingly where grazing had reduced structural diversity one study found no overall differences in avifauna richness between managed broadleaved and coniferous stands, possibly due to becoming structurally similar when grazing removes the shrub layers and young regeneration (Donald et al., 1998). An invertebrate study found richness of carabids and syrphids was best correlated with vertical stand structure, high canopy cover values (shading/density) reducing diversity (Humphrey et al., 1999). Bats will also favour woodlands with diverse structure and presence of features such as old trees, water courses and regeneration thickets (Boye and Dietz, 2005), such diversity being beneficial in providing feeding areas and roosts sites (Forestry Commission, 2005). A recent review of temperate forest studies concluded that structurally complex forest stands promote diversity of canopy dwelling species (Ishii et al., 2004).

Woodland flora In semi-natural woodlands only minor effects of stand structure were seen on ground-flora species composition, the effects attributable to the availability of light and to a lesser degree the vertical structure of the canopy (Thomsen et al., 2005), suggesting that shading may impact on diversity. Managed forests however show a much wider range of effects. Studies in managed forests found plant richness was positively associated with diverse local neighbourhoods surrounding a forest stand (Skov, 1997). Ultimately stand diversity can be considered to be high where there is a high richness of tree species and a high variation in age and size of trees (Lahde et al., 1999). However it is important to recognise that while generalisations can be made, exceptions occur. Examination of designated Atlantic Upland Oakwood sites revealed that for mosses and lichens uniform structure may not necessarily indicate low biodiversity as rich carpets may occur in humid areas beneath such stands (Thompson, 2005).

5.4.7 Topography, aspect and soils

Topography and aspect may impact on botanical diversity through their associations with hydrology, fertility and levels of disturbance. Studies of ground-flora in secondary woods showed landform influenced the relationship between management and resulting diversity, topography having a strong impact on plant richness (Okubo et al., 2005). Research has noted the importance of steep or inaccessible topography in allowing the retention of rare or restricted woodland species (Brunet, 1993), probably due to the association between topography and lack of disturbance and associated woodland longevity. Certain topography such as cliffs and steep slopes with infertile soils may be associated with higher diversity due to being associated with presence of ancient woods or less intensively managed or inaccessible land (Stahle and Chaney, 1994, Therrell and Stahle, 1998, Larson et al., 2000). Extremes of topography may therefore represent possible indicators of high diversity in woodland sites. In Oakwoods rock outcrops and steep ravines are less likely to have been disturbed in the past and may retain locally humid conditions allowing growth and retention of bryophytes and specialist species, enhancing diversity (Thompson, 2005). Studies of ancient woodland indicator species have found the richest woodland sites to be those with the richest soil types (Peterken, 1974). Additional work examining ground-flora diversity have also found effects attributable to soil type or diversity (Dzwonko and Loster, 1992).

5.4.8 Hydrology and presence of watercourses

Although linked to soil diversity and topography, the presence of watercourses also influences diversity, principally within the flora, but with associated effects on invertebrates and avifauna. Several Upland Oakwood flora NVC communities and sub-communities are associated with areas of damp or flushed ground, or the presence of streams (Rodwell, 1991). Woods containing such features as flush-lines, where breaks of slope occur or different geology types meet, or containing watercourses are thus likely to hold rich communities in comparison to sites occurring on uniform, drier soils or lacking watercourses. Research, including Upland Oakwood stands, found presence of streams enhanced woodland plant diversity and that plant species richness decreased with distance from streams (Coroi et al., 2004). Within Upland Oakwoods stream banks and ravines may allow growth of bryophytes and specialist species due to higher humidity and lack of disturbance in the past (Thompson, 2005). Presence of streams and riparian zones in woodland are recognised as of high value for biodiversity, and authors have noted such areas should therefore be sensitively managed (Ratcliffe and Peterken, 1995).

5.5 Woodland creation, restoration and conservation

5.5.1 Introduction

This section summarises several areas of research in the creation, restoration and conservation of woodland communities ranging from observational studies of practical works (Radford, 1998) to reviews of the effectiveness of recent woodland creation schemes (Currie, 2001). Advice or guidance inferred from several larger landscape-scale and modelling studies are also included, where these provide insight at the site scale. Areas examined are therefore: woodland site creation, woodland "restoration", and the maintenance and enhancement of woodland ecological interest.

5.5.2 Woodland creation: ecology and practice

New woodlands arise through planting or natural colonisation. Although often even woodlands created by natural colonisation are consciously created by man, through actions taken to create the necessary conditions for natural colonisation, such as reduced grazing levels. A potential conflict in creation at currently unwooded sites is the trade-off between ecological value of the current habitats and future woodland. This is considered to occur more in upland situations which are already rich in habitats of conservation interest, as opposed to lowland areas where the creation of new woodland is typically on intensive ex-agricultural land (Good et al., 1997). Woodland developing on low quality upland habitats such as *Molinia caerulea* (purple moor grass) or *Pteridium aquilinum* (bracken), may take significantly longer to develop diverse communities than woodland developing on current ecologically rich sites such as unimproved pasture or mixed heathland. However in such situations it can be considered that woodland developing on the former habitats may result in a higher net conservation value than woods developed on the later habitats as the pre-cursor vegetation has lower initial value (Good et al, 1997). For typical Upland Oakwood sites such distinction between pre-cursor communities may impact most on the timescale of development of oakwood habitats. However for more specialist Upland Oakwood features such as the small areas of richer ash or alder (W7, W9) woodland that may occur along flush lines or streams such communities may be limited in their creation to the location of already relatively diverse vegetation types.

In recent decades the methods of woodland creation have been those developed for the rapid development of closed-canopy conditions through planting. Only recently have ecological methods been considered, recognising the importance of natural regeneration. Guidelines now exist where the aim of woodland creation is the replication of native woodland communities (Rodwell and Patterson, 1994) (Table 5.9).

Table 5.9
Guidelines listing factors of importance to successful native woodland creation. Reproduced from Currie 2001 and Rodwell and Patterson 1994.

Sites on similar soils types to nearby native woodlands will be the most appropriate
Riparian areas should be included as they may act as potential sources of native woodland species
Sites should include areas of open space habitats which can maximise biodiversity potential
Natural regeneration is stressed as being most appropriate for woodland creation in preference to planting, particularly when close to existing sources, but long-distance dispersal of some species can occur.
Where planting is necessary it should be designed in a natural way, and comprise suitable native species with regard to species in the locality. Local provenance stock should be used where possible, while rare trees and shrubs were not recommended for planting
Planting mixes within separate clumps should comprise 2-3 species at most, while diversity may be increased by planting single species clumps of varying size up to 50m across, gaps between clumps should be between 7 – 20m. Open space areas should be retained at least 25m across.
Recommendations for planting adjacent to moorland edge highlights a preference for widely spaced clumps of close spaced trees rather than widely spaced trees in larger clumps, with landscape benefits and shelter effects for the trees.

In discussing the use of natural colonisation Harmer (1999) considered the process to be relatively unreliable and patchy, except where conditions were particularly suitable and occurred close to parent trees. He noted successful regeneration only up to a maximum 100-200m from parent trees, depending on species (Table 5.8). However such guidance remains concerned with achieving relatively high tree densities over short terms of 5-10 years and therefore the distances over which natural regeneration may be considered to be "successful" can be significantly extended where timescales are longer. Historically creation grants in England have required dense stocking within 2-5 years (Anon, 2003, Watkins, 1984), although this has recently been relaxed for native woodland sites to 10 years (Forestry Commission, 2006). It is acknowledged that woodland created adjacent to ancient woodland and incorporating semi-woodland habitats, will be richer in species (Peterken, 2000b), but that even then it may take approximately 100 yrs to be colonised by ground-flora (Wulf, 2003). Therefore new woodland sites occurring in isolation or distant from such rich and diverse ancient woodland sites may be expected to take considerably longer to acquire a full complement of woodland species.

Harmer provides further advice for encouraging natural regeneration, highlighting the benefits of sites close to parent trees, downwind for wind dispersed species, providing perches for, or planting trees that will encourage, birds and that operations should coincide with seed input (Harmer, 1999). While the patchy distribution of natural regeneration may be detrimental to forestry interests this can lead to more ecologically diverse and interesting communities, where mixed aged stands with variable canopy cover develop, allowing a wider range of associated species to colonise and exist during woodland development. Additionally trees and shrubs arriving by natural colonisation may not be available from cultivated stock, also increasing potential diversity levels. There also may be concerns that planted stock may be of non-local genetic origin, perhaps originating from European timber stock. Local seed may be adapted to local sites and areas and so should be used in native wood schemes (Ennos, 1998). However the distances quantifying "local" range from several to several hundred km's, and are largely unknown and will differ between species (Ennos, 1998). The Forestry Commission have produced guidance on local provenance source zones for use in both existing woods and in woodland creation (Herbert et al., 1999) (Fig 5.2). Importantly the guide notes in certain cases "it will be preferable to delay planting until local stock is available, rather than use sources outside the seed zone" (Herbert et al., 1999). However the composition species used in native woodland creation schemes remain heavily influenced by availability of local provenance stock. Initiatives have recently been undertaken to encourage the supply of true local provenance tree and shrub seeds such as the "Moor Trees" charity in Dartmoor (http://www.moortrees.org/). These typically involve the collection of seed from known remnant local populations, with at least 20-30 interbreeding parents from each remnant population to ensure genetic variability (Ennos, 1998).

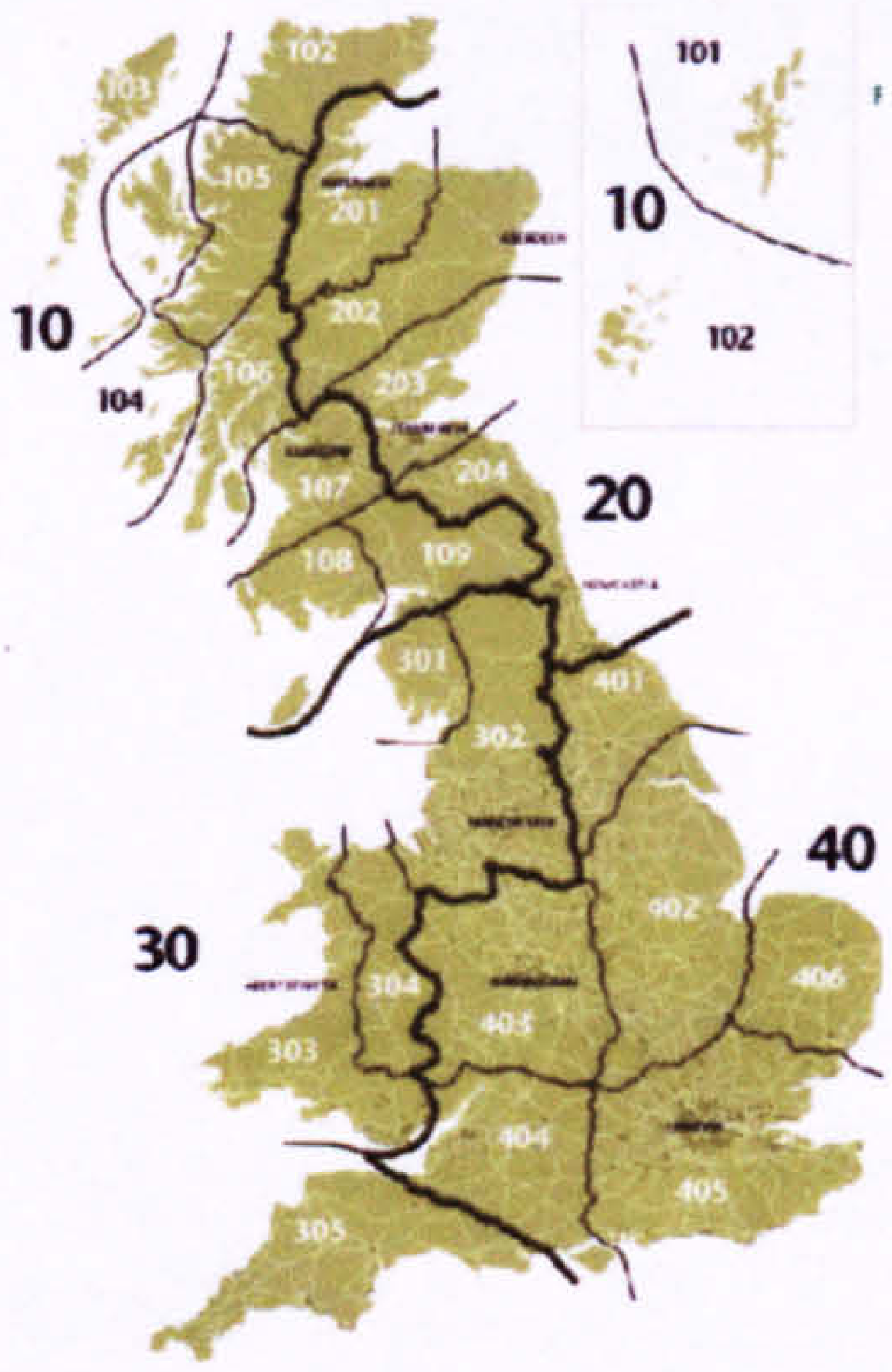


Figure 5.2
Forestry commission Local Provenance seed zones (Forestry Commission, 2003).

Unfortunately the species most likely to be available as planted stock are also those most likely to colonise naturally. Schemes principally comprise oaks, birches or ash while rarer species such as aspen, whitebeam, and limes remain rare. Some authors have questioned the reliability of basing woodland creation and restoration on only species still found locally, or from projected future natural communities, instead suggesting more use should be made of historical data in the pollen record to determine how diverse woods were in the past, using these to inform potential woodland composition (Tipping et al., 1999).

Native wood creation schemes in English national parks (Currie, 2001) have focussed on the replication of NVC communities (Rodwell and Patterson, 1994). A review of the Challenge Fund, supporting such work, highlighted important several factors (Currie, 2001). Application was hampered by restrictions encouraging rapid establishment, emphasis on planting and on quick visible results and a lack of ecological expertise in native woodland creation (Currie, 2001). However in terms of connectivity 86% of schemes examined either incorporated or occurred adjacent to existing semi-natural woodland (Currie, 2001). The report cited lack of time to allow appropriate design and sourcing of local provenance stock, the majority of sites were planted with trees within 1 year of the scheme entry date (Currie, 2001). The report made several recommendations for future creation scheme enhancements: sites over 10ha should be subject to analysis by the Ecological Site Classification (ESC) system to ensure appropriate matching of suitable native woodland communities, site locations should be better targeted at catchment or sub-Natural Area level, there should be more encouragement for locally sourced stock planting, while existing standards on tree spacing, open ground and timescales for creation should be relaxed (Currie, 2001). Several of these recommendation were subsequently reflected in changes made to native woodland creation guidance within the current England Woodland Grant Scheme (Forestry Commission, 2006). These recommendations also reflect research in

Scotland which concluded that creation sites varied greatly in establishment cost and ecosystem potential score but generally expenditure was negatively correlated to the ecosystem value. The most cost-effective woods being those created near to existing woodlands using natural regeneration (MacMillan et al., 1998).

5.5.3 Woodland restoration: ecology and practice

5.5.3.1 Introduction.

Restoration involves the re-establishment of native cover to sites where it has previously existed, but has been removed or significantly altered. Sites include woods degraded where through heavy grazing and plantation on ancient woodland sites (PAWS). The shift in conditions during replanting and the subsequent heavy shade cast by maturing conifers cause changes to occur in woodland flora and sensitive species may be lost (Fig 5.3). Restoration of PAWS is of prime importance to woodland conservation, and has been the focus of UK restoration research. However the principles may also be applied to restoration of native cover on secondary woodland sites.

Conifer planting results in a reduced field layer community in PAWS compared to semi-natural stands (Radford, 1998). PAWS restoration comprises "securing features from the former ancient semi-natural woodland, removing introduced species of trees, shrubs and other plants, encouraging the re-establishment of native species and initiating or enhancing ecological processes which may be absent or damaged (such as appropriate grazing regimes)" (Thompson et al., 2003). The aim therefore has been noted as to "manipulate the canopy to create the conditions in which the remnant ancient woodland communities can recover" (Pryor et al., 2002). Other authors have considered the genetic integrity of sites and believe woodland restoration should "produce a resource which is capable of adaptation in response to environmental change" (Ennos, 1998).

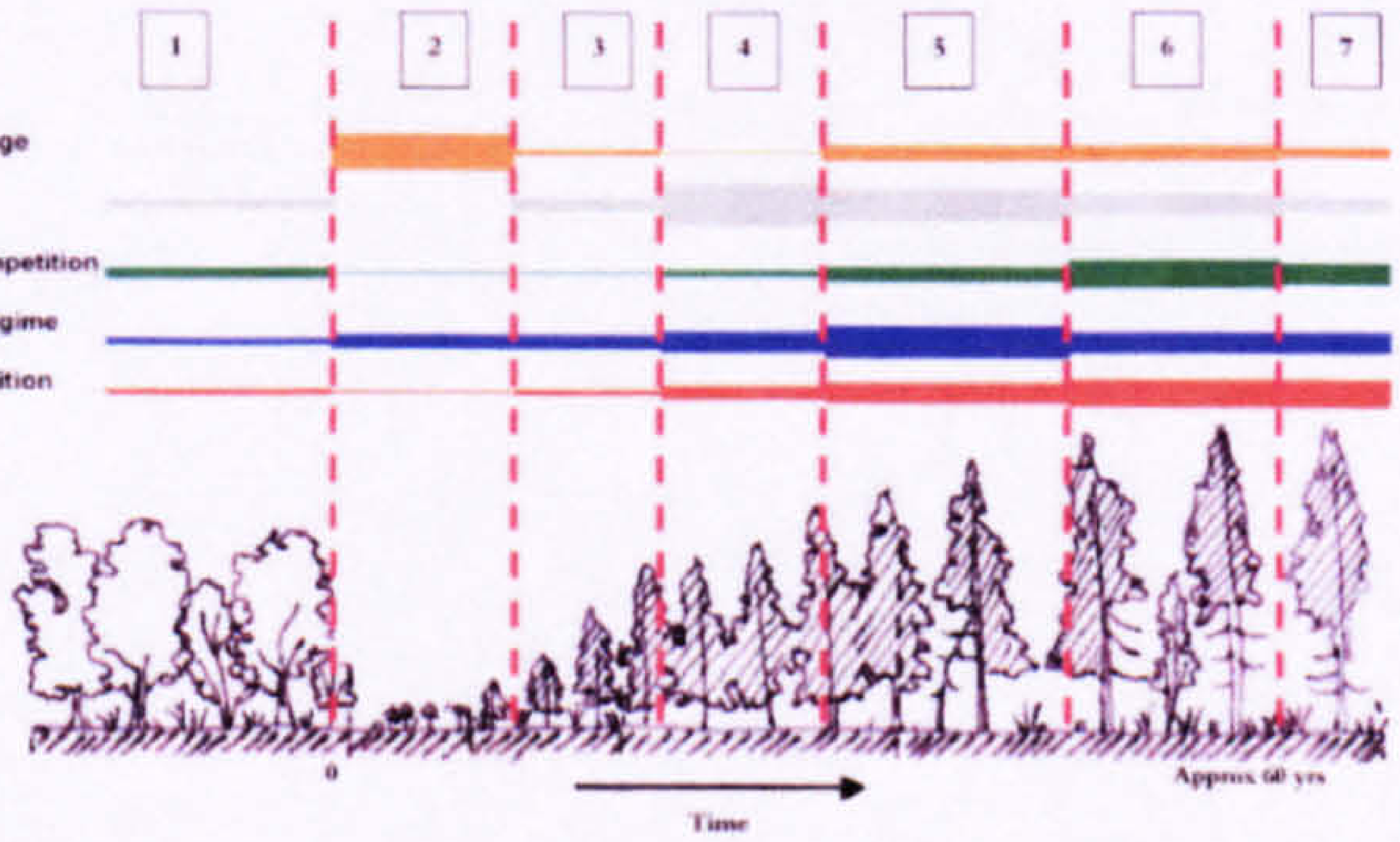


Figure 5.3
Time line detailing potential detrimental impacts during the development of a PAWS site. Reproduced from (Pryor et al., 2002). Stage: 1= pre-conversion to PAWS, 2= clearance and planting, 3= thicket, 4= pole, 5= middle rotation, 6= mature, 7= over-mature. Thickness of line indicates weight of effect.

5.5.3.2 Assessing remnant ecological interest

Restoration of flora from seedbanks is considered unlikely, especially where species have disappeared from the vegetation (Bossuyt and Hermy, 2001, Bossuyt et al., 2002). Therefore management that fosters the growth of remnant pockets of ancient woodland vegetation is critical in achieving successful restoration. While in some sites restoration may involve planting, in general the aims is to expand areas of current interest. Colonisation of native species to existing woodland patches will differ than colonisation to new woodland where existing woodland structure is lacking.

Guidance has been produced for woodland restoration schemes (Thompson et al., 2003) (Table 5.10, Fig 5.6), while a number of studies of ancient woodland sites and restoration methods have been undertaken: by the Forestry Commission (Spencer, 2002), English Nature (Radford, 1998) and the Woodland Trust (Pryor et al., 2002, Pryor and Jackson, 2002, Pryor and Smith, 2002). A number of properties of PAWS are considered to increase restoration potential. Kirby and Reid noted the restoration of replanted ancient woodland is most likely to be successful in woods that have only recently been replanted (last 30 years), where some of the former broadleaved cover and/or ground-flora survived within the crop, with a species-rich ride system and those in the east of England, where site history appears to be strongest (Kirby and Reid, 1997). Pryor (2003) recorded 4 methods by which remnant features may originate; standards retained when the conifer crop was under-planted (typically oak), pole-stage trees retained at the time of planting due to their potential timber values (oak or ash coppice poles, or birch canopy shelterwood), native re-growth from coppice stools (ash, sycamore, oak, hazel, goat willow, holly) and a range of prolific native broadleaves which regenerated at the site and had not been removed during thinning (birch, willow, sycamore, ash) (Pryor, 2003). Due to varying levels of interest remaining within sites, information should be collected allowing remnant interest to be assessed and the appropriate option for restoration chosen (Table 5.10, 5.11, Fig 5.6) (Thompson et al., 2003).

Table 5.10
Ecological survey information relevant to PAWS restoration planning. Source: (Thompson et al., 2003).

Richness and condition of ancient woodland features (e.g. veteran trees, coppice stools)
The number, species, condition (including seed-bearing potential) of native trees and shrubs
Patch size, distribution and species composition of remnant ground-flora
Size, species, stability and quality of plantation trees
Presence of advanced regeneration (native and non-native species)
Presence of invasive species within the site and in the adjoining landscape
A broad assessment of the structural and species diversity within adjoining native woodlands

Table 5.11
Features of ecological interest surviving within PAWS stands, reproduced from (Pryor et al., 2002), where the original references are detailed.

Surviving component	Characteristics	Distribution	Value
Ground vegetation and shrub layer	Includes "indicator" species usually associated with AW, plus other woodland specialists and sometimes non-woodland species	Often concentrated into "hotspots" where plantation conditions and operations were less intense.	Provide food source e.g. fruit and nectar (Marren, 1990) and micro=-habitat: may have popular appeal (e.g. bluebells).
Semi-natural trees and underwood	Overstorey trees (including stored coppice) retained from previous stand: underwood and naturally regenerated trees that have re-grown with the planted crop or from retained overstorey trees	Their distribution patterns may be natural or reflect past management, but have usually been altered by the conversion process and subsequent plantation treatment	They provide genetic continuity with previous semi-natural stand: semi-mature trees also will provide the next cohort of mature and eventually senescent woody habitat (Read, 2000)
Veteran trees, deadwood and associated communities	Large, old trees (often standards) surviving from previous stand: stumps and root systems of standards, stubs and stools: plus coarse woody debris (stems and limbs) left from original felling up to 50 years earlier.	Their location is often historically meaningful (e.g. boundary trees) (Rackham, 1976).	They provide direct genetic continuity with previous stands, and may be particularly rich in associated communities particularly fungi, epiphytes, invertebrates, birds and bats.
Soil profiles	Never having been cultivated, ancient woodland soils have been described as "archive soils" (Ball, 1981), often retaining undisturbed profiles (Peterken, 1993) along with low and spatially variable nutrient levels	Profiles and nutrient levels are spatially variable (Wilson et al, 1997), reflecting both Management and history and subtle edaphic variation	This variation and diversity forms part of the site's "habitat quality" (Honnay and Hermy, 1999) and many woodland specialists are adapted to specific soil conditions.
Small-scale topographical features	Drainage ditches, gullies, ravines, wet patches, streamsides, rock outcrops, buildings, boundary banks, walls, hedges, ride edges, and wood margins, rides and glades.	They are often associated with, or cause, gaps and "anomalies" in the plantation crop which in turn leads to pockets of surviving communities or "hotspots" of biological diversity	These produce a diversity of habitat niches, and can also be of cultural or geomorphological interest.

One study of PAWS restoration, examining "woodland specialists", found the majority of PAWS had retained some level of interest with only 16% of stands without surviving veteran trees or significant woodland ground-flora (Pryor et al., 2002). Remnant interest features fell within 5 broad categories: ground vegetation and shrub layer, semi-natural trees and underwood, veteran trees / deadwood and associated communities, soil profiles and small topographical features (Pryor et al., 2002) (Table 5.11). In assessing the importance of these a "scattering of veteran trees and ancient coppice stools may be as valuable as the ground-flora" in providing sources for future restoration (Pryor et al., 2002). The authors also noted the frequent survival of ground-flora in hotspots such as watercourses, rides or areas of plantation failure where either ground-flora or aspects of native canopy had managed to survive (Pryor et al., 2002). Pryor et al grouped ASNW and PAWS into 4 classes of remaining semi-natural tree cover or natural regeneration. Generally levels of ground-flora survival, decreased with increasing plantation character, however the distinction between categories was not as large as may have been expected (Pryor et al., 2002). Studies therefore suggest that significant interest survives within PAWS, from which to develop during restoration.

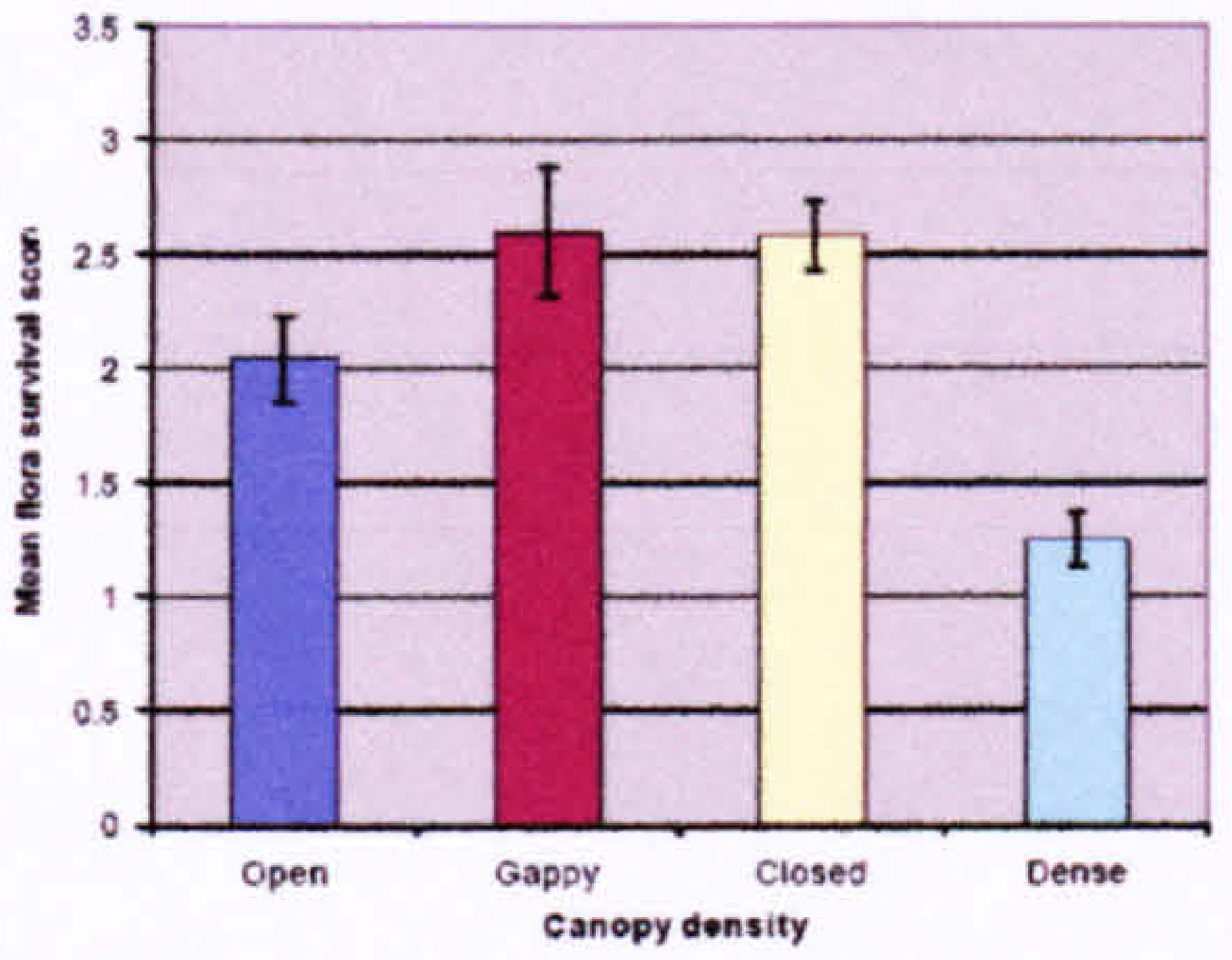


Figure 5.4
Mean flora survival scores under 4 canopy density classes. Error bars denote 95% CI. Canopy cover classes: Open= <30%, Gappy= 30-60%, Closed= 60-90%, Dense= >90%. Reproduced from (Pryor et al., 2002).

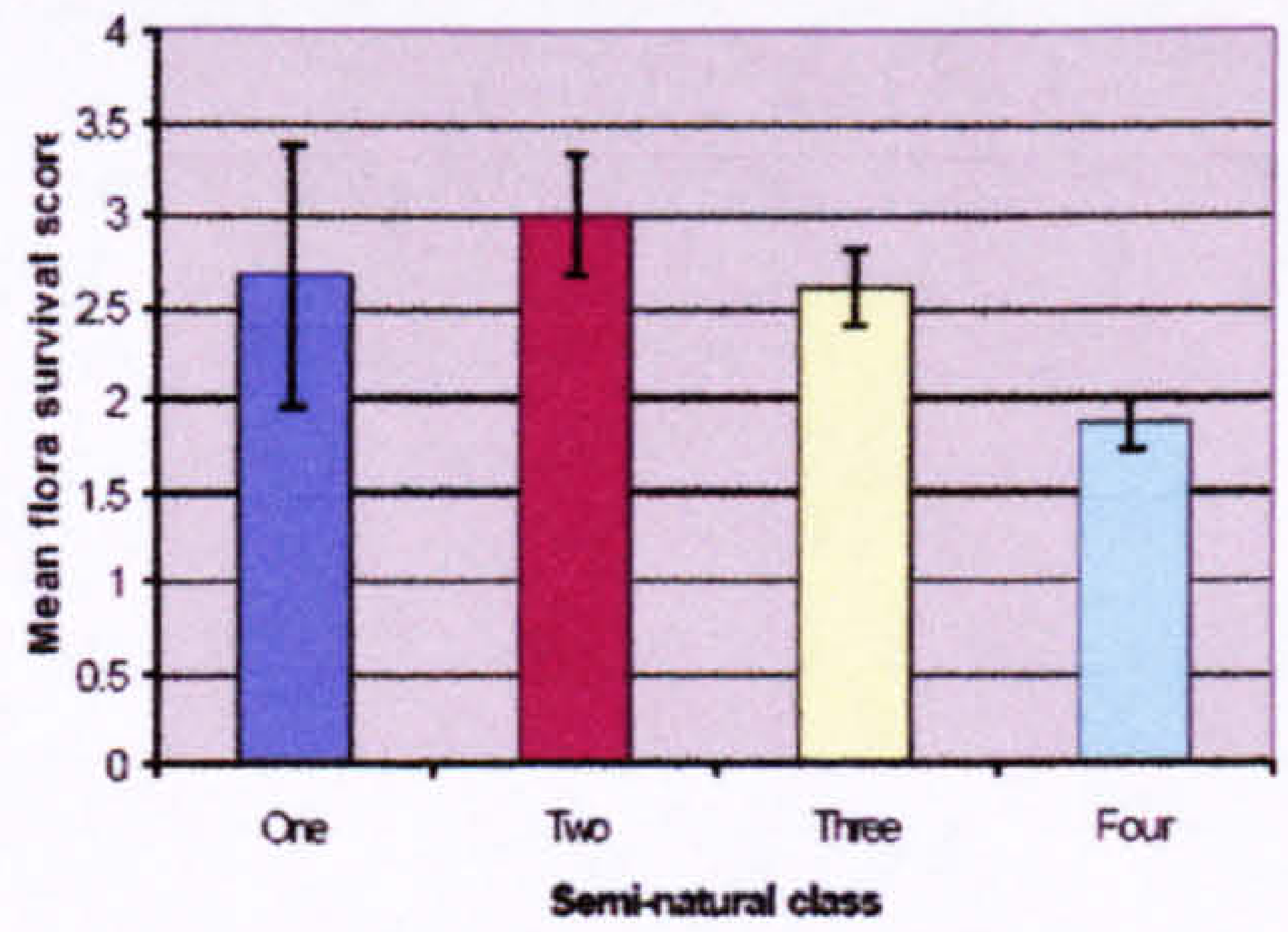


Figure 5.5
Woodland specialist flora survival within different semi-natural stand classes. Error bars represent 95% CI. Data for class one were considered unreliable due to the small sample size. Classes: One = >80% semi-natural, Two = 50-80%, Three = 20-50%, Four = <20% semi-natural. Reproduced from (Pryor et al., 2002).

Criterion	Potential gains for biodiversity		
	High	Medium	Low
Designations (ASSI, SSSI, NNR, SAC, SPA)	• Site or adjoining area is designated for semi-natural woodland • Restoration will contribute significantly to safe-guarding and enhancing adjacent designated site	• Restoration site within close proximity to designated woodland • Restoration will make the management of adjacent designated site easier	• Site is not near to a designated woodland • Designated feature will not benefit from restoration • Designated feature will suffer from restoration
Potential contribution of site to UKBAP habitat strategies at local, regional and national levels	• Potential woodland types will contribute to area UKBAP targets for restoration	• Area UKBAP strategies not yet fully developed, but potential woodland type likely to be prioritised	• Potential woodland types not prioritised within area UKBAP strategies where the latter exist
Potential of site to contribute to a forest habitat network	• Restoration of the site will contribute significantly to the forest habitat network • If site is left unrestored, non-native trees will restrict the ecological functioning of the network	• Restoration will have a minimal or neutral impact on the forest habitat network	• Site is isolated and restoration will not improve connectivity
Potential contribution of site to conservation and enhancement of populations of SAP, Red Data Book (RDB) or other notable species	• Restoration will secure or benefit population of nationally scarce or threat-ened species (SAP or RDB) • Species is at the edge of its range	• Species is of local importance • Species already has extensive existing habitat • A viable population of the species can be maintained by partially restoring stand	• There are no scarce or threatened species likely to benefit from restoration • Nationally important species will suffer from restoration
Rarity of potential native woodland or stand type	• Nationally rare stand type (e.g. aspen stands) • Regionally rare woodland type (e.g. upland oakwood in SE England)	• Regionally rare stand type (e.g. slope alder woods; yew woods)	• Woodland type and stand type are common
Diversity of different habitats and features within the site	• A range of site types and habitat features exist	• Some variability in site and habitat diversity	• Uniform; mostly one site type, little habitat diversity
Size of site (relative to other woodland within the region).	• Large site	• Average sized site for region	• Small site for region

Criterion	Potential for native woodland ecosystem development		
	High	Medium	Low
Number, condition and diversity of mature remnant semi-natural trees and shrubs on site	• High density of mature trees (>20/ha) • Some veteran trees • Good range of different native tree and shrub species (some locally or regionally scarce) • Remnants of epiphytic communities	• Remnant trees all of one species • Remnant trees at a low density (e.g. <5/ha) • No remnants of epiphytic communities • Limited range of native tree and shrub species	• No mature remnant native trees • Very limited range of native tree and shrub species
Presence of specialist woodland ground flora species on site	• High diversity of species in robust patches • Some locally or regionally scarce species	• A few patches • Small populations of one or two species	• No specialist species present
Adjacency of existing semi-natural woodland	• Restoration site completely surrounded by semi-natural woodland	• Semi-natural woodland adjacent to some parts of restoration site	• No semi-natural woodland adjacent to the restoration site
Quality and type of adjacent semi-natural woodland	• High structural diversity • Presence of old large trees and deadwood • Adjacent wood is of the same type to that of the potential restored wood	• Some structural diversity (e.g. two age-classes) • Some deadwood • Adjacent wood of similar type to that of the potential restored wood	• Even-aged stand • No deadwood, no old trees • Adjacent wood is of significantly different type to that of the potential restored wood
Adjacency of other non-wooded semi-natural habitats	• Adjacent semi-natural vegetation is closely related to the potential woodland type (e.g. calcicolous grassland adjacent to ash woodland)	• Adjacent semi-natural vegetation will facilitate the migration of some species into the restoration site	• No semi-natural vegetation adjacent to restoration site

Figure 5.6
Aspects utilised to assess the potential value of restoring PAWS sites to native cover.
Source (Thompson et al., 2003)

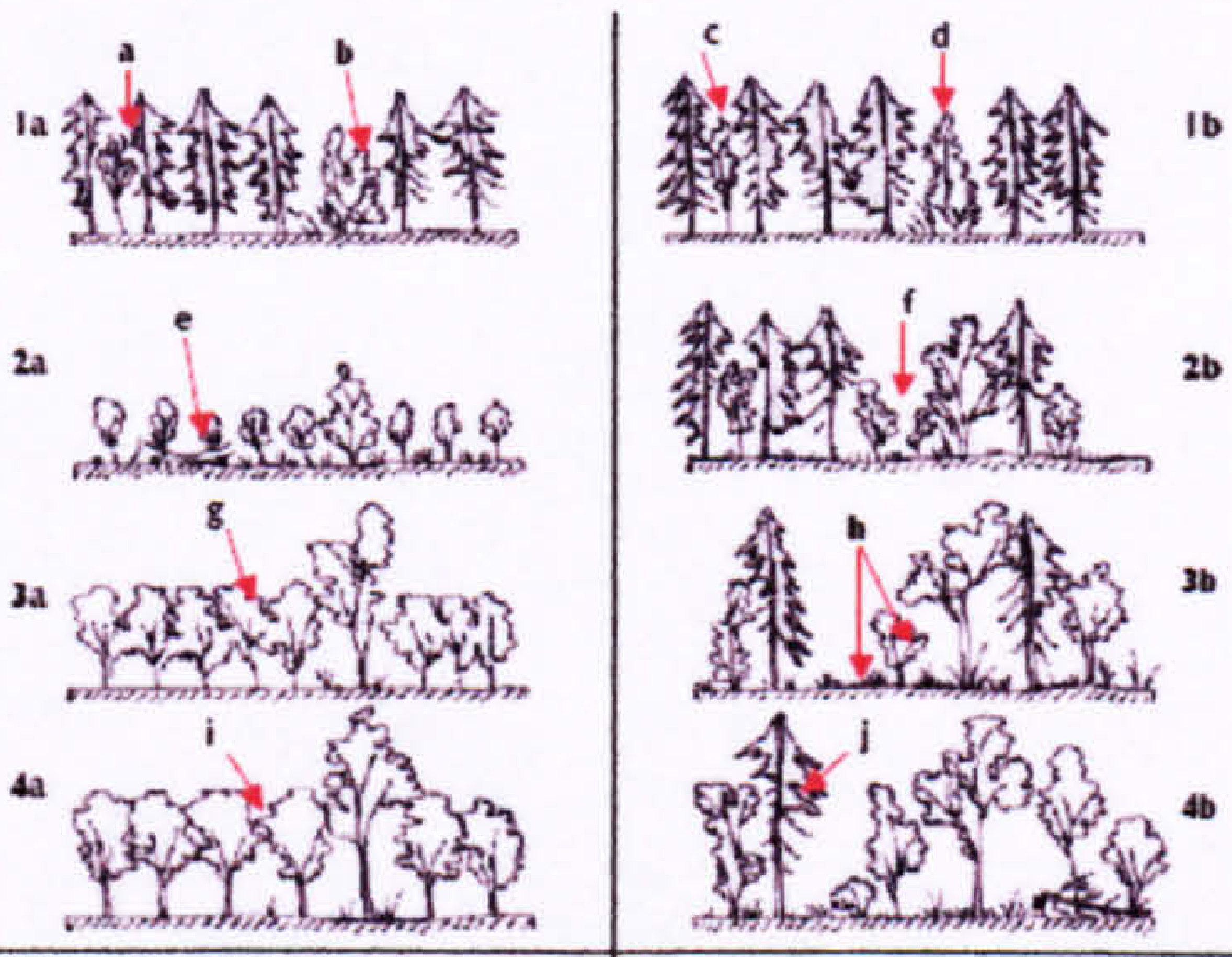
Research also reported canopy type affected remnant interest levels. Presence of planted broadleaves led to an improved survival of ground-flora in comparison to conifer sites, while among conifers, larch and pine typically held higher survival scores than heavily shading conifers such as spruce and western hemlock (Pryor et al., 2002) (Fig 5.5). However there was high variability between sites, indicating additional factors were important in determining flora survival (Pryor et al., 2002). In another analysis of PAWS stands with high ground-flora cover tended to have high species richness (Kirby, 1988b). Canopy density has also been shown to be associated with poor ground-flora survival (Pryor et al., 2002) (Fig 5.4). However canopy cover cannot always reflect the range of past regimes, for example due to past thinning and harvesting. This was considered to account for levels of variability seen where several currently open sites showed low survival scores that may be due to past levels of dense cover prior to clearfelling and additionally the low levels of difference in survival between gappy and closed-canopy woods (Pryor et al., 2002). When assessing levels of remnant interest Pryor et al noted that consideration should also be given to the interest remaining in areas adjacent to the site, where valuable features may remain or exist (Pryor et al., 2002).

Current research therefore highlights the wide range of features surviving within PAWS, and notes that restoration must be based on a full assessment of stand features rather than on broad assumptions from species or age (Pryor et al., 2002). In particular the value of recording the features outlined in Table 5.10 has been noted, with full species surveys not being necessary (Pryor et al., 2002). However biodiversity assessment will always be limited by the species or features targeted during the assessment and some organisations have suggested that all PAWS sites will retain some level of ecological interest, allowing their restoration (The Woodland Trust, 2005).

5.5.3.3 Restoration management

Where restoration is a priority management will depend on the nature of the interest remaining, together with an assessment of strategic management priorities relating to restoration costs (Appendix A5.3). Pryor et al considered PAWS restoration comprised three main elements: removal of the plantation crop, restocking or re-establishment of the native species and retention and management of semi-natural components of the existing stands (Pryor et al., 2002). When levels of remnant interest are high, full restoration is recommended (Appendix A5.2). Where lower levels exist, or are localised, then partial restoration is suggested to maintain or gradually enhance levels of interest with options for full restoration being possible in the future (Thompson et al., 2003). When restoration management is initiated and conifers are removed semi-natural tree cover may develop if suitable seed source trees are nearby but other detrimental effects may also occur, such as the natural regeneration of unwanted conifers (where mature conifers occur) and the growth of ruderals (Radford, 1998), therefore monitoring is required.

In order to monitor restoration success guidance recommends placing targets, e.g. "to establish locally native tree species ecologically suited to the restoration site" (Thompson et al., 2003). A suggested target for the flora of upland sites being "to re-establish some of the key species (e.g. invertebrate food plants or species that define the woodland type) over half the site within 40 years" (Thompson et al., 2003). Management to achieve these goals and targets principally revolve around the use of gradual "continuous cover" methods which avoid the detrimental effects of clearfelling and which take considerable time to achieve success (**Fig 5.7, Appendix A5.1, A5.4**) (Thompson et al., 2003, Pryor et al., 2002). Selective thinning is recommended around areas of remnant native trees and ground-flora to encourage expansion, while recommendations are given on a wide range of topics from extraction methods to levels of brash remaining following felling (Thompson et al., 2003).



	Clearfell approach	Continuous cover approach	
1a	Initial even aged conifer crop with drawn semi-natural tree a, and small hotspot b in canopy gap.	Same initial stand with semi-natural tree c, and small hotspot d in canopy gap.	1b
2a	Site is clearfelled and replanted. Hazards include windthrow, e, and lack of shade causing excessive weed growth and necessitating herbicide use.	Some thinning around hotspot. Shade is retained but semi-natural tree released f. Natural regen and recovery of remnant flora encouraged.	2b
3a	Young even-aged broadleaf stand enters thicket stage, creating unbroken canopy g with a slight break near tree retained from before.	As semi-natural component develops more conifers are removed, creating a spreading patch of broadleaves interspersed with open ground and conifers h.	3b
4a	Semi-mature even-aged broadleaf crop i with single mature tree.	Uneven aged stand with gaps, shade and occasional senescent conifer elements j.	4b

Figure 5.7
Comparison of continuous cover and clearfell restoration of PAWS sites. Reproduced from: (Pryor et al., 2002)

Sites with intermediate levels of remaining ecological interest may represent high priority sites for restoration as they are at risk from continued negative effects of the plantation (Pryor and Smith, 2002) (**Appendix A5.3**). The majority of guidance therefore recommends long-term and gradual restoration, but initiated rapidly through immediate thinning to allow existing interest to expand and then providing stable canopy conditions to encourage spread. While most restoration favours enhancement of existing interest, some studies have indicated that dispersal-limited flora may never reach restoration sites, and therefore may have to be re-introduced to achieve restoration within acceptable timescales (McLachlan and Bazely, 2001, Bellemare et al., 2002).

In considering the financial cost of restoration works studies have shown similar results to examination of native wood creation. Often the "economic value will be inversely correlated with ecological value, which makes restoration far less painful for owners to contemplate" (Pryor, 2003). Therefore the worst PAWS sites with the lowest current economic value are the more interesting ecologically and are priorities to restore. Key financial factors relating to restoration are the potentially high costs of thinning and felling operations and that incentives for restoration should be much higher compared to the generous funding given to woodland creation (Pryor, 2003). Perhaps most notably gradual restoration to native broadleaves using a shelterwood system was found to be "significantly more profitable than conventional clearfelling and replanting" although premature felling of the stand is noted as being very expensive (Pryor and Jackson, 2002).

5.5.4 Woodland conservation (enhancement and maintenance): ecology and practice

Native woodland and Upland Oakwood sites occur within managed landscapes. While some very large sites are suitable for non-intervention (Mountford, 2000), the composition, size and location of most sites ensures that management is necessary. In long-established natural woodland a range of conditions occur that promote ecological diversity. These include the fall of large mature trees opening canopy gaps, grazing effects of wild herbivores and the action of periodic storms and gales causing wind-throw and disturbance of mature trees. Because many native woodland sites do not occur in sufficiently large or old stands for these processes to occur, management may mimic these actions to enhance ecological value. Even in existing oakwoods thinning is usually possible to improve the structure, release potential future crop trees and promote development of future veteran trees in uniform stands. In Atlantic oakwoods one study found 10%-25% of the basal area could be removed without significantly affecting the biodiversity value (Thompson, 2005). Management activities can be summarised as management of the grazing impacts of woods, parkland and pasture woods, and management of the density of trees and shrubs in mature woodland or woodland within which timber production is occurring.

Typically continuous cover systems are most appropriate to foster conservation interest (Section 5.5.3). In discussing management to create or maintain "favourable condition" within SAC Atlantic Oakwoods on the west coast of Scotland, Peterken and Worrell noted potential long rotation management could allow rotations of 150-200 years, restocking 10-13% every 20 years. Felling patches of up 2ha were considered acceptable in large sites, retaining deadwood at 40m^3/ha, >10% shrub layer cover and reduced grazing to allow regeneration periods of 15 years within every 100 years over most of the site. Thinning operations were defined to remove no more than 30% of the canopy crown at any one time (Peterken and Worrell, 2001). Normal forestry thinning however may impact on existing interest such as veteran trees which are important features in oakwoods, and guidelines have been set to identify suitable trees for retention in thinning operations (Thompson, 2005).

Grazing is critical in woodland management. While naturally occurring in open woodland, grazing may impact on regeneration and ground-flora composition. Chatters and Sanderson (1994) note the majority of trees and shrubs "are capable of maintaining themselves under grazing so long as the full range of natural process is allowed to occur", by which they refer to the occurrence of areas of windblown trees and periods of high and low grazing pressure fluctuations over time (Chatters and Sanderson, 1994). Studies have shown that successful regeneration can occur in upland and oakwood sites with low grazing pressures and locally unshaded or lightly shaded conditions (Hester et al., 1996, Kelly, 2002), and that regeneration of tree species can be affected by the timing and intensity of grazing pressure in upland woods (Hester et al., 1996), or where good quality grazing was also available, reducing the impact of browsing (Pollock et al., 2005).

Many upland Oakwoods fall between two extremes of grazing pressure, either receiving none (through being fenced or enclosed within conifer plantations), or open to adjacent grazed areas (where they may be used for stock shelter and heavily grazed). The effects of grazing may be scale and patch size dependent, but the small size of Upland Oakwood sites means any localised effects of grazing are accentuated. Grazing impacts and woodland conservation interest have been reviewed, identifying a variety of effects on ground-flora, bird communities, invertebrates, and tree regeneration (Mayle, 1999). The physical action of stock moving through woodland, particularly cattle, can cause trampling and compaction of the soil surface which may have detrimental effects on soil organisms and mycorrhiza (Adams, 1975 in Mayle, 1999), an effect seen even with sheep grazing at a site in the Dark Peak (Piggott, 1983). In many upland woods the benefits of trampling in producing seed establishment micro-sites is thought to outweigh detrimental effects of soil compaction, although effects depend on frequency and intensity of trampling (Mitchell and Kirby 1990 in Mayle, 1999). Ground-flora diversity is generally highest at light or moderate grazing levels. Grazing will encourage the retention of open space areas within woodland which have been shown to hold a significant range of species contributing to overall ground-flora and woodland biodiversity (Peterken and Francis, 1999). Reduced grazing can also allow leaf-litter to develop, within which germination may be more successful, and will also prevent direct damage to seedlings and saplings (Ovington and MacRae, 1960, Shaw, 1968b, Shaw, 1968a, Watt, 1919, Hulme and Borelli, 1999). In terms of timing of grazing an increased proportion of seedlings tended to reach sapling status in winter grazed plots such that winter grazing may have less long-term damaging effects on the seedlings (Hester et al., 1996). Generally overall growth and survival to sapling stage was much better in winter grazed than summer grazed plots (Hester et al., 1996).

Intermediate grazing levels are also thought to be best for fauna diversity. Invertebrates and birds benefit from the higher structural and floral diversity at such sites. Small mammals are typically more abundant in sites with abundant ground cover and low shrub vegetation (Petty 1998 in Mayle, 1999). Voles may be limited by trampling and leaf litter disturbance of grazing as well as reduced structural diversity (Mayle, 1999). Indirectly bats, shrews, hedgehogs, badgers and foxes benefit from the grazing as they feed on the invertebrates which are associated with dung (Mayle, 1999).

Several authors have defined suitable grazing levels: 4-7 red deer/km^2 have been suggested to avoid detrimental effects of deer grazing in Scotland (Gill, 2000) while 5 red deer/km^2 was recommended for Exmoor oakwoods (Langbein, 1997 in Gill, 2000). How these translate to smaller deer species abundances is unclear. In lowland pasture woods (which are likely to be more productive than native upland pasture woods) grazing rates of one cow per 5 or 10ha have been suggested (equates to 0.2–0.1 LU/ha) (Chatters and Sanderson, 1994). In considering lowland grazed sites a practical minimum of 30ha was recommended with 100ha preferred to ensure adequate grazing dynamics (Chatters and Sanderson, 1994). Due to the need for groups of cows to be kept in groups of a minimum 10-30 adult cows that the use of such grazing may be limited to woods in excess of 30ha (Mayle, 1999). Broad guidance figures have been compiled for the maintenance of ancient and semi-natural woodland sites by grazing (Table 5.12).

Table 5.12
Recommended grazing levels for maintaining Upland Oakwood sites. Source (Mayle, 1999).

NVC	Community	Stocking levels Max density / ha	
		Cattle	Sheep
W11 / W17	Upland oakwoods Oak dominated	Summer only = 0.2 – 0.5 Continuous = 0.05	Continuous = 0.33
	Upland birchwood Birch dominated	1st 5-10 yrs = 0.07 10yrs onwards = 0.1	1st 5-10 yrs = 0.5 10yrs onwards = 0.7
W16	Lowland oakwoods	1st 5-10 yrs = 0.07 10yrs onwards = 0.1	1st 5-10 yrs = 0.5 10yrs onwards = 0.7

5.6 Discussion: woodland classification, site history, ecology and strategic woodland conservation priorities

5.6.1 Introduction

The characteristics of woodland species, and how they interact with woodland spatial and site abiotic characteristics influences resulting site biodiversity and is thus of importance when considering woodland conservation issues at the landscape scale.

5.6.2 Ancient woodland and woodland classification

Woodlands can be classified by their dominant canopy type, estimated longevity and levels of "naturalness", which can be used to assess a wood's conservation interest. Ancient woodland sites are of prime importance to woodland conservation, with semi-natural ancient woodland sites (ASNWS) being the most valuable. The ancient woodland inventory records the location of ancient woodland sites which can be considered to act as sources of woodland specialists within woodland conservation strategies. Ancient woodland sites may have characteristic locations, e.g. topography and landscape position, due to historic land management. Such characteristics may therefore be useful in examining areas of land that may hold long established woodland, but have not been recorded on the ancient woodland inventory. Ancient woodland sites are typically the only woodland in a landscape with known history, although other areas of woodland may also be as old, but may lack historical records.

5.6.3 Woodland species ecology and dispersal: influences on woodland site diversity

Upland oakwoods are typical of particular environmental and soil conditions. Established or high quality examples hold a typical woodland flora and fauna tolerant of these infertile conditions, occurring in upland and upland fringe environments. Characteristic oakwood flora and fauna are well described within NVC and BAP texts (Peak District National Park Authority, 2002, The UK Biodiversity Steering Group, 1995b), with a range of information available on species preferences and tolerances. Component Oakwood species hold different life-history strategies ranging from generalists to oakwood specialists, traits which affect the precise relationship between woodland type, quality and species occurrence and abundance. While the exact habitat requirements of some species are known in detail the associations and requirements of many species remain unknown, or can only be generalised.

Specialist woodland species are expected to occur in woodlands with certain key characteristics, based on conditions such as topography, soils, and landscape position which impact on woodland longevity. Long-established and ancient woodland sites are considered to act as sources of woodland specialists, especially when they are semi-natural. However additionally a range of woodland species will occur across non ancient woods where they will be affected by woodland abiotic features, landscape position and woodland classification and canopy type. Woodland species may also occur in semi-woodland habitat in the landscape matrix. When considering conservation strategies and woodland networks, the relative dispersal ability of characteristic Upland Oakwood and other specialist species are of interest to compare to woodland isolation and connectivity levels. The composition of newly created or restored woods will depend, for their resulting diversity, on the colonisation of species from nearby sites, will be influenced both by the relative quality of nearby sites and by the relative isolation distances, compared to the dispersal ability of different groups.

A variety of factors impact on woodland colonisation. Dominant factors are the dispersal strategy and distance ability of individual species, the relative importance of short-distance versus sporadic long-distance dispersal, the timescales over which dispersal and colonisation occur and the relative favourability of the colonised site to growth and further recruitment. Species will be strongly affected by local dispersal mechanisms. Species dispersed by birds or ants for example being at a disadvantage when these species are absent or at low densities. Studies suggest population size and seed crop will influence dispersal events, the probability that at least one seed will disperse to a colonisation site will increase with the size of the source population and the number of propagules released, thus larger source populations (e.g. from more diverse or higher quality woods) will have increased chances of colonising sites (Wilson, 1993, Clark et al., 1999, Portnoy and Willson, 1993), and will also have increased chances of occasional long-distance dispersal events. The dispersal ability of trees and ground-flora with smaller populations in smaller or lower quality patches is therefore likely to be lower than larger populations from larger or higher quality patches. Guidance from the Forestry Commission suggests that in the uplands very small fragments of woodland, as those remaining in cloughs and steep banks will only be able to successfully regenerate across approximately 10-15m wide areas surrounding a site, in the short term (Thompson, 2004). The limited powers of plant dispersal is emphasised by research where events beyond 100m are often classified as "long-distance" for flora (Cain et al., 2000).

Estimates of the colonisation / dispersal potential of some principal British tree and shrub species are available (Table 5.8). However these are based upon achieving dense regeneration by natural regeneration over short timescales for owners objectives and to meet grant requirements (Thompson, 2004). Insight into the extent of the dispersal "tail" can be gained by examining estimated long-distance dispersal during the Holocene (Table 5.8). Studies have suggested that where leptokurtic long-range dispersal occurs then distances of the order of 1-10km may be important (Clark, 1998), and these approximately match the estimated Holocene rates. If stochastic long-distance dispersal events, and longer timescales, are considered then the relative dispersal strategy and life history of different flora species may be less relevant to predicted dispersal (Clark, 1998). The relative importance of including estimates of mean local dispersal or sporadic long-distance dispersal ultimately remains ambiguous and determined by the timescales over which research is being considered (Hanson et al., 1990, Clark, 1998, Cain et al., 1998, Cain et al., 2000).

In considering broad approximations based upon short term "local" dispersal over a period of 5-20 years typical Upland Oakwood tree and shrub flora may be able to disperse / colonise 50-200m while ground-flora species may be limited to 5-20m. Ultimately such distances would be highly variable and affected by factors such as the landscape matrix, spatial context and occurrence of local fauna, as for example occurrence of widespread deer populations could lead to increased incidences of dispersal of even ground-flora species across distances of kilometres (Eycott et al., 2004). Immobile invertebrate fauna may be expected to show similar distances to the woodland ground-flora while mobile fauna such as birds and bats are expected to regularly move and colonise sites across distances of several kilometres. If colonisation were considered over timescales of 100-200 years increased distances of 100–500m for tree and shrub species and 20-50m (100m) for ground-flora species may be more realistic. Ultimately when conservation strategies are examining networks over human centred timescales then it is reasonable to utilise the shorter term estimates. By using broad estimates of colonisation ability it is possible to model potential likely colonisation events and by relating distances to broad potential groups to be able to analyse whether newly created or restored woods are likely to be able to be colonised, or to exchange woodland species with nearby woods.

5.6.4 Woodland diversity: determinants of composition and diversity at the site scale

Woodlands and the communities they support are variable. Under semi-natural conditions flora and fauna develop characteristic communities that are strongly related to local site factors, allowing particular communities to be recognised (Rodwell, 1991). In managed woodlands the effects of management operations and the imposed woodland composition may have strong impacts on the species present. Key factors identified as influencing woodland species richness are: regional location, climate, elevation and geology, woodland longevity, form of pre-cursor land-use and length of any agricultural activity, canopy composition, woodland management, woodland maturity, occurrence of deadwood, veteran trees and structural diversity, topography, aspect, soils, hydrology and the presence of watercourses (Table 5.13).

A recent study of British woodlands found woodland vegetation composition was strongly determined by climate and nutrient availability, largely resulting from variability in rainfall, soil pH and accumulated temperature (Corney et al., 2004). Across Britain regional ground-flora composition in ancient woodland sites is also mainly explained by climate and soils, with soil pH most important in effecting species richness (Watkinson et al., 2001). Additional research found ancient woodland composition was largely determined by topography, soils and edge-effects, with only minor effects of canopy type (Thomsen et al., 2005). These factors result in the development of the different natural woodland types seen within the NVC classification system across the country (Rodwell, 1991). However local studies have shown that large parts of the variation in ancient woodland ground-flora composition may not be explained by such recorded factors, and may be due to different aspects operating at the stand scale such as levels of grazing activity between sites (Watkinson et al., 2001). Flora typical of Upland Oakwoods sites may be restricted to micro-niches within woods or where management has not altered natural conditions. Woodland management operations, and in particular canopy manipulation can strongly impact on woodland flora and fauna.

A range of research has woodland site features known to be associated with biodiversity levels in a number of groups. These relate to underlying abiotic characteristics of woodland sites such as geology, hydrology and topography, and to current woodland habitat conditions such as woodland density, canopy type and structure. Interesting research potential exists in examining the links between these two areas, e.g. where the abiotic factors also influence within-stand features, such as topography being linked to likelihood of woodland longevity, management intensity and naturalness.

Woodland communities are particularly likely to be diverse where woods are long established, or for secondary woods, where woods have developed on habitats that have only been subject to low intensity land-use or experienced short periods of non-woodland land-use, or where such usage was historic and not recent (Peterken and Game, 1984, Wulf, 2004, Honnay et al., 1999c, Dupouey et al., 2002). Broadleaved woodlands are expected to be more diverse than coniferous, while mixed stands hold intermediate communities and diversity (Donald et al., 1998, Donald et al., 1997, Bibby et al., 1989, Coroi et al., 2004, Kirby, 1988b). Woodlands with diverse canopies of shrubs and trees hold higher fauna diversity than more impoverished stands (Table 5.6) (Diaz, 2006, Donald et al., 1998, James and Wamer, 1982). Stands that experience dense or heavy shading from canopy species hold lower diversity. A diverse structural composition increases biodiversity levels, with a high variation in sizes and ages of trees and several canopy / shrub layers being present, additionally a mature canopy cover and presence of frequent deadwood and veteran trees are associated with higher biodiversity. Woodlands that occur across or incorporate areas of more extreme topography, multiple aspects and include cliffs, ravines, steep slopes a diversity of soil types and presence of streams are particularly likely to be associated with high biodiversity level, this is due to the diversity of soils, hydrology and nutrient and pH conditions and the likely presence of ancient or remnant communities unaffected by management or grazing in areas of extreme topography (Stahle and Chaney, 1994, Therrell and Stahle, 1998, Spencer and Kirby, 1992, Thompson, 2005). Individual wood flora diversity is therefore likely to be highest where several NVC communities or transitions occur within a wood, promoted by features such as the presence of streams, along which areas of wet woodland, or flushed woodland may occur (NVC: W4, W7, W8, W9) (Rodwell, 1991). One study noted that patch habitat diversity was critical for promoting high botanical diversity, and that this was the principal reason larger stands were typically more diverse, that area was simply associated with habitat diversity which was the driving cause (Honnay et al., 1999b). Studies have also indicated high biodiversity levels are associated with high variation or extremes in topography, aspect, soils, hydrology and the presence of watercourses as these promote occurrence of a range of NVC communities and associated fauna.

The diversity of woodland fauna is to a great extent linked to that of the flora. Many species, including invertebrates and woodland birds have preferred hosts from which feeding or other resources are gained (Table 5.6), therefore where botanical diversity in the ground-flora, shrub and canopy layers are high, faunal invertebrates and avian diversity are also expected to be higher. The presence of these features are strongly affected by forestry management methods and by the dominant tree species within the canopy and woodland habitat type.

Overall past research suggests a range of features promote woodland diversity at the site scale, but that many of these may be correlated. Features leading to diverse and natural flora, with a varied structure, and likely to hold diverse fauna, and therefore hotpots of biodiversity across a number of species groups, are expected to occur where conditions promote high woodland biodiversity in a site. Research thus suggests several characteristics driving biodiversity may be able to be extracted and modelled for use in network and landscape-scale modelling. Opportunity exists to examine the range of variables potentially predictive of site-scale diversity and examine their use in landscape scale conservation planning.

Table 5.13
Generalised factors affecting richness of component or characteristic species of established Upland Oakwood communities at a woodland / "site" scale

Factor	High biodiversity	→	Low biodiversity
Climate / elevation	Central, core zone of climate and elevation tolerances	→	Reduced at extremes of climate at elevation
Site longevity	Old, long established wood sites	→	Recent, young woodland sites
Agricultural use prior to woodland	Low intensity, short duration and historic agricultural use	→	High intensity, long duration and recent agricultural use
Canopy Composition	Broadleaved stands. Multi-species stands	→	Coniferous stands. Single species stands
Woodland management	Infrequent + low intensity, but frequent enough to maintain some open ground areas	→	Frequent high intensity operations
Woodland maturity	Old, mature, established woods	→	Young, developing woodlands
Deadwood + veteran trees	Frequent, large diameter deadwood and frequent veteran trees	→	No veteran trees and no large deadwood
Structural diversity	Complex, multi-layered woodland structure	→	Simple, single-layer woodland structure
Topography, aspect, soil	Changes in topography and multiple aspects. Multiple soils types	→	Uniform level topography and aspect or uniform consistent slopes. Single soil type
Hydrology + watercourses	Variation in hydrology. Presence of multiple watercourses	→	Single or uniform hydrology conditions. No watercourses

5.6.5 Woodland conservation, restoration and creation

A range of methods and expertise exist for the creation of native woodland communities. Due to the dynamics of woodlands the management required may be gradual and long-term, not necessarily fitting easily with incentive based grant schemes. Schemes for woodland creation can be designed based upon woodland ecology, maximising the input of woodland colonisation to enrich site diversity, or using planting composition and design to mimic and replicate diverse natural woodlands (Table 5.9). Either way the resulting diversity will be maximised where site conditions are varied and the local landscape promotes woodland species dispersal or colonisation. Natural regeneration at creation schemes can be achieved by a number of design considerations and attention to site location and context. Research indicates the previous land-use will also have a strong impact on developing woodland site quality.

Species of conservation interest may be restricted to long established or ancient woodland sites. Species are not always able to colonise or re-colonise sites successfully. High importance exists within ancient and semi-natural woodland as current pools and sources of woodland species. Opportunities for enhancing woodland conservation interest occur with the restoration of ancient woodland sites to enhance the range of species occurring, both locally to enrichen individual sites and to increase the regional pool of potential colonists. Guidance exists on the assessment of remnant interest within restoration sites. There is sufficient knowledge of ecological aspects to enable woodland conservation of existing sites to foster enhanced ecological interest, for example by modifying grazing or woodland densities. The range of research and techniques associated with restoration of ancient woodland PAWS sites will also be of value in restoring / converting existing non-ancient plantation sites to native cover.

The ecological interest of existing woodland sites can also be manipulated and maximised by management. Critical factors are canopy composition and density, through felling and re-stocking regimes. Low intensity and more naturalistic regimes result in higher richness. Grazing levels and management of open ground habitat may strongly impact on biodiversity. Management that results in a wide range of tree species and ages, with a diverse canopy structure, incorporating areas of permanent open space, with light grazing regimes, that periodically introduces disturbance, but allows growth of sapling and shrubs, is of maximum benefit. The potential for positive management, and in particular appropriate grazing, of existing woods may be determined by factors such as their location, landscape context and ownership.

Table 5.14
Comparison of management to create native woodland habitats through new native woodland creation or restoration of PAWS sites. Source: (Pryor, 2003).

Feature	Restoration of PAWS	Creation of new native woodland
Woodland Habitat / conditions	Gradual restoration can sustain woodland/shaded conditions which may suit many woodland interior species.	Sites are essentially open-ground habitats for the first 5-10 years, and scrub thickets for the next 5 years, not achieving a woodland canopy until 15-20 years after planting.
Species assemblages	True woodland communities may have survived, albeit in a depleted or eroded state.	New communities are a combination of planted species, species surviving from the previous land-use and ruderals.
Plant strategies	Sedentary species, including ancient woodland indicator species, may have survived and respond well to restoration.	The dominant species appearing naturally on the site are likely to be pioneers and ruderals, which are often relatively abundant in other habitats.
Interior vs marginal habitat	The context of PAWS means they are often part of larger forest blocks and restoration will often lead to substantial gains in "core forest area" (i.e. woodland interior habitat, well buffered from non-woodland land-uses).	In England and Wales new woodlands are frequently small and isolated, giving a high edge:area ration and relatively large amounts of woodland margin habitat. Only in the Scottish highlands are much larger and contiguous areas being created.
Isolation	Parts of larger woodlands were frequently converted to plantations, and hence PAWS often adjoin surviving ASNW, which assists re-colonisation.	New woodlands are frequently isolated from ASNW and may thus accumulate woodland species relatively slowly. In the Scottish highlands, contiguity with semi-natural woodland is more likely.

5.6.6 Overview of woodland conservation ecology insights for landscape planning

A wide range of information and research exist in relation to woodland ecology, site classification and regeneration that are valuable to the consideration of woodland conservation strategies. Current information suggests that different benefits can be gained through strategic management aimed at woodland conservation, restoration or creation. Conservation works are a long-term process, new woodland will take many years to develop woodland conditions truly suitable for woodland species colonisation. Strategies should consider the occurrence of existing levels of species interest, and therefore could initially focus upon woodland restoration, conversion and the conservation of existing woodland within network and landscape-scale strategies, prior to considering woodland creation. Pryor (2003) gives a variety of ecological features promoting the ecological value of restoring PAWS in comparison to creating new woodland (Pryor, 2003) (Table 5.14). The addition of a management option of native woodland creation from existing non-ancient plantations shares several of the benefit of PAWS restoration.

Woodland biodiversity within individual sites and the broader network will depend on the balance between occurrence and diversity of species already occurring at sites (existing woodland, restoration and creation sites), and the availability of local colonists, and how this range of biodiversity can be enhanced. Multiple species must colonise a site, or must be encouraged to spread from areas where they currently occur as remnant populations. Research suggests this can be achieved by examining the features that promote woodland biodiversity at both the site and landscape scale. Studies have shown the importance of hotspots of vegetation remaining during conversion to plantations, while several research has also shown that ancient ground-flora species may survive periods of woodland clearance or conversion to agriculture and then re-colonise woodland sites from small and marginal remnant habitats such as hedges, banks, ditches, given time. In some environments such as the uplands the existence of "semi-woodland" habitats can serve a similar purpose to remnants hotspots within plantations, allowing pockets of woodland species to survive in the landscape, thus enhancing incidences of woodland expansion when suitable conditions arise. Species mobility is a key factor. A range of studies have shown certain characteristic woodland flora and fauna have low dispersal ability and will only colonise over short areas, or whose conservation must examine the occurrence of such remnant hotspots of species within plantations, or in open ground or semi-woodland habitats. A range of abiotic factors have strong influence on woodland biodiversity at the site scale and impact on existing woodland diversity and occurrence of remnant woodland population in plantation woods and open ground, potential woodland sites. Woodland communities are also strongly affected by biological features such as woodland richness itself, and structure, and many measures of woodland quality may therefore be inter-correlated. At the landscape scale research suggests that in the shorter term local range estimates of species dispersal of different woodland species groups can be of use in examining potential levels of colonisation or dispersal between sites, and their resulting effects on biodiversity levels. Key areas for conservation planning research are thus an examination of the driving abiotic causes of woodland site diversity, the associations between the many site biodiversity measures and the modelling of potential dispersal / colonisation for species groups. Further challenges are then how to use such knowledge in landscape planning. A range of these research areas have already been analysed in relation to the formulation of woodland conservation strategies at the landscape scale, and together with strategy research inspired by broad landscape ecology theory (Chapter 3), these are reviewed in the following chapter (Chapter 6).

5.7 Chapter Summary

Woodland classification and Ancient woodland status

- Woods can be classified by canopy type, longevity and "naturalness"
- Ancient woodland, particularly semi-natural, are of prime importance to woodland conservation, acting as known sources of specialist woodland species
- A large proportion of the ancient woodland resource has been converted into PAWS
- Site distribution and frequency vary, but tend to be small, fragmented and clustered
- Woodland classification type can be used to indicate relative biodiversity levels
- Woodland naturalness and ancient wood status may be associated with topography

Woodland species assemblages

- High quality oakwoods hold a woodland flora and fauna tolerant of infertile conditions, occurring in upland and upland fringe environments
- Characteristic oakwood flora and fauna are well described within NVC and BAP texts
- Oakwood species range from generalists to specialists, traits which affect the precise relationship between woodland type, quality and species occurrence and abundance
- While the habitat requirements of some species are known in detail, the requirements of many species can only be generalised using life-history types or broad species groupings.

Woodland colonisation, regeneration and dispersal

- New or restored woods increase in diversity with colonisation from nearby sites
- Some species readily colonise, while others such as woodland ground-flora may be poor colonists
- Seed dispersal declines with distance from seed sources
- Distances are likely to be proportional to the size of the source population
- Trees and shrubs may colonise sites over short distances of 50-200m
- Herbaceous species may colonise 5–20m, although some ground-flora are almost unable to colonise unless sources are directly adjacent to a site
- Species dispersed by animals may result in longer colonisation events, e.g. kilometres
- Typically fauna have stronger colonising ability than flora
- Species occurrence in patches can result from the effects of local dispersal limited by the spatial arrangement of habitat
- Critical in estimating colonisation is the differences between local, short-distance and stochastic long-distance events
- Short-term dispersal are more affected by dispersal mechanisms, species mobility and life-history
- Long-range dispersal effect landscapes over long periods of time, in the short-term, local colonisation events dominate
- Due to lack of dispersal data for most species, broad approximations by species life-history type or species groups may be used for modelling

Woodland history and ancient woodland indicator species

- Ancient woodland indicator species are typically dispersal-limited flora indicative of site longevity, although several faunal groups have also been proposed
- Sites with numerous Ancient Woodland Indicator species are considered likely to be ancient

Woodland diversity and species richness

- Key factors influencing woodland species richness are: regional location, climate, elevation and geology, woodland longevity, pre-cursor land-use, length of agricultural activity, canopy composition, management, maturity, deadwood, veteran trees, structural diversity, topography, aspect, soils, hydrology and the presence of watercourses
- Species richness, particularly flora and less mobile fauna, are promoted by site longevity, and negatively affected by the duration and intensity of any previous agricultural land-use
- Woodland flora is largely determined by soil type, hydrology, topography, and can be heavily influenced by management
- Upland Oakwood flora may be restricted to micro-niches within woods or where management has not altered natural conditions
- Many woodland diversity features may be inter-correlated
- Richer and more natural communities occur where woodlands hold a broadleaved canopy comprised of native species, with low intensity management that causes disturbance levels (canopy changes: felling, re-stocking) similar to levels experienced under natural conditions
- Intensive operations, use of non-native, especially dense conifers under short rotations, negatively affect species richness, mixed stands can hold intermediate biodiversity levels
- In established woods biodiversity is associated with canopy maturity, increased presence of deadwood and veteran trees
- Canopy maturity is associated with increased structural diversity which itself is positively associated with diversity by providing niches for feeding and nesting among the avifauna and creating different shading and competition effects in the flora, promoting diversity
- In Upland Oakwoods there will be a strong association between total biodiversity and presence of component NVC communities; where multiple communities occur, floral diversity will be high
- Where floral diversity is high faunal diversity will be high due to increased availability of niches, occurrence of food plants, hosts or nesting sites
- High biodiversity levels are associated with variation or extremes in topography, aspect, soils, hydrology and the presence of watercourses

Woodland creation

- A range of methods exist for native woodland creation, ideal management involves natural regeneration and the management required may be gradual and long-term
- Creation can be designed based upon woodland ecology, maximising the input of colonisation to enrich site diversity, or use planting composition and design to mimic and replicate natural woodlands
- Previous land-use will also have a strong impact on developing woodland site quality

Woodland restoration

- Existing ancient woodland, including PAWS are important sources of woodland species
- PAWS restoration enhances the local species-pool and increases colonists at the regional scale
- Guidance exists to assess levels of remnant interest within PAWS
- Most PAWS retain some level of biodiversity interest from former native communities
- Surviving features are often associated with biological features, such as higher broadleaved cover
- Research on PAWS restoration will also be of value in converting existing non-ancient plantations
- High quality restored sites represent sources of specialist species to colonise or move among adjacent woodlands
- Restoration of existing woods will increase woodland species richness and population size quicker than creating new woods

Woodland conservation

- Woodland ecological interest can be maximised by appropriate management
- Management has a positive influence on diversity, non-intervention is only rarely suitable
- Critical factors are appropriate canopy composition and density levels resulting in higher richness
- Management resulting in diverse tree species, ages, diverse canopy structure, that periodically introduces disturbance but also allows growth of sapling and shrubs is of maximum benefit
- High quality sites can represent sources of specialist and generalist species to colonise nearby woods

Overview of woodland ecology review insights for landscape planning

- Due to the different levels of existing woodland biodiversity, strategies must typically address separate goals for woodland conservation, restoration and creation
- Achieving high diversity / natural sites at new woodland creation sites is likely to be most difficult due to the limitations of woodland species dispersal, but opportunities can be maximised by selecting sites where existing levels of future potential diversity may be highest, and where local woodland landscapes promote colonisation
- The management option of native woodland creation from existing non-ancient plantations shares many of the benefits of PAWS restoration
- Studies have shown the importance of woodland biodiversity hotspots, both regionally as the existence of ancient woodland sites in the landscape, and within individual woodlands, where multiple features of woodland biodiversity are often correlated
- Key aspects of woodland conservation ecology for landscape planning are the abiotic conditions that promote diversity in existing and newly created woods and may correlate with levels remaining in plantation woods
- Woodland species are known to have widely ranging dispersal ability, and in order to address planning over relatively short timescales, e.g. decades, then approximation of species dispersal ability by species group / life history type are required for use in landscape planning

Chapter 6

Wooded landscape conservation strategies:

A literature review

This chapter reviews current landscape-scale woodland conservation strategies from the academic and professional conservation literature. Many such strategies have now been undertaken, typically involving landscapes scales from 10,000 ha (100 km^2) to 500,000 ha (5,000 km^2) in extent. These studies have utilised a range of methods. The following review considers this research separately within categories of strategy formulation (Section 6.3) and strategy implementation (Section 6.4) methodologies. Strategy formulation methods are classified as: landscape based, community/habitat based, or species based. Strategy implementation methods are classified as landscape design/guidance, conservation zone mapping, and land allocation targeting / optimisation methods. The chapter concludes by discussing strategy merits and limitations, highlighting areas requiring future research and potential applications and developments in the UK.

6.1 Drivers and scales: conserving woodland species, habitats, or networks?

Across a number of years conservation activity has been expanding to address broader scales, culminating in the range of landscape-scale initiatives practiced today. In multi-use habitats, such as woodland, drivers include economic factors in addition to biodiversity aims. Studies have highlighted the potential for focussing landscape planning using an environmental accounting approach (Haines-Young, 2003), while in addition to biodiversity spatial woodland conservation planning methods may include economic considerations, with reserves selected on cost-benefit basis (Siitonen et al., 2002), whilst strategies to locate new urban woodlands have incorporated recreation in priority scoring (van Elegem et al., 2002). Other conservation activity drivers may be the benefits of tree cover in reducing runoff from farmland (Carroll et al., 2004), capturing atmospheric ammonia deposition for pollution control (Sutton et al., 2004), and the need to conserve sufficient levels of genetic diversity in forest ecosystems to allow systems to respond to future global environmental change (Ehrlich, 1996). Concerns for landscape-scale conservation is increasingly needed due to the effects of agricultural intensification in the landscape matrix in addition to potential effects of climate change (Peterken, 2002b). Although conservation may attempt to cover such a wide range of factors it has been recognised that conservation science still needs to address methods to simply retain species within degraded and fragmented forest habitats rather than additional factors such as functional integrity (Simberloff, 1999). The focus of much current woodland conservation planning has been the maintenance of biodiversity and possible future enhancement by expanding total forest area. At a local scale woodland site management may be driven by species biodiversity enhancement (Greenaway and Hill, 2004, Stowe, 1987), being examples of fine-filter conservation (Schwartz, 1999) (Section 3.3.2). At larger scales woodland conservation initiatives may address the extent and quality of woodland habitats across landscapes.

Woodland spatial conservation strategies have resulted from integration of a number of sources including spatial planning within the forestry sector, county and Natural Area Biodiversity Action Planning (BAP), nature reserve selection methodologies and a range of landscape ecology based research. Due to the dramatic landscape changes caused by forestry and the need to effectively plan resource use over large areas the timber sector holds a long history of visual landscape and environmental spatial planning. Early studies examined cutting patterns and rotations and suggested spatial management strategies to maintain high levels of landscape biodiversity while accommodating forestry use (Haris, 1984, Franklin and Forman, 1987) based upon island biogeography theory (MacArthur and Wilson, 1967). North American authors revisited the impact of cutting strategies and aspects of spatial landscape design to aid conservation (Li et al., 1993). Forestry planning therefore has been developed that includes consideration of spatial aspects and structures in optimization planning (Kurttila, 2001, Zavala and Oria, 1995, Baskent and Keles, 2005). Geographic Information Systems (GIS) are increasingly used within such planning and may include visual interfaces allowing easy interpretation of alternative forestry management scenarios (Naesset, 1997). Within the UK forestry use and woodland creation may be targeted through spatial "indicative forestry strategies" (DOE, 1992), which incorporate guidance on nature conservation benefits in addition to visual concerns of forestry locations (Kirby and Miller, 1996). Recent studies have extended beyond investigations of the impacts of forestry practices or attempts to limit the potentially negative locations of plantation forestry to examining the pro-active conservation potential of spatial woodland planning. Such studies recognise that in addition to the conservation of existing woodland resources successful strategies may require the restoration of additional woodland cover (Good et al., 1997, Jerram, 1998, Thompson et al., 2001b).

Under the UK BAP process biodiversity enhancement targets have been set for the conservation, restoration and creation of woodland habitats within national (The UK Biodiversity Steering Group, 1995b) and regional landscapes (Peak District National Park Authority, 2002). These initiatives recognise woodland species and habitat rarity as driving forces for conservation and emphasise issues of site representativeness for habitats of conservation concern. Increases in the areas of favourably managed woodland and in woodland extent are expected not only to maintain presence of these key habitats but also to foster dependant species populations, a process aided by the identification of "BAP species" typical of, or dependant upon, priority BAP habitats. Research has examined the potential for planning BAP conservation and restoration work to avoid conflicts (Lee et al., 2001a) or to maximise efficiency of woodland conservation work (Latham et al., 2004, Purdy and Ferris, 1999). Within the UK woodland conservation is also being encouraged by changes in management practices within the forestry and farming sectors, aiming to foster positive woodland management within the wider countryside (Forestry Commission, 2006, DEFRA, 2006). Biodiversity related reform and uptake of these schemes (England Woodland Grant Scheme, Environmental Stewardship) are necessary to raise biodiversity levels to meet government sustainable development commitments including wild bird populations trends (woodland and farmland birds) recognised as national sustainability indicators (Anon, 1999, Anon, 2005d).

Additional drivers of landscape-scale projects have been the increase in interest in re-wilding projects and re-introduction of naturalistic grazing, especially in the uplands, following the hypothesis of Vera (Vera, 2000, Worrell et al., 2002, Rogers and Taylor, 2003). Vera emphasises the importance of large herbivores in driving woodland dynamics raising the potential for large-scale experiments where natural grazing in woodland and semi-woodland habitat are allowed to drive woodland dynamics. However the importance of large herbivores for natural woodland conditions, their relevance to the English uplands, and the practicality of current re-introduction to create such natural woodland conditions have been contested and are not universally accepted (Kirby, 2005, Hodder and Bullock, 2005).

As a result of the range of drivers and BAP initiatives within the UK, and similar experience globally, conservation initiatives are increasingly addressing the landscape-scale. Where management aims expand beyond sites to address conservation across regions and landscapes factors relating to the spatial arrangement and occurrence of habitat become important. Where conservation aims to enhance or re-create diverse habitats, then opportunities for species flow through the landscape and for colonisation of restored sites must be provided. The methodologies used to achieve this in wooded landscapes are varied.

6.2 Landscape-scale woodland conservation strategies

A wide range of studies have examined landscape-scale woodland conservation within the UK, Europe and North America, driven by conservation targets, theoretical research and "proof of concept" studies. These have utilised landscape ecology and conservation biology theory, modelling studies and organisational experience. Distinctions exist between activity aimed at the practical conservation and restoration of woodland habitats, theoretical aspects of landscape-planning and activity examining situations in which forestry management can be integrated with conservation at the landscape-scale. Key themes emerge as the use of GIS technologies to plan conservation action and the development of methodologies and techniques to address the spatial configuration and occurrence of woodland patches. Strategy scales have ranged from national

(Hampson and Peterken, 1998), regional (Worrell et al., 2002), and county (Ray et al., 2004a), to catchment based (Glimmerveen, 2003).

The problems inherent in attempting to reconcile the development of landscape-scale woodland conservation prescriptions and determining how particular landscape configurations may affect the conservation status of different woodland species can be illustrated by the ongoing evolution of the "Forest Habitat Network" (FHN) concept. Peterken et al 1995 first introduced the concept comprising core forest areas, links and nodes, noting:

> "the aim of a forest habitat network is not simply to increase the area of habitat, but to develop a pattern which both maximises opportunities for species requiring many habitats or large territories, and the successful movement of all species over both short and long distances. It requires shifts in the emphasis of nature conservation policy from species to habitats, from sites to ecosystems, and from local to national and international measures" (Peterken et al., 1995).

In addition to the concept of FHN a range of terms have become frequent in the landscape planning literature and include "habitat opportunity mapping", "vision" mapping / planning and "future landscape" planning (RSPB, 2004, Saunders and Parfitt, 2005, Hewston and Scott, 2000). Whilst allowing consideration of land-use across landscapes (catchment, county, regions) these strategies also allow the consideration of alternative future land-use options, increasing flexibility of planning. This ability to consider future landscapes under different management scenarios and to contrast alternative disturbance regimes has been considered to allow opportunity for linking forest managers and scientists in addressing biodiversity conservation (Boutin and Hebert, 2002).

The following sections examine a range of examples of landscape-scale woodland conservation and associated studies. Initially the research methodologies and examined, followed by review of the strategy implementation and delivery mechanisms. Additionally a broader range of studies and discussion papers have been included where they provide guidance or insight suitable to woodland conservation strategies. Specifications for the inclusion of studies were that they incorporated a substantial element of spatial planning, or applied conservation advice, at a "landscape" scale and they addressed one, or more, of the three key areas of consideration in current BAP planning: woodland conservation, restoration and creation.

6.3 Strategy methodologies

A variety of methodologies have been employed to landscape-scale woodland conservation. These have been categorised into three approaches, although individual studies may overlap, they are discussed separately for clarity. "Landscape based" studies take a top-down approach by focusing on consideration of landscape ecology theory and landscape structure within the area of conservation, possibly analysing existing composition and using analysis to provide conservation guidance. Examples include studies that invoke landscape ecology to examine how strategies may be planned and designed (Smithers, 2000, Hampson and Peterken, 1998). "Environment / Habitat / community based" studies examine landscape with regard to the potential occurrence of the woodland habitat of conservation concern, concentrating on identifying environmental / geo-botanic factors relevant to habitat occurrence and distribution and using this information to give conservation guidance. Examples include models to predict the woodland community that could potentially develop at different sites, for use in planning at various scales (Towers et al., 2001, Hester et al., 2003, Pyatt et al., 2001). Finally "species-based" approaches take a bottom-up approach utilising detailed knowledge of species ecology or may examine the landscape from a species perspective typically using a range of surrogate, indicator or focal species to examine relationships between woodland species–habitat–landscape use and upon analysis of these in relation to woodland occurrence to provide conservation and enhancement advice (Ratcliffe et al., 1998, Brooker, 2002).

6.3.1 Landscape based

Two areas of conservation can be categorised that are "landscape based", these are distinguished by an analysis of landscape structure and pattern with an absence, or only minor consideration of species or habitat analysis in outlining conservation priorities, which may be solely devised from broad ecology rules and analysis of existing landscape structure.

6.3.1.1 Landscape ecology

Several studies have recognised the potential for incorporating landscape ecology principles into forest design and woodland conservation planning (Hampson and Peterken, 1998, Haris, 1984, Franklin and Forman, 1987, Smithers, 2000, The Woodland Trust, 2002). The use of landscape ecology in the design of restored wooded landscapes was enthusiastically taken up by proposals for "Forest Habitat Networks" (Hampson and Peterken, 1998, Peterken, 2000b, Peterken, 2003). These have been applied at a national scale in Scotland (Hampson and Peterken, 1998, Peterken, 2003), more locally in lowland England (Peterken, 2000b), North York Moors (Peterken, 2002a) and with increasing levels of detail and incorporating species-habitat interactions in the Cairngorms (Ratcliffe et al., 1998, Cairngorms Partnership, 1999), West Lothian (Ray et al., 2004a), Scottish Borders (Ray et al., 2003b) and Wales (Latham et al., 2004). The broad FHN approach initially derived from an assessment of the landscape ecological literature and the application of derived rules examining levels of woodland cover and impacts of edge-effects, core area and isolation. Early FHN studies noted a targeted approach was necessary to address past woodland habitat fragmentation and isolation and therefore focussed on designing landscapes that were ecologically "connected" (Hampson and Peterken, 1998). While increased woodland connectivity was considered desirable the strategies recognised potential dangers of increased connectivity being the potential for spread of pests, disease or fire (Hampson and Peterken, 1998). These strategies applied the 30% rule, where wooded landscapes were recommended to cover 30% of available land area (Peterken, 2000b, Peterken, 2002b), in order to promote connectivity, as highlighted by percolation theory and landscape ecology studies (Andren, 1994). Within these strategies woodland distribution was examined with the aim of increasing cover and identifying existing and potential areas of "core forest areas" and "major landscape links" where work could be prioritised (Hampson and Peterken, 1998), with similar suggestions in England for the creation of "forest districts" and "woodland districts" (Kirby and Reid, 1997).

Other UK examples have been the national strategies applied to ancient woodland sites and woodland creation by the Woodland Trust (The Woodland Trust, 2002), based upon island biogeography and landscape ecology (Smithers, 2000). The strategy identified a number of factors critical to woodland biodiversity: ancient woodland habitat, old growth habitat, size, core area, woodland edge adjacent to other semi-natural habitats, density of semi-natural habitats and linkage to open-ground habitats (Smithers, 2000, The Woodland Trust, 2002). A large edge-buffer width of 100m was used to identify core areas and edge-effects and four measurable, mapped, surrogates were developed for woodland biodiversity: density of ancient woodland cover, percentage of Ancient Woodland which is semi-natural, cumulative core-area of semi-natural habitats and area of old-growth (Smithers, 2000, The Woodland Trust, 2002). The strategy was based upon percolation theory and landscape thresholds and defined a sustainable landscape rich in biodiversity as containing 30% semi-natural woodland, 30% other semi-natural habitats and 40% low-intensity land-use (The Woodland Trust, 2002). The strategy then produced priorities based upon analysis of existing land-use and structure.

In addition to studies explicitly using landscape ecology to derive strategies a series of publications have provided guidance on landscape-based woodland conservation as a result of single-species landscape-based, or advisory studies not explicitly devising new landscape-planning methods. These include strategy advice for applied conservation planning (Humphrey, 2003), relevance of landscape ecology to woodland conservation planning (Bailey and Pryor, 2004), strategy implications of woodland species dispersal ecology (Dolman and Fuller, 2003), studies examining English habitat fragmentation (Kirby, 1995) or following examinations of the landscape ecology of species groups including woodland birds (Fuller et al., 1995) and woodland plants (Wulf, 2003).

6.3.1.2 Landscape assessment strategies

Several studies have undertaken analysis of local woodland character through GIS based landscape assessment or by examining alternative conservation strategies in order to develop strategies that can be applied at the county or catchment scale. These may be relatively simplistic and based on an examination of the regional context and woodland character in order to set conservation guidance. English Nature examined local priorities for woodland conservation among Natural Areas (Kirby and Reid, 1997, Reid et al., 1996) using assessments of woodland distribution and quality. In Wales a more detailed approach involved identification of mapped "Ecological woodland units" (EWU) (Latham, 2003). While Natural Areas reflect broad ecological zones and have been designed to be utilised for landscape planning (English Nature, 2005), identification of EWU's focuses on woodland planning. EWU's have been identified based on landscape composition factors including physical proximity of woods, corridors and stepping stones between woods, woodland types, topographic positions, physical separation and management history (Latham, 2003). The scale of individual EWU's were delimited so they were administratively practical but also of a size that ensured woodlands within each EWU were ecologically linked (Latham, 2003). Boundaries were based on river catchments, separation by ground above 300m and major floristic and geological divisions (Latham, 2003). Also in Wales research has refined and updated woodland creation and conservation targets based on woodland assessment within local Landscape Description Units (LDU) areas to create targets that are more regionally targeted than similar Natural Area or county based BAP targets (Griffiths et al., 2004b). Additionally studies have examined the effects that different conservation strategies may have on different landscapes, in terms of structure and typical woodland character attributes (Buckley and Fraser, 1998). Several studies have involved detailed examination of current woodland cover and landscape structure within Landscape Types (Section 3.4.5), enabling targeting at a local landscape-scale (Cairngorms Partnership, 1999, Peterken, 2002a, Griffiths et al., 2004b).

A further range of methods share a range of common GIS based methodologies, but take a hierarchical classification approach to examining and defining woodland conservation areas. These use techniques to identify "opportunity" and "constraints" to woodland occurrence and potential. Such "opportunity" mapping has recently been reviewed by English Nature and the RSPB (RSPB, 2004, Saunders and Parfitt, 2005). Good et al (1997) used a GIS methodology to examine the potential for woodland creation in the English uplands, including the Peak District as a case study area (Good et al., 1997). The study used GIS and expert opinion to examine potential woodland creation areas and potential conservation benefits and constraints. The methodology compiled ITE landcover, OS topography data and Ancient Woodland Inventory data. A study by Jerram (1998) in the Forest of Bowland further developed this methodology. The study utilised Phase 1 habitat maps and AWI data to highlight areas considered suitable for woodland creation, at a local catchment scale.

6.3.2 Woodland environment / Habitat / community based

6.3.2.1 Representativeness and rarity

Several programmes addressing woodland conservation issues have incorporated measures or assessment of habitat based rarity or "representativeness". In considering existing occurrence of habitats simple measures of habitat type, occurrence and rarity may be used to select suitable areas for conservation action. This rationale was used to select reserve networks across the UK in the Nature Conservation Review (Ratcliffe, 1977), and more locally in English Nature's SSSI network, such that examples of rare woodland types are included where they are high quality and typical, but also such that a representative selection of woodland types are included in regional SSSI networks. These networks however were not spatial in their original design and the extent to which they are efficient or optimal is questionable.

6.3.2.2 Environmental envelopes / defining potential habitat occurrence / limits

As an extension from approaches using current levels of woodland cover and existing habitats to delimit conservation potential, several studies have investigated the physical and environmental variables that define the location of naturally occurring woodland habitats, in order to predict woodland zones. By investigating underlying processes that determine woodland distribution these studies have enabled mapping of the woodland habitats that could potentially occur across landscapes. This allows consideration of past-natural, present-natural and future-natural woodland composition, following the work of Peterken (Peterken, 1993, Peterken, 1996). Such distributions can be compared with other land-uses or conservation opportunities when examining conservation priorities. These studies examine factors such as soils, climate and altitude with, or without, a consideration of existing landcover and management in order to define the current environmental and ecological limits to woodland cover and different woodland types. Studies have included the Native Woodland Model (NWM) in Scotland (Towers et al., 2001, Hester et al., 2003) and the Ecological Site Classification (ESC) program and its GIS extension (Ray et al., 2003a, Pyatt et al., 2003). Conceptually similar models include GIS based models mapping the current environmental "niches" of Eucalyptus tree communities in Australia (Austin et al., 1990) and of bracken in Scotland (Birnie et al., 2000). Some of the principally landscape assessment methodologies carrying out opportunity mapping for woodland creation have also undertaken GIS modelling work that partly identifies the extent of zones within which native woodland may be predicted to occur (Good et al., 1997, Jerram, 1998).

The NWM was developed to aid strategic woodland planning for the UK BAP using expert knowledge and biophysical data to combine a national soils map (1:250,000 scale) and landcover map (1:25,000 scale) to predict "potential woodland distribution for current environmental conditions" (Towers et al., 2001). The output was considered applicable at scales from 1:50,000 to national level (Hester et al., 2003, Towers et al., 2001). Within the NWM the authors assumed the use of soil data captured sufficient climatic and environmental determinants of woodland cover that direct investigation of the climatic limits of individual woodland species was unnecessary. In contrast the Ecological Site Classification (ESC) developed by the FC to aid forestry and native woodland planning was based on direct examination of climatic variables, combined with analysis of site conditions in order to plot potential for native woodland occurrence (Ray et al., 2003a, Pyatt et al., 2001). ESC utilised accumulated temp above 5degrees, maximum summer moisture deficit, windiness, continentality, soil moisture regime and soil nutrient regime, all with a typical resolution of 1ha. The ESC model uses fuzzy membership classes to map NVC woodland types based on an ordination of all the species data from each NVC type using Hill-Ellenberg values to establish species-environment links. The ordination gives continuous scores for suitability based on climate and soils which can then be mapped. Similar studies have mapped landscape "niches" of bracken in Scotland (Birnie et al., 2000) and different Eucalyptus species communities in Australia (Austin et al., 1990). Birnie et al (2000) used a rule-based model to examine the potential habitat niche of bracken by applying decision rules to GIS datasets and comparing predicted distribution against independent datasets. A key assumption was that the niche related to current environmental conditions (Birnie et al., 2000). Topographic, edaphic and climatic distribution limits were derived from the literature (Birnie et al., 2000). The rules were combined within a GIS based on 1:50,000 OS DTM and 1:250,000 soil survey data to define optimal locations for bracken occurrence (Birnie et al., 2000).

Model based approaches are important in that they can ignore current management and land-use restrictions and map the "potential" occurrence of habitats. Therefore such distributions differ from exercises that have simply examined the range of site conditions occurring across a landscape that are similar to the current distribution of native woodland cover as a tool for future woodland planning (Good et al., 1997). The modelling of habitat-environment links enables the full potential spread of habitats to be examined.

6.3.3 Species based

A number of studies have undertaken "species based" strategies where species conservation issues are explicitly addressed and modelled. While the broad occurrence of a habitat type may be defined with relative accuracy, when species based conservation is examined a range of additional technical and value based judgement factors apply. Within a single woodland habitat a myriad of species occur, beyond the range that can be recorded, or even identified. Species may use a habitat at varying degrees of scale and with varying levels of specialism and dependency. With the recognition that any assessment or consideration of the full species complement of a habitat is impossible, a range of approaches are taken to utilising species information to target spatial conservation action. "Reserve selection" methods in the sense of Pressey et al (1993) are utilised and include: flagships, umbrellas, indicators, hotspots, minimum sets, gap analysis and gap analogs (Schwartz, 1999) and more recently include focal and eco-profile species approaches (Lambeck, 1997, van Rooij et al., 2004) and various "keystone", "priority" or "representative" species (See Section 3.3.2). These methods often use species records or knowledge of species requirements as surrogates for uncollected information, on which action is based (Caro and O'Doherty, 1999, Noss, 1999).

Methodologies differ depending on which of these species approaches are taken, while some studies show mixed methodologies. The approaches taken reflect the importance of either applied conservation action or research based problem examination. While the former may undertake mixed methodology approaches or may fail to clearly justify, explain and categorise their approach the latter are more likely to focus on clear definition of conservation rationale, approaches and methodologies and to be more focussed on single approaches and examination of their relative efficiencies (Table 6.2). Two main study categories exist. Approaches using existing information on species occurrences to prioritise sites, and methods examining a species-based view of the landscape to examine species-environment interactions, and make suggestions on priorities for conservation. The latter may not necessarily be based on existing or collected records of species occurrence.

6.3.3.1 Species occurrence based methods

When considering conservation programmes for a woodland habitat, the occurrence of existing species (e.g. atlas records, or collected specifically for the purpose), may be used to prioritise conservation or restoration action across a network of sites. When potential sites are known and mapped this may be a valuable method. Methods used may select a range of species such that a network contains a "representative" selection of species typical of the habitat type. Species may be selected by some form of threat or "priority" assessment where the occurrence of species identified as being rare or priority for conservation action are selected above other species. Planning utilising "priority" or "representative" species acknowledge the goal as conservation of these species, previously identified as being priority (e.g. species recognised as under threat or in decline, classified as BAP "priority" species). These approaches assume selected species hold higher importance and are valid targets for direct conservation action. While conservation directed at these species may be considered to maintain the conservation interest of the habitat there is no stated aim that such conservation action will ultimately conserve the wider associated biodiversity, or maintain its long-term integrity. At a simple level species may be used to plan and track the effects of forest management, e.g. the US forest service uses lists of "management indicator species" in each national forest to assess habitat quality and species trends enabling conservation planning (Zavala and Oria, 1995). At more complex levels increased use of spatial aspects of species life history and needs are included.

Alternatively a range of surrogate and indicator species based approaches may be taken. Peterken proposed a method of using woodland ground-flora indicators of ancient woodland status / longevity to prioritise the importance of woodland sites for conservation (Peterken, 1974). Other methods include the use of flagship, umbrella, focal species and GAP analysis to promote woodland conservation action (Section 3.3.2). These methods are frequently used with existing records to select priority conservation areas, although other information such as environmental variables can also be used. These methods are often applied at large spatial scales and either examine the overlap in species occurrence to define biodiversity "hotspots" for conservation action, or examine current reserve networks against species or hotpot distribution and suggest gaps in such conservation networks that could be filled. Species-based strategies utilising gap analogs and similar strategies have been less frequently applied to wooded landscapes at local / regional scales, as such analysis are typically carried out at larger national and international scales and involving multiple habitat types e.g. the GAP analysis program in America. Researchers using GIS to examine the barred owl as a woodland umbrella species in America found varying success depending on definitions in identifying bird habitats from modelled habitat data (Rubino and Hess, 2003). When tested the model based on barred owl occurrence held land that contained from 0% to 75% of other species groups species, so showed variable accuracy as umbrella species. UK studies have also found poor association between taxa (Prendergast et al., 1993, Prendergast and Eversham, 1997).

6.3.3.2 Species ecology based methods

Species ecology based approaches represent a shift away from structural consideration of landscape form in earlier woodland landscape ecology and landscape assessment studies, or simple examination of species occurrence data to a functional ecology, functional connectivity approach where consideration is given to how species utilise landscape structure and experience connectivity (With et al., 1997, Wiens and Milne, 1989). UK examples of this approach include the study by Ratcliffe (1998), examining a range of species in a broad "ecosystem approach" to identify spatial woodland conservation considerations. Early UK FHN studies noted that in attempting to examine species that may benefit from a FHN, that surrogates could be used to assess biodiversity such as deadwood, but also noted that practically it was difficult not to consider individual species in order to assess possible fragmentation effects (Ratcliffe et al., 1998). Ratcliffe's approach selected a range of species with which to assess fragmentation by examining their area requirements, dispersal ability and potential to benefit from increased connectivity, which were then used to design network requirements (Ratcliffe et al., 1998). In an ambitious program Ratcliffe initially based design on requirements of both current and extinct priority species to allow for potential re-introductions. Species chosen included those that were threatened, rare, endemic or keystone species using local expert knowledge and literature. However species generally fell into two types, wide-ranging species with narrow niches, and species with poor powers of dispersal and narrow niches. Ultimately a list of local priority species developed all of which were assessed for area and isolation limits using literature and expert knowledge. The clear advantage of this approach is its ability to address conservation of potential benefit to species by considering landscape design, but not being limited to existing records of species occurrence. Similarly in lowland England Peterken noted strategies may be implemented by consideration of species dispersal characteristics, and identified three types of thresholds that may apply: a minimum area of individual woods, separation distances between woods and overall woodland density (Peterken, 2000b). The evolution of the FHN concept shows a move away from broad assessments or examination of landscape thresholds such as the 30% rule in earlier studies e.g. (Hampson and Peterken, 1998, Peterken, 2000b, Peterken et al., 1995, Peterken, 2002b, Peterken, 2002a) to an examination of areas utilising a functional network approach incorporating species functional ecology rather than an approach based on structural connectivity of the landscape. These have involved the use of focal species methods (Lambeck, 1997, Brooker, 2002, Watson et al., 2001), whereby species profiles are chosen that represent different dispersal and area requirements (Ray et al., 2004b, Ray et al., 2003b). Rather than targeting species of conservation concern, different classes of focal species are chosen, based on their ecological characteristics within the local landscape, and categorised by "threat" factors to their persistence. The focal species approach (Lambeck, 1997) is an extension of the umbrella species concept and aims that through addressing the conservation of particular focal species the conservation of many other associated species will also be fostered. Through knowledge of focal species ecology landscape design guidelines are able to be created, and the method may be applied to landscapes in the absence of species records, due to linkage between species ecology and landscape form, structure or dynamics. Lambeck suggested grouping species by categories of "threat" to their persistence then utilising the most sensitive species within each category (e.g. area, isolation) to define minimum acceptable levels of that threat that should be allowed to be present in the landscape (Fig 6.1).

Early FHN studies (Cairngorms Partnership, 1999, Ratcliffe et al., 1998, Hampson and Peterken, 1998) broadly fit with the focal species theory in their intent, although the methodology is not cited. In its original form Lambeck considered the focal species approach to be a realistic application for conservation, in that although ideally based on full species survey assessment and knowledge in reality it would be applied through expert knowledge, existing data and assessed through ongoing monitoring (Lambeck, 2002). However detailed methods have been examined in order to select focal species. Brooker (2002) used a modelling approach based on woodland birds in Australia to investigate landscape presence effects similar to the studies of Hinsley and Bellamy (Hinsley et al., 1994, Bellamy et al., 1996a). Focal species were identified for relevant threats and landscape conditions and "focal communities" chosen (Brooker, 2002). Other application of the concept also exist (Watson et al., 2001), while similar approaches have been applied, merging the original definition and function of large bodied umbrella species with Lambeck's focal species approach (Sanderson et al., 2002, Coppolillo et al., 2004) (Fig 6.2), or using species "eco-profiles" to define appropriate landscape forms (van Rooij et al., 2004).

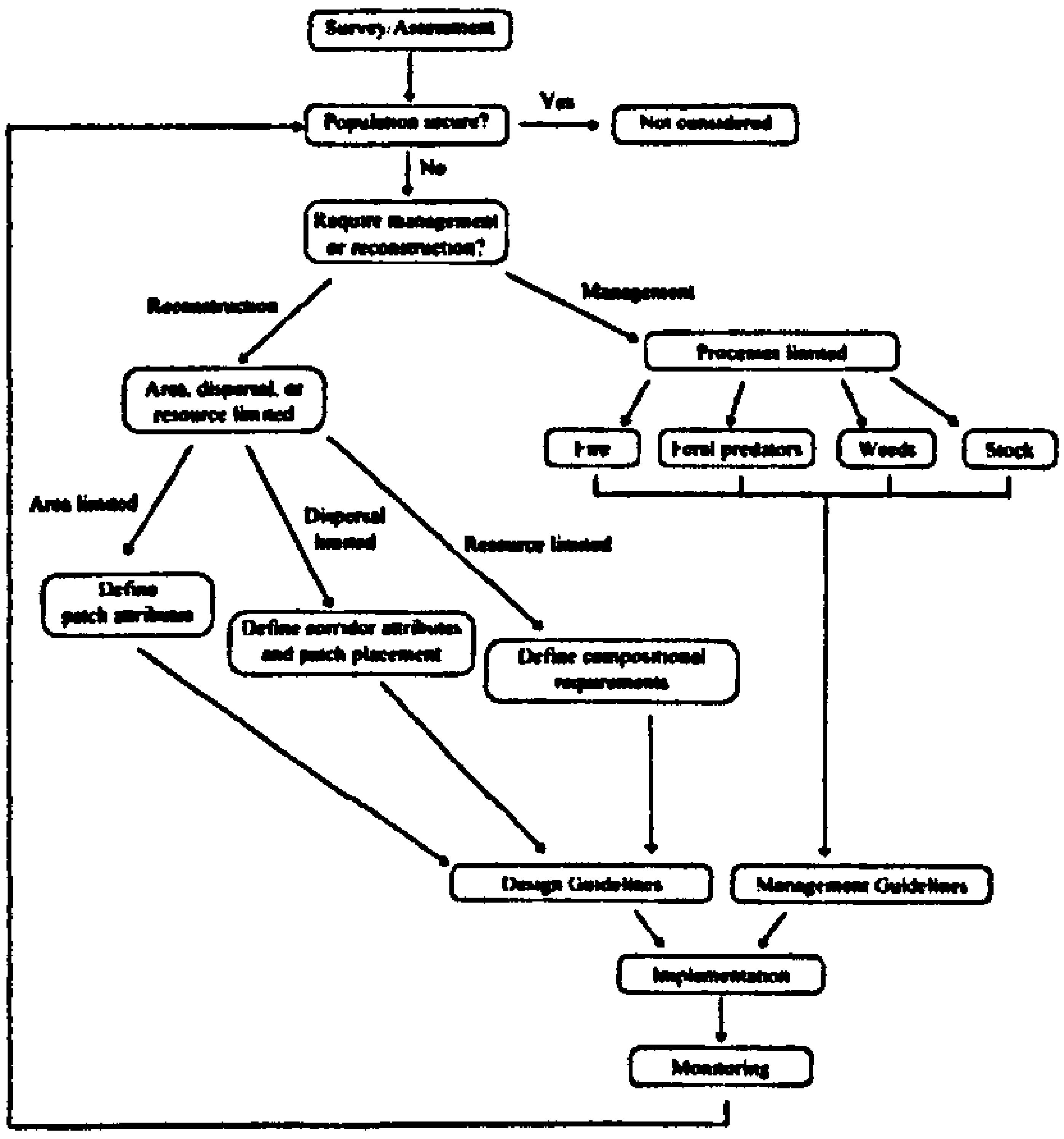


Figure 6.1
Procedure for identifying focal species. Reproduced from (Lambeck, 2002)

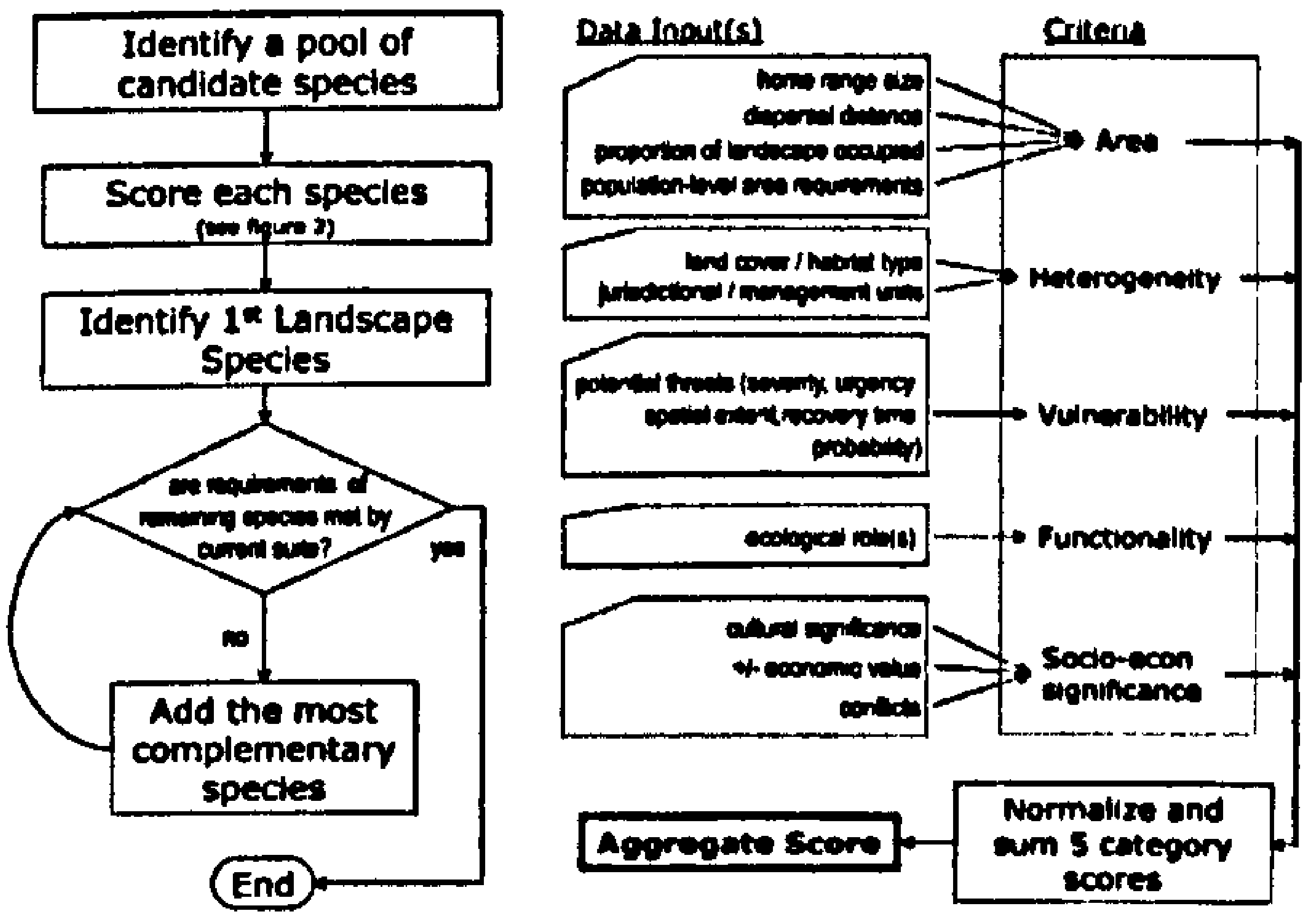


Figure 6.2
Methods to select and score focal "landscape species". Reproduced from (Coppolillo et al., 2004).

The application of focal species has largely been restricted to birds within woods in agricultural matrixes. This reflects the knowledge of bird species biology, their high conservation priority and ease of distinguishing woodland patches within a matrix. A study examining species other than birds was conducted in Cheshire using "eco-profiles", similar to focal species, where "each species represents a number of species with similar spatial characteristics and habitat requirements" (van Rooij et al., 2004). Although eco-profiles were used these were interpreted through named species, such as dormouse, without identification as to whether the example species used were the species most threatened by particular features, as with focal species, or were example species in need of conservation, or were flagship species (van Rooij et al., 2004). The model developed examined landscape structure in relation to "eco-profiles", each chosen to represent the requirements of similar species, such that landscape planning could be undertaken.

Recently focal species have been extended to a strategic level, arguably allowing a more effective treatment of all species groups, by the use of "generic focal species" (Watts et al., 2005, Ray et al., 2004a, Ray et al., 2003b, Ray et al., 2004b, Latham et al., 2004). While the original focal species approach aimed to use species grouped by threat to identify the most detrimental characteristics of the landscape, in reality many species requirements will be unknown. It may therefore be useful to justify conservation strategies by referring to broad species area and dispersal activity categories and thus define "generic focal species". These authors utilise this method to identify areas of the landscape that may be functionally connected for different generic focal species. The authors note the methods' potential to examine the functional range of biodiversity that could be maintained in a landscape (Ray et al., 2004b). In these studies a network is defined as a landscape structure through which focal species can disperse freely through numerous habitat patches (Ray et al., 2003b, Ray et al., 2004a). In southern Scotland generic focal species were based on the minimum area required to maintain a viable population and the maximum recorded dispersal distance (Ray et al., 2004b). In Wales generic focal species were developed based on area and dispersal values defined in consultation with habitat specialists, defining landscape permeability values and dispersal distances in relation to 50 year timescale events (Latham et al., 2004). Application of weighted cost buffers around habitat patches to the maximum focal species dispersal distance allowed functionally connected patches to be identified, where the area of these clusters was above the minimum requirement of a generic focal species, then a viable population was assumed to occur. However the method remains based on assessment of area and structural landscape features, where in reality habitat quality factors would apply (Ray et al., 2004a). Although the method aimed to examine and map core areas for woodland biodiversity, it also enabled an assessment of possible detrimental effects of woodland expansion by investigating open-ground generic focal species (Ray et al., 2004a, Ray et al., 2004b).

Table 6.1
Example classification of focal species. Source (Ray et al., 2004a). Woodland specialist - (restricted to broadleaved habitat, affected by edges, limited travel through other wood types). Woodland generalist - (utilises all woodland habitat, not affected by edges, travels through all woodland types)

	Dispersal ability	
Area requirements	Low – 1km	High – 5km
High – 10ha	Woodland specialist Woodland generalist	Woodland specialist Woodland generalist
Low – 2ha	Woodland specialist Woodland generalist	Woodland specialist Woodland generalist

6.4 Plan / strategy implementation

The variety of methods applied to study methodologies addressing woodland conservation at the landscape-scale are also reflected in a variety of methods implementing recommendations for conservation action. Although individual studies or examples may overlap, these will be discussed within three categories: production of landscape design rules and guidance to be applied to conserve wooded landscapes, mapping of broad conservation zones and prioritization of land parcels / sites to conservation activity.

6.4.1 Landscape design rules / guidance and landscape ecology guidance

Several studies have used different methodologies to produce "landscape design guidance" for woodland conservation, by stating a range of actions or relative priorities for woodland conservation, but not actually mapping or scoring / prioritising sites or areas. Typically these guidelines relate to woodland extent / proportion of cover, woodland linkage, minimum woodland sizes and minimum isolation levels (Table 6.3) (Kirby and Reid, 1997, Ratcliffe et al., 1998, Peterken, 2002b).

Landscape ecology studies in the UK produced guidance based on preferred levels of woodland cover in a region and highlighted the benefit of core areas and linkage between woodland sites / areas (Hampson and Peterken, 1998, Smithers, 2000). Within the Woodland Trust's approach the principal strategy was to increase cumulative core area, by selective woodland buffering or expansion, noting that this would improve the resilience of species to change (habitat sustainability) but also acknowledging this did not directly promote woodland linkage (Smithers, 2000, The Woodland Trust, 2002). Scottish proposals for "core forest areas" and "major landscape links" (Hampson and Peterken, 1998) were echoed by proposals in England for "Forest districts", defined as areas where woodland cover could be enhanced to achieve "connected" woodland areas of over 5,000 ha covering at least 30% of an area, and "woodland districts", areas where at least 750 ha of connected woodland cover at 30% cover could occur among a larger area of up to 2,500 ha (Kirby and Reid, 1997). The main recommendations of the national FHN strategy within Scotland were to: retain existing woods, expand woods, develop existing clusters of woods and develop connections between linear clusters of woods (Hampson and Peterken, 1998). Principal "landscape links" suggested were the riparian network, valley sides and the treeline zone, of which the riparian network was noted as holding considerable potential to enable long-distance links from lowland floodplain forests to the semi-natural habitats of the uplands (Hampson and Peterken, 1998). Other opportunities for linkage sites were noted as the conversion of existing plantations, where they occurred on steep valley sides (Hampson and Peterken, 1998). In lowland England Peterken summarised FHN priorities as: retain ancient woods, create large woods, create well-wooded districts, locate new woods adjacent to existing woodland and improve the matrix by restoring semi-woodland habitats between woods (Peterken, 2000b). The Clyde Valley FHN identified priority design rules to: consolidate the main wooded river network, link minor tributaries to the main river network, extend main river network to headwaters, extend main river network to plateau farmland, re-design upland plantation forests, develop wooded habitats in urban open spaces (Peterken, 1999).

A further range of studies noted that while broad conservation guidelines can be valuable, that the suggested priorities and design guidance was landscape context dependent, and thus priority actions would differ between regions and areas. The guidance produced by regional studies may thus not be able to be generalised to other areas. Kirby and Reid (1997) categorised Natural Areas by woodland creation or conservation potential and suggested priorities should be related to the cover and quality of remaining woodland. Landscapes rich in woodland cover, where woods were densely spaced or clustered, would already hold high connectivity levels and management should focus on ensuring such woods were in favourable condition and on expanding woodland size rather than creating new woods or linking sites (Kirby and Reid, 1997). Additionally Peterken noted the difference between the flora in ancient woodland sites and secondary woodland sites was less in lowland English districts with higher overall covers of woodland, where woods were less isolated, indicating isolation was not hampering secondary wood colonisation to the extent that it does in more fragmented landscapes (Peterken, 2000b). Kirby considered patch linkage, for example with corridors, should not be a high priority in areas with high covers and diversity of semi-natural habitats, unless local reduction in edge-effects was also an aim (Kirby, 1995). Within landscapes where woodland patches were isolated it was suggested that new woods were added to woods that already exist to link up woods (Kirby in Dover, 1994, Kirby and Reid, 1997). In such regions where linkage is planned the authors suggested a strategy adopts the "stepping-stone" approach rather than using long thin corridors (Kirby, 1995, Kirby and Reid, 1997). Utilising a range of approaches and strategy guidance BAP targets have been produced for Natural Areas and Counties (English Nature, 1998, Kirby and Reid, 1997). In Wales woodland focused planning units, Ecological Woodland Units (EWU), have been identified from woodland landscape analysis. These small-scale networks, similar to LDU landscape character planning zones (Swanwick and Land Use Consultants, 2002), are intended to be used to prioritise conservation work based upon an examination of woodland structural states (natural woodland, high-forest, coppice, wood-pasture) resulting in specific conservation recommendations for each EWU (Latham, 2003). Similar detailed woodland management and conservation guidance have been produced for individual "Landscape Types" following work in the North York Moors (Peterken, 2002a). These methodologies recognise that priority action can differ between areas depending on the current landscape context and woodland character, guidelines for conservation being able to be tailored to increasingly local scales.

While one method of tailoring design advice to more local areas was to analyse existing woodland form and character, in local areas other approaches have used knowledge on various forms of target species, known to occur, (or aiming to be re-introduced), such that design guidance can be based, not only on existing landscape structure, but also on species functional ecology. Early FHN studies used an "ecosystem approach", selecting a range of species (initially both current and extinct priority species to allow for potential re-introductions) with which to assess fragmentation, and examining their area requirements, dispersal ability and potential to benefit from increased connectivity. These were then used to design requirements for the network (Ratcliffe et al., 1998). These requirements were then utilised in a relatively simplistic way by comparing modelled species requirements (core area, isolation distances, density occurrences) against an analysis of current woodland cover and configuration to provide broad indications of potential species occurrence / population sizes, and in order to make recommendations for different landscapes areas and woodland types. Priority areas were not scored, mapped or prioritised. The end results of the strategy were thus broad guidelines that may be applied to the landscape that were believed to aid species conservation. Ultimately these were based upon, red squirrel, capercaillie, pinewood plants, pinewood invertebrates and crested tit. Recommendations were discussed based upon individual species / groups, under individual habitats. Using such methodologies, guidance such as creation of core areas, and reduction of isolation or creation or large woodland patches can be quantified. Studies have noted species or landscape-specific values for woodland design. In the Cairngorms broad targets for several priority species were initially developed, although the report noted more work on implementation was needed to identify requirements of key species (Ratcliffe et al., 1998), and then expanded into a multi-species strategy (Cairngorms Partnership, 1999). Guidelines considered woods less than 5ha of limited value, patches generally being expected to be increased to be more than 5ha or 25ha, and for some areas to be above 500ha (Cairngorms Partnership, 1999, Ratcliffe et al., 1998). New woodland patches were required to be separated by less than 1km, based partly upon likely dispersal distances from natural regeneration of 500m (noted as a compromise between 100m recommended by FC and potential for pine to colonise up to 1-2km) (Cairngorms Partnership, 1999).

In discussing the creation of FHN in lowland England Peterken referred to his studies of species richness in ancient woodland flora (Peterken, 1974, Peterken and Francis, 1999, Peterken and Game, 1984), describing patch area and isolation thresholds (Peterken, 2000b). Peterken noted patch thresholds in ancient woodland sites, at 3ha sites include some areas of open ground habitats while above 25 or 30 ha sites include a wider diversity of structural growth stages, noting that these features were critical for woodland flora diversity (Peterken, 2000b). Peterken also noted, assuming edge-effects of 50m, that patches would have to be a minimum of 0.8ha (assuming a circular shape) to avoid edges-effects (Peterken, 2002b). Peterken also noted that separation distances of 50m between woods would significantly impair plant colonisation while distances above 200m may effectively isolate woods from colonisation sources (Peterken, 2002b). However while acknowledging such effects on colonisation Peterken considered plant species loss from isolated woods, following the theory of "relaxation" not to be a problem due to the ability of woodland flora to persist in small patches (Peterken, 2002b). Therefore isolated ancient woods will tend to remain examples of ancient woodland flora and thus be potential sources of colonists, while isolated secondary woods will tend to remains species-poor, due to lack of colonisation.

Species-focussed design methods were expanded and clarified by Lambeck who considered focal species in relation to different categories of threat, applying landscape creation guidelines such as increasing patches to a minimum size and ensuring patches are within minimum dispersal distances (Lambeck, 1997). Example applications include studies of woodland birds in Australia, where from analysis of current landscape condition and structure and the needs of the focal community, recommendations were made for local woodland management and design (Brooker, 2002). The final strategy involved several products: designed corridors to enhance connectivity, plan areas most suitable for new woodland creation focussed on increasing existing patch size or adding sites between existing patches (Brooker, 2002). The authors avoided determining how much extra habitat to add due to difficulties of actively linking this to population persistence values (Brooker, 2002).

Additional conservation guidance advice and landscape design guidance have resulted from a range of broader ecological works addressing landscape ecological issues. Dolman and Fuller reviewed species colonisation in wooded landscapes and noted several strategy recommendations. Where the aim was to "restore biological communities, including specialist woodland species" the authors recommended emphasis be placed on establishing large core areas, reducing edge-effects and highlighted the value that spatial and structural heterogeneity within and between landscapes would hold to resultant biodiversity (Dolman and Fuller, 2003). The use of habitat corridors to aid dispersal through landscapes or into new woodland sites was considered to be of doubtful value for many species (Dolman and Fuller, 2003). The authors also noted that where conservation strategies aimed to "maximize ecological richness at a regional scale", suitable strategies would include the creation of separate woodland blocks across a diverse range of soil and environmental conditions (Dolman and Fuller, 2003). Another study reviewing topics at a conference on the restoration of woodland landscapes identified four broad priorities "to maximize future gains for woodland biodiversity". These were, in decreasing importance: to restore planted stands, expand existing woods to buffer core woodland areas, integrate adjacent "naturalised" plantations with native woodlands and to link existing woodlands by creation of wildlife corridors (Humphrey, 2003). Following a review of

European studies on woodland flora Wulf recommended: maintenance of current forest area, especially ancient forest sites, maintaining current habitat quality, that afforestation, wherever feasible, should be adjacent to ancient woodland sites and, in order to maintain genetic diversity in woods, afforestation should occur across a range of soil conditions and not just be restricted to marginal soils, and addition of forest herbs to new woodland should be investigated (Wulf, 2003). In another review of woodland species, examining woodland birds Fuller et al (1995) noted guidelines for the creation of woods in lowland landscapes to aid species diversity and persistence. These noted that while woodland proximity would be important for some species, especially sedentary birds, it was less important than for less mobile species groups than birds (Fuller et al., 1995). The recommendations were; plant a mixture of small and large woods ideally with some larger than 5ha and not less than 2ha, avoid creating isolated woods, especially small <2ha woods, create clusters of woods where possible, include a variety of tree and shrub species, including native species, include shrub/under woods species, use natural regeneration to assist planting, and create and maintain areas of scrub, especially at woods edges (Fuller et al., 1995).

Bailey and Pryor reviewed the relevance of landscape ecology theory and studies to prioritising woodland conservation within England and noted a variety of important factors ranging from the poor dispersal powers of ancient woodland flora to the potential effects of future climate change (Bailey and Pryor, 2004). They concluded that conservation activity should be focused by the varying levels of threat towards different woodland habitats and suggested the following priorities: protect threatened high-value ancient and semi-natural woodland, restore PAWS, improve the condition and quality of ASNW, improve other secondary semi-natural woodland and finally to create secondary woodland (Bailey and Pryor, 2004). Pryor indicated how different emphasis should apply to strategies where conservation aimed to increase overall woodland biodiversity or for the conservation of ancient woodland communities (Pryor, 2003). Woodland creation sites would aid the former, but to achieve the latter the focus should be on enhancement or restoration of ancient woodland sites (Pryor, 2003). In discussing the conservation of English Ancient Woodlands Thomas et al (1997) noted the first priority should be the maintenance and enhancement of all remaining large sites that are all or part semi-natural, due to their scarcity (Thomas et al., 1997). Peterken (2000) recommended the expansion of existing woods. Dolman and Fuller (2003) noted the "importance of retaining existing habitats and refugia as a source of colonists into the future. In this respect the importance of small and isolated remnant patches of ancient woodland should not be underestimated".

6.4.2 Conservation zone mapping

A frequent outcome of landscape-scale woodland conservation planning has been the production of mapping, often resulting from GIS analysis, outlining woodland conservation zones and areas of broad priority. These methodologies therefore move beyond the production of landscape conservation guidelines to provide visual and map identification of target zones. The relative importance or priority of areas within these targets zones may not be quantified however. Studies may indicate that within mapped priority zones that local economic or bio-geographic factors will influence the actual uptake of policy (Hampson and Peterken, 1998), or that further work to implement the strategy is then required by targeted delivery mechanisms such as agri-environment schemes or forestry schemes (Cairngorms Partnership, 1999).

Zone mapping has been carried out by several studies, and typically produces maps identifying areas for woodland conservation, expansion, restoration or creation, although connectivity and potential network maps have also been produced. Once produced these maps may be utilised in various ways. Map zones may be used to visually identify and locate areas, within which work can be planned or co-ordinated. Alternatively zones may be used to carry out further analysis, for example using patch occurrence within a mapped zone to classify existing woodlands as high quality conservation sites, potential restoration sites or classifying existing land of certain land-use within zones as potential woodland creation sites. While some mapping can be carried out by the simple implementation of landscape design rules, such as woodland planting within 1km of existing woodland, most woodland zone mapping is more complex.

Mapping has been produced both by the Native Woodland model (NWM) and Ecological Site Classification to produce maps that detail the potential current distribution of particular native woodland types, allowing such ranges to be visualised and examined (Towers et al., 2001, Pyatt et al., 2001) (Fig 6.4). Additionally some habitat creation / conservation studies have integrated several conservation aims with the production of habitat "vision" maps, e.g. in the Sherwood forest initiative (Hewston and Scott, 2000), detailing aims for habitat creation and conservation targets for the Natural Area. In Scotland the NWM was used to plan conservation by examining potential current woodland distributions. The model combined soil and landcover data to describe "present site conditions" in Scotland (Hester et al., 2003). Current landcover was utilised as it was noted that past levels of woodland cover (from paleobotanical studies) are of limited use current woodland restoration, due to the many changes of conditions over time, such as climate change and soil modification through anthropogenic activity. The model output used the NVC for descriptive purposes and contained three main types of woodland cover, single types, mosaics, and interchangeable types (Hester et al., 2003) (Fig 6.4).

Within the Woodland Trust's landscape ecology approach the principal strategy was to increase "cumulative core area" (Fig 6.3) by selective woodland buffering or expansion, noting this would improve the resilience of species to change (habitat sustainability) but also acknowledging this did not directly promote woodland linkage (Smithers, 2000, The Woodland Trust, 2002). The strategy mapped national priorities for PAWS restoration and woodland creation utilising AWI data, NIWT and the Landcover Map of Great Britain (The Woodland Trust, 2002). Creation was considered most favourable in areas where current cumulative core area of semi-natural habitats was low, and focused on areas where Ancient woodland formed more than 5% of the land in 10km squares, areas which were then mapped and further analysed (Smithers, 2000). The Trust promoted PAWS restoration in areas where the % of ancient woodland sites that were semi-natural was low (Smithers, 2000). Targets were required to be met within a 50 year timescale (The Woodland Trust, 2002).

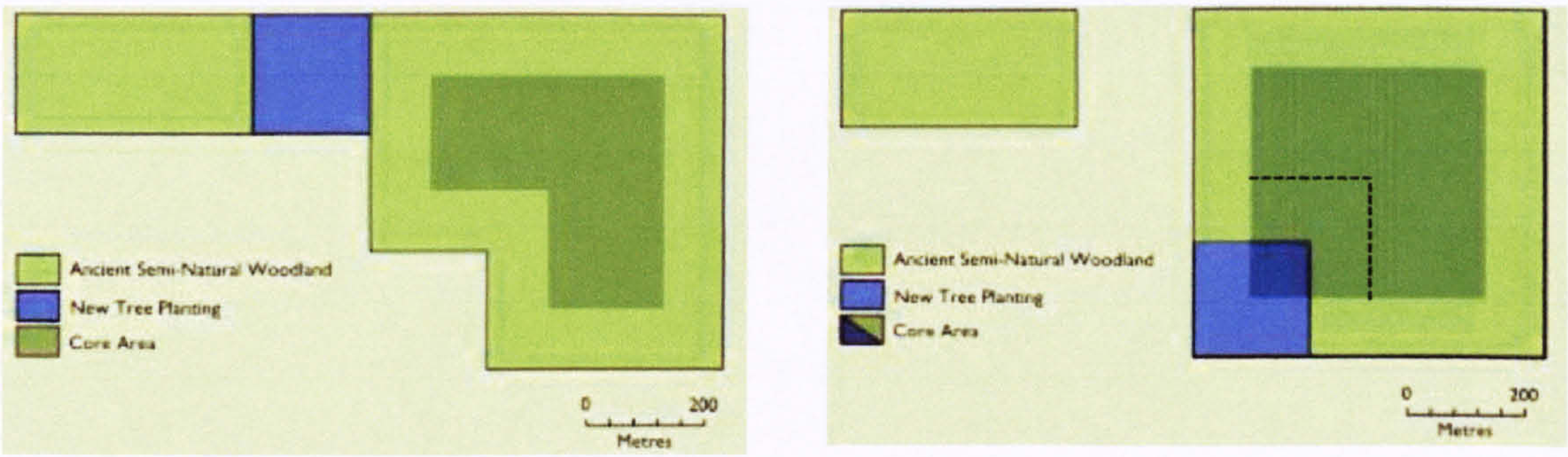


Figure 6.3
Buffering / woodland creation to increase cumulative core area. Reproduced from (The Woodland Trust, 2000b).

Research in the English uplands also resulted in identification of mapped priority zones from GIS analysis. Zones considered "potentially suitable" for native woodland were identified (Good et al., 1997). Potential woodland creation areas were identified by selecting all non-woodland habitat between 200m and 600m in elevation and excluding areas of moorland and heath above five hectares together with areas of peat soils (Good et al., 1997). The study generally found no shortage of land considered potentially suitable for native woodland expansion, although the importance of making final creation decisions on the basis of local Natural Area guidelines was highlighted (Good et al., 1997). In order to select priority areas the existing woodland network (woodland habitat + ancient woodland sites) was analysed in relation to topography. Areas were identified that held similar combinations of site characteristics (aspect, slope, elevation) to existing woods. These areas were principally slopes and valleysides, where bracken was noted as a potential pre-cursor habitat. Their case study within the Dark Peak noted most opportunity areas lay adjacent to existing deciduous or mixed woodland, noted as high potential due to being less likely to affect the core moorland character (Good et al., 1997). The authors also noted the potential for expanding woodland out from the cloughs (Good et al., 1997). Although aims included addressing woodland isolation and fragmentation, landscape ecology theory was not explicitly mentioned and key methods were derived from a replication of current site conditions, avoiding unacceptable levels of landscape change (Good et al., 1997). Similar work was also carried out by Jerram (1998) in the Forest of Bowland which further developed this methodology. Areas considered suitable for woodland creation were: open ground between existing semi-natural woods, especially when connecting separate AW sites, areas with relic ground flora, including PAWS and cleared ancient woodland sites (CAWS), areas of scattered scrub / trees, bracken stands and fish spawning rivers (Jerram, 1998). The project defined constraints as land above 400m, deep peat and blanket mire, sites of existing ecological interest (waders interest, species rich flushes, grassland and mires) and sites of archaeological or historic interest (Jerram, 1998). Additionally where survey data was available flora species were used as being indicative of past woodland cover: *Vaccinium myrtillus, Hyacinthoides non-scripta, Pteridium aquilinum, Dryopteris felix-mas, Viola canina, Digitalis purpurea, Luzula sylvatica, Geranium robertianum, Hedera helix, Ranunculus ficaria, Dryopteris dilitata, Primula vulgaris, Silene dioica, Anemone nemorosa, Teucrium scorodonia, Oxalis acetosella, Galium odoratum* (Jerram, 1998).

The results of focal and eco-profile species analysis have also been mapped in order to identify priority areas for action. The eco-profile method was based on consideration of networks, assuming different species were subject to different effects of key area requirements and sensitivity to barriers. Notably the method considered that the presence of a larger "key area" patch in a local network reduced the overall size required by the species to form a functional network compared to a network area lacking a large key patch. By applying species requirements to the landscape, possible viable populations could be identified and areas where habitat creation or enhancement could benefit populations can be visually identified (van Rooij et al., 2004). In generic focal species planning functionally connected networks can be identified for different calculated for different generic focal species. These authors utilise this method to visually identify areas that may be functionally connected, for different generic focal species, and note the methods potential to examine the functional range of biodiversity that can be maintained in a landscape (Ray et al., 2004b). Such mapping can be utilised to examine which groups of species in a landscape will experience woodland as functionally connected. The aim of these studies was to "identify key areas for native woodland restoration and expansion in order to functionally link .... core woodland habitat for focal species" (Ray et al., 2004a). Results were examined in relation to visually identified "core ancient woodland areas", where local concentrations of ancient woodland were high. The method was used to visually select priority areas for woodland conservation or expansion. In one study these areas were restricted to within 300m or 500m of existing high quality woodland sites (ancient woodland and semi-natural broadleaved woodland) (Ray et al., 2003b). In West Lothian priority areas were considered around clusters of existing ancient woodland sites, where the mapping of functional networks was used to recommend the location of corridor or buffer strips of woodland of at least 150m in width to link woodland areas (Ray et al., 2004a). These FHN studies and other examples from the literature tend to result in either broad mapped areas, within which work is suggested to occur, or broad management guidelines. The exact priority areas or sites within these zones are not scored or prioritised. This can be seen as an advantage in allowing the goals to be achieved under several different combination of landownership uptake of grants, or can be seen as a disadvantage as meaning a lack of focus in potentially approaching landholdings on which to achieve the goals.

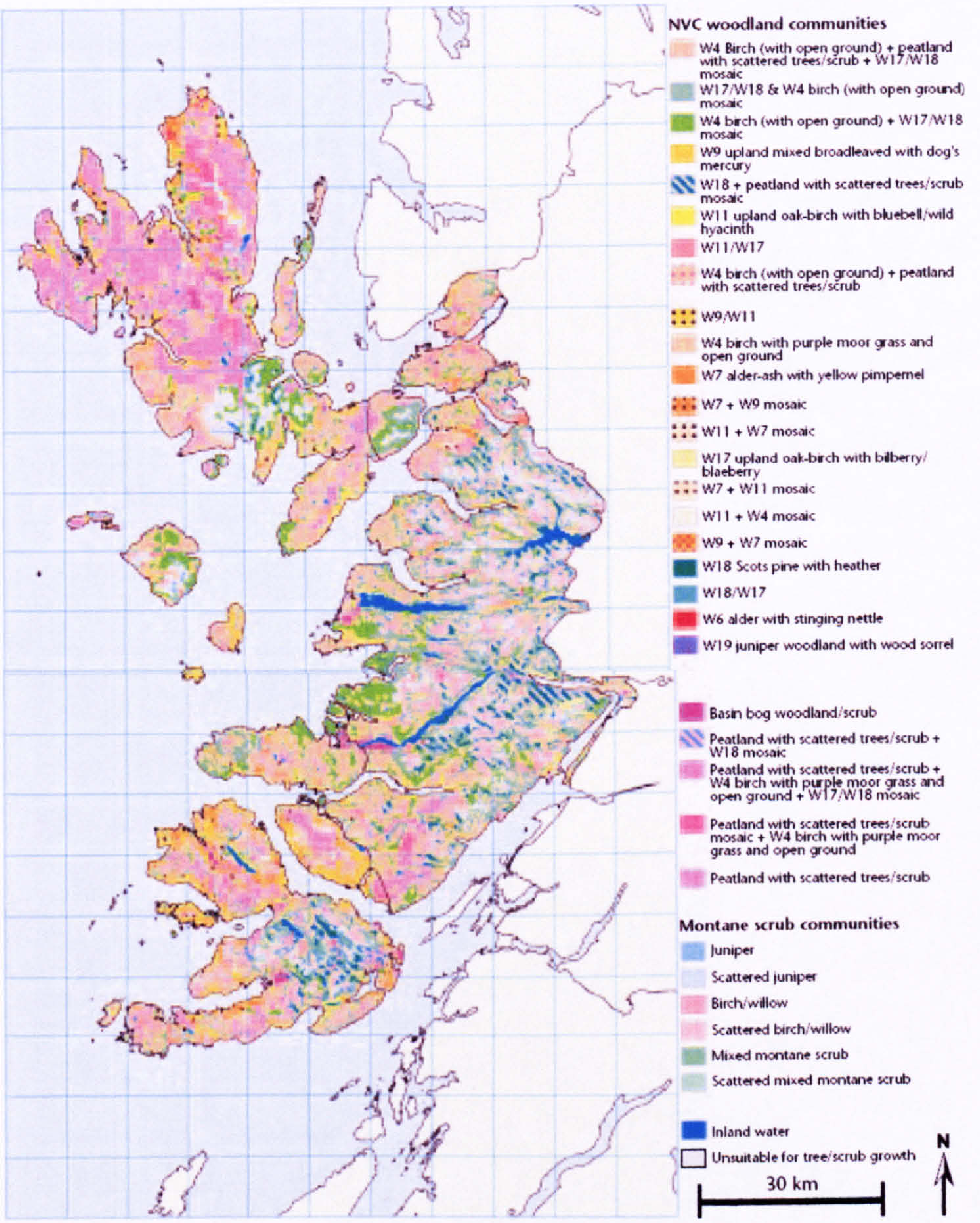


Figure 6.4
Output from predictions of the Native Woodland model (NWM) for woodland and scrub suitability on the Isle of Skye, Scotland. Reproduced from (Hester et al., 2003).

6.4.3 Land allocation targeting / optimization

Several conservation methodologies, from design guidelines, to environmental mapping and focal species result in lists of sites or land parcels needing to be prioritised for conservation action. Where targets for land uptake need to be met this problem is an example of the area-covering or set-covering reserve selection dilemmas (Margules, 2005, Baskent and Keles, 2005, Margules and Pressey, 2000). In this sense reserves are taken as areas of land addressing conservation in the sense of Pressey et al (1993) and not implying formal nature reserve designation. A large literature exists on reserve selection dilemmas, studies examining the various issues with regard to the SLOSS debate (Ovaskainen, 2002, Simberloff and Abele, 1982, Gilpin and Diamond, 1980, Diamond, 1975, Higgs and Usher, 1980), but many concentrate on various issues of methodology development and how to prioritise sites or land parcels when varying levels of biological or environmental information exist (Gaston and Rodrigues, 2003, Rodrigues and Gaston, 2002b, Schwartz, 1999, Possingham et al., 2000, Pressey et al., 1996, Arponen et al., 2005, Williams and Araujo, 2002, Pressey, 2004). The ultimate goal of such research is to identify the most optimal and efficient network or selection of sites that fulfils the conservation problem with minimum cost, in terms of either area of land, number of sites or economically (Siitonen et al., 2003, Pressey et al., 1996). Additionally research has examined approaches where the number of sites or areas to be included are set and selection must choose the best sites to fulfil these targets. The methods to prioritise sites may differ. Ultimately the methods of allocating areas of land are achieved by: ranking or prioritising sites by importance criteria, running selective algorithms to choose optimal solutions, or by utilising heuristics. Many examples exist in the conservation literature, including woodland studies, and at a variety of scales (Bayliss et al., 2003, Palik et al., 2000, Russell et al., 1997, Possingham et al., 2000, Siitonen et al., 2003, Arponen et al., 2005). Recent reserve planning research has incorporated examination of additional spatial and connectivity factors (Williams et al., 2005, Briers, 2002). Studies most applicable to the current project study area, carried out at similar scales, with similar data sources, or aimed at Biodiversity Action Plan implementation are discussed.

Some types of spatial woodland planning have used mathematical optimization techniques to find the most appropriate solution to particular spatial planning problems or queries (Baskent and Keles, 2005, Kurttila, 2001). These include linear programming, simulation techniques, meta-heuristic techniques and integer-programming, methods may be selected that are efficient in both cost and computing time and return one or several feasible options to a planning problem (Baskent and Keles, 2005). These techniques are essentially similar to the range of mathematical techniques used in the nature reserve selection literature. While such spatial modelling may be complex it may be essentially multi-disciplinary, incorporating many factors such as timber quality and economics in addition to ecology. Therefore the level of detail and precision given to the landscape ecology planning element in such complex models may not be high (Baskent and Keles, 2005). In forestry planning the additional inclusion of spatial considerations can increase the size of the optimization problem (Kurttila, 2001). The principal techniques used are integer-programming and heuristics (Kurttila, 2001). Integer and mixed integer programming result in a global optimum solution without splitting decision variables, while heuristics may be preferred when the number of variables and constraints increase to the extent that integer programming become too slow (Kurttila, 2001). Heuristics may provide several possible solutions but may not provide the optimum (Kurttila, 2001)

UK conservation prioritisation studies exist from a variety of habitats including grasslands (Bayliss et al., 2003, Thompson et al., 1999c, Lee et al., 2001b) and woodland habitats (Thompson et al., 2001b, Lee and Thompson, 2005, Gkaraveli et al., 2004, Purdy and Ferris, 1999). Strategies have been based on consideration of broad woodland abiotic characteristics across the landscape (Griffiths et al., 2004b, Lee et al., 2001b), the prioritisation of ancient woodland sites (Purdy and Ferris, 1999) or of particular woodland BAP habitat within areas (Gray and Stone, 2003).

Following consideration of the potential for establishing Forest Habitat Networks (FHN) (Hampson and Peterken, 1998, Peterken et al., 1995) the Forestry Commission undertook a study to examine how available GIS data may be used to prioritise areas for forest habitat network creation and to enable monitoring of BAP and favourable condition targets (Purdy and Ferris, 1999). This used GIS to prioritise areas for woodland creation or restoration. For woodland creation a raster analysis scoring system was developed that promoted creation of woodland close to existing woodland and scored higher for areas close to existing native or high quality woodland. Woodland quality assessment was based on interpretation of woodland category: broadleaved, conifers, mixed or by designation (e.g. SSSI). The raster analysis scoring system allowed pixel size and search radius to be varied and meant that scores were assigned to defined areas of land rather than whole land-use patches or areas. Woodland creation desirability comprised a standard score (0-100) calculated from the combined scores of each woodland polygon present in the search radius, where each wood was assessed and given a standardised score based on type, area and isolation distance. Such a scoring system allowed the rapid assessment of areas of land and could be flexibly altered where new information became available. This scoring system was then utilised to map local BAP targets by assigning planting areas to the highest priority grid squares in order of score until all desired planting had been allotted. This method although efficient and practical was limited by the way in which scores were standardised within each search radius. This resulted in a masking of the relative difference between pixel and search radius areas that would have affected the accuracy of the planting schemes. Additionally the area and distance figures utilised were arbitrary and not based on any real or focal species or literature evidence, just based on the assumption that woods created closer to larger woods were preferable and would reduce fragmentation. Purdy and Ferris noted the potential of their methodology to be applied at both broad strategic levels and again at finer local scale using different pixel and radius sizes (Purdy and Ferris, 1999). A similar methodology addressed PAWS woodland restoration, by prioritising sites close to existing ASNW, based upon existing patch habitat, designation, size and potential restored size.

A similar method to allocate BAP targets to individual PAWS site was then undertaken. This research is notable in illustrating how woodland conservation and creation targets can be allocated to parcels of land based on GIS scoring. However the method was severely limited by its lack of justification for the scoring rules used and the way in which standardisation of scores may have affected the values applied to different combination of area and isolation scores for different woods. The scoring for the study was also limited by a lack of understanding of how it would react to different combinations of woodland characteristics, where for example different combinations of large or distant woods may create similar scores. Purdy and Ferris did note the benefits that applying real species requirements to the landscape would have in allowing an assessment of the changes proposed by their scoring system allowing the impact of different conservation site choices to be assessed.

Another method to score woodland sites for conservation has been applied to ancient woodlands in the Chilterns by Thompson et al (2001). The study used a ranking system of physical patch characteristics within a GIS to prioritise woodland conservation and enhancement (Thompson et al., 2001b). Patch area, shape, mean nearest-neighbour and surrounding land-use were ranked for each ancient woodland site, assuming larger sites, compact shapes and sites with more surrounding favourable habitat were higher priority. GIS patch variables were compared against different species lists for 48 sites in order to assess whether such characteristics affected patch species quality. Positive correlations occurred between alpha diversity and patch area (0.32 to 0.48) and patch shape (0.27 to 0.46), but no correlations occurred to patch isolation values (Thompson et al., 2001b). When relationships were examined using multiple regression between 15-22% of the variation in species richness were accounted for by the patch variables. To prioritise sites woodland patches were ranked, depending on the amount of potential expansion land around sites, patch size and by examining the amount of woodland perimeter shared with land classed into different classes of restoration potential suitability (Thompson et al., 2001b). The authors noted the potential use of this approach allowing the targeting of conservation and enhancement work, when detailed site based information was not available. Although noting the potential of such abiotic surrogates to prioritise sites the study was severely limited by several broad generalisations. A relatively low number of woods were examined within small study areas and woods were assumed to hold homogenous habitat, additional aspects of site quality and management were not investigated, while the species lists used to attempt to validate the broad assumptions of abiotic patch quality were based on combined species records from numerous sources and dates which may be highly unreliable, and may not represent the current species occurring within sites.

Another prioritisation study on the Isle of Mull, Scotland developed a GIS assessment strategy for a single BAP habitat. The strategy used woodland cover data, woodland designation categories and modelled potential Upland Ashwood distribution to prioritise conservation, restoration and creation (Gray and Stone, 2003). The GIS strategy involved use of the Native Woodland Model (Towers et al., 2001) to map areas of land considered likely to support Upland Ashwoods. Sites were defined as high quality upland ashwoods where ancient or other semi-natural woodland occurred within the predicted ashwoods zone. High priority restoration sites were identified where PAWS sites occurred within the potential ashwoods zone and within 100m of the previously identified high quality ashwood sites. Lower priority restoration / conversion sites were then identified as PAWS and other plantation woods still occurring within the predicted ashwoods zone, but beyond 100m from existing high quality ashwood sites (Gray and Stone, 2003). Expansion sites, comprising open ground or plantation habitats, were also classified within two importance categories within the predicted ashwoods zones where they occurred within or beyond 100m of existing high quality sites (Gray and Stone, 2003). The authors note the potential of this approach in providing rapid prioritisation for conservation action, which in the future, with increased availability of data, may be able to be applied across Scotland for other BAP habitats (Gray and Stone, 2003). Such a strategy would allow local BAP conservation area targets to be applied to high priority sites.

A study within Snowdonia examined a potential woodland conservation strategy across the National Park (Gkaraveli et al., 2004). The authors identified priority areas for native woodland expansion, following policy aims and expert advice. Landcover map data (20 m pixel raster data) were classed as suitable or unsuitable for woodland conversion. Additional data included sites of designations and NIWT data. Local climate data models were developed using data from local observation stations. Scoring gave higher priority to sites close to semi-natural ancient woodlands (ASNW) and to creation sites near existing woodlands. Woodland of importance were identified as the woodland SSSI's. Scoring criteria were combined in a weighted linear combination method in IDRISI GIS to identify priority areas (Gkaraveli et al., 2004). The restoration of PAWS were assessed by scoring systems that examined their characteristics at each site, such as distance to existing ASNW (Gkaraveli et al., 2004). Cleared ancient sites were also examined and prioritised for conversion (Gkaraveli et al., 2004). Restoration potential was further assessed by examining the current topographic and climate distribution of woodland cover and developing a map where such conditions occurred. This map was then used to further prioritise PAWS sites in these areas (Gkaraveli et al., 2004). This study is of interest in providing a practical example of GIS assisted landscape planning. Site designations and current broad woodland or scrub habitat type were essentially used as indicators of ecological quality, for the development of the scoring system. The scoring however was based on broad and unjustified assumptions, not related to any specific, hypothetic species or species group, indeed target organisms were not mentioned, the strategy being developed to follow policy rather than species conservation. As such the distances used <300 from ASNW sites, 300-900 and >900 were not justified or related to species, while the use of current climatic and topographic distribution of ASNW in choosing future priority sites may not indicate potential high quality sites more than indicating those areas that were not cleared or converted and so only a potential portion of the potential niche of this woodland type may have been selected. Also in Wales Griffiths et al (2004) developed a prioritisation / targeting strategy that examined land parcels and scored sites for woodland creation using site area, distance to existing woodland and the amounts of semi-natural habitat surrounding the potential create site (Griffiths et al., 2004b). These were then able to be used to produce maps and lists of priority creation sites.

6.5 Discussion: woodland conservation planning at the landscape-scale: progress, limitations and opportunities

The use of ecological spatial landscape planning is now embedded in the woodland conservation activities of organisations such as the Forestry Commission (Ferris et al., 2000, Watts et al., 2005), Woodland Trust (Smithers, 2000), English Nature (Kirby, 1995, Kirby and Reid, 1997, Good et al., 1997, Buckley and Fraser, 1998), Scottish Natural Heritage (Peterken et al., 1995) and the Countryside Council for Wales (Good et al., 2000, Latham et al., 2004). Woodland conservation may be driven by a diverse range of factors, the desire being to increase general woodland cover, to maintain the integrity and interest of ancient woodland sites or to promote the conservation of particular woodland BAP habitats. A variety of strategies and methodologies exist that have been tested or developed to meet these aims. These have been based on factors ranging from the use of general landscape ecology principles and the use of abiotic factors to infer site quality, while other strategies have attempted to design woodland landscapes around the requirements of species known to use the current wooded landscape but under conservation threat. Further work avoided the ambiguity of the need to design landscapes for particular selected species by examining the conservation potential for hypothetical "generic" focal species. The choice of conservation strategy developed will reflect the local agenda and conservation drivers and may be driven by the organisational structure of the body aiming to implement the strategy.

This review has identified a number of categories of methodology and means of implementation of woodland landscape-scale planning, with an emphasis on biodiversity conservation. These categories are similar to the range identified by Kurttila (2001) in his review of forestry spatial planning. Kurttila recognised three main types; (1) adjacency constraints, where clear cuts and harvesting limits are set by sizes of distances between patches or in relation to common boundaries, this can result in efficient planning of harvesting, (2) landscape level approaches where landscape metrics and analysis of factors such as core area and connectivity are used to plan alternative harvesting strategies, non-spatial biodiversity indexes such as deadwood volumes and structure may be incorporated in such assessments and planning, (3) species-specific where information and studies may be used to parametize models to co-ordinate forestry spatial planning, e.g. in relation to patch size and isolation to protect or promote key species (Kurttila, 2001). Additionally the methodologies have much in common with the conservation biology minimum area planning and set covering reserve selection methodologies. With the uptake of spatial woodland planning in the academic literature and in conservation practise much variation exists in the methodologies applied, and the direction in which the methodologies are progressing, these are discussed below.

6.5.1 Strategy methodologies: successes, limitations and research priorities

Landscape ecology Several studies have applied insight from broad landscape ecology theory. These studies utilise results from the landscape ecology literature and allow rapid assessment and insight into conservation priorities and goals in a region or landscape. Studies can be undertaken with minimal data collection, perhaps utilising existing datasets and can be rapidly planned. In this respect a lack of consideration of exacting habitat or species requirements can be seen as beneficial, allowing rapid strategy development without becoming bogged down in detail. Such strategies therefore hold important positions in forming policy and advising on aspects such as agri-environment schemes development and scoring. However the lack of explicit consideration of habitat and species issues limits the fine scale application of these strategies in local landscapes. Issues of scale, the calculation of factors such as core area, which may be species and context dependent, limit the precision of this methodology. A further area which has recently come under much criticism is the focus of these methods on structural connectivity – where thresholds, linkage and isolation may be pre-occupied with structural linkage between areas of habitat, when recent work has promoted the benefits of functional examination of connectivity, where higher importance is given to species-specific experience of connectivity and higher value is given to the consideration of matrix effects (Brooker, 2002, Cerdeira et al., 2005, Watson et al., 2001, With et al., 1997). Key future developments in the application of these methodologies are likely to be a move away from prescriptive consideration of generalised rules to more flexible landscape ecology goals, such as increased connectivity and diversity at a range of scales, assessment of multiple associations between species and patches at multi-scales and less focus on structural connectivity, how such aims can be expressed clearly without recourse to broad generalisations or to uptake of habitat or species based methodologies is unclear however. Ultimately use of these strategies should follow a clear assessment of existing landscape form to ascertain how these landscape ecology insights should be applied.

Landscape assessment The consideration of existing landscape form and composition allows strategies to be more reliably focussed on landscapes at a local or regional scale. While early approaches simply examined the potential for woodland conservation within mapped landscapes (e.g. Natural Zones) (Kirby and Reid, 1997), recent approaches have included detailed assessment of structure at a finer scale (Latham, 2003). Such strategies allow assessment of the potential within discrete local areas to be assessed and compared. Existing woodland context within the landscape matrix can be examined in detail, and conservation targets be applied to the most appropriate areas. These methods allow incorporation of detail, for example where local topography and landscape structure means limitations will apply in the application of percolation theory to connectivity and isolation. Such approaches also have the benefit of being able to incorporate additional factors such as visual landscape assessment, such that woodland expansion can be maintained at levels that do not adversely alter landscape visual character. However these methodologies may focus too much on where woodland can be accommodated and not detail the exact ecological value such woodland conservation and expansion may have, or which species it will benefit. Such approaches may therefore benefit broad woodland biodiversity, but where particular key species or habitats are known to be at risk, or are considered a priority, they are unlikely to be fully addressed. Therefore such methods, and their growing use in conservation "opportunity" mapping (Good et al., 1997, Jerram, 1998) may efficiently detail where woodland conservation (especially creation) can occur, but they may not detail why certain areas are more valuable than others. These methods do not adequately address habitat quality, typically concentrating on structural occurrence rather than functional or qualitative approaches. However, these do allow advice to be tailored to defined local areas, and can allow consideration of the levels of functional connectivity that already exists in a landscape on which to build. Key developments in these methods would be an increased application of analyses at increasingly local scales, down to ecological neighbourhoods, while methods to apply multi-scale assessment could also prove beneficial in allowing overlap and integration with species and focal species based methods (Vos et al., 2001, Lambeck, 1997). The incorporation of woodland site structural states, as initially suggested in early FHN studies (Hampson and Peterken, 1998, Ratcliffe et al., 1998, Peterken, 2002a) and also proposed by Latham (2003) would also enhance and increase the detail of these assessments.

Environment / Habitat Several methodologies have been applied that examine the woodland habitat being addressed by conservation, either through analysis of its rarity and quality or through detailed assessment of environmental requirements, and community niche space. The approach allows methods to move beyond a current analysis of landscape structure, detailing where woodland could occur (based on current habitat distributions), to a more objective approach presenting potential future distributions, defining hypothetical extents and allowing these to be compared to current landscape forms. The relative objectivity of these methods is an advantage. However limitations occur in that while selected environmental variables may have been key drivers of woodland occurrence in history their application can be limited in a modern landscape, where soils or habitat have been very extensively modified. Additionally these studies may be severely limited by scale and by availability of data. Where only coarse data are available the insights may only be applicable to broad areas and regions and not appropriate for fine-scale planning. While some of the limitations of this approach are normally overcome by linking such methodologies with a landscape assessment approach key opportunities in the development of these methodologies would beneficially be the incorporation of factors driving not just woodland habitat occurrence, but also relative diversity and habitat "quality". In some studies such aspects can be inferred from areas where mapping indicates that several woodland types (e.g. NVC) may occur, but such outputs are rarely expressed in diversity or structural terms. Further insights can be gained from analysis of associations between woodland species diversity and woodland structural or environmental factors in the woodland ecology literature (Chapter 5). This method also has the potential advantage over broader landscape ecology and landscape assessment methods in that when habitats are modelled, for example woodland NVC communities the modelling can be indicative of elements of woodland rarity and possibly quality, over that possible through broad assessment of general woodland cover.

Species methods A number of species-based spatial woodland conservation strategies have been implemented, ranging from presence based methods to those involving detailed assessment and modelling of real or hypothetical species requirements. Such methods provide some level of clarity for managers over the specific objectives and aims of woodland conservation and restoration activities, beyond the perhaps more nebulous activities of broader biodiversity conservation per se. Species approaches allow conservation to be directly targeted to a species under current threat. The limitations of these methods are that due to the large number of woodland species the choice of species chosen as the target or "focal" species for conservation may be open to criticism, and could be largely arbitrary. Indeed much discussion and debate has occurred over these points resulting in research to examine the efficiency of different species based selection methods (Lambeck, 2002, Caro, 2002, Caro and O'Doherty, 1999, Lambeck, 1997, Simberloff, 1998, Lindenmayer et al., 2002). Potential resolutions have been suggested as species that cover large areas (umbrella species), ensuring species are selected for different threats (Lambeck, 1997), or that multiple species of "focal communities" are chosen (Brooker, 2002). However the association between these surrogate based conservation methods and achieving representative biodiversity conservation is far from established (Lindenmayer et al., 2002, Prendergast et al., 1993, Caro and O'Doherty, 1999). An additional factor is that where species currently occurring in the landscape are chosen to foster conservation of a woodland habitat this will ignore very rare or extinct species because they will either be so rare they cannot be related to any form of woodland structure, or they will be absent. Therefore authors have noted that even extinct species should be included in species methods so that landscapes can be designed with a view to potential future re-introduction (Ratcliffe et al., 1998). Additionally such methods, being reliant on species occurrence data either require a large investment in collection of such data, or rely on expert knowledge of species requirement, each of which have limitations. These limitations led in part to the suggestions for generic focal species application, where generic focal species (or eco-profiles) are applied to landscapes and hypothetical examples of species life-history traits (van Rooij et al., 2004, Ray et al., 2004a). Future development of these species methods could focus on increased investigation of the choice of target focal species, focal community or guilds for different habitats. Additional methods of focal species could be development to include analysis with different structural woodland states and to incorporate associations with levels of woodland diversity, habitat quality and general species richness at different, multiple, scales. A critical factor in the use of these methodologies is that future studies clearly state the rationale and approach in species and surrogate selection. The consideration of species as focal, flagship or umbrella species is critical in considering the relative success of schemes when outcomes are monitored.

Strategy methodologies Pure landscape ecology approaches have proved useful for advancing consideration of spatial issues, and will continue to provide input to development of agri-environment schemes and grants. However due to the complexities of landscape form, differences in species composition between areas, and varying strategy priorities, the use of additional methodologies is required. The methodologies of landscape assessment studies thus provide the opportunity to fully address the current habitat resource and examine the landscape matrix, either at a range of scales or in a hierarchical manor, addressing driving landscape forces on the woodland habitats in question. Such approaches therefore allowing targeting at finer scales. Woodland environment / habitat modelling has widespread application in spatial planning but methods must be clearly related to spatial scales, data availability, and be linked with existing assessment of current land-use patterns. The relative value of landscape assessment, woodland "opportunity" mapping and methods derived from woodland habitat / environment / community mapping can depend heavily on data availability. At fine scales when detailed soils mapping is available the results of modelling such as ESC and NWM can be preferable to existing landscape assessment methods of current habitats. However the reverse may be true when detailed land-use data is available but soils data are coarse or lacking. Species methods suffer from difficulties in collation of the data required to parametize, models or lack of expert knowledge. Suitable species may be difficult to justify and methods of choice of focal species, communities and guilds need to be advanced. The use of focal species as a rapid method involving reduced need for species data merits further research and investigation in relation to expansion to include measures of broad biodiversity / species richness and focal species requirements for different woodland structural states. Authors have acknowledged however the benefits of applying a mixed methodology approach to applied conservation planning combing the benefits of different strategies and risk spreading of any potential limitations (Lindenmayer et al., 2002). A further range of important points arise in relation to conservation strategy aims. Different strategies may give varying importance to woodland conservation, restoration or creation. For example strategies also aiming to address existing woodland conservation should ideally incorporate existing woodland management in their assessment due to its important influence on habitat quality, whilst strategies addressing restoration must consider a range of factors in relation to existing woodland quality and restoration potential that purely creation based strategies need not.

6.5.2 Strategy implementation: successes, limitations and research priorities

Woodland landscape design guidance A range of studies have produced landscape design guidance promoting biodiversity in the target landscape. Guidance has been applied at different scales and with widely differing levels of detail. At national or large regional scales design guidance from landscape ecology theory and broad analysis of landscape structure may foster enhancement in biodiversity through increasing woodland connectivity, but are unlikely to allow fine scale targeting for species or specific woodland habitats. At local scales however guidance can more accurately reflect local priorities by examining detailed woodland form and structure or by incorporating assessments based on design guidelines for local priority, target or focal species (Ratcliffe et al., 1998, Latham, 2003, Brooker, 2002). Such landscape design planning may therefore be most affective at small catchment or local ecological neighbourhood scale, although broader approaches applied in the North York Moors showed how wide ranging assessment of opportunities and constraints can result in detailed text based guidance across landscape zones across many square kilometres (50 – 200km^2) (Peterken, 2002a).

When landscape design relates to current landscape features it may be mapped into zones, to which the guidance advice applies. However generally guidance is not prescriptive, and is open to later analysis and interpretation "on the ground". It may be advantageous to allow individual owners to take up different options for woodland conservation, creation / expansion without prioritising areas of land, allowing greater flexibility. However this may still risk some of the problems of random uptake of schemes and result in a lack of spatial targeting. E.g. more prescriptive and definitive allocation of options to set areas of land may be desired.

However even without exact prioritisation of land and mapping of conservation or expansion areas, design guidance can focus conservation action. Proposals for FHN areas, core forest areas and large landscape links can focus efforts within broad geographic areas where finer scale work can identify actual areas of land on which to realise the strategy. Additionally a frequent outcome of such strategies are design guides in terms of minimum isolation and patch size for new woodland creation. Such values are then easily used in agri-environment schemes and woodland grant scoring. Additionally lists of regional or local priority actions and guidance for woodland conservation can be used by local conservation staff to give background to other land-use decisions within scheme applications.

A criticism of broad design and guidelines approaches is that, in attempting to generalise or summarise action across a landscape, compromises will inevitably be made. The minimum patch size and isolation values will favour some species more than others. Compromises may be made over colonisation distances from those initially suggested from guidance, e.g.(Cairngorms Partnership, 1999), while other studies have noted guidance may be altered and compromised where initial design guidance is considered too strict to be likely to be implemented by landowners (Watson et al., 2001). Ultimately design advice given should be applicable to the landscape in question. Therefore such guidance will have most value where the existing landscape has been analysed to ascertain if guidance, such as enhanced connectivity and progression towards cover thresholds, is applicable. Additionally guidance will be affected by scale, both of data available, and in any consideration of potential benefits. Ideally guidance and design should apply at multiple scales to ensure coverage of multiple species. Such future enhancement to the methodology could also include clearer consideration of habitat and focal species requirements, perhaps resulting in guidance divided among outcome categories such as potential benefits for broad biodiversity, target habitat or species.

Woodland conservation zone mapping Mapping of zones classified as priority for conservation activity, perhaps additionally classified separately for conservation, restoration and creation has the advantage that such areas can be examined and quantified, landowners can be contacted and measures initiated, following which design considerations of individual sites and management issues can be considered. The mapping can be the focus on which action is built. However such mapping can be very generalised, may only be applicable at certain scales, and therefore only indicative of broad strategic priority areas. Mapping of potential native woodland distribution for example may identify large areas as being potentially suitable but beyond the levels that are practically likely to be achieved. Never-the-less this mapping is considered useful and is frequently being employed with the scope of "habitat opportunity mapping" for a number of habitats (RSPB, 2004, Saunders and Parfitt, 2005). Implementation of conservation action in such areas may benefit from further refinement and prioritisation or scoring of such priority zones. Some studies have explicitly avoided defining exactly what habitat should be restored or created, noting that defining for example the appropriate habitat amount in a landscape is difficult due to issues of attempting to define habitat and species persistence (Brooker, 2002).

Several enhancements could usefully be made to mapping strategies. Within the area of priority zone mapping opportunity exists for further integration with the results of design guidance and species-based methods. Mapping at finer scales will increase the realism and use of such mapping, while incorporation of further subdivision and mapping of priority zones for different woodland management or structural types could also increase its uptake. Additionally some of the developments of analysis in generic focal species and functional connectivity analysis could be mapped, such that potential locations of expansion or linkage areas could be identified, so as to maximise the functional connectivity of the landscape. There is also scope in examining broad "opportunity" and "constraints" areas to consider, in more detail, the relative benefit of different land-use options and how they affect resultant habitat quality in contrast to simple assumptions of habitat considered suitable or unsuitable for woodland conservation. Different pre-cursor habitat will for example affect developing woodland types and will affect the stock of habitats remaining in the landscape matrix and thus need consideration.

Scoring / prioritisation Scoring based implementation of strategies vary widely in their complexity. Strategies have employed relatively simple ranking methods e.g. Peterken's (1974) method to select priority woods for conservation based on the presence of ancient woodland indicator species, a system found successful in ranking priority sites. Strategies have examined the use of abiotic patch features to score ecological interest or potential (Lee et al., 2001b, Thompson et al., 2001b) while a range of woodland studies have applied GIS scoring approaches to define optimal selections of sites from potential sites or networks (Purdy and Ferris, 1999, Gkaraveli et al., 2004). However much research in this area is international and rarely examines woodland or single habitat issues (Gaston and Rodrigues, 2003, Rodrigues and Gaston, 2002a). These studies have the advantage of actually highlighting the sites of most importance and clearly locating them. However this requires accurate location information and the lack of such data, of sufficient resolution, can limit this approach. Although where exact site patch detail information is lacking the methodology can still be applied to raster grids (Purdy and Ferris, 1999). While these methods may allow high degrees of accuracy, if input data is of low resolution the outcome may still be very speculative. The methodology and scoring system chosen for implementation of a strategy can have effects on its accuracy, speed and optimality. In the areas of systematic conservation planning, many methods are available. Recent work considers heuristics, greedy and simulated annealing may not find optimal most efficient results to site selection and that integer programming may find optimal solutions (Crossman and Bryan, 2006). Therefore while heuristics may be more frequently used linear / integer programming are considered preferable and can now be carried out at adequate speeds (Rodrigues and Gaston, 2002a, Crossman and Bryan, 2006). Such optimization / scoring methods vary in the way they implement methods to select priority sites or areas. These may be based on patch size, isolation or on various measures of representativeness utilising species records. The relative importance given to site abiotic characteristics, spatial network values or such species records within the scoring system varies. Existing methods for considering proposed reserve networks may incorporate elements to consider connectivity, but workers have criticised the lack of consideration of connectivity in such methods and have proposed a new approach to consider reserve connectivity as part of initial model constraints (Cerdeira et al., 2005). Connectivity can be included within initial opportunity and constraints "exclusion", or can be included as part of a detailed reserve selection algorithm. Ultimately different options exist, some studies note that to speed up selection areas of land can be pre-selected and classified as unsuitable, thus speeding up selection algorithms. It would also be possible to pre-select non-connected land as unsuitable or to pre-define conservation zones within which priority sites will be scored, such that connectivity is effectively incorporated in the optimization. While examples of selection algorithms in the literature differ greatly, a number of software products have been created to perform landscape-scale reserve planning as standardised selection problems. These have resulted from both practical conservation implementation work and research programs and variously implement constraints mapping, greedy heuristics, simulated annealing and use of irreplaceability scores. These include: C-PLAN (Anon, 2001a), Sites V.1.0, a GIS adaptation of the earlier SPEXAN reserve planning software (Anon, 2001b), MARXAN (Possingham et al., 2000, Ball and Possingham, 2000) and CLUZ, an implementation of MARXAN in ArcView GIS which incorporates analysis of land parcel irreplaceability scores and introduces elements of iterative reactive conservation planning in examining alternative land-use scenarios (Smith, 2004). The use of such methods is likely to continue to increase. The value of standard products versus specialist applications depends on the conservation problem at hand, the scale of conservation and the data available. These reserve planning software are limited to standard problems and so are most suited to planning new reserve networks, of multiple habitat types in large bio-diverse areas, and are less suitable to address conservation of a single habitat at a specific local scale.

While a large range of scoring methodologies exist a more important area ecologically is the reasoning utilised to define the relative scoring and priorities given to different sites, patch or landscape features. The scoring utilised must be fully justified and related to the landscape, habitat and species in question. While broad scoring strategies such as those based upon selecting representative networks utilising existing species records acknowledge these hotspots are hoped to be representative of other unmeasured biodiversity, other strategies based upon patch abiotic values or measures of clustering, aggregation and isolation, may be less clearly justified. The effects of assigning different relative values to patch or area-based scoring and the reasoning utilised in assigning these scores should be investigated and tested as part of the scoring process. Often in UK woodland examples woodland type or designations have been utilised as representative of levels of habitat quality, although these have not always been justified or based upon literature (Purdy and Ferris, 1999, Gkaraveli et al., 2004). Broad site characteristics have been assumed rather than measured in such studies. No UK studies have justified their scoring methods based on thorough assessment of existing woodland networks and analysis of habitat–environment–structure relationship in established or ancient wood sites.

In considering future priorities for woodland conservation scoring and optimization, further information is required to prioritise within broad targets and categorises such as ASNW or PAWS sites (Bailey and Pryor, 2004). Important areas for research being: how the presence of key species within particular sites affects this prioritisation scale, and the extent to which recommended priorities between woodland groups can be changed by quality factors, for example whether some high quality secondary woods be more important for enhancement or expansion than heavily degraded ancient woodland sites (Bailey and Pryor, 2004). Algorithms can be easily created and GIS based reserve selection automated but these selection rules must be clearly justified by prior landscape or species analysis.

Summary A variety of implementation strategies have been employed in landscape-scale woodland conservation projects in the UK and similar strategies worldwide (Fig 6.5, Table 6.2). Differences between the methods used relate to the precision with which areas of priority land are identified, and therefore the scale of action which can be implemented. Benefits and limitations exist across the strategies. Simple design guidance and rules can be utilised by existing conservation organisations and multiple landowners to be applied immediately in individual land management and land-use decisions. However targeted approaches and scoring strategies create inequalities. Different values between areas of land, where particular landowners or groups of landowners must be approached to discuss management alterations, while owners in supposedly low priority areas may be less inclined to undertake positive conservation works where such areas or ownerships are apparently considered unimportant by the mapping or scoring methods. Ultimately the method employed must be justified and grounded in local and focussed ecology methods. The scale and precision of advice must be related to the strategy's end usage and potential.

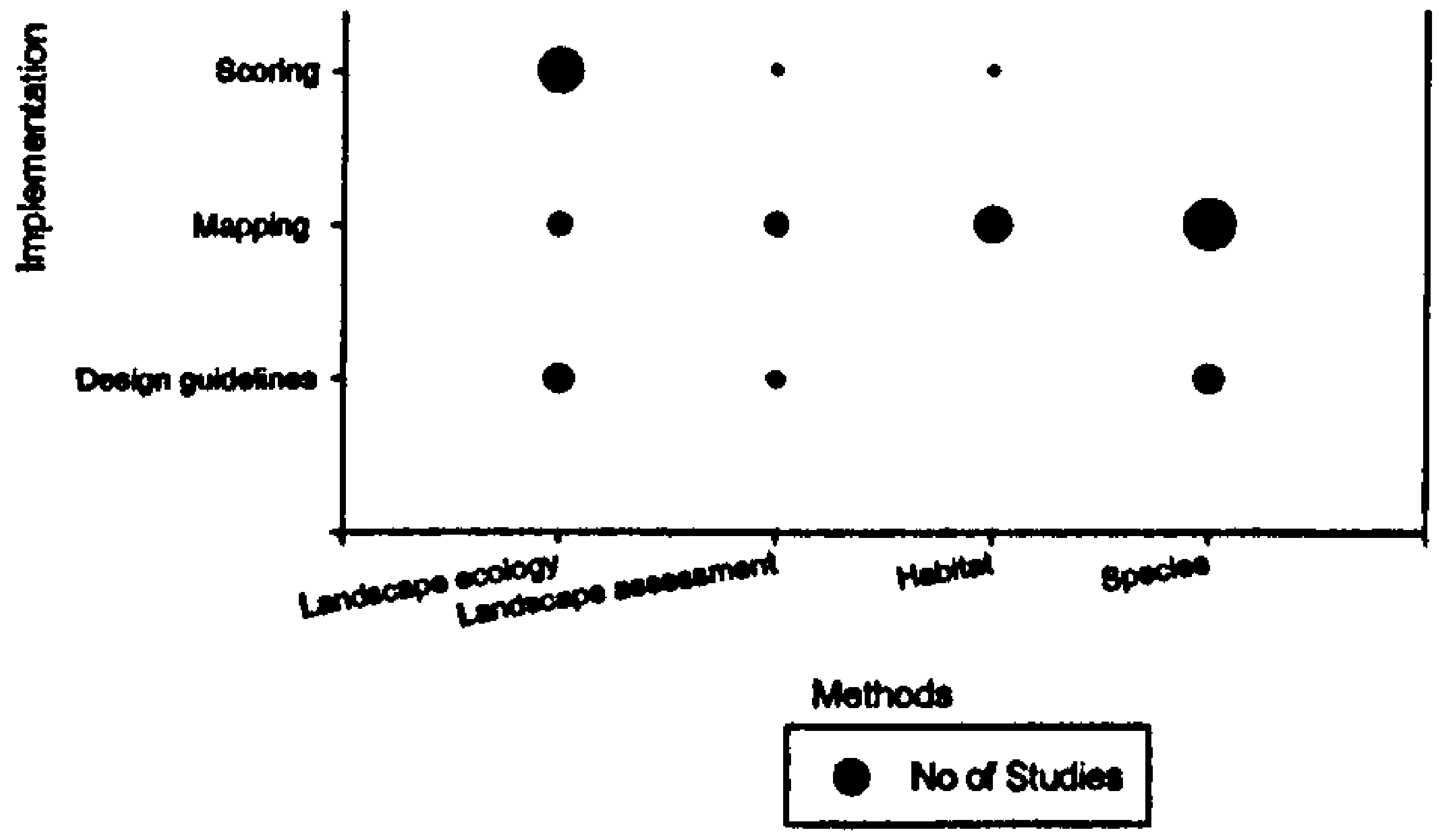


Figure 6.5
Association between study methodology and study implementation strategy of reviewed literature presented in Table 6.2.

Table 6.2
List of research and applied conservation work examining targeted woodland conservation planning at the landscape scale in the UK, and associated relevant studies from other habitats and locations. Each reference is classified by study aim, highlighting where targeting of agri-environment schemes (Agri-Env) or Biodiversity Action Plan (BAP) was a driver. The focus of the study in terms of addressing conservation (C), restoration (R), or creation (Cr) is noted. The methods and implementation strategies used are noted following the categories detailed in the test. Main = ••• Minor = •. LE = landscape ecology, LA= landscape assessment, Hab= habitat / environment, Sp= species.

Reference	Habitat	Location	Study Area (km^2)	Aims / Drivers	Focus			Methods				Implementation			Theory / Case study	Academic Professional
					Con	Res	Cr	LE	LA	Hab	Sp	Design	Map	Score		
(Bayliss et al., 2003)	Neutral grassland	England	270	BAP	•		•	•••	•		•		•	•••	Case study	Ac
(Birnie et al., 2000)	Bracken habitat	Scotland	78,772	-	-	-	-			•••		-	-	-	Case study	Ac
(Brooker, 2002)	Native vegetation	Australia	c.375	-	•		•				•••		•••	•	Case study	Ac
(Brown et al., 1998)	Wetland grassland	England	35	Agri-Env	•					•••			•••		Case study	Ac
(Coppolillo et al., 2004)	Multi-habitat, regional	S. America, Africa	-	-	-	-	-				•••	-	-	-	Case study	Ac
(Gkaraveli et al., 2001)	Woodland and scrub	Wales	2x 100	-			•	•	•••				•••		Case study	Ac
(Gkaraveli et al., 2004)	Native woodland	Wales	2142	BAP	•	•	•	•••		•			•	•••	Case study	Ac
(Gray and Stone, 2003)	Upland Ashwoods	Scotland, Mull	c.250	BAP	•	•	•			•••			•••		Case study	Ac
(Griffiths et al., 2004b)	Broadleaved woodland	Wales	2 x 100	BAP	•	•	•	•••	•••					•••	Case study	Ac
(Hampson and Peterken, 1998)	Woodland	Scotland	-	-				•••				•••			Theory	Ac
(Lambeck, 1997)	N/A	-	-	-	-	-	-				•••	•••			Theory	Ac
(Latham, 2003)	Woodland	Wales	20,779	-	•	•	•		•••			-	-	-	Case study	Ac
(Latham et al., 2004)	Woodland	Wales	20,779	BAP	•		•				•••		•••		Theory	Ac
(Lee et al., 2001b)	Chalk grassland	England	833	BAP	•		•	•••					•	•••	Case study	Ac
(Lee et al., 2002)	Native woodland	England	100	BAP			•	•••	•				•••		Case study	Ac
(Nikolakaki, 2004)	Native woodland	England	c.100	-			•				•••		•••		Case study	Ac
(Palik et al., 2000)	Multi-habitat	USA	114	-		•				•••				•••	Case study	Ac
(Peterken, 1974)																
(Peterken, 2000b)	Woodland	England	-	-		•	•	•••				•••			Theory	Ac
(Ray et al., 2004b)	Woodland	Scotland	945 + 4,750	BAP, Agri-Env	•		•				•••		•••		Case study	Ac
(Russell et al., 1997)	Riparian habitats	USA	1500	-	•	•				•••			•••	•	Case study	Ac
(Sanderson et al., 2002)	Multi-habitat	A. America, Africa	-	-	-	-	-				•••	-	-	-	Proposal / theory	Ac
(Smithers, 2000)	Native woodland	England	-	-	•	•	•	•••					•••	•	Case study	Ac
(Thompson et al., 1999c)	Chalk grassland	England	125	Agri-Env	•	•				•••			•••		Case study	Ac
(Thompson et al., 2001b)	Native woodland	England	100	BAP	•		•	•••					•	•••	Case study	Ac
(Towers et al., 2001)	Native woodland	Scotland	78,772	BAP		•	•			•••		-	-	-	Proposal / theory	Ac
(van Rooij et al., 2004)	Woodland	England, Cheshire	2,200	-	•		•				•••	•••	•••		Case study	Ac
(Watson et al., 2001)	Native woodland	Australia	2,700	-	•		•				•••	•••			Case study	Ac
(Buckley and Fraser, 1998)	Native woodland	England (lowlands)	4 x 100	-			•	•••				-	-	-	Case study	Pro
(Cairngorms Partnership, 1999)	Woodland	Scotland	6,490	BAP, Agri-Env			•	•		•••		•	•••		Report	Pro
(Good et al., 1997)	Native woodland	England (uplands)					•		•••				•••		Case study	Pro
(Jerram, 1998)	Native woodland	England	c.4,000	BAP, WGS	•		•		•••				•••		Case study	Pro
(Peterken, 1999)	Woodland	Scotland	840		•	•	•	•	•••			•••			Report	Pro
(Peterken, 2002b)	woodland	Britain	-	-	•	•	•	•••				•••			Report	Pro
(Peterken, 2002a)	Woodland	England	1,638	-	•		•	•	•••			•••			Report	Pro
(Purdy and Ferris, 1999)	Native woodland	England	5,000	BAP	•	•	•	•••					•	•••	Case study	Pro
(Ratcliffe et al., 1998)	Woodland	Scotland	6,490	-	•	•	•	•••			•••	•••	•		Case study	Pro
(Ray et al., 2003b)	Woodland	Scotland	4,677	-	•		•				•••	•	•••		Report	Pro
(Ray et al., 2004a)	Woodland	Scotland	464	-	•		•				•••	•	•••		Report	Pro
(The Woodland Trust, 2002)	Native woodland	England	130,395	BAP			•	•••					•••	•	Report	Pro

Table 6.3
Methodology issues from UK landscape-scale woodland conservation research. Target cover: All= all landscape, Loc= core areas, or locally. nw = new woods, mw = managed woods, mia = minimum intervention areas,

Reference	Current % wood Cover and target [ ]	Data res or min area	Edge distances	Area Threshold (ha)	Isolation / dispersal thresholds (m)	Matrix classified	Habitat constraints	Habitat opportunities	Topographic / altitude issues	Clusters / cores	Value / ecological quality inferred from ?	Forest Structure Considerations
(Gharaveis et al., 2001)	15%	40m raster	20, 50, 70, 100, 130, 150	-	-	Yes	-	Conifers, agricultural land	-	-	Woodland cores	-
(Gharaveli et al., 2004)	15%	20m raster	-	-	<300, 300-900, >900	Yes	Woodland, rock, open water, urban areas, important semi-natural habitats 200m buffers of important habitats	Land without nature conservation constraints	Above 650m not suitable	-	Woodland class, AW classification, designations	-
(Gray and Stone, 2003)	-	1 50,000 raster 0 1ha wood data	-	-	100	Yes	-	Rough grassland, bracken, plantations in potential zones	-	-	Modelled woodland zone co-occurrence	-
(Griffiths et al., 2004b)	-	1:10,000 10m raster	-	<2, 2-5, 5-25, 25+	<=20, <=100, <=250, >250	Yes	High quality habitat, artifical habitats	Non high quality conservation habitat	Yes by comparison to LDU zones	-	Abiotic	-
(Hampson and Peterken, 1998)	-[30%]	-	-	-	-	-	-	Existing plantations on steep slopes	Steep slopes on valleysides Riparian, key potential	Yes	-	-
(Jerram, 1998)	-	-	-	-	-	Yes	Species rich habitats. Habitats with important sp records	Open ground between semi-natural woods +AW sites, Areas with select woodland flora, Scrub / scattered trees, Bracken, Spawning rivers	Above 400m unsuitable	Yes	Abiotic, Some species records	-
(Latham, 2003)	-	-	-	-	-	Yes	-	-	Yes, considered	-	-	Yes
(Latham et al., 2004)	-	0 01 ha, vector	-	-	-	Yes	High dispersal cost habitats, highly modified, artifical habitats	Low dispersal costs, highly natural habitats	-	Yes	AW site status, connectivity,	-
(Lee et al., 2002)	-	10m raster	-	100	200 – 750	Yes	Calcareous grassland	Deciduous woodland, arable, improved and agricultural land	-	-	-	-
(Nikolakaki, 2004)												
(Peterken, 1974)												
(Peterken, 2000b)	11% [30%Loc]	-	-	3, 25, 30,	50, 100-200	-	-	-	-	Yes	Abiotic	Yes
(Ray et al., 2004b)	20%	1:10,000 10m raster	-	3, 10	2,000, 5,000, 10,000	Yes	High dispersal / resistance costs	Low dispersal / resistance cost	-	No	-	-
(Smithers, 2000)	-[Loc]	-	100	-	-	-	-	-	-	Yes AW	Abiotic, AW classification	-
(Thompson et al., 2001b)	20%	10m raster	-	-	-	Yes	Non-coniferous woodland, chalk grassland, scrub, water,	Arable, set-aside, improved grassland			Abiotic and species data	-
(Towers et al., 2001)	-	100m raster?	-	-	-	Yes	-	-	-	-	-	-
(van Rooij et al., 2004)	-	-	Barriers	varied	varied	Yes	Classified for different eco-profiles	Classified for different eco-profiles	-	key patches	Connectivity	-
(Buckley and Fraser, 1998)	-	-	25	-	-	-	-	-	-	-	-	-
(Cairngorms Partnership, 1999)	12-15% [30%Loc]	-	-	5, 25, 100, 500,	500, 1000- 2000	Yes	Some unsuitable habitat identified		Effects of altitude considered Riparian areas a priority	-	Abiotic	-
(Good et al., 1997)	Varied [No]	100m raster?	-	-	-	Yes	Peat, woodland, scrub, >5ha blocks of upland native vegetation	Various semi-natural habitats including grassland, bracken and arable land	Only between 200m and 600m Slopes, valleys, cloughs and riparian areas promoted	-	-	-
(Jerram, 1998)	-	1:50,000 1.25,000	-	-	-	Yes	Peat, blanket bog, high interest nature conservation sites	Open ground between Aw sites, select flora, scrub / scattered trees, bracken, spawning rivers	Limited at 400m	-	Some species records, connectivity to AW sites	-
(Peterken, 1999)	14%	-	50	3 (nw), 20 (mw)	200 (flora)	-	-	-	Riparian network	-	-	-
(Peterken, 2002b)	-[15% All 30% Loc]	-	-	3, 25 (mw), 50 (mia)	50, 200,	-	-	-	Riparian network noted as priority	Yes	Abiotic, AW classifications	Yes
(Peterken, 2002a)	21%[30% Loc]	-	100 (50)	<5, 5-30, 30+	100, 200	Yes	-	-	Riparian a priority	Yes	Abiotic, AW classification	Yes
(Purdy and Ferns, 1999)	-[30% Loc]	Raster and vector	-	<2, 2-5, >5	<30, 30-400, >400	Partly	SSSI designations	-	-	Yes	Abiotic, AW classification, habitat classification	Yes
(Ratcliffe et al., 1998)	12-15%[30% Loc]	2ha min	20 (flora), 100 (fauna)	5, 500	1000, 1000-2000	Yes	-	-	-	Yes	Abiotic, connectivity, deadwood, AW and naturalness classification	Yes
(Ray et al., 2003b)	19%	-	-	<10, >10	330, 500	Yes	-	-	Riparian noted as priority	Yes	Abiotic, connectivity, AW classification	-
(Ray et al., 2004a)	14%	10m raster	50	2, 10	300, 1000, 5000	Yes	-	-	-	Yes	Abiotic, connectivity, AW classification	-
(The Woodland Trust, 2002)	-[30% Loc]	1km raster, + vector	100	-	-	Yes	Current semi-natural areas	-	-	Yes	-	-

6.6 Chapter Summary

Landscape Ecology

- Landscape ecology based methods can rapidly be applied to devise broad landscape design rules and guidelines of benefit to woodland diversity
- Landscape ecology methods are limited by issues of scale, an overemphasis on structural connectivity measures and a lack of consideration of individual habitat and species requirements
- Interpretation of patch size, core areas and connectivity levels set by such methods may be highly anthropocentric and lack species realism
- The application of broad generalised landscape ecology theories is limited where consideration is not given to existing landscape structure
- Levels of existing woodland cover, aggregation and clustering can significantly influence the relevance of broad theories to woodland conservation, the importance of insights from percolation theory for example lessening when woodland aggregation levels are high
- Landscape form and structure should be assessed prior to consideration of applicable landscape ecology theories

Landscape Assessment

- Landscape assessment can be undertaken for existing woodland cover and allow suitable conservation methods to be applied
- Advice can be targeted to fine scales within landscapes, where different priorities and opportunities may be identified to those across broader landscape areas
- Assessment methods allow advice and research to be informed by such existing landscape form and structure
- Assessing opportunities and constraints to woodland conservation allows realistic constraints to visualised to strategy implementation

Environmental / Habitat Modelling

- Environmental envelopes / habitat modelling allows conservation areas to be targeted more clearly to particular woodland habitat types, rather than broad assessment of woodland creation or restoration potential by landscape assessment based methods
- Habitat modelling can incorporate potential habitat distributions and as such may be of more use in considering visionary landscape change than research limited by relation to current landscape structure
- Modelling has potential to address the core driving processes operating in relation to woodland biodiversity beyond factors simply affecting habitat distribution
- Environmental / habitat modelling is limited in application by the resolution and quality of available data, or lack of information on relationships between habitat or communities and driving environmental factors, affecting the scale of research outputs

Species based methods

- Species based methods allow the target of conservation action to be clearly defined, but choice of surrogate species must also be clearly defined and the methodology and use of the selection methods must be clearly stated
- It may be difficult to assess the success of surrogate based methods where conservation of known or hypothetical focal species is intended to benefit broad unmeasured woodland biodiversity
- Research suggests surrogate measures may have varying success in capturing associated biodiversity, depending on the scale of application and the co-occurring groups investigated
- Promising developments are the adoption of threat based focal species and focal community approaches, while generic focal species approaches warrant further investigation and research

Design guidelines and rules

- Design rules allow concise and clear expression of strategy priorities ready for implementation and uptake, and may be used by a range of organisation and individuals, not necessarily restricted to larger conservation projects
- Due to generalisations such guidance may be limited to broad biodiversity measures and limited in its potential application to broad biodiversity enhancement effects
- Design rules allow flexible uptake of schemes and priorities by landowners
- Design / guidance will be most robust and valuable when resulting from detailed assessment of current structure and focussed on key habitat requirements and insight from species or habitat planning, rather than broad landscape ecology focussed methods
- Work remains to develop guidance applicable at multiple-scales and to clearly address separate priorities for broad biodiversity conservation versus species-specific or habitat focussed benefits

Conservation zone mapping

- Mapping of zones clearly identifies areas for conservation action rather than relying on text based identification of priorities
- The lack of relative prioritisation within broad mapped zones may be considered benefit or limitation in discussions with landowners
- Lack of exact identification of blocks of land allows increased flexibility and ignoring ownership boundaries may promote broader multi-owner uptake of landscape-scale conservation
- Mapping can allow an initial quantification of the areas of land suitable or appropriate for conservation action, upon which strategy considerations and further targeting and optimization can be based

Conservation targeting / scoring / optimization

- Scoring / optimization can increase precision and clarity and identify top priority sites to pursue for conservation
- Can be perceived as a more rigorous and detailed approach due to results being quantified
- Exact identification of sites may cause issues with landowner contact and may be seen as too prescriptive and inflexible

- Wide variety of mathematical options exist to apply scoring techniques
- Resolution of data will affect the scale of output
- Different GIS based methods exist that can apply scoring based on raster GIS grid cells, effectively ignoring patch land-use boundaries on the ground, or may be based on actual patch polygon boundaries to score individual, pre-existing sites

Overall summary

- Landscape assessment based methods are particularly valuable as they allow strategies to be tied more clearly into the study landscape
- Habitat / environment modelling must be clearly related to study scale, accuracy and clearly state intended conservation as present-natural, past-natural or future-natural predictions of woodland cover
- Generic focal species and mixed methodology approaches require further consideration
- Text design guidelines are best implemented in association with mapped conservation zones and applied to sub-division of a landscape resulting from landscape assessment
- The relative value of broad conservation zone mapping and quantified prioritisation of exact areas of polygons for conservation action may be variable depending on the landscape in question and the perceived acceptance of landowners or conservation agents.
- Few studies have addressed all three BAP actions of conservation, restoration and creation
- Abiotic values (isolation and patch size) considered indicative of current or future ecological value are rarely fully substantiated within strategies
- Several strategies have highlighted the importance of topography and noted the potential importance of the riparian network in spatial woodland network strategies
- Work remains to link assessment of woodland structure, habitat quality and heterogeneity