COMMISSIONED REPORT

Commissioned Report No. 104

A review of approaches to developing Lowland Habitat Networks in Scotland

(ROAME No. F02AA102/2)

For further information on this report please contact:

Claudia Rowse
Scottish Natural Heritage
2 Anderson Place
EDINBURGH
EH6 5NP
Telephone: 0131–446 2432
E-mail: claudia.rowse@snh.gov.uk

This report should be quoted as:

Humphrey, J., Watts, K., McCracken, D., Shepherd, N., Sing, L., Poulsom, L., Ray, D. (2005). A review of approaches to developing Lowland Habitat Networks in Scotland. Scottish Natural Heritage Commissioned Report No. 104 (ROAME No. F02AA102/2).

COMMISSIONED REPORT

Summary

A review of approaches to developing Lowland Habitat Networks in Scotland

Commissioned Report No. 104 (ROAME No. F02AA102/2)
Contractor: Forest Research, Scottish Agricultural College, Forestry Commission Scotland
Year of publication: 2005

Background

Habitat fragmentation, coupled with habitat loss and degradation has had a detrimental impact on the biodiversity of lowland agricultural landscapes in Scotland, especially over the last 50–60 years. Site-protection measures alone are insufficient to conserve biodiversity and a wider landscape scale approach is needed which fosters connectivity between habitats through the development of ecological networks.

Current ecological theory and approaches to landscape evaluation for biodiversity are reviewed and tools offered for developing habitat networks in Scottish lowland agricultural landscapes, focusing at the sub-catchment scale (~200km^2).

Main findings

- Two contrasting ecologically-based approaches to landscape evaluation can be identified: the first focuses on landscape structure (eg metrics); the second on landscape processes (eg species dispersal and habitat usage).
- Focal species modelling integrates both structural and species-based approaches and is recommended as a practical, ecologically robust method for constructing and evaluating habitat networks. Sources of data on lowland habitats and species are identified and examples given of ecological profiles for focal species.
- Landscape Character Assessment (LCA) is recommended as a tool for assessing the aesthetic and visual impacts of network development but needs to be combined with a Historic Land use Assessment.
- Computer-based visualisation packages are available to help with communicating LCAs.
- Work is required to integrate the LCA boundaries with Natural Heritage Future boundaries (the latter provide the ecological framework for networks).
- A GIS-based approach to assessing recreational impacts on habitat networks is proposed involving analysis of buffer distances around access and recreational facilities.
- Network construction and analysis tools which combine ecological, cultural and aesthetic attributes can be developed but need to be tested within case-study areas.
- A number of factors may constrain the practical implementation of habitat networks such as climate change, cultural resistance and the economic uncertainties facing the agricultural sector.

For further information on this project contact:
Claudia Rowse, Scottish Natural Heritage, 2 Anderson Place, Edinburgh EH6 5NP. Tel: 0131-446 2432

For further information on the SNH Research & Technical Support Programme contact:
The Advisory Services Co-ordination Group, Scottish Natural Heritage, 2 Anderson Place, Edinburgh EH6 5NP.
Tel: 0131–446 2400 or **ascg@snh.gov.uk**

Acknowledgements

The authors are grateful to Colin Stewart, Daniel Gotts, Claudia Rowse and John Hood for providing information and commenting on earlier versions of the report, and to Kirsten Hutchison for help with library and literature searches. Lachlan Renwick provided information on the Scottish Paths Record and Angus MacDonald helped with discussions on delineating lowland areas. Nigel Buchan provided an update on progress with testing landscape character assessments. Chris Quine, Peter Freer-Smith and Richard Brand-Hardy also made helpful comments on the report.

Extended Summary

Habitat fragmentation, coupled with habitat loss and degradation has had a detrimental impact on the biodiversity of lowland agricultural landscapes in Scotland, especially over the last 50–60 years.

There is recognition that site-protection measures alone are insufficient to conserve biodiversity and a wider landscape scale approach is proposed which fosters connectivity between habitats through the development of ecological networks.

The theory and development of Forest Habitat Networks (FHNs) is well advanced in Scotland. As government policy is now encouraging better integration of forestry, agriculture and other land-uses, there is a need to assess the applicability of the network concept to agriculturally-dominated landscapes and explore ways of generating a positive interplay between Agricultural Habitat Networks (AHNs) and FHNs.

This report reviews current ecological theory and approaches to landscape evaluation for biodiversity and offers tools for developing habitat networks in Scottish lowland agricultural landscapes. The intention is to provide support for the strategic planning of biodiversity enhancement in combination with other landuse objectives such as landscape design, recreation and access.

The Scottish lowlands are defined with reference to SNH's Natural Heritage Futures (NHFS). Seven areas of predominantly lowland character are identified using climate, geology and land-use criteria (Figure 6).

A variety of spatial scales are relevant to the development of networks ranging from country/regional scales through landscape/catchment scales to individual farms. End-user needs/questions, ecological requirements of key species, and data availability combine to determine the scale of interest. Sub-catchment scale (~200km^2) is suggested as a practical starting point for developing the network approach.

Two contrasting ecologically-based approaches to landscape evaluation can be identified from the literature and case-studies. The first focuses on assessing landscape structure, and includes the use of landscape metrics/indices; defining threshold values for certain structural elements, or spatial targeting of structural change either within a network concept (ie emphasising physical connections between elements) or by prioritising changes in area or quality of individual elements.

The drawback of purely structure-based approaches is that they take no account of landscape function (eg dispersal processes) or the needs of specific species. Species differ in their spatial preferences and movement capacity and require an array of different types of network.

Species-based approaches to landscape evaluation can be classed into empirically-based habitat suitability modelling; metapopulation modelling; focal species modelling and spatially-explicit population modelling. All have their advantages and disadvantages. Focal species modelling sits mid-way on the modelling continuum between simple structure-based models and detailed single species-based models and is recommended as the most practical approach for constructing multi-species based networks.

The GIS-based modelling tool BEETLE (Biological and Ecological Evaluation Tools for Landscape Ecology) integrates land cover and focal species (either real or generic) data to evaluate habitat suitability and functional connectivity within landscapes at a variety of spatial scales. Connectivity is modelled on the

dispersal ability of a focal species and the ease of movement through the landscape surrounding suitable habitat patches.

In lowland agricultural landscapes, remnant semi-natural habitats such as woodland, scrub, hedgerows and unimproved grassland provide the permanent framework for network development, but need to be integrated with short-term habitats such as arable crops, field margins and set-aside.

Species of agricultural landscapes vary greatly in their habitat preferences and sensitivity to fragmentation. Some are highly mobile; others are very restricted in their movements. Some are habitat generalists; others are specialists, or require specific habitats at specific times of the year or during their lifecycle. A range of typical focal species are identified for potential use in network modelling.

The availability of autecological information is limited for some of these focal species but general assumptions can be used for certain parameters based on expert opinion. Examples are given of ecological profiles for a woodland species and an open ground species. Generic species profiles can be constructed where data is lacking for specific species.

The BEETLE model is recommended as a tool for constructing lowland habitat networks based on focal species analysis. Sixteen spatial datasets are identified as potential sources of land cover data (Table 7). Data on individual fields is held within the Scottish Integrated Agricultural Control System (SIACS), but the value of this system as a source of habitat information needs to be tested.

Landscape Character Assessment (LCA) is recommended as a tool for assessing the aesthetic and visual impacts of network development, but needs to be combined with a Historic Land-use Assessment to give an improved understanding of the historic element of landscape character and hence an assessment of cultural attributes.

Computer-based visualisation packages are available to help with communicating the visual effects of landscape change to communities and stakeholders and should form part of the LCA tool.

Unfortunately the boundaries of the Scottish LCAs (Figure 17) do not coincide with the NHF boundaries which are used as the basis of the ecological analysis. This makes integration difficult and it is recommended that SNH try and combine the two approaches.

The construction of habitat networks could have an impact on access and recreational use of the landscape through the risk of increased disturbance to wildlife. Rules relating to buffer distances around recreation facilities, paths and roads are proposed which will allow iteration with the ecological analysis to identify ways of accommodating both people and wildlife within the landscape

There is considerable scope for combining focal species-modelling with landscape character and recreational impact modelling within an integrated network development and analysis tool. However, before this approach can be recommended as a general method of constructing lowland habitat networks, it needs to be tested within case-study areas. Ideally three to four different case-study landscapes should be selected representing contrasting land-use mixes.

Looking beyond the case-study evaluation phase, there are a number of factors which constrain the practical implementation of habitat networks. Ecological constraints such as landscape dynamics, succession and climate change may limit the long-term viability of particular network designs. Biophysical constraints such as soil and topography will reduce flexibility for locating particular types of habitat. Lack of knowledge of habitat creation/restoration techniques, poor quality data, ownership, cultural and policy and planning issues can restrict the potential for network development. Finally, given the uncertain economic environment facing agriculture it is possible that significant "reactive" changes in land use practices could rapidly override "pro-active" strategic approaches to managing land use change.

Contents

List of Figures

List of Tables

1 INTRODUCTION

There is widespread acceptance that habitat fragmentation, coupled with habitat loss and degradation has a detrimental impact on biodiversity (Anon, 1995; Fahrig, 1997; Henle *et al.*, 2004a; Hudgens and Haddad, 2003; MacDonald, 2003). At both European (Bennett, 2003; Bouwma *et al.*, 2002; Foppen *et al.*, 2000; Jongman, 1995) country (Anon, 2000, 2002; Theobald, 2003; van Rooij *et al.*, 2001) and regional levels (Hodcroft and Alexander, 2004; Thompson *et al.*, 1999) there is increasing recognition that site-protection measures alone are insufficient to conserve biodiversity and a wider landscape scale approach is needed which fosters connectivity between habitats as well as improving general landscape quality (Angelstam and Andersson, 2001; Nowicki, in prep; Young *et al.*, 2003; Young *et al.*, in press). The Scottish Biodiversity Strategy (Anon, 2004a) emphasises the importance of adopting a landscape perspective to biodiversity conservation and to focus efforts on creating ecological networks. Further, the Scottish Executive report *Custodians of Change* (Anon, 2002) states (page 7):

> *"Scottish Natural Heritage's (SNH) Natural Heritage Futures programme and the production of Local Biodiversity Action Plans (LBAPs) provide a sound basis for the protection and enhancement of biodiversity, but greater emphasis should be given to the creation of ecological networks and the generation of co-operative action at a bio-regional or catchment scale"*

Like other areas of Britain, the Scottish lowlands has a long history of intensive land-use which has resulted in the loss and fragmentation of semi-natural agricultural habitats (eg species-rich grassland) and reduction in biodiversity (Robinson and Sutherland, 2002). In more recent times there has been an increase in woodland cover (mostly of introduced tree species) and the creation of linear wooded features such as shelterbelts and small plantations (Forestry Commission, 2002). Semi-natural woodland is often restricted to river valleys and non-economically productive agricultural land (Roberts *et al.*, 1992; www.scotlandswoods.org.uk).

Agri-environment measures have been introduced (Kleijn and Sutherland, 2003) to protect and enhance biodiversity through provision of incentives for habitat creation and management (Scottish Executive, 2004; Scottish Natural Heritage, 2001). Historically the focus within agri-environment schemes has been on individual habitats and ecosystems (eg wetlands, agricultural land, grassland, arable land, woodland) in isolation from each other. Where the existence of other habitats or features have been considered it is generally simply in relation to proximity to, or broad influence upon, the habitat/ecosystem forming the primary target. Increasingly however, agriculture, forestry and other land uses are no longer seen as separate entities, but considered together as part of the ecology of the wider landscape (Tattersall *et al.*, 2002). There is now a real need to consider the links and inter-relationships between the different components of the landscape. Although some species may be very habitat specific, others can and do exist across a range of different habitats demanding a more large-scale strategic perspective.

The concept of reversing the effects of habitat fragmentation through establishing habitat networks has developed rapidly throughout the last decade since the signing of the Convention on Biological Diversity (Jongman, 1995; UNCED, 1992). This has prompted a new international acceptance and emerging agreement of the need to conserve biological diversity using an approach which includes the planning, establishment and adaptive management of protected-area networks (UNEP, 2003). The landscape ecology paradigm of patch, matrix and corridor first introduced by Forman and Godron (1986) provides a theoretical backdrop to the development of networks, mirrored by the evolving ideas of 'greenways' (Smith and Helmund, 1993) from within the discipline of landscape architecture.

Whilst the creation of habitat networks is considered to be desirable from the perspective of biodiversity conservation, a variety of theoretical approaches to network development have been advocated derived from contrasting starting points and rationales. Research on defining ecological networks in relation to the landscape patch mosaic has covered a range of spatial scales, focusing initially on structural measures as indicators of connectivity. More recent work has explored the concept of functional connectivity of habitat (as opposed to simple physical connectedness) as a better measure of network integrity. Using this approach, habitat networks for wide-ranging species have been assessed in the design and planning of nature reserves (Theobald, 2003) and conservation areas at the regional scale (van Rooij *et al.*, 2004). However, there is a need to recognise that there are a number of possible solutions to the problem of how best to design networks, and a thorough review of approaches and application of theory is needed.

Work is beginning in some parts of lowland Scotland (Peterken, 2003; Ray *et al.*, 2004a) to develop Forest Habitat Networks (FHNs) in conjunction with the delivery of Local Biodiversity Action Plans (LBAPs) through the Scottish Forestry Grant Scheme. There is now a pressing need to extend this work to agriculturally dominated landscapes, to explore ways of generating a positive interplay between Agricultural Habitat Networks (AHNs), and FHNs and to offer guidance as to how agri-environment scheme resources might be best targeted to enhance lowland biodiversity within the context of other land-use pressures (eg landscape design, recreation and access). The purpose of this report is to offer an approach to developing habitat networks in Scottish lowland agricultural landscapes which will aid in strategic planning for biodiversity enhancement.

2 OBJECTIVES AND METHODS

The objectives of the work described in this report are:

i) to review current theory, and document evidence of the impacts of habitat fragmentation on the viability of habitats and populations of focal/key species within lowland landscapes

ii) to evaluate different theoretical approaches to reversing habitat fragmentation focusing on the development of habitat networks

iii) to establish ecological principles for the development of lowland habitat networks based on existing knowledge and on-going research

iv) to integrate ecological principles with other aspects of sustainable land-use, such as landscape character and recreation, and outline a practical and cost-effective approach to building habitat networks

The approach used in this study is predominantly desk-based, reviewing current ecological theory together with examples of recent approaches to implementing landscape-scale solutions to tackling habitat fragmentation. Relevant national and international literature was consulted through journal and library searches. In addition all relevant reports were obtained from country agency web-sites. Web searches were also undertaken to identify grey reports/literature. Very little work was identified concerned with the development of non-wooded habitat networks within agricultural landscapes relevant to the Scottish lowlands. Therefore a lot is inferred from studies of other ecosystems.

While linkage is made between ecological theory and to practical implementation through agri-environment schemes and other mechanisms, no recommendations are made as to how these schemes and mechanisms might be modified. Similarly, no assessment is made of the role of current agri-environment measures in creating and maintaining habitat networks. The focus of the report is primarily on outlining general principles and a practical methodology for designing habitat networks to be used by strategic planners as a tool to aid decision making.

3 DEFINITION OF "LOWLAND" AREAS

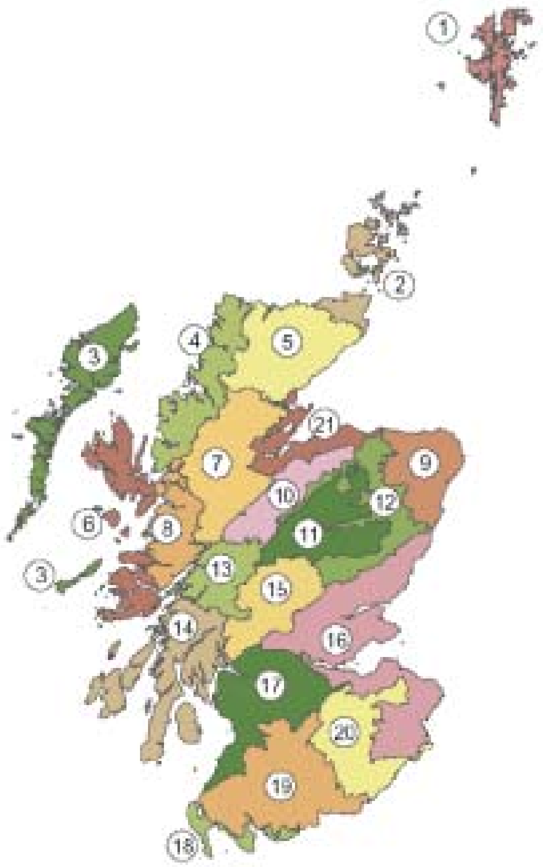
3.1 Principles

The work is focused on the Scottish lowlands and considers both agricultural (non-woodland) and woodland habitats as potentially being part of the habitat network. Arriving at a definition of lowland is not an exact science, since there are land use issues and cultural perceptions etc. to consider as well as climatic and edaphic aspects. In an effort to derive a straightforward definition, there is a danger of being too simplistic, for example the Upland HAP UK Co-ordination group define the uplands as being all land at elevations higher than 250–400m. The implication of this is that by default all land lower than this should be classified as lowland. Clearly a definition such as this would include low lying areas of north-west Scotland/Shetland which in terms of climate, geology and land-use would typically be considered as part of the uplands.

The SNH Natural Heritage Futures (NHFs – Scottish Natural Heritage, 2002a) provide a potential biogeographical framework (Figure 1) for arriving at a definition of lowland. NHFs are based on an amalgam of climatic, biological, aesthetic and cultural criteria, including land use patterns, and so provide a wider set of criteria for determining lowland character.

Whilst the individual descriptions of the NHF areas provide some indication of their relative "lowlandness" in character, the overlaying of climatic and geological attributes provide additional filters to help identify relative proportions of lowland area within each NHF.

Figure 1 Natural Heritage Futures map reproduced from Scottish Natural Heritage (2002a)



Key
1. Shetland
2. Orkney & Northern Caithness
3. Coll, Tiree and the Western Isles
4. North Western seaboard
5. The Peatlands of Caithness & Sutherland
6. Western Seaboard
7. Northern Highlands
8. Western Highlands
9. North Eastern Coastal Plain
10. Central Highlands
11. Cairngorms Massif
12. North East Glens
13. East Lochaber
14. Argyll West & Islands
15. Loch Lomond, The Trossachs & Breadalbane
16. Eastern Lowlands
17. West Central Belt
18. Wigtown Machars & Outer Solway Coast
19. Western Southern Uplands & Inner Solway
20. Border Hills
21. Moray Firth

3.2 Climate

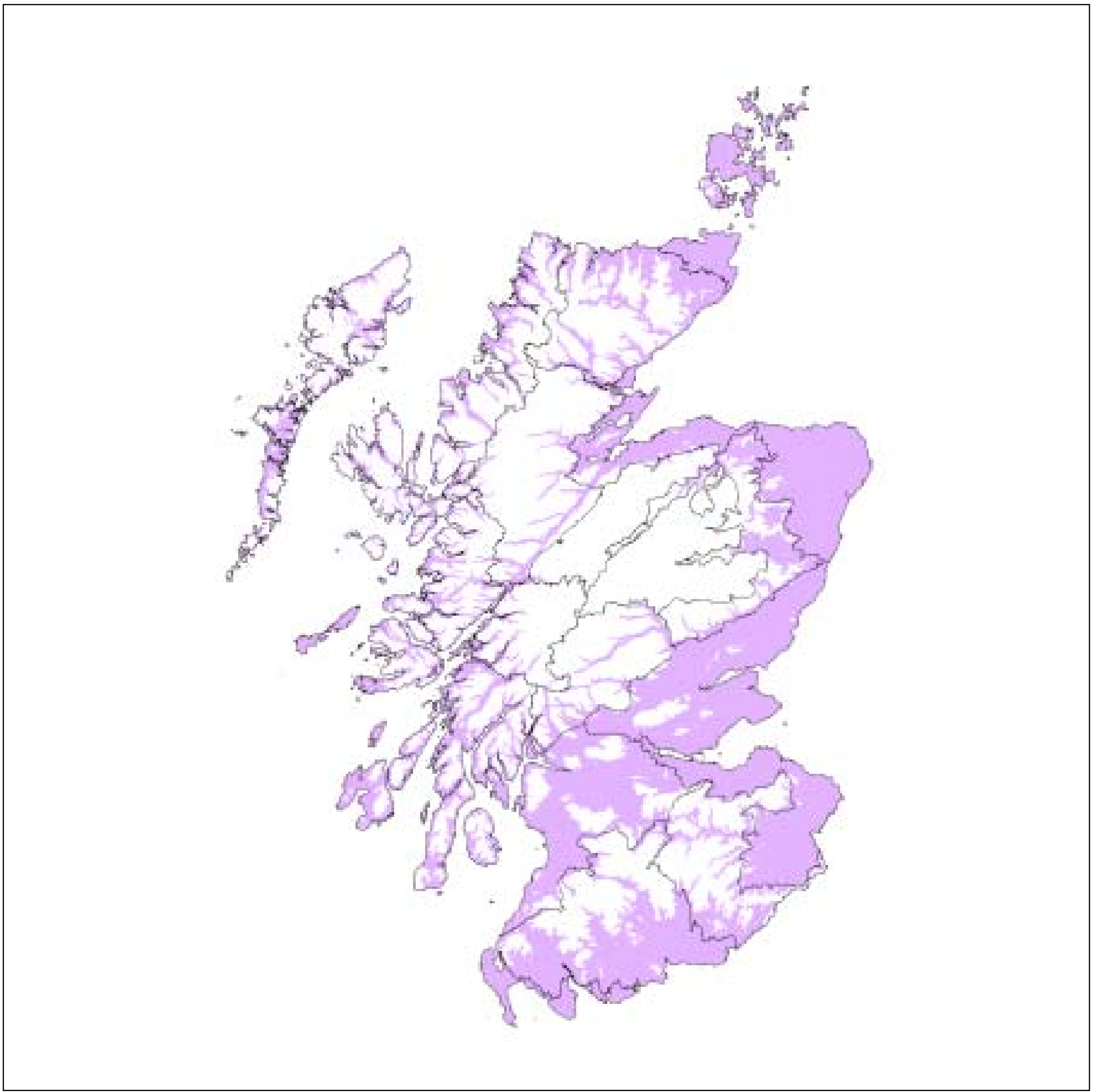
Climate data were obtained through the Forestry Commission's Ecological Site Classification. ESC (Pyatt *et al.*, 2001) is a tool for objectively assessing and classifying sites and landscapes in terms of their ecological potential for the suitability and yield potential of a given species of tree, or the ecological suitability of the National Vegetation Classification (NVC – Rodwell, 1991) woodland communities (Ray *et al.*, 2003). Similar methodologies have been pioneered in Scandinavia (Cajander, 1926) and in North America for mainly natural forest types (Klinka *et al.*, 1989). In Britain, ESC has been developed for use with plantation and semi-natural forest stands as well as open semi-natural communities (Pyatt and Suárez, 1997). ESC combines climatic influence with soil quality of the site. Climate data are predicted from models for any grid reference in Britain. The four variables used to describe the climate are: Accumulated Temperature (AT), Moisture Deficit (MD), Windiness (DAMS) and Continentality (Conrad Index) which are measures of climatic warmth, wetness or droughtiness and wind exposure. ESC provides a model to derive each of the indices for a given location; the climatic warmth index accumulated temperature (AT) above 5°C, moisture deficit (excess of summer evaporation over rainfall) and wind exposure using DAMS (Detailed Aspect Method of Scoring – Quine and White, 1993) are calculated from the national grid reference easting, northing and elevation. The data are available from an Ordnance Survey 50m resolution DTM (digital terrain model), allowing ArcView GIS spatial analyst (ESRI) to model the climate factor at a 50m grid resolution on the ground.

The climate variables are combined in ESC to yield seven climate zones (Pyatt *et al.*, 2001): alpine, sub-alpine, cool wet; cool moist; warm wet and warm dry. The cool moist; warm moist and warm dry zones are considered to fall within the wider "lowland climatic envelope", delineated by moisture deficit values of 90mm or more. ArcView GIS was used to overlay the three lowland zones on the NHF map. Table 1 shows the area covered by each zone within the 21 NHFs. Seven of the NHFs (highlighted in yellow) are clearly predominantly lowland in terms of climate (ie over 50% lowland) and four are predominantly upland (shaded green). The one anomaly is Shetland which although 98.4% lowland in a climatic sense, could be classed as upland in terms of vegetation and land use. Indeed, in a recent analysis of the occurrence of a montane scrub zone in Scotland, most of Shetland was predicted to fall within this zone (A. Macdonald *pers. comm.*). It has therefore been excluded from the lowland zone for the purpose of this report. The remaining 10 NHFs contain upland and lowland climate zones in varying proportions (composite zones). The lowland zones were combined in ArcView to produce one lowland climate data layer. This was overlaid on the NHF map (Figure 2).

Table 1 Occurrence of ESC lowland climate zones (Pyatt *et al.*, 2001) within NHFs. NHFs with predominantly lowland climate shaded yellow; those with an upland climate are shaded green. Areas are in hectares rounded to nearest hectare.

NHF	Total NHF Area	Cool moist zone	Warm moist zone	Warm dry zone	Total Lowland Area	% lowland
Argyll West and Islands	545745		267613		267613	49.0
Border Hills	413017	50652	60475		111127	26.9
Cairngorm Massif	403571	915			915	0.2
Loch Lomond the Trossachs & Breadalbane	355369	9852	46462		56314	15.8
Central Highlands	273213	5540	935		6476	2.4
Western Southern Uplands & Inner Solway	670968	3378	365778	57	369214	55.0
Eastern Lowlands	867437	130161	663788	1967	795916	91.8
East Lochaber	243507	186	23232		23419	9.6
Moray Firth	204493	60534	111157		171691	84.0
North East Coastal Plain	325086	211653	105434	44	317132	97.6
North East Glens	376661	117067	8504		125572	33.3
North West Seaboard	376329	68209	37123		105332	28.0
Northern Highlands	549080	21911	31622		53534	9.7
Orkney & Northern Caithness	186689	180867			180867	96.9
Shetland	150364	148011			148011	98.4
Peatlands of Caithness & Sutherland	522053	176176	5710		181886	34.8
West Central Belt	522052	14917	356919		371836	71.2
Western Highlands	274374	246	71099		71345	26.0
Coll, Tiree & Western Isles	355270	20030	106686		126716	35.7
Western Seaboard	331344		110496		110496	33.3
Wigtown Machairs and Outer Solway Coast	79611		79497		79497	99.9
TOTAL	8026242	1220312	2452538	2069	3674920	45.8

Figure 2 Lowland climate zone (light purple) in relation to NHFs. The lowland climate zone includes the Cool Moist, Warm Moist and Warm Dry zones within ESC (Pyatt *et al.*, 2001).



3.3 Geology and soils

Further delineation of the lowlands within the composite NHFs was achieved by removing areas with metamorphic or igneous geology (Figure 3). Both these geological types tend to give rise to acid soils supporting vegetation of upland character. Although a relatively blunt tool, the exclusion of areas with acid geology effectively reduces the proportion of lowland area within western and northern NHFs (ie those north of the Highland Boundary fault), (Figure 4) and identifies the NHFs shown in Figure 5 as being predominantly lowland in character. Area figures for Figure 4 are given in Table 2.

Figure 3 Location of metamorphic/igneous geology (shaded blue/green) in relation to NHFs

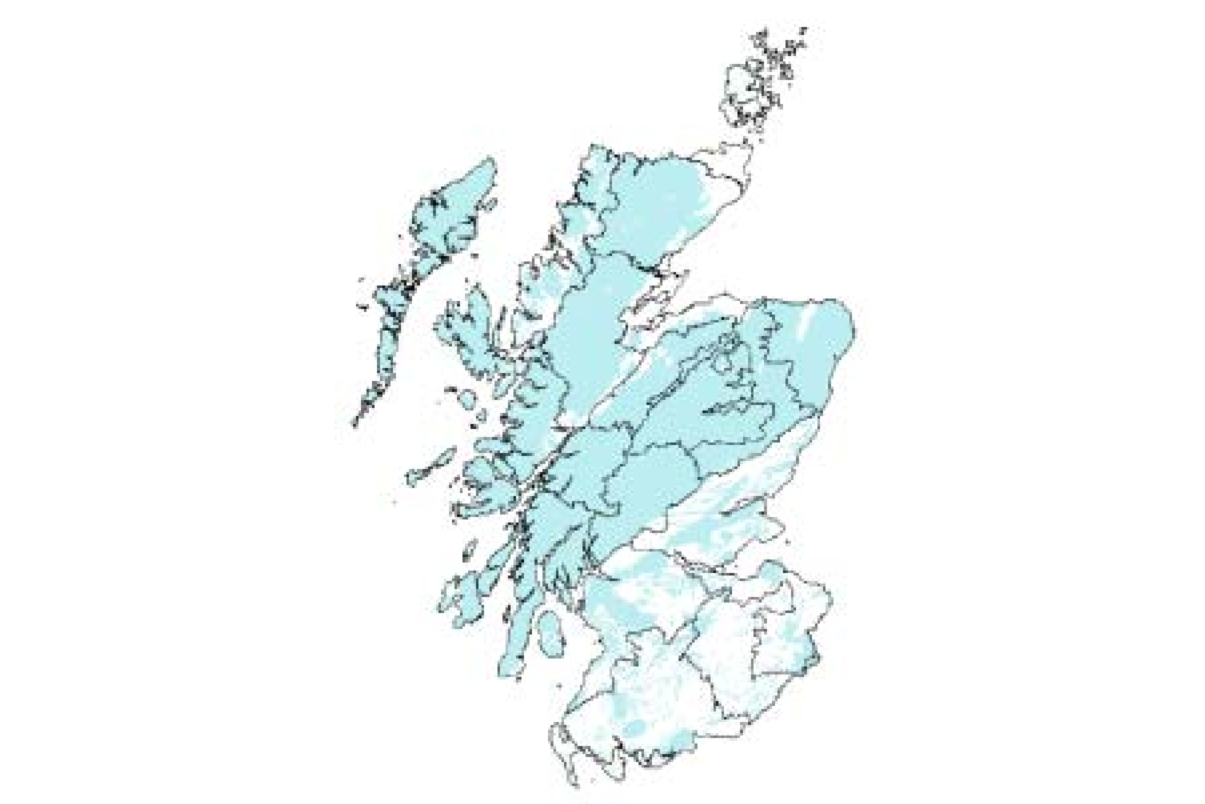


Figure 4 Lowland zone (shaded purple) based on a composite of climate and geology. Upland geological associations have been removed in NHFs north and west of the Highland boundary fault.

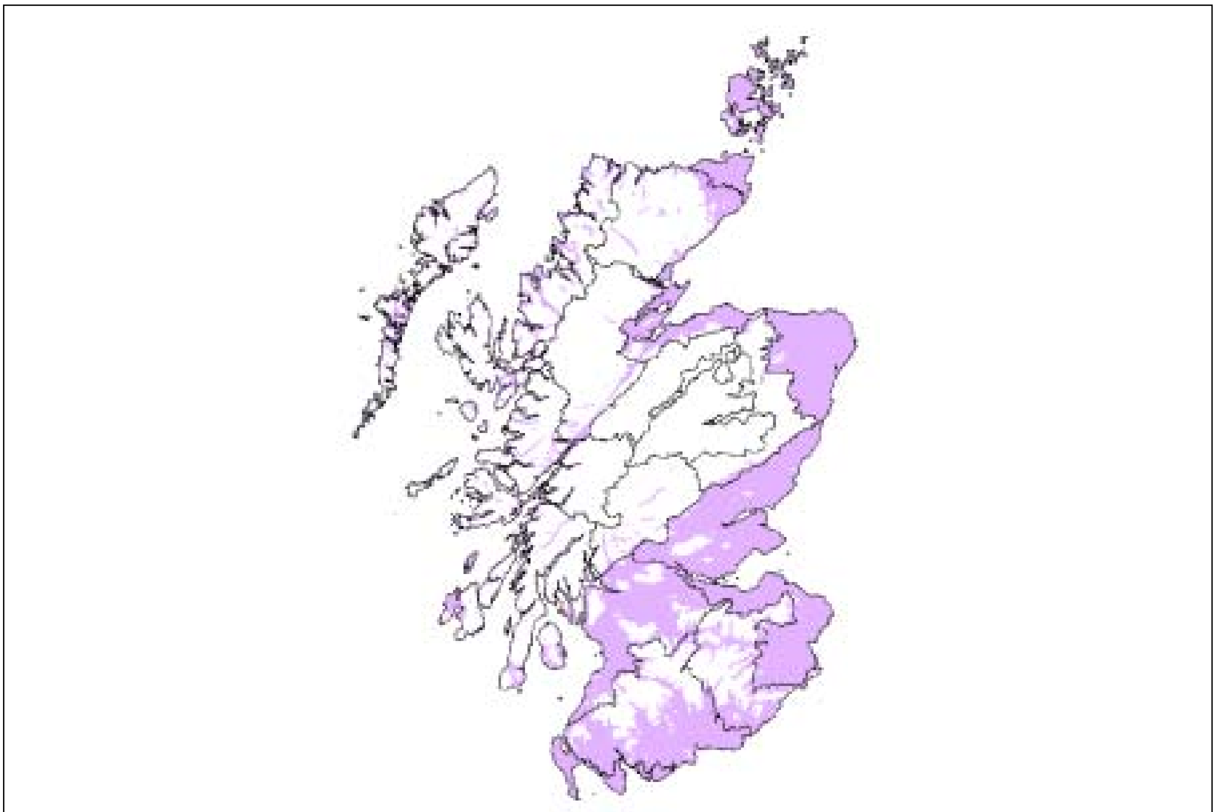
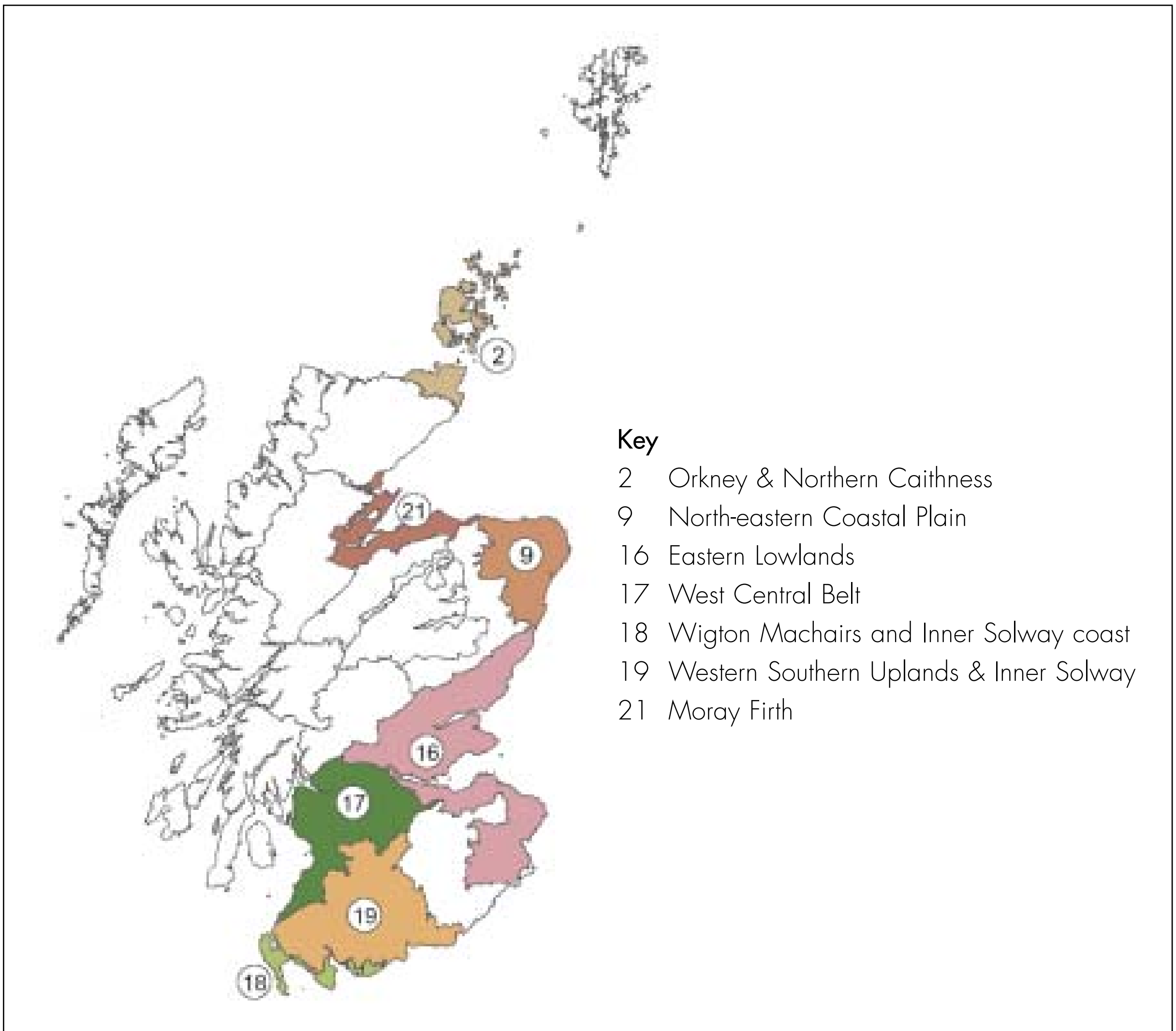


Figure 5 Location of lowland NHFs providing the context for this project



The seven NHFs shown in Figure 5 provide the geographical context for this project. However, although predominantly lowland in terms of character and ecology, it should be noted that these zones will all contain upland habitat as a minor component, and it would be unnecessarily prescriptive to exclude any consideration of these habitats and those adjacent in different NHFs from any network analysis. In many situations upland and lowland habitats exist in intimate association often as a result of land-use rather than of climate or edaphic influences (eg Orkney).

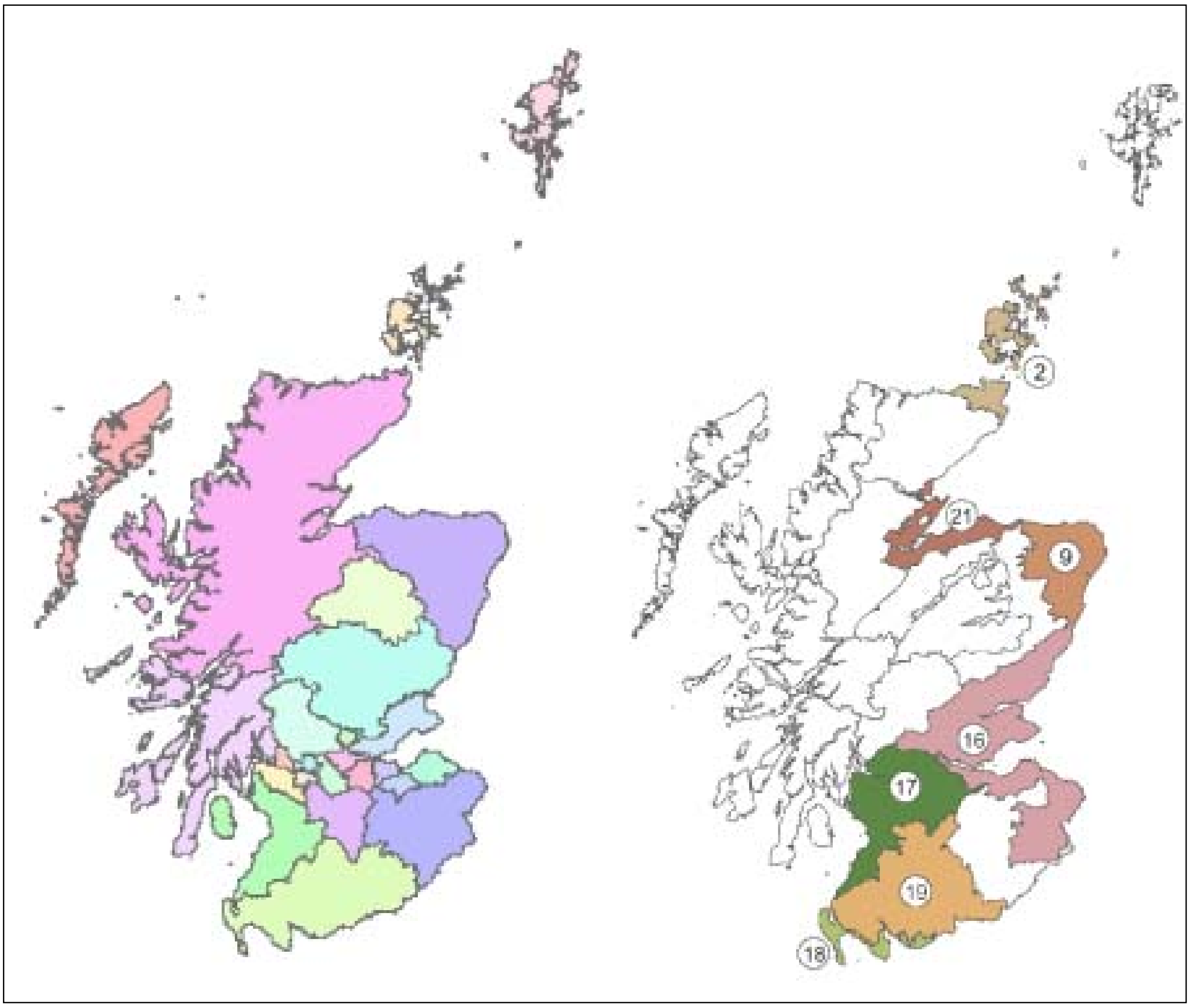
Table 2 Percent area of NHFs with lowland climate and edaphic character. Upland geology was removed from NHFs north of Highland Boundary fault. Lowland NHFs are shaded yellow. Shetland has been excluded

NHF	Total NHF Area	Total Lowland Climate Zone from Table 2 Climate zone	Total upland geology within Lowland	Total lowland ecological character	% lowland
Argyll West and Islands	545745	267613	179853	87761	16
Border Hills	413017	111128		111128	27
Cairngorm Massif	403572	915	915	0	0
Loch Lomond the Trossachs & Breadalbane	355369	56315	43786	12529	4
Central Highlands	273214	6476	5595	882	0
Western Southern Uplands & Inner Solway	670969	369215		369215	55
Eastern Lowlands	867437	795917		795917	92
East Lochaber	243508	23420	18692	4727	2
Moray Firth	204493	171691	20644	151047	74
North East Coastal Plain	325086	317133		317133	98
North East Glens	376661	125573	112364	13209	4
North West Seaboard	376330	105333	35795	69538	18
Northern Highlands	549080	53535	31891	21643	4
Orkney & Northern Caithness	186689	180868	2947	177920	95
Peatlands of Caithness & Sutherland	522053	181886	104588	77298	15
West Central Belt	522053	371837		371837	71
Western Highlands	274375	71346	37997	33348	12
Coll, Tiree & Western Isles	355270	126717	65224	61493	17
Western Seaboard	331344	110497	60584	49913	15
Wigtown Machairs and Outer Solway Coast	79612	79497		79497	100
TOTAL	7875877	3526912	720875	2806035	46

3.4 Relationship with LBAP regions

The spatial relationship between Local Biodiversity Action Plan (LBAP) regions and NHFs was assessed to ascertain whether further delineation of lowland character could be achieved. However, LBAP regions are aligned to local authority boundaries rather than to NHFs (Figure 6), and are therefore not particularly useful in helping to define the lowland zone. The advantage of linking LBAP regions to NHFs in any decision support system is that there is a direct link through the LBAP internet site (www.ukbap.org.uk) to the NBN which provides information on key habitats and species of use in the network modelling process (see section 9).

Figure 6 LBAP regions in Scotland (from www.ukbap.org.uk) compared to lowland NHFs



4 DEFINITION OF SCALE

4.1 Moving from site to landscapes and regions

In recognition of the increased threats from habitat fragmentation Jongman and Pungetti, (2004a) describe how new philosophical directions have emerged, moving from isolation to connection and from a concentric to a peripheral approach. Nature conservation, accordingly, is moving from small to large spatial scales.

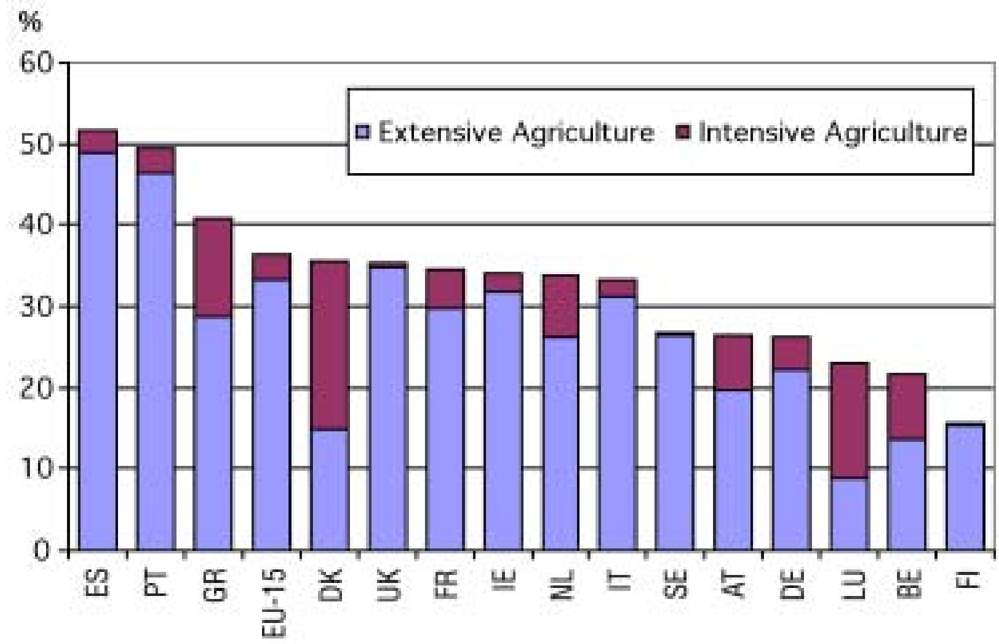
> *If the previous focus was primarily on areas of high nature conservation, eg national parks, now the focus is moving towards linkage between them and linkages between nature and the human environment such as greenways, ecosystem coherence and ecological networks. These concepts have become familiar in ecological language at both the scientific and the public level (Jongman and Pungetti, 2004b, p.1).*

There is recognition that strategies focused on protecting key sites have failed to reverse fragmentation and the loss of species diversity in the countryside (Hawkins and Selman, 2002; Lee *et al.*, 2002; Lee *et al.*, 2001; Thompson *et al.*, 1999; Thompson *et al.*, 2001) and in future site conservation measures will be increasingly complemented with actions to sustain habitat quality and wildlife in the wider landscape (Humphrey *et al.*, 2003a; Townshend *et al.*, 2004). At the EU level considerable progress has been made towards implementing a landscapes-scale approach to biodiversity conservation through development of the NATURA 2000 site network (http://europa.eu.int/comm/environment/nature/) and the Pan-European Ecological Network (PEEN – Bouwma *et al.*, 2002) which forms part of the Pan-European Biological and Landscape Diversity Strategy (PEBLDS – http://www.strategyguide.org/index.html).

It is intended that the Natura 2000 network will build on the proposed sites of communal interest (pSCIs) that have been listed by the Member States. Out of the 198 habitat types listed in Annex 1 of the Habitats Directive (European Commission, 2000), 65 have been shown to be threatened by the intensification of agriculture practices, whilst 26 grazed pasture habitats and six mown grassland habitats are threatened by the abandonment of pastoral management practices (Ostermann, 1998). However, despite the dominance of farmland across Europe and its importance from a biodiversity perspective, agricultural habitats only form about 35% of the total area listed as pSCIs across the EU-15 and only three countries (Greece, Portugal and Spain) have included a greater proportion of such habitats within the pSCIs they have listed (Figure 7). It would therefore appear that the site protection measures employed to-date have not been targeted at areas of high farmland biodiversity potential within the more intensively managed agricultural landscapes.

The aim of the PEEN is to protect important core areas for surviving wildlife, enlarge these through habitat creation, connect them through corridors to encourage dispersal migration and protect them against negative influences by establishing buffer zones.

Figure 7 The amount of agricultural land included in pSCIs proposed under the Habitats Directive within each of the EU-15. The amount of extensive and intensive agricultural habitats is shown as a proportion of the total area designated as pSCIs within each country. Source ETC/NPB in McCracken (2004).



One of the central themes of the PEEN is the development of the "ecosystem approach" which is holistic approach to large-scale management, incorporating social and economic factors with ecological processes and biodiversity (*sensu* – Franklin, 1989). The BIOFORUM project (a pan-European Concerted Action: www.nbu.ac.uk/bioforum/) is exploring ways of implementing the ecosystem approach in Europe. The focus is on the reconciliation of human activities and biodiversity conservation (Young *et al.*, 2003), operating from the premise that once knowledge is gained about the conflicts generated in terms of biodiversity conservation objectives, the resolution of these conflicts becomes possible. The methods for conflict reconciliation proposed by BIOFORUM have to be applied in a temporal and spatial framework; for this reason the ordered application of conflict resolution strategies requires reference to spatial planning. BIOFORUM participants are currently drafting a report recommending principles and guidelines for applying an ecosystem approach to spatial planning across Europe (Nowicki, in prep).

Within the UK, country biodiversity strategies (eg – Anon, 2004a) emphasise the importance of management of the wider landscape in conserving biodiversity including UK Biodiversity Action Plan (Anon, 1995) priority species and habitats. Scottish Natural Heritage's approach (Scottish Natural Heritage, 2002b) to improving the wildlife value of the farmed landscape recognises implicitly the need to work across sectors and scales. This is further reinforced in the Scottish Executive report *Custodians of change* (Anon, 2002) – see section 1).

4.2 Importance of policy and planning frameworks in determining scale

Despite the recent focus on landscape-scale conservation, there has been fewer attempts at quantifying the appropriate size of spatial-planning framework and this has led, in part, to slow take up of landscape planning by local authorities (eg – Hodcroft and Alexander, 2004). However, important lessons have been learned from English Nature's "Lifescapes" approach (Porter and Wright, 2003) which was a practical attempt at landscape planning. Lifescapes originated from initial work reviewing the effects of habitat fragmentation on nature conservation (Kirby, 1995) and subsequent pilot projects on habitat restoration at the landscape scale (Thomas, 2000). The objective of Lifescapes was to find ways of securing shared action to deliver the nature conservation objectives set for English Nature's Natural Areas (http://www.english-nature.org.uk/science/natural/role.htm) in a way which also contributed to social and economic objectives. The work addressed concern over processes and impacts that operate above the site scale, including fragmentation of habitats, species that operate at landscape scale, ecosystem functioning, climate change, and how to link biodiversity and geodiversity to socio-economics. The vision was about influencing a new shape for the English countryside, delivering biodiversity targets and changing unsustainable landuse practices through building partnerships, providing information and informing decisions making. Pilots were set up in within four Natural Areas: Forest of Bowland, Suffolk Coasts and Heaths, the Chilterns and the South Downs.

In a review of the outcome of the pilots Porter and Wright (2003) concluded that **a common vision across all sectors/organisations involved in the projects was impossible to achieve and an options approach was most desirable. It was also concluded that Natural Areas were too large for effective engagement of local communities in the planning process suggesting that smaller catchment scale analyses were more appropriate for the purpose**. Greater emphasis is now being placed on a programme of "Landscape Delivery" which takes the Lifescape concept further in particular emphasising the need for a spatially-explicit planning approach which allows ecosystem managers to consider actual and potential effects of their activities on adjacent and other ecosystems.

Such an approach has been adopted in the Life ECOnet project. This is a European LIFE-Environment funded project, led by Cheshire County Council through a partnership of local authorities, Government agencies, practitioners and research centres in a number of countries (www.lifeeconet.com). A series of models were identified to evaluate options for future land use. The outputs will help local authorities and others to integrate environmental issues into land-use planning and management. The project has used GIS data on habitats and species to model how the current pattern of habitats and land use supports or prevents species dispersal. Existing semi-natural habitats have been mapped, and other areas identified that could link these habitats and allow species to move between them. In three study areas (Cheshire, Abruzzo and Emilia-Romagna regions in Italy), a wide range of local people and groups have worked together to restore or create suitable habitats in the network, using existing sources of funding. The project has successfully encouraged the adoption of this idea of ecological networks within sustainable development strategies and planning policies in north west England and **demonstrates the importance of selecting a scale of study which is both ecologically and socially/culturally relevant**.

4.3 Importance of species-based approach

The spatial planning projects described above demonstrate the importance of ecological attributes in determining relevant scale of enquiry. A range of scales for developing habitat networks can be envisaged from the individual farm, through catchment scale farm groupings, to the regional and country levels. **As will be made clear in this review, the spatial scale of interest is determined to a considerable extent by the species of interest ie connectivity for birds is best considered at the regional scale, invertebrates at the catchment scale etc. or less**. For example, viable populations of a very wide range of invertebrates could potentially be accommodated satisfactorily within an area covering only 10s of hectares (Settele *et al.*, 1996) while maximising the number and abundance of larger species such as birds and mammals would require consideration of 100s of hectares (Goodman, 1987). Differences in the scale required will mean that it will not be feasible to provide for the requirements of all species. Trade-offs will be required between the species of interest and practicality of spatial planning from the end-user perspective. It is important therefore to take care with species choice and ensure that selected species are collectively truly representative of wider biodiversity.

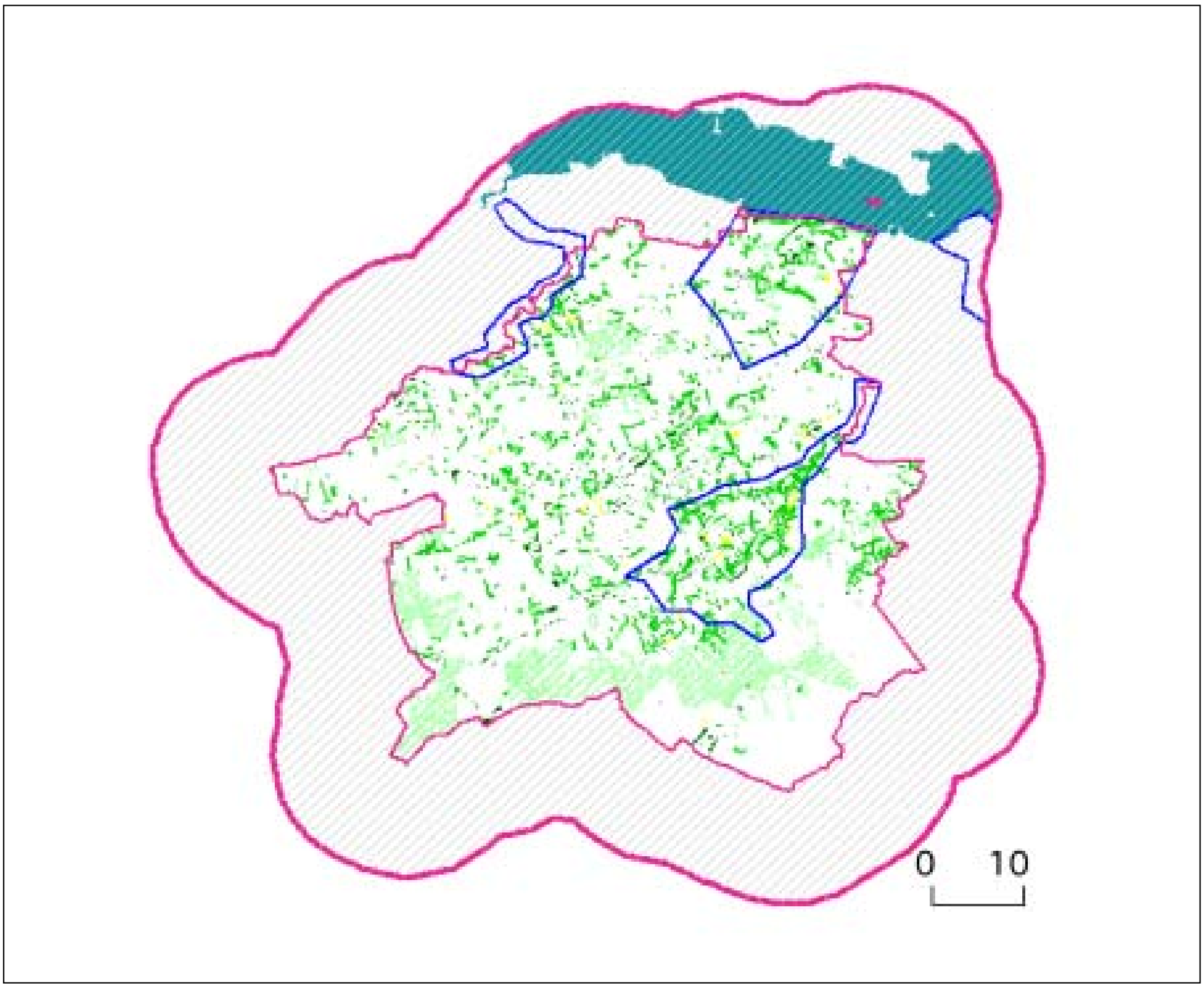
4.4 Importance of end-users

Scale is also determined in part by the nature of the end-user, the questions that the network is being asked to address, and the ability and interest of that end-user to engage in the planning process. Strategic planners engage with the process of network development at a regional level (eg 1000–2000km^2) whereas individual land-owners and those using local surroundings find it easier to engage at smaller spatial scales. As part of its review of agri-environment schemes and as a consequence of CAP reform, SEERAD will be promoting land management contracts and catchment scale management plans involving groups of farms. Thus catchment scale may be an appropriate starting point for considering network development in this study, although the regional scale will be relevant when it comes to integrating AHNs with FHNs. In their development of local forestry frameworks in southern Scotland (Environmental Resources Management, 2000) suggested that water catchments provided an appropriate sub-area for analysis as:

- They are of a suitable scale;
- They are discernible physical areas which are understandable to local residents, landowners and land managers;
- They do not change;
- They are not subjective; and
- They relate to the EC Water Framework Directive (http://europa.eu.int/comm/environment/water/water-framework/index_en.html) which will require land management plans to be prepared on a catchment basis.

For a catchment scale analysis Humphrey *et al.* (2004a) suggest that an upper area threshold of 200km^2 is advisable (for example see Figure 8). This provides a benchmark for characterising landscape grain, structure and pattern. Beyond this threshold, species responses are less likely to be driven by local landscape structure and more by regional pattern (Humphrey *et al.*, 2004a). In addition, as indicated earlier, Porter and Wright (2003) conclude that the sub-catchment scale is the most appropriate scale at which to engage end-users. Woodland and non-woodland habitats can be integrated at a variety of different ways at this scale, with varying consequences for different taxa.

Figure 8 Map of West Lothian (Ray *et al*. 2004b) illustrating typical spatial scale within which a number of landscape/catchment scale analysis can be undertaken (areas outlined in blue).



5 THEORETICAL FRAMEWORK FOR LANDSCAPE EVALUATION

5.1 Introduction

A number of key ecological theories and scientific approaches have influenced our understanding of the impacts of habitat fragmentation on biodiversity at the landscape scale and the relationship between habitat (patch) area, habitat isolation and species viability. Many of these theories and approaches form the building blocks in the understanding and design of landscape scale solutions to the problem of habitat fragmentation. In this section we review the main theoretical approaches, and go on in sections 6 and 7 to illustrate how they have been applied in practice and discuss shortcomings.

5.2 Ecological consequences of habitat fragmentation

The basic fragmentation process involves the breaking up of a few large patches of habitat into an increased number of smaller patches (Kirby, 1995; Saunders *et al.*, 1991). This process poses two challenges for biodiversity: first, there is a reduction in the area of available habitat, and; second, the remaining patches suffer from increased isolation (Fahrig, 1997; Henle *et al.*, 2004a). This also results in a greater amount of edge habitat for a given area resulting in a reduction in the amount of core habitat area as the frequently detrimental impact of increased edge conditions permeate the relatively undisturbed habitat core (Kirby, 1995; Ozanne *et al.*, 2000).

According to a number of scientific theories, the reduction in area may lead to increased local extinctions, whilst increased isolation may cause a reduction in the exchange of individuals between isolated patches, threatening their long-term viability (Saunders *et al.*, 1991). Many species have become adapted to a highly connected and extensive habitat, and fragmentation has inevitably had a major impact on them (Olff and Ritchie, 2002). Those with very large home-ranges will have become extinct rapidly, whilst the chronic interruption of dispersal, migration and metapopulation dynamics of many species will have caused a slow attrition of biodiversity (Henle *et al.*, 2004b). There are concerns that climate change will compound these effects, as species will not be able to track the movement of their climatic niches across landscapes and will become more susceptible to extinction (Opdam and Wascher, 2004).

5.3 Species-area relationships and island biogeography

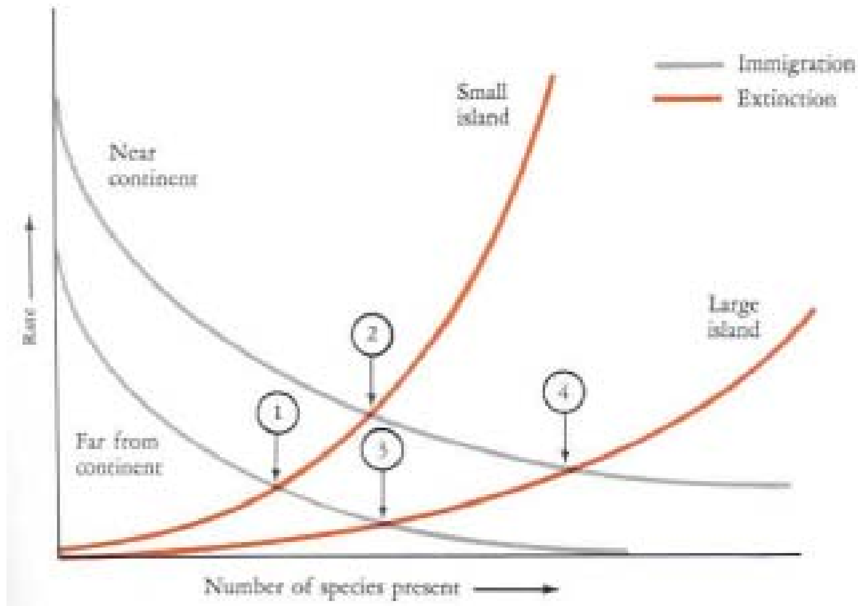
Theories of species-area relationships and island biogeography provide a fundamental starting point in the development of our understanding of the effects of landscape structure and process on biodiversity.

The **species-area** relationship is a formalisation of the observation that large areas usually contain more species than small areas of comparable habitat, but with a decreasing rate of increase in the number of species as areas become larger. Plant ecologists first attempted to elucidate the exact form of the curvilinear relationship early in the 20th Century (Arrhenius, 1921). The relationship between species and area has an extensive literature; indeed, Connor and McCoy (1979) suggest an awareness of the basic species-area relationship dates back to 1835. Shafer (1994) stresses that the basic idea of the species-area curve predated the theory of island biogeography by over 120 years.

The theory of **island biogeography** (MacArthur and Wilson, 1967) which attempted to explain the variations in species diversity on oceanic islands, is an especially important component of the landscape ecology approach. Simply stated, the theory holds that the number of species and the species composition of an island is dynamic, and is determined by the equilibrium between the immigration of new species and the extinction of those already present. According to the model, rates of immigration and extinction depend on the size of an island and its distance from a mainland species reservoir, which allows the construction of a general equilibrium model (Figure 9). Four equilibrium points are shown on the model representing different combinations of large and small islands near and far from continental shores. In this illustration, point 1 is the worst scenario and point 4 is the best.

Owing to the continuing fragmentation and isolation of habitats, an analogy soon formed between the true 'oceanic islands', upon which the theory of island biogeography is based, and 'terrestrial habitat islands' which were surrounded by an apparent 'sea' of inhospitable domesticated or urbanised landscapes. The theory of island biogeography enabled ecologists to relate island size to the range and viability of species, indicating larger habitat islands would be more likely to sustain a larger number of species.

Figure 9 Theory of island biogeography after MacArthur and Wilson (1967)



The idea that such habitat islands could be treated by the same theories as real islands was initially very popular and led to several suggestions as to how such theories could aid conservation, culminating in proposals for designing and acquiring nature reserves (Diamond, 1975). Selman (2000) p61 describes how the theory of island biogeography was "highly influential on nature conservation policy, where it led scientists to debate the respective merits of protecting several small sites as opposed to a large single one within a particular area" (the **SLOSS concept** – 'single large or several small'). Diamond (1975) used the theory of island biogeography and species-area relationships to propose certain optimal design principles for nature reserves in order to maximise their species richness and viability (Figure 10).

The principles behind the six designs were:

A A large reserve is better than a small reserve, as the large reserve can hold more species at equilibrium, and it will have lower extinction rates.

B The reserve should generally be divided into as few disjunctive pieces as possible, for essentially the reasons underlying principle A.

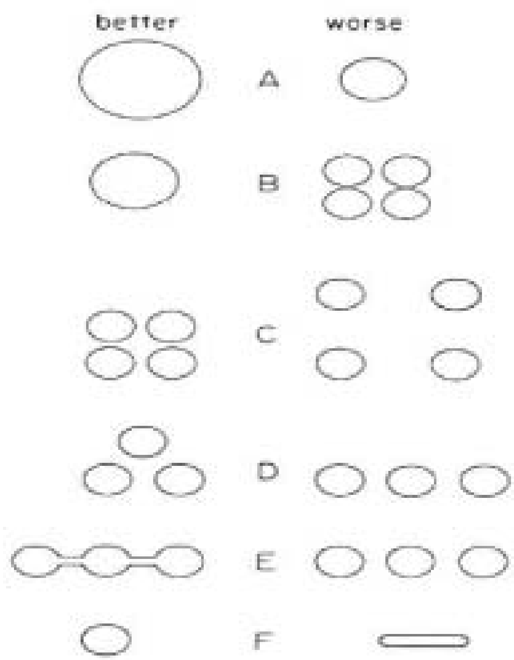
C If the reserve is broken up, the pieces should be as close to each other as possible, to increase immigration rates.

D The reserve pieces should be grouped equidistant from each other, rather than grouped linearly, as in linear arrangement the terminal sites become isolated with reduced re-colonisation.

E Connect several disjunct reserves with strips of protective habitats, which will increase the ability to disperse between reserves.

F Reserves should be as nearly circular in shape as possible, to minimise dispersal distances within the reserve.

Figure 10 Nature reserve design principles after Diamond (1975)



The application of island biogeography theory to terrestrial habitat islands is an appealingly simple idea, but the relationships between the population dynamics of species, and the qualities of core and intervening habitats, is far more complex. As a result, both the theory of island biogeography and its subsequent applications are often criticised for being too simplistic and not recognising the actual reality of designing and acquiring protected areas (Gilbert, 1980; Margules *et al.*, 1982; Reed, 1983). However, Peck (1998) points out that the principles proposed by Diamond (1975) are an important step in the development of the field, identifying several ideas that proved fundamental for reserve design:

> *For example, large reserves are clearly valuable for most reserve systems. His principles regarding the size and shape of reserves addressed the impact of edges and the importance of maintaining interior habitat for sensitive species. By advocating reserves located close together, or connected by corridors, he highlighted the value of connectivity for species dispersal (Peck, 1998, p.92).*

5.4 The emergence of landscape ecology

Landscape ecology is defined as "the study of the interactions between the temporal and spatial aspects of a landscape and its flora, fauna and cultural components" (Dover and Bunce, 1998). The term landscape ecology was first coined by the German biogeographer Carl Troll at the end of the 1930s (Farina, 1998). Troll hoped that a new science could be developed that would combine the spatial, 'horizontal' approach of geographers with the functional, 'vertical' approach of ecologists. Landscape ecology also occupies an important bridge between pure and applied ecology, with great potential for the integration of emerging theories (eg island biogeography, metapopulation models).

Landscape ecology provides a basis for understanding the nature and dynamics of the landscape, based on the key principle that landscapes contain an inherent ecological infrastructure or network (often based on elements such as patch, corridor, matrix) that is conducive to different levels and types of species diversity. Landscape ecology appears to be able to help us explain, predict and plan change in the landscape, by focusing on the wider ecological structures and functions. Land use plans and indicative strategies based on these ecological principles are finding increasing application, especially in Europe and the USA, and more recently the UK. This reflects a growing maturity in landscape ecology, enabling it not only to inform theory, but also offer solutions to 'real world' planning problems (Hawkins and Selman, 2002).

The landscape ecology paradigm provides a useful framework for reviewing approaches to addressing biodiversity conservation at the landscape scale. **Two contrasting, yet complementary approaches can be identified (after Opdam *et al.*, 2002):**

1. **focusing on modifying landscape structure and implying benefits for species and ecological functioning (eg species dispersal and population dynamics)**

2. **focusing on the needs of different species as a means of prioritising changes in structure.**

6 STRUCTURE-BASED APPROACHES TO LANDSCAPE EVALUATION

6.1 Use of landscape metrics

Attempts have been made to derive measures (metrics or indices) of landscape structure/pattern (eg Gustafson, 1998; Turner and Gardner, 1991) in order to imply suitability for different species groups. In essence the metrics are treated as indicators or surrogates of biodiversity although they are rarely tested in this way (Humphrey and Watts, 2005). Landscape metrics are easily calculated using spatial statistical packages, and are used to measure the distribution, shape and proximity of habitat patches within a landscape. The use of landscape metrics in landscape evaluation has been criticised recently (Li and Wu, 2004; Lindenmayer *et al.*, 2002). Li and Wu (2004) identify three drawbacks of metrics: (1) conceptual flaws in landscape pattern analysis, such as, unwarranted relationships between pattern and process, ecological irrelevance of particular landscape indices based on mathematical constructs etc.; (2) inherent limitation of landscape metrics (eg variable response of indices to changes in spatial pattern; difficulties in interpreting behaviour); (3) improper use of indices (eg inappropriate inference from a single landscape analysis).

Perhaps the most important drawback of the metrics approach is that it amplifies the importance of structure over function, and, it is very difficult to infer function from structure (Wu and Hobbs, 2002). It is necessary to make the fundamental distinction between 'structural connectivity' and 'functional connectivity' (Gergel and Turner, 2002). Structural connectivity is the degree of physical connection between elements of the same type; ie an attribute of landscape pattern. Functional connectivity, on the other hand, is an attribute of landscape connectivity that is defined by landscape processes such as species movement and dispersal between patches. Indeed, it is possible to have high functional connectivity in a physically fragmented landscape with low structural connectivity, as long as the wider matrix supports the particular ecological process (Farina, 1998).

The use of landscape metrics in practical spatial planning has never really taken off because of these theoretical shortcomings (Li and Wu, 2004). However, metrics appear to have a use in landscape evaluation when they are to be linked specifically to known species requirements (Lindenmayer *et al.*, 2002). For example, Moser *et al.* (2002) found that landscape patch complexity can predict species-richness in vascular plants. Tischendorf *et al.* (2003) found that patch isolation became a good generic predictor of immigration when the nature of the intervening matrix was taken into account and data incorporated on the dispersal ecology and behaviour of organism of interest. The use of metrics may be useful at regional scales where simple rules can be linked to general policy. For example, Petit *et al.* (2004) found that variation in ancient woodland indicator plant species richness within woodlands in the British lowlands was explained by patch area; the amount of woodland within 500m of the woodland and two measures of connectivity (the length of adjacent hedgerows and lines of trees). In contrast, species richness in upland woods was determined by habitat quality rather than attributes of landscape structure. This suggests that different policy approaches to woodland conservation would be appropriate in different areas.

The use of metrics at the landscape/catchment scale to predict biodiversity values appears to have limited value. Lee *et al.* (2001) found that alpha (species) richness in calcareous grassland was not significantly correlated with a range of metrics (patch area, shape, nature and proximity of surrounding land use) although key aggregations of habitat patches were identified to allow some targeting of resources to improve landscape quality. However, in a similar study, Thompson *et al*. (2001) found that the alpha diversity of patches of ancient semi-natural woodland in the Chilterns was positively correlated (although r-square values were low) to the physical attributes of woodland patch size, patch area, and patch shape but there was no effect of proximity of "other" woodland or nature of surrounding land-use.

Summary Box

Landscape metrics

- provide measures of landscape structure (eg patch size and distribution) as a surrogate for ecological function and biodiversity value
- are often ecologically irrelevant unless calibrated by species requirements
- are of limited value in landscape scale analyses where there is a need to assess landscape function
- have a role in broad-based regional scale evaluations as long as shortcomings are recognised

6.2 Spatial targeting and landscape thresholds

In contrast to the metrics approach which evaluates landscape quality as a whole, spatial targeting approaches tends to focus on the protection and creation/enhancement of particular structures/habitats identified as important surrogate measures of biodiversity value (Thompson *et al*., 2001). The concept of spatially targeting conservation measures in agricultural landscapes is not new (Thompson *et al*., 1999; Wilson, 1997). For example Webster and Felton (1993) suggested that agricultural policies should reflect regional differences in biodiversity and habitat quality and take account of the farm systems present, to ensure habitat creation/restoration measures are targeted to most appropriate places. With the development of GIS, and greater accuracy and availability of large-scale spatial datasets, there has been an increasing emphasis on exploring different methods of targeting resources.

In connection with English Nature's "Lifescape" project (Porter and Preston, 2001) a number of different spatial targeting methods have been tested in the Chilterns (eg Lee *et al*., 2002; Thompson *et al*., 1999). Thompson *et al*. (1999) combined land cover data with a set of rules based on slope and altitude to identify priority areas for chalk grassland creation and restoration which could then feed into modification of ESA agri-environment funding mechanisms. Three tiers of 'restorability' were identified ranging from improved grassland areas on flat slopes with limited potential, to areas with high potential on dry, acid soils with slope gradients of more than 1:10. Over 12500ha of suitable ground for restoration was identified although only 275ha is currently good quality habitat. It seems unlikely that all the suitable area would be restored and further prioritisation would appear necessary.

In a similar study Lee *et al*. (2002) developed a model for targeting the expansion of native woodland in the Chilterns. On the basis of optimising biodiversity value, they set an area-target of 100ha for semi-natural woodland patch size, and then identified which woods it would be most cost-effective to expand (ie least requirement for planting) to achieve the target size. They identified patches of 20–50ha as cores for expansion and classified surrounding land use in terms of its suitability to support woodland and its proximity to these core areas. Woods which were surrounded by other woods were therefore targeted for expansion.

No account was taken of the "ecological ease" in which new woodland could be created on adjacent non-wooded ground. New woodland was targeted to areas of low conservation value (eg arable) rather than habitats with existing conservation value such as species-rich grassland.

In this study, no account was taken of the known difficulty in establishing woodland ecosystems on intensively managed arable sites (Harmer, 1999; Peterken, 2000; Worrell and Francis, 2003) nor attempts made to consider what appropriate balance of different habitats might be at the landscape scale. In addition, no species-based justification is made as to why 100ha might be a suitable target for woodland area. Peterken (2002) recommends minimum areas of 25ha for managed woodland and 50ha for minimal intervention reserves. Despite the shortcomings of this approach, the authors maintain that in the absence of good species data, a habitat-based method of evaluation is the most pragmatic alternative.

Gkaraveli *et al*. (2004) used a slightly different approach to spatial targeting of new woodland areas in a study based in north Wales. They identified land cover constraints on woodland expansion (eg water, semi-natural habitats of existing value for nature conservation), treated SSSI and other ancient-semi-natural broadleaved woodland as nodes within a proposed network. They then used a weighted scoring system to indicate potential biodiversity value of areas for native woodland expansion (eg open ground, areas of plantation on existing ancient woodland sites, and areas previously cleared of ancient woodland). Using this method 83000ha out of a potential area of 150000ha were identified as a priority for woodland expansion. The authors clearly identified the need for further targeting within this priority area by steering woodland away from other (non-designated) open ground of conservation value. Gray and Stone (2003) adopted a similar approach on Mull by combining land cover, forest history and forest composition data to identify priority areas for woodland expansion.

In a recent study in West Lothian Ray *et al*. (2004b) identified similar problems associated with implementing simple rules for woodland expansion. They tested the effect of buffering all existing semi-natural woodland to a distance of 300m following rules set out in the Scottish Forestry Grant Scheme (Anon, 2003). The results showed that virtually the whole landscape would end up being converted to woodland if the rule was followed on every woodland patch. Focal species modelling (see section 7.4) was advocated as a method to allow further prioritisation of woodland expansion.

In an alternative to spatial targeting of changes in landscape structure (Peterken, 2000; Peterken *et al*., 1995) introduced an influential approach based on **land cover thresholds** derived from the analysis of random (neutral) landscapes (see – Andren, 1994; Franklin and Forman, 1987; Gardner *et al*., 1987; Gardner and O'Neill, 1991). According to this approach there are potentially a large number of small, isolated woods within a landscape with 10–20% woodland cover, edge habitats are relatively minimal and there is little or no core area. As the woodland cover reaches 30% small woods clump together to form larger woods, ecological isolation is reduced as patches start to coalesce and edge habitat becomes substantial. As 60% cover is reached, edge habitats have reached their maximum, core area increases rapidly and woodland forms the matrix within which other habitats sit.

However, recent work by Watts and Griffiths (2004) in relation to constructing habitat networks in Wales has shown that these land cover thresholds do not generally hold true when different types of random landscapes are used (eg fractal v aggregated). In addition, when "real" landscapes are modelled the threshold values do not have any meaning and are entirely dependent on initial landscape structure.

Summary Box

Spatial targeting and landscape thresholds

- focuses on the protection, enhancement and creation of particular structures/habitats as surrogates for biodiversity value
- threshold values (amount, patch size etc.) are set for the structure or habitat within the landscape
- approach has been used to help prioritise areas for locating new woodland or semi-natural grassland
- criteria used to prioritise areas are invariably insufficient to allow effective prioritisation and bear no relevance to species requirements such as dispersal

6.3 Spatial targeting and the development of ecological networks, habitat networks and greenways

The concept of ecological networks forms a specific set of spatially targeted measures for reversing habitat fragmentation through modifying landscape structure. **Ecological networks** provide a framework of ecological components which comprise the structural elements of a landscape which are deemed necessary to maintain biological and landscape diversity. These include:

1. **Core areas/habitat patches** secure stable, permanent habitat for native species. These comprise remnants of natural or semi-natural areas.
2. **Buffer and development areas** surround the core areas which prevent negative impacts from intensively used landscapes upon the core areas. Buffer and development areas can have conservation value in themselves, or may have the potential to develop towards semi-natural habitat.
3. **Connectivity elements** are areas which ensure or facilitate genetic exchange among the populations of animals and plants in the core areas, as well as migration, dispersal and re-colonization processes. These can be 'stepping stones' or corridors.

In a similar fashion, the development of the **greenway** concept is based around the principle that the identification of key structural components of the landscape will assist environmental functions, such as species dispersal and hydrological processes (Ahern, 1995; Barker, 1997; Hawkins and Selman, 2002; Smith and Helmund, 1993; Thorne, 1993). However, an important quality of the greenway is that it is essentially a multi-benefit device and, whilst the initial motivation may be ecological, it also supports other objectives such as recreation, visual appreciation, scenic highways and pollution buffering. Jongman and Pungetti (2004a) suggest that although ecological networks and greenways show a distinction in focus they show a similarity in concept and structure.

> *"While greenways came initially from the need to create connections and paths for people to access the American countryside, ecological networks came from the need to conserve European species and habitats. In their later stages, however, the two concepts have come closer, having both been recently recognised as fundamental frameworks for the survival and movement of species populations, including humans" (Jongman and Pungetti, 2004a, p.4).*

There have been a number of approaches to taking forward the concepts of ecological networks in the UK. For example, Barker (1997) illustrates an approach to the design of multifunctional networks within an urban setting. The focus is on providing green space and landscape and connectivity for both people and wildlife. Taking a more wildlife-focused view Quadrat Scotland (2002) produced an inventory of potential corridors and stepping stones for biodiversity in the East Dumbartonshire local authority area. They recommended a set of priority actions including the protection and buffering of existing corridors and features, closing gaps in the network and creating new links where needed. However, no attempt was made to link any of the identified features to the needs of specific groups or species. The authors make the intriguing statement that strategic corridors are:

> *"… important for their linkage value to the wider environment and not necessarily for their intrinsic ecological value."*

A lot of recent work on ecological networks has tended to focus on forest and woodland habitats. Fragmentation of woodland is regarded as a particular problem for biodiversity conservation within the UK (Peterken, 2003) even though many woodland fragments have been protected by considerable site-scale conservation measures. Recent increases in woodland area (from a low of 4% of land area in 1900 to 11.7% today – Forestry Commission, 2004) are not thought to have ameliorated the effects of habitat fragmentation for two reasons. Firstly, new woodland may not have been located in the right places to improve functional connectivity, and secondly the woodland may be of insufficient quality to provide habitat for ecologically exacting woodland species (Peterken, 2002).

Early work by Peterken *et al*. (1995) and Hampson and Peterken (1998) has led to the adoption throughout Britain of the Forest Habitat Network (FHN) approach as a means of targeting native woodland creation and expansion. This is being achieved through implementation of the Country Forestry Strategies (Anon, 2000), publicity (Fowler and Stiven, 2003) and incentives (eg SFGS – Anon, 2003). Development of the national network is being followed up by local studies to apply guiding principles tailored to local biophysical conditions and existing woodland cover, for example the Cairngorms (Ratcliffe *et al*., 1998), the Clyde Valley (Peterken, 1999) and Highland Perthshire (Worrell *et al*., 2003).

The long-term goal of FHNs strategies is to enlarge and reconnect woodland habitats without the need for a large-scale expansion of woodland (Peterken, 2003; Peterken *et al*., 1995). Network proposals are often based on two of the basic structure elements of the landscape mentioned earlier: **core areas (**or **nodes)** and **linkages**. Core areas are retained, expanded and developed within existing clusters, while linear woodlands are developed into linkages to connect these core areas. Although it is emphasised that woodland should not take precedence over other scarce or important habitats (Peterken, 2002). Inherent in these cases is the assumption that a FHN as a simple network of woodland habitat patches will facilitate the dispersal of <u>all</u> woodland species. This leads to general principles interpreted as simple rules for woodland patch design. Generalisation will tend to mean that some species will benefit from the policy while others will not.

Summary Box

Spatial targeting and ecological networks

- ecological networks are a specific set of spatially targeted measures for reversing habitat fragmentation through modifying landscape structure
- basic structural elements of networks include: core patches, buffer areas and connectivity elements
- often seen as having multiple benefits i.e. improving environment for both people wildlife
- the focus to date in the UK has been on designing forest habitat networks following a simple set of untested assumptions about improving connectivity for species

6.4 Problems with approaches based on landscape structure

One of the main functions of spatial targeting, structure thresholds, ecological networks and greenways is to protect and enhance biodiversity (Verboom and Pouwels, 2004). However, a significant deficiency of this approach has conventionally been its tendency to produce a single optimal network design, principally based upon landscape structure without reference to how different species or species groups might use the network (Hawkins and Selman, 2002). Functionality needs to be linked specifically to known species requirements (Lindenmayer *et al.*, 2002). Different species and habitats have different requirements and hence those occurring within an area would be expected to be affected by landscape changes in different ways; some may be adversely affected, some may benefit while others may remain unaffected. Hence it will generally not be possible to provide one simple statement as to whether a particular change will be good, bad or indifferent to the totality of biodiversity occurring within an area. The potential impact will vary depending on the species, groups of species or habitats under consideration (Bolck *et al.*, 2004; Opdam, 2002; Opdam *et al.*, 2003; Ratcliffe *et al.*, 1998; van Rooij *et al.*, 2001; Verboom *et al.*, 2001). As (Hawkins and Selman, 2002) p.214) state

> *"one of most serious practical difficulties facing landscape ecologists when advising on the re-design of landscape elements is that there is no single optimum design that suits 'biodiversity' generally, as each species has distinctive spatial requirements".*

7 SPECIES-BASED APPROACHES TO LANDSCAPE EVALUATION

7.1 Types of species-based modelling approaches

For the purposes of this review species-based modelling approaches to landscape evaluation are classified into 4 categories.

(1) Habitat suitability modelling for selected species;
(2) metapopulation models;
(3) focal species modelling;
(4) spatially-explicit population modelling. Short reviews of these approaches are presented together with illustrations of recent applications relevant to lowland agricultural habitats.

7.2 Habitat suitability modelling

Habitat suitability modelling (HSM) is an empirically-based approach which involves constructing relationships between measurable habitat variables (eg patch size) and species occurrence. There are numerous examples of approaches to constructing suitability models for different species (Bender *et al.*, 1996; Engler *et al.*, 2004; Gibson *et al.*, 2004; Tamis and VantZelfde, 1998). HSM has some similarities to the landscape metrics concept, but differs in that the focus in usually on single species (usually wide ranging – Donovan *et al.*, 1987) and there is an attempt to ensure that the measured habitat variables have actual ecological relevance. The classic analytical approach is to use data for a wide suite of habitat variables and relate these to species occurrence through multivariate analysis and general linear modelling techniques. Significant relationships established are then used to determine the most important habitat variables for predicting occurrence. In turn, these are used to model suitability for the species across the landscape (Weyrauch and Grubb, 2004).

The advantage of HSM is that it can give relatively quick estimates of landscape suitably for specific species over quite large areas. For example, Eyre *et al.* (2004) used land cover data to predict the occurrence of carabids (ground beetles). In a lowland agricultural mosaic landscape in north Wales Cowley *et al.* (2000) found good correlations between habitat variables and the occurrence of butterflies and day-flying moths over a 35km^2 area.

In a river catchment in central Scotland devoted to agriculture, Dennis *et al.* (in press) estimated the relative effects of agricultural management compared with landcover patterns, soils, and physiography on the distribution of farmland biodiversity. Data from national spatial datasets, farm management surveys and stratified sampling of habitats and species were collated in a GIS. Multiple regression analysis was used to identify the associations between environmental or management variables and the distribution of species diversity of five different groups of biota (vascular plants; cryptogams; ground beetles; spiders; farmland birds). The multiple regression procedure produced six models, one for each group of biota and one for combined taxa, containing from three to five variables in each to provide an estimate of species numbers. All models included physiographic, land cover and management variables, but they differed amongst species groups. Spiders were positively correlated with altitude, in contrast to summer birds which were inversely correlated with the number of pesticide inputs and soil phosphorus concentration. The results highlighted the importance of land cover pattern and physiography in maintaining species in the current agricultural landscape and illustrate the value of extending the HSM method to a range of species groups and including a number of management and other ecologically relevant variables.

Despite its popularity as a technique, HSM does have some significant drawbacks. Firstly collinearity between explanatory (habitat) variables and spatial autocorrelation can hamper the detection of key environmental factors affecting species-habitat relationships (Heikkinen *et al.*, 2004). Although, this problem can be addressed to some extent by new statistical approaches (Heikkinen *et al.*, 2004) there is the underlying assumption that correct variables have been included in the model at the outset.

Other problems associated with the HSM approach are that no account is taken of species dispersal ecology or population dynamics nor is account taken of temporal changes in habitat suitability. Focal species modelling approaches tackle the first of these shortcomings, but there has been little work addressing the integration of HSMs with landscape dynamics and temporal changes in habitat suitability. To address this concern, Hope and Humphrey (2004) piloted a method of integrating a habitat suitability model for a rare lichen with a forest landscape dynamics model. Although the integrative techniques used in this study are of relevance, the approach to modelling landscape dynamics was developed for a semi-natural upland landscape, and would have to be significantly adapted to be applicable to lowland managed landscapes.

Summary Box

Habitat suitability modelling

- uses an empirical approach that establishes relationships between measurable habitat variables (eg patch size) and species occurrence.
- is useful in that it can give relatively quick estimates of landscape suitability for a range of specific species over quite large areas.
- disadvantages are that collinearity between explanatory variables and spatial autocorrelation can hamper detection of key environmental factors affecting species-habitat relationships
- does not take into account species dispersal ecology, population dynamics or temporal changes in habitat suitability

7.3 Metapopulation modelling

Metapopulation models (Hanski, 1999; Hanski and Gilpin, 1997; Levins, 1969, 1970) play an increasingly important role in landscape ecology and the study of habitat fragmentation. Levins (1969) first used the term metapopulation to describe a population of populations which are actively in contact with each other. One of the key factors which determines whether or not species form metapopulations is the underlying structure of the landscape. Landscapes supporting metapopulations are typically mosaic-like where discrete patches of habitat are separated by a matrix of non-habitat (Lee *et al.*, 2001; Ovaskainen and Hanski, 2004).

The metapopulation concept assumes that essential life-cycle processes operate between these dynamically linked sub populations, with the risk of local extinction and the probability of re-colonisation depending on the ability to maintain an exchange of individuals. When populations living in a heterogeneous environment become isolated by hostile or less favourable conditions, contact between them is ensured only by emigration or immigration. Within a metapopulation sub populations may undergo periodic extinction and colonisation, while the metapopulation as a whole persists.

The three main factors thought to govern persistence of metapopulations are: patch size and shape; patch isolation and adjacent land-use (Andren, 1994). Within the UK, many species with a formerly continuous distribution are being turned into possible metapopulations by habitat fragmentation acting through these three main factors (Peterken, 2002). The subsequent isolation of these fragmented populations increases the probability of local extinction on small habitat patches, and reduces the exchange of individuals on isolated patches. In tandem, modification of the matrix (adjacent land use) has reduced the ability of species to disperse between areas of suitable habitat (Lee *et al.*, 2001; Peterken, 2002). Metapopulation models are becoming increasingly important in understanding the dynamics of such fragmented populations, and extremely useful when applied to biodiversity conservation in a fragmented environment (Verheyen *et al.*, 2004). Hanski and Gilpin (1991) strongly emphasise the importance of metapopulation models to future biodiversity conservation strategies:

> *Metapopulation ideas have become vogue in conservation biology, and with most environments becoming increasingly fragmented, it seems clear that much of the metapopulation research in the future will be motivated by and applied to conservation biology (Hanski and Gilpin, 1991 – p.13).*

However, it is important to note that not all patchy or spatially structured species populations necessarily form metapopulations (Freckleton and Watkinson, 2003; Pannell and Obbard, 2003). Some patchy populations (eg of plants) may be able to persist under conditions of very low rates of colonisation and extinction (Freckleton and Watkinson, 2003). In contrast patchy populations may also exist where there is considerable movement of individuals between patches (Ovaskainen and Hanski, 2004). Although these distinctions may appear academic, they have important implications for how defragmentation efforts are targeted (Pannell and Obbard, 2003). For example in patchy populations with zero or low rates of colonisation (often termed **sink** populations (Dolman and Fuller, 2003), the focus might be on buffering existing habitat patches. For **classic metapopulations**, the focus would be on conserving existing habitat and strengthening links between habitat patches either by creating stepping stones, corridors, or modifying the intervening matrix. In patchy populations with high turnover and exchange of individuals (**source** populations – Dolman and Fuller, 2003) the focus might be on providing additional habitat, without being particularly concerned about conserving existing habitats, modifying the matrix, or improving spatial contiguity.

Clearly in any particular landscape, it is important to identify the spatial and dynamic properties of the species populations of key concern, in order to inform the approach to network construction (Frank, 2004; Henle *et al.*, 2004b; Hudgens and Haddad, 2003). Table 3 shows an approach (Vos *et al.*, 2001; Watts *et al.*, 2004) which translates these principles into simple criteria for assessing the risk of extinction due to habitat fragmentation for different species.

Practical applications of metapopulation modelling often use Population Viability Analysis (PVA) to determine the impact of differing management scenarios on likelihood of extinction (Hope, 2003). PVAs are closely linked to detailed studies of metapopulations and model changes in population size by using a range of measures such as mortality, fecundity, dispersal etc. estimated at the population level. For example, tree colonisation models have been constructed which incorporate empirical data on seed dispersal (Greene *et al.*, 2004). Also, Akçakaya and Atwood (1997) used a commercial metapopulation modelling package, RAMAS GIS (Akçakaya, 1994) to model the metapopulation dynamics and risk of extinction for the Californian gnatcatcher.

Table 3 Ecological profiles of species illustrating varying sensitivity to fragmentation

	Short dispersal distance	Medium dispersal distance	Large dispersal distance
Large area requirement	High extinction risk low colonisation; highest fragmentation sensitivity		High extinction –good colonisation; medium fragmentation sensitivity
Medium area requirement	Moderate extinction – medium colonisation	Medium fragmentation sensitivity	
Small area requirement	Low extinction – low colonisation Medium fragmentation sensitivity		Low extinction – good colonisation Lowest fragmentation sensitivity

Summary Box

Metapopulations

- are populations of populations which are in active contact with each other, eg through species dispersal and migration
- occur naturally but are also created by habitat fragmentation
- metapopulation models are important in understanding the dynamics of fragmented species populations
- metapopulations vary in their spatial and dynamic properties and these properties need to be understood when designing networks for different species

7.4 Focal species modelling approach

The adoption of the metapopulation approach into species ecology has spawned a number of species-based modelling approaches. The **Focal species** concept builds on the idea of umbrella/flagship species (Simberloff, 1998) whose requirements are believed to encapsulate the needs of other species and ecological processes (Lambeck, 1997). Rather than promoting a narrow species-based approach, focal species are designed to represent various habitat types and particular ecological processes and vary in their sensitivity to habitat modification and fragmentation (eg – Bolck *et al.*, 2004). According to underlying theory, the sensitivity of a species to habitat fragmentation is basically linked to their area requirements, dispersal ability and sensitivity of dispersal characteristics to differing matrix quality (Vos *et al.*, 2001), (Table 3).

One of the essential elements of the focal species approach (and in individual modelling approaches – see section 7.5) is the need to model connectivity. Connectivity is expressed in terms of the ability of species to move or disperse between areas of suitable habitat. It is becoming accepted that the surrounding matrix has a significant impact on connectivity for many woodland species. Semi-natural and extensive habitats are considered to be more conducive to species movement, whereas, intensive land uses are predicted to reduce connectivity and increase ecological isolation (Peterken, 2002). The relative ecological permeability of intervening habitats is based around the degree of modification and structural diversity (Table 4).

Permeability is expressed in terms of 'ecological cost' which represents the probability of movement through the surrounding landscape matrix (Adriaensen *et al.*, 2003; Chardon *et al.*, 2003; Sutcliffe *et al.*, 2003).

It will always be important to ensure that an appropriate range of species are considered in the spatial planning process – not only with regard to the scale of the requirements of different biota but also with regard to trying to cover a range of roles, functions and interactions within the landscapes and ecosystems under consideration.

Opdam *et al.* (2003) stress that focal species, or their ecological profiles, should be regarded as part of the evaluation toolkit and not direct targets in themselves. Therefore in many cases, it may be desirable, or at least necessary, to create a number of **Generic Focal Species (GFS)** profiles (Table 4) in order to reinforce the focus on landscape processes, and to represent the bulk of species for which insufficient autecological knowledge exists, rather than focus on single species conservation.

In contrast to ecological networks and greenways, the focal species approach does not advocate an optimal landscape design. The approach is intended to act as an aid to integrated landscape planning by assessing the relative merits of a landscape for particular representative species. The focal species approach has received much attention being easy to understand and readily applicable to management problems. (eg – Brooker, 2002; Gaston *et al.*, 2002; Kintsch and Urban, 2002; Lambeck, 2002; Lindenmayer and Fischer, 2002; Noss *et al.*, 2002).

Recognising the shortcomings in the original FHN approach (section 6.3), Ratcliffe *et al.* (1998) in the development of a forest habitat network for the Cairngorms recognised the importance of using focal species. They stress that:

> *"it is difficult to proceed beyond the theoretical and general in making use of surrogates, or by applying our knowledge of fragmentation, without considering at least some individual species".*

Table 4 Example of ecological profiles for generic focal species. The value of the profile parameters can be varied to test effects on model outputs

Profiles	**Woodland specialist dispersal limited**	**Woodland specialist mobile**	**Agricultural habitat specialist dispersal limited**	**Agricultural habitat specialist mobile**
Area requirements	10ha	10ha	10ha	10ha
Max dispersal distance	1km	5km	1km	5km
Matrix types + Costs	Cost = Resistance to dispersal			
semi-natural broadleaved woodland	1	1	20	10
planted coniferous woodland	1	1	20	10
acid grassland	20	20	1	1
bracken	10	10	10	10

The focus of this study is clearly on the development of a functional, species-based habitat network strategy rather than an ecological network based on landscape structure principles. However, their study revealed a considerable shortage of information for many important species, particularly in terms of habitat requirements and dispersal ability.

Buckley and Fraser (1998) also adopted a focal species approach in order to examine the implications of new woodland planting strategies. They assessed strategies based on random, envelope, buffering and linking planting strategies in four contrasting regions of lowland England. JIGSAW challenge funding has provided the financial incentive to link fragmented woodlands in England. JIGSAW aims to reverse the fragmentation of existing native woodlands to ensure their long-term survival and conserve priority species (Forestry Commission, 2001).

The focal species approach is particularly applicable to large areas. For example Bruinderink *et al.* (2003) describe a network analysis of the Netherlands, Belgium, and adjacent parts of France and Germany, performed with the LARCH landscape ecology model. The aim was to identify the structure of the ecological network for red deer and the spatial connectivity of the landscape. The resulting maps show areas that could support viable populations and indicate habitat areas that will support persistent populations if they are in a network of linked habitats The results of the analysis were used to inform policy decisions on nature conservation and spatial planning.

Based on the LARCH modelling principles, a number of focal species modelling studies have been undertaken in Britain. Van Rooij *et al.* (2004) designed a long-term vision for a woodland ecological network in Cheshire based on the requirements of three focal species. Watts *et al.* (2004) used a combination of different generic focal species to identify different types of woodland habitat network in Wales; Humphrey *et al.* (2004a) evaluated the effects of woodland expansion on both woodland and open-ground focal species in two upland case-study areas. Finally, Ray *et al.* (2004a) used both generic and specific focal species modelling to help target the location of new woodland in two contrasting landscapes in southern Scotland. This suite of studies has served to demonstrate the practicality of the approach, and its usefulness in aiding the strategic planning process.

Similarly, in a recent review of the use of the focal species approach in agricultural landscapes in Australia, Freudenberger and Brooker (2004) concluded that the approach made a useful contribution to setting nature conservation priorities in that it provided explicit recommendations (eg in relation to threatened birds of shrubland) rather than general principles with no spatial context.

There are a number of inherent assumptions within the focal-species modelling approach, such as habitat preferences, area requirements, dispersal distance and matrix permeability which undoubtedly have an impact on the model outputs. Perhaps the biggest assumption is the extent to which individual species can act as umbrellas or surrogates for others. In a critique of the focal species approach Lindenmayer and Fischer (2002) stressed the need to test ability of one taxonomic group to provide guidelines for another and to assess the relative importance of managing the ecosystem process in the landscape against managing for the persistence of species in the landscape.

As focal species do not represent "real populations" it is difficult to test model outputs (Freudenberger and Brooker, 2004; Lindenmayer *et al.*, 2002; Melbourne *et al.*, 2004). For example, if field survey revealed that species were not present in the habitat they were predicted to occur in would this be a failure of the model, or simply an indication of poor understanding of the habitat requirements of the species? Melbourne *et al*. (2004) suggest that the best way of testing the models is to run a range of simulations to test the sensitivity of the model parameters to changes in value. In conclusion, it is important to consider focal species modelling as only one of a suite of evaluation tools available to support management action, rather than a tool to model and predict actual species dispersal and viability.

Summary Box

Focal species modelling

- focal species are those which are considered to encapsulate the needs of other species and ecological processes
- species are selected which represent a range of habitat types, processes and sensitivities to fragmentation
- both species habitat requirements and dispersal abilities are taken into account in the modelling process
- least cost modelling is used as a measure of connectivity – this allows the effect of the intervening matrix on connectivity to be evaluated
- the approach has been used extensively as tool for aiding spatial planning for conservation – a range of different landscape scenarios can be evaluated
- shortcomings of the approach include – difficulty in testing model outputs, simplistic representation of species attributes and lack of consideration of landscape and species population dynamics

7.5 Spatially-explicit population modelling

Like metapopulation models, spatially explicit population models (SEPMs – Dunning and Stewart, 1995) take into account the birth, mortality and/or movement of individuals within landscapes (Murrell and Law, 2000). However, in contrast to metapopulation models no assumptions need to be made about the population structure of the species in question (Rushton *et al*., 1997). Generally, SEPMs model population dynamics directly on a spatially explicit representation of the landscape (usually raster grid or hexagonal tessellation) whereas GIS-linked metapopulation models derive a connected graph (a set of vertices connected by edges) structure from a habitat suitability map, then model population dynamics according to the graph structure (Vos *et al*., 2002). Additionally, SEPMs may be individual based rather than population-based models, ie the location of each individual of the target species is explicitly modelled. For a review of individual-based species models see (Melbourne *et al*., 2004).

The advantage of SEPMs over HSM and focal species modelling approaches is that they are in theory a better reflection of reality, since they include a wide range of parameters many of which are obtained from field observation and experiment. There are a range of variants of SEPMs and Stephens *et al*.(2003) have reviewed the usefulness of different models in predicting the response of farmland bird populations to changing food supplies. They concluded that these types of modelling techniques have outstripped availability of data required to parameterise them. This is particularly true when large scale analyses are required and land cover and species data may not be available in sufficient detail. Nevertheless Stephens *et al.* (2003) recommend use of models that incorporate parameters such as food availability for evaluating responses of individual species to agriculture, and stress the need to obtain more data. This highlights one of the main disadvantages of the SEPM approach in that it is very "data-hungry" and can only be conducted on single species thus limiting its use in large-scale multi-species studies (Vos *et al.*, 2001). However, research is underway in Scotland to construct better spatially explicit models of biodiversity (see Appendix 1).

Summary Box

Spatially explicit population models

- take account of birth, mortality and/or movement of individuals within landscape as a means of predicting population change and persistence
- model population dynamics directly on a spatially explicit representation of the landscape
- take account of landscape and population dynamics
- Are a better reflection of reality and should have improved predictive power over focal species modelling and habitat suitability modelling
- Require large amounts of detailed habitat and species data for model parameterisation
- Have limited application in large-scale studies or where modelling is required for a range of species

7.6 Conclusions – integrating species and structure approaches to evaluating landscapes

Based on this short review of different approaches to evaluating landscapes for biodiversity, there appears to be clear merit in integrating structural and species-based methods in the spatial planning process. Currently landscape planning focuses on making changes to structure, either through expansion or restoration of priority habitats (Gkaraveli *et al.*, 2004) and/or by increasing connectedness between priority habitats (Fowler and Stiven, 2003). Despite the rapid development of species-based approaches, there are few examples of where landscape-scale changes are driven primarily by species requirements. Although landscape-scale approaches to species conservation are being pursued for a small number of priority species (eg capercaillie – Kortland, 2003) the focus is still on improving the quality of individual habitat patches rather than thinking in terms of developing species-calibrated networks.

Spatially explicit population models provide useful tools for detailed analysis of landscapes, but are limited by data availability and by scale. For example, there is little point in developing complex models whose functioning relies on detailed land cover data when those data are not available. Focal species modelling sits mid-way on the modelling continuum between simple structure-based models and detailed species-based models. In this respect it offers a practical approach applicable at a range of scales based on robust theoretical assumptions. Recent work has shown that it is possible to develop network modelling tools which are sufficiently flexible to incorporate both structure (Humphrey *et al.*, 2004a) and species (Watts *et al.*, 2004) approaches. In the next section we describe one such network modelling tool, BEETLE (The Biological and Environmental Evaluation Tool for Landscape Ecology).

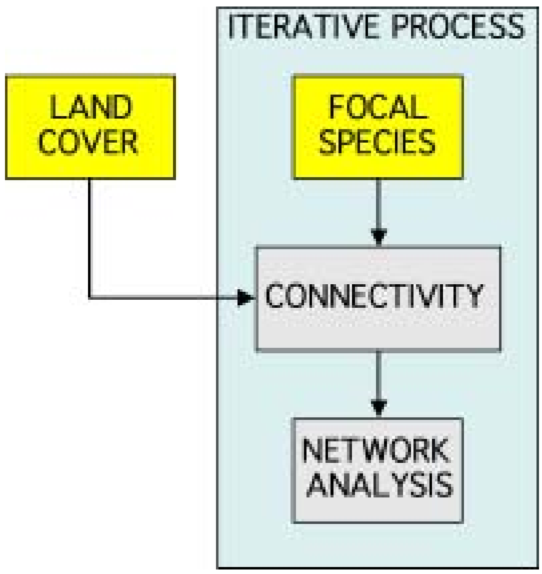
8 DESCRIPTION OF THE BEETLE MODELLING TOOL

8.1 Outline of BEETLE

The BEETLE model is currently being developed by Forest Research as part of a suite of tools being constructed within FRs Landscape Ecology Project (Ferris *et al*., 2000; Watts, 2003). Instead of using metrics as surrogates for understanding the meaningful processes of species-landscape interactions, BEETLE tests the landscape pattern against ecological profiles for 'focal' species (van Rooij *et al*., 2001). The model runs within ArcView GIS (ESRI) allowing integration of a wide range of land cover data within the modelling process.

BEETLE is implemented through a set of modules that represent and process data, as illustrated in Figure 11. There are two input data elements: a **land cover module** and a **focal species module**. Both modules can be varied and contribute equally to model behaviour. The **connectivity module** models the interaction between land cover and focal species. This module outputs areas which are considered as habitat and indicates the probability of movement across the landscape. This analysis then allows the **network analysis module** to identify habitat patches within functional networks, within an iterative environment.

Figure 11 BEETLE modelling process



8.2 Land cover and focal species modules

A wide range of land cover data can be used to construct the land cover module such as woodland cover (eg National Inventory of Woodland and Trees – Forestry Commission, 2002), remote sensed data (eg Land Cover Map 2000 – Fuller *et al*., 2001; Land Cover Scotland 1988 – MLURI, 1993) or habitat information obtained from ground survey such as Phase 1 (NCC, 1990) or NVC (Rodwell, 1991). The land cover data are amalgamated into one layer which is then classified in terms of habitat preference and permeability (to dispersal) for the focal species selected for analysis.

The use of focal species is fundamental to the BEETLE approach in exploring habitat networks. As discussed in section 7.4 either real or generic focal species can be used in the modelling process. Given the problems with availability of autecological information for the number of specific species needed to fully test the range of landscape structures and processes possible in spatial analyses, the use of generic focal species (GFS) has been favoured (Ray *et al.*, 2004a; Watts *et al.*, 2004). The GFS are defined to be representative of a number of species groups, priority habitats and key ecological processes. GFS profiles allow an iterative analysis to be carried out which aids understanding of the landscape from various species perspectives and enables the exploration of the range of potential networks. The parameters that control the focal species module include: habitat preference; habitat area requirements; dispersal preferences (permeability); and dispersal distance. Generic species are selected which represent a range of values within these parameters (eg see Table 4).

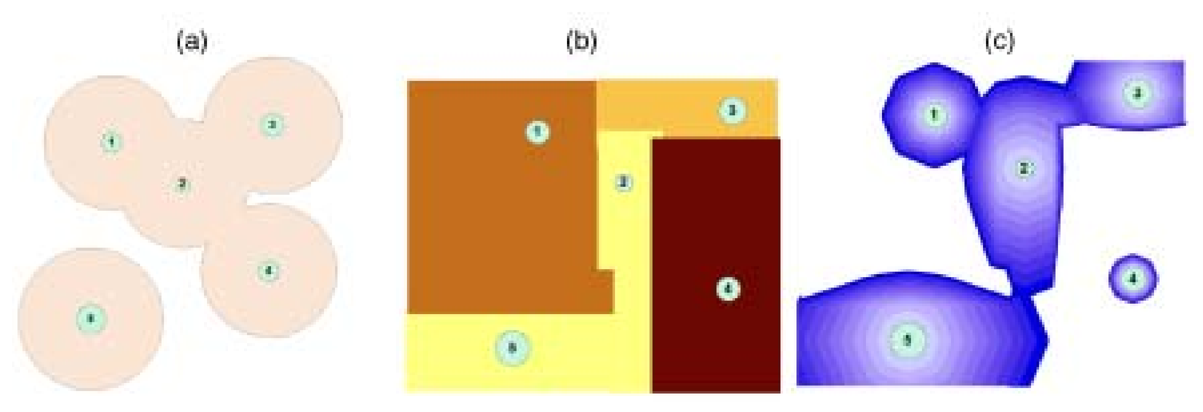
8.3 Connectivity module

In order to assess ecological isolation, one of the fundamental threats to biodiversity from habitat fragmentation, it is necessary to model connectivity. Connectivity is a functional attribute of the landscape related to an ecological process, as opposed to connectedness which is based on physical distance.

In this context, connectivity is modelled on the dispersal ability of a focal species and the ease of movement through the surrounding landscape (as discussed in section 7.4). It is becoming accepted that the surrounding matrix has a significant impact on connectivity for many woodland species (Ricketts, 2001). Semi-natural and extensive habitats are considered to be more conducive, or permeable to species movement, whereas intensive land uses are predicted to be less permeable, thereby reducing connectivity and increasing ecological isolation (Humphrey *et al.*, 2004a). The ease of movement through, or permeability of, different land cover types is expressed in terms of 'ecological cost'. The connectivity modelling process is illustrated in Figure 12.

Four of the five habitat patches in Figure 12a appear to be connected (1–4), using just the dispersal distance for a focal species but disregarding the permeability of the surrounding landscape. Figure 12b represents the permeability, or ecological cost, of the surrounding landscape based on the potential ease of movement for woodland species. In this illustration the ecological cost ranges from yellow (high permeability), light brown (moderate permeability) to dark brown (low permeability). Figure 12c which takes into account ecological cost, indicates that patch 1,2,3 & 5 are potentially connected. The link between patch 2 and 5 demonstrates the concept of a permeable corridor, whereas patch 4 appears to be isolated within a hostile landscape. Through this process it is possible to have high connectivity in an apparently fragmented landscape, with low connectedness, as long as the wider matrix supports the particular ecological process. For example, the habitat patches may be species-rich grasslands supporting bee populations; the more permeable matrix types could be improved grasslands, whereas the impermeable matrix might be a conifer plantation.

Figure 12 Illustration of connectivity modelling: (a) habitat patches (1–5) buffered (pink); (b) cost surface of matrix – yellow low cost to dark brown high cost; (c) modelled connectivity between patches



8.4 Implementation process

Figure 13 demonstrates the modelling implementation process. Data from the **land cover module** is represented in Figure 13(a). The **focal species module** is used to define the habitat areas, in this example broadleaf woodlands are selected (Figure 13b). This module also provides the necessary data for use in the **connectivity module** to assess landscape permeability, in this example high permeability is illustrated with light colours whilst dark colours signify low permeability (Figure 13c). This allows the identification of potential networks, defined by different colours, which can be examined within the **network analysis module** (Figure 13d).

8.5 Outputs and interpretation

The basic outputs of the modelling process are sets of maps showing networks for the species used in the modelling process. Figure 13 shows a number of networks for a generic woodland species. Landscape quality for each focal species is assessed in terms of the number of networks supported by the landscape, the size of those networks, the size of the habitat patches within each network and the proportion of habitat in large networks. Habitat patches within large networks will be more robust than small isolated patches surrounded by a hostile landscape. Overall "biodiversity value" of the landscape is assessed by comparing network quality across the range of focal species. Often this process will illustrate trade-offs between different species. For example, Figure 14 shows the effect of adding native woodland to a landscape on networks for focal species with differing habitat requirements (Humphrey *et al.*, 2004b). The landscape in question is Glen Affric in northern Scotland where management is encouraging the expansion of pine and birch woodland from a current average cover of 37%–75% over a long time-scale. The modelled expansion of woodland cover had contrasting effects on the species networks. While networks for the woodland specialist species witch's hair lichen (*Alectoria sarmentosa*) increased in number (Figure 14c), networks for the open ground species large heath butterfly (*Coenonymphya tullia*) and the woodland edge species pearl-bordered fritillary (*Boloria euphrosyne*) became smaller and more fragmented (Figure 14).

The outputs generated by the modelling of woodland expansion serve to highlight the differential effects that landscape change can have on networks for different species. Although the model does not provide the answers to effective land-use planning it illustrates the potential consequences of different management options. On the back of the initial outputs shown in Figure 14, (Humphrey *et al.*, 2004b) went on the model the effects of adding woodland to the Glen Affric landscape in a differing spatial configuration (more dispersed woodland blocks) and found that both woodland and open ground species networks could co-exist more effectively than in the initial model run.

Figure 13 Example of the BEETLE modelling approach (Watts *et al.*, 2004): (a) output from the land cover data; (b) core habitat for focal species; (c) permeability of matrix to dispersal; (d) identification of habitat networks

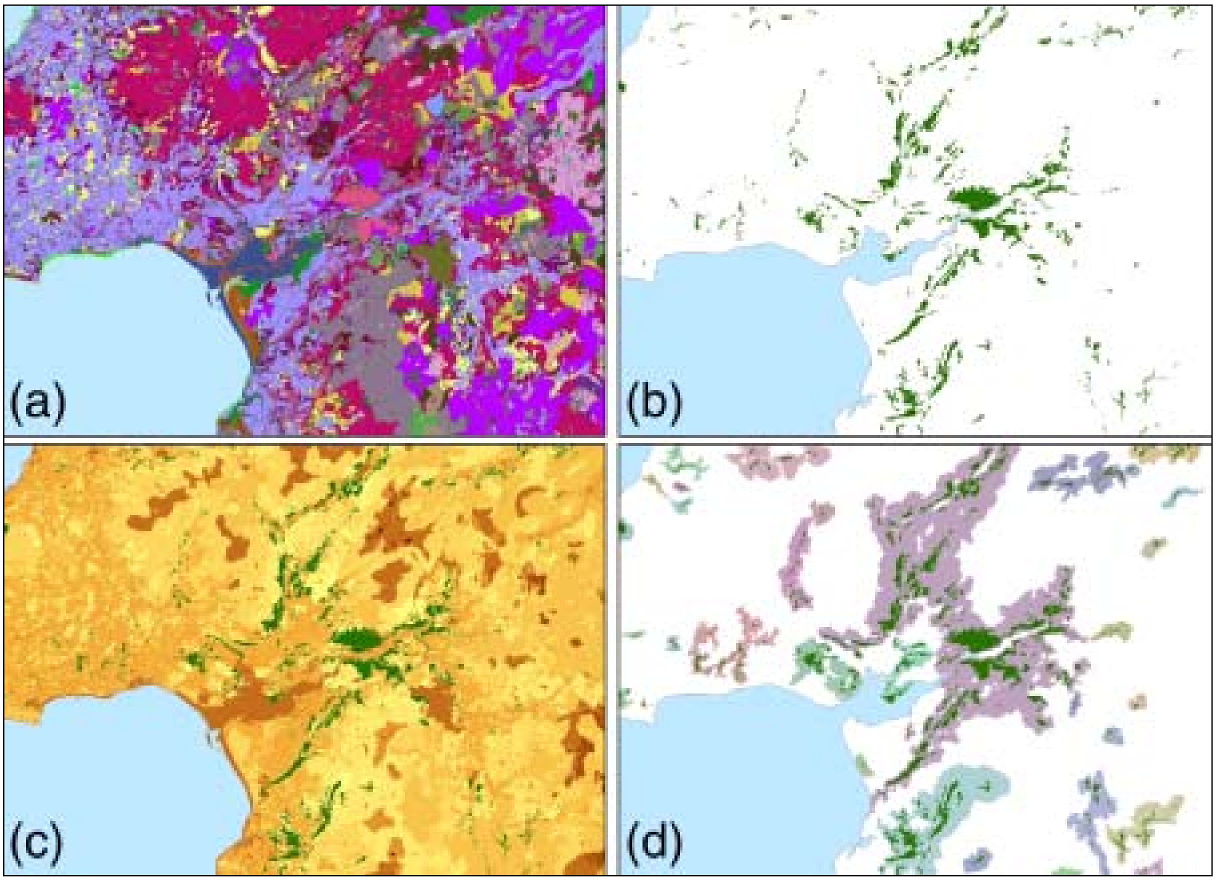
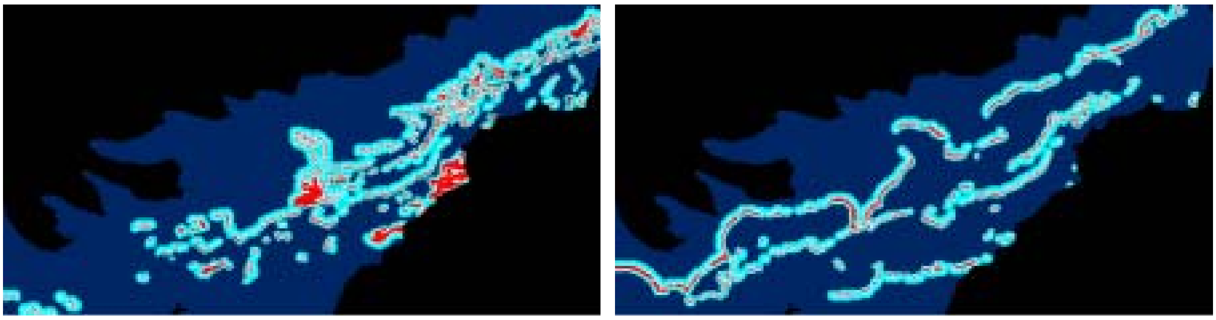


Figure 14 Habitat networks for (a) pearl-bordered fritillary; (b) large heath butterfly and; (c) witch's hair lichen in Glen Affric. Maps on the left show networks at 37% (current) native woodland cover; maps on right show networks at 75% woodland cover. Dark blue = simulation area (200km2); Red = habitat. Maximum dispersal distance = light blue in (a), light green in (b) and yellow in (c). Reproduced from Humphrey *et al*. (2004b)

(a)



(b)

(c)

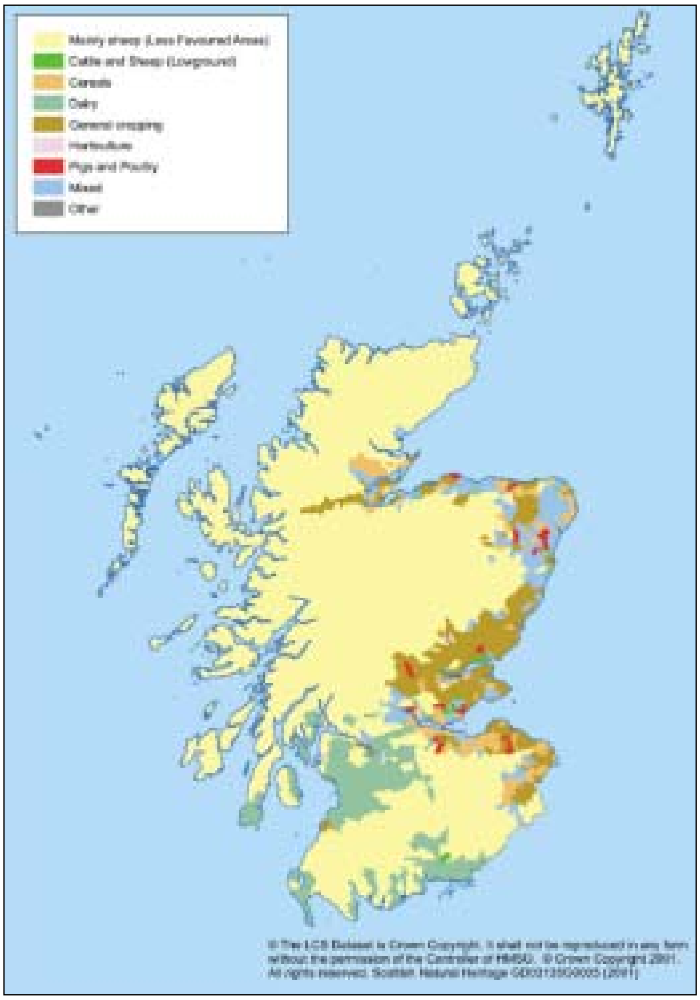
9 SPECIES AND HABITATS THAT WOULD BENEFIT FROM DEVELOPMENT OF NETWORKS IN THE SCOTTISH LOWLANDS

9.1 Ecological character of lowland NHFs

Two-thirds (or nearly 5.2 million ha) of Scotland's land area is classified as agricultural land. Although much of this is upland grassland, heath and bog, over 1.7 million ha consists of the managed and cultivated fields which surround farm steadings and contribute to the characteristic patchwork landscape of lowland Scotland (McCracken and Bain, 2000; Scottish Natural Heritage, 2002b). Since the second world war these arable and grass fields have undergone broadly similar changes to that seen on farmland south of the border and elsewhere in Europe with increasing intensification and loss of habitats and biodiversity (Donald *et al.*, 2000; McGowan *et al.*, 2002; Robinson and Sutherland, 2002).

There has been a move to concentrating on growing crops in the east and grass in the west of the country (McCracken and Bain, 2000), (see Figure 15). In parts of west coast Scotland there has been significant losses of arable land from farms and crofts. Despite this relatively high degree of specialisation, most lowland farmers still retain some mixture of arable and grass on their holdings. On average even those farmers concentrating mainly on arable cropping still have over 25% of their farm under grass, while a typical Scottish dairy farmer will still grow at least one or two fields of cereals.

Figure 15 Distribution of farm types in Scotland. Reproduced from McGowan *et al.* (2002)



9.2 Habitats that should be considered part of lowland habitat network

A number of surviving habitats and structures that are important to retain in landscapes can be identified (see Table 5 for examples). In lowland Scotland (as in the rest of the UK), these may derive from more or less original land cover such as woodland or be the product of human land uses such as field systems and parkland. Human modification of these elements may in turn create variations on the original theme such as wood pasture (Quelch, 2000, 2001). In addition, particular features may be associated with these habitats such as ditches, banks and verges.

Generally, the closer the habitat, variation or feature to any original natural land cover, the more it is held that it should be retained in modern landscapes. However, such is the overlay of human processes, that even highly modified systems such as semi-natural grassland are highly prized in both cultural and ecological terms. Also, whole landscapes may be made up of several such habitats, interdependent and creating a unity which is itself to be valued (Nowicki, in prep). It is, however, the spatial location of these features in the landscape and the overall relationship of one to another which will influence the biodiversity value associated with the habitats involved.

An important ecological characteristic of the lowland farmed landscape is the distinction between habitats and features which are permanent over the medium to long-term and those which are temporary in time and space and have only short term persistence (Table 5). The approach suggested for constructing networks focuses on the permanent habitats (those with medium/long-term persistence), but takes account of the important interactions with the more transient elements. In the following sections we describe a selection of those permanent habitats likely to be of key importance in the construction of habitat networks. The descriptions are based on Ellis and Munro (2004); UKBAP priority habitat action plans (Anon, 1995; UK Biodiversity Group, 1998); Mackintosh *et al*. (2004) and descriptions in the Rural Stewardship Scheme for Scotland (Scottish Executive, 2004).

Table 5 Habitats and features important to include in habitat networks created in lowland agricultural landscapes

Main category	**Subsidiary habitats**					
Woodlands	Semi-natural woodland	Plantations	Orchards	Shelterbelts	Hedges	Individual trees
Wetlands	Lochs	Ponds	Riparian zones (in part)	Lowland bogs	Rush pasture	
	Streams	Canals	Ditches	Water margins	Rivers	
Coastal	Sand dunes	Saltmarsh	Coastal Heathlands	Machair		
Grasslands	Wet grasslands	Meadows	Improved agricultural grasslands	Amenity grassland	Dry grasslands	
Scrubland	Roadside verges	Railside verges	Industrial wasteland	Spoil heaps	Quarries	Woody scrub
Transient habitats	Set-aside	Beetle banks	Grass margins	Intermittently flooded areas	Arable crops	Ley grassland

9.2.1 Unimproved grassland habitats

The following BAP priority grassland habitats have been recorded in Scotland: lowland calcareous grassland; lowland dry acid grassland; lowland meadows; purple moor grass and rush pastures (upland hay meadows occur but are a minor component of lowland landscapes). All of these habitats are thought to have undergone considerable loss and fragmentation over the last 50–60 years. The average size of most unimproved grassland patches in England is less than 2ha (English Nature, 2002); in Scotland the average is around 7ha (J. Mackintosh *pers. comm.*). Management often determines the grassland type, whether grazed (pasture) or managed for fodder (meadow). Underlying geology and soil type are also influential and are usually reflected in the name of the grassland habitat.

Lowland hay meadows and other species-rich grasslands occur typically on well drained and/or unproductive soils. As a consequence, the sward is generally characterised by a colourful variety of relatively low-growing plant species such as bird's-foot-trefoil *(Lotus corniculatus)*, red clover (*Trifolium pratense*), autumn hawkbit (*Leontodon autumnalis*), lady's bedstraw (*Galium verum*) and meadow vetchling (*Lathyrus pratensis*). The main form of unimproved neutral grassland to occur in Scotland is classified as the crested dogstail (*Cynosurus cristatus*) and black knapweed (*Centaurea nigra*) type (Mackintosh *et al.*, 2004).

Such floristically-rich grasslands were once common on farms throughout the UK lowlands. However, a decline in the perceived agricultural value of these meadows and pastures within the context of modern farming systems has led to a marked decline in their quantity and quality over the past 50 years. It is estimated that between 5,000–10,000ha survive in England and Wales with an additional 2,000–3,000ha in Scotland (Mackintosh *et al.*, 2004). The latter are scattered throughout the country but with a particular concentration in the crofting areas of Lochaber, Skye and the Western Isles.

These declines have largely been a result of agricultural improvement involving drainage, reseeding, increased fertiliser application and a shift away from hay-making to silage production. In other instances the abandonment of cutting and grazing has led to these grasslands becoming rank and overgrown and the subsequent encroachment by bracken (*Pteridium aquilinum*) and scrub.

In addition to their importance for plants, these relatively unimproved grasslands support a number of declining or rare birds such as corncrake (*Crex crex*) and skylark (*Aluada arvensis*). Fields shut up for hay or silage making can provide attractive cover for nesting. However, the timing of mowing is an important factor since nests may be destroyed and chicks exposed to predators after an early cut.

Lowland acid grassland typically occurs on nutrient-poor, free-draining soils overlying acid rocks or sand and gravel. This type of grassland contains a range of plant species such as heath bedstraw (*Galium saxatile*), sheep's fescue (*Festuca ovina*), common bent (*Agrostis capillaris*) and tormentil (*Potentilla erecta*). Dwarf shrubs such as heather (*Calluna vulgaris*) and blaeberry (*Vaccinium myrtillus*) are also often present but at low abundance.

Grassland of this type in Scotland occurs mainly in management enclosures associated with the upland fringe and in the coastal regions of the north and west. The extent of these grasslands and variety of vegetation structures in close proximity to one another provide important breeding areas for waders such as lapwing (*Vanellus vanellus*) and curlew (*Numenius arquata*). The abundant populations of small mammals and birds such as meadow pipits (*Anthus pratensis*) and skylark also mean that these grasslands form important feeding and hunting areas for birds of prey such as hen harrier (*Circus cyaneus*) and merlin (*Falco columbari*). Many of the invertebrates that occur in this type of grassland are specialist species which do not occur anywhere else. The open swards on sandy soils in particular can support a considerable number of ground-dwelling and burrowing insects such as solitary bees and wasps.

Surveys of lowland acid grassland have been limited to-date but it is estimated that Scotland contains less than 5,000ha in total (Mackintosh *et al.*, 2004). No figures are available on the rate of loss, but lowland acid grassland is known to have undergone a substantial decline across the UK in the last 25–50 years. Both agricultural intensification and management neglect have resulted in a general depletion of the resource, with overgrazing and the associated sward damage sometimes being a more localised problem.

The objectives and targets under the Lowland Dry Grassland Habitat Action Plan (www.ukbap.org.uk) are similar to those for Lowland Meadows in that they also cover habitat conservation, restoration and expansion. As with all lowland grassland types, fragmentation of the resource is a major concern and the Plan also highlights the need to consider mechanisms whereby lowland acid grassland within common land can be brought under sympathetic management. Research is also required to establish the most appropriate grazing management regimes to implement on such sites.

Covering an area of 8000ha purple moor grass and rush pastures are again typical of marginal upland areas, although non-grazed purple moor grass and rush habitats are common on the mountains of western Scotland. This habitat includes a range of vegetation types dominated by purple moor-grass and tall rush species, mainly on poorly-drained peaty gleys, shallow peats and acidic mineral soils in lowland areas with high rainfall. It is a distinctive type of species-rich fen-meadow and rush pasture vegetation, often found as part of a mosaic with wet heath, dry grassland, swamp and scrub. Other species may include wavy St. John's wort (*Hypericum undulatum*), whorled caraway (*Carum verticillatum*), marsh hawkweed (*Crepis paludosa*), meadowsweet (*Filipendula ulmaria*), globeflower (*Trollius europeaus*) and the lesser butterfly orchid (*Platanthera bifolia*). The habitat supports species such as the marsh fritillary butterfly (*Euphydryas aurinia*), (Fowles, 2003) which feeds on devil's-bit scabious (*Succisa pratensis*).

9.2.2 Species-rich hedgerows

Species-rich hedgerows are rare and concentrated in southern and central lowland areas of Scotland (Hepburn, 2000). However, the resource is thought to have been under-recorded. A species-rich hedge is defined as having four or more native woody species, on average, per 30m stretch (Anon, 1995). Any associated banks, ditches or trees are considered part of the hedge habitat. Hedgerows are like woodland edges in terms of plant species they support (McCollin *et al.*, 2000) containing generalist rather than specialist woodland species. Hedges are particularly important as a source of food and as a habitat structure for small mammals, butterflies, moths, bats and farmland birds including owls (Usher *et al.*, 2000). For example, carabid beetles use hedgerows as places to overwinter, moving into adjacent crops to predate aphids during the early growing season (Joyce *et al.*, 1999). Hedgerows of over 15m in width will attract both farmland and woodland generalist species. For a general review see Maudsley (2000).

9.2.3 Lowland heathland

There seems to be some debate as to whether lowland heath can be separated ecologically from upland heath. The former is thought to form on acid impoverished mineral soils below 300m where heather and other ericoid shrubs cover at least 25%. Ellis and Munro (2004) list the main NVC types associated with the habitat. Dry heathland communities are species-poor being dominated by dwarf shrub, usually heather (*Calluna vulgaris*) and bell heather (*Erica cinerea*). It is estimated that there is around 6000ha of lowland heath occurring in the Borders, Lothian and around Aberdeen. There is increasing evidence that lowland heath plant species are strongly affected by fragmentation and would benefit from the creation of lowland heathland habitat networks (Piessens *et al.*, 2005).

9.2.4 Field margins

Cereal field margins are a UK BAP priority habitat forming strips of land lying between cereal crops and the field boundary, extending for a limited distance into the crop and which are deliberately managed to create conditions to benefit key farmland species. They can be as wide as 6 or 12m. They include 'wildlife strips', 'conservation headlands' and game crops. Cereal fields account for about 44% of arable fields in Scotland. Their margins are usually covered with grass species such as false oat-grass (*Arrhenatherum elatius*) and couch (*Elytrigia repens*). The flora may be characteristic of disturbed margins and may include weed species such as thistles (*Cirsium* spp.) and ragworts (*Senecio* spp.).

Cereal field margins can provide nesting and feeding sites for game birds, passerines and many species of butterfly, grasshopper and plant bugs. A considerable number of insect species breed in crops but spend the winter in grassy banks at the field margins. Excluding soil invertebrates, some 2,000 species of invertebrate are commonly found in cereal fields. Unfortunately, many species of arable flora, such as corn marigold (*Chrysanthemum segetum*) and cornflower (*Centaurea cyanus*) occur extremely rarely and are unlikely to reappear from the seedbank which has been impoverished through intensive land management during the twentieth century. Field margins also form important habitat for small mammals; in one study bank vole populations were found to be much higher in linear field margins than in non-linear farmland and woodland blocks (Tattersall *et al.*, 2002), although habitat quality was found to be more important than shape. Maintaining permanent field margins appears to be of key importance for many species groups, as this allows time for semi-natural character to develop including colonisation of native plants which form food plants for many invertebrates (Hutton and Giller, 2003; Tudor *et al.*, 2004).

Grass margins and beetle banks form a more transient sub-set of field margins and are designed to provide temporary rotational habitat for species such as: grey partridge (*Perdix perdix*), linnet (*Carduelis cannabina*), bullfinch (*Pyrrhula pyrrhula*), spotted flycatcher (*Muscicapa striata*), corn bunting (*Miliaria calandra*), purple ramping-fumitory (*Fumaria purpurea*), cornflower and a range of other invertebrates. Insects using grass margins can over-winter and breed early in the season. This allows them to effect a useful form of biological control by attacking aphid populations in adjacent crops. Grass margins are usually 1.6–6m wide.

9.2.5 Wetlands and riparian zones

Wetland habitats are critical for a variety of flora and fauna including wading birds and aquatic invertebrates. Wetlands can be permanently wet or, as within a flood plain, periodically immersed. Wetlands are very important as breeding and feeding areas for waders, particularly where associated with unimproved pasture. Different species of wading bird require differing levels of water. For example, snipe (*Gallinago gallinago*) need wet conditions to probe for invertebrates while lapwing will inhabit drier areas. Reed buntings (*Emberiza schoeniclus*) will nest in a variety of wetland vegetation types including sedges, rushes and other tall, thick vegetation. Alder, willow and other trees growing around wetlands and watercourses are important in stabilising river banks. Water voles inhabit earth banks alongside open water, ditches, and marshes. Water margins are also a key aquatic habitat. For example, good marginal habitat with tall vegetation and high water quality will benefit bats (Wickramasinghe *et al.*, 2003).

Ditches and drains in arable land often provide some of the richest habitats available in intensively managed areas, although land managers are seldom aware of their wildlife value. These watercourses are usually regarded purely as a means of flood prevention, or as an aid to maintaining optimum soil moisture conditions for crop growth, and their full biodiversity potential is rarely achieved. Important features of the habitat include diversity of water depths, marginal vegetation and the occurrence of mud and bare areas. Ditches and drains can provide important connections between streams and riparian areas. Water courses and riparian zones in general provide important elements of connectivity in the landscape for their dependent wildlife (Peterken, 1999). For example, Petersen *et al.* (2004) found that stream corridors (10–20m either side of the stream) provide the main habitat for aquatic insects (stoneflies, caddis flies and mayflies) and a linear "highway" for adult dispersal.

Intact lowland raised bogs are one of Europe's most threatened habitats (Patterson and Anderson, 2000). They occur in the lowlands of central Scotland and are recognised by gently sloping domes of peat that have accumulated to a depth of many metres over thousands of years. The surface of the bog is raised well above the influence of groundwater so that the vegetation is dependent almost entirely on rain and snow for its source of nutrients. Only plant species specially adapted to live in such waterlogged, nutrient-poor conditions can survive, and this results in a specialised plant community supporting unusual insects. The vegetation is usually dominated by bog mosses, heathers and cotton grasses (*Eriophorum* spp.). Blanket bogs also occur to some extent in the lowlands. Considerable work has been undertaken prioritising methods for restoring bogs (Anderson, 2001) including taking into account landscape context.

9.2.6 Woodland and scrub

This is a broad category that includes ancient semi-natural woodland, recent semi-natural woodland, plantations of native and introduced species, shelterbelts and scrub. The biodiversity value of semi-natural woodland has been well documented (Marren, 1992) as has the value of plantations of introduced species (Humphrey *et al.*, 2003b). There is ample evidence that addition of small woods/shelterbelts will benefit a range of generalist species. For example, in mixed farm landscapes, song thrushes (*Turdus philomelos*) selected field boundaries and woodland as territories avoiding arable crops (Peach *et al.*, 2004). Woodlands, scrub and tall grassland tend to provide more of the moist invertebrate bare soil habitats required for feeding; these being rare in arable fields.

Woodlands can act as barriers to species dispersal (eg the large heath – Hofmann and Marktanner, 1995) and in some instances are known to harbour predators which can increase the risk of predation of open ground species (Parr *et al.*, 1995). In contrast the dispersal of other open ground species is not limited by the occurrence of woodland. For example, bumble bees (*Bombus terrestris* and *B. pascuorum*) appear able to cross 600m wide strips of forest during foraging trips (Kreyer *et al.*, 2004).

9.2.7 improved grassland and arable fields

Excluding rough grazing (both sole right and common), permanent (>5 years old) and temporary (< 5 years old) grassland comprises 19% of the total agricultural land area in Scotland (McCracken and Tallowin, In press). Lowland grasslands can be utilised by a range of bird species. For example, moist grassland soils are especially important for waders such as lapwing, snipe, curlew and redshank (*Tringa totanus*) and passerines such as starling (*Sturnus vulgaris*). These rely on invertebrates such as earthworms and leatherjackets in the soil, beetle adults and larvae on the soil-surface and sawfly larvae and plant bugs on the vegetation (eg – Barker, 2004; Holland, 2004). In addition, grass seeds are utilised by a number of bird species such as starling, house sparrow (*Passer domesticus*) and yellowhammer (*Emberiza citrinella*) while the seeds of broad-leaved plants in the sward are consumed by the adults of other species such as skylark, greenfinch (*Carduelis chloris*) and linnet. However, there is now strong evidence that habitat quality for farmland birds has declined markedly throughout grassland dominated landscapes (Robinson *et al.*, 2001; Chamberlain and Fuller, 2001). Changes in the populations of farmland birds appear to be linked to large-scale temporal changes in invertebrate numbers and seed resources (Vickery *et al.*, 2001) and especially the loss of ecological heterogeneity at multiple spatial and temporal scales caused by agricultural intensification (Benton *et al.*, 2003).

Arable farmland can consist of small individual fields surrounded by grassland and other habitats (as in many areas of the north and west of Scotland) to large areas dominated by a wide range of different crop types (as in the productive areas of the north-east and south-east). Across Scotland as a whole, about 0.6 million ha (or 13% of the total agricultural land) is under arable crops, excluding rotational grassland, each year. Although the area grown has decreased somewhat in recent years, barley is still the major crop grown on Scottish farms. In addition, the move to autumn-sown cereals has not occurred to anything like the same extent as in southern Britain. Over half of the Scottish cereal crop is still planted in the spring although there is increasing economic pressure on farmers to shift towards autumn planting. Spring sowing enables the retention of stubble fields which are essential for birds to forage in during each winter, and the later harvest dates associated with spring crops means that many late-breeding birds are able to raise their young successfully (McCracken and Bain, 2000). However, arable and cereal crop production in Scotland has by and large followed the general trend of mechanisation and intensification as exhibited in the remainder of the UK. In particular, there has been a reduction in the rotation of crops with other land covers and an associated decline in the practice of undersowing cereal crops to produce a grass ley after harvest. As a consequence, there has been a substantial reduction in the attractiveness of arable crops to a wide-range of plant, invertebrate, mammal and bird species. For example, the main factor responsible for the decline of grey partridge across the UK has been identified as reduced food supplies for chicks caused by the use of insecticides and herbicides.

Improved grassland and/or arable crops constitute the dominant features in most if not all lowland agricultural landscapes in Scotland. Although there can be a wide range of variation in the characteristics of individual fields, the majority are subject to rather intensive management and relatively uninteresting from a biodiversity perspective. For example, Cole *et al.* (in press) found that intensively managed grasslands and arable crops were very similar in species assemblage composition for both ground beetles and spiders across a range of farms in central Scotland. However, their dominance of the landscapes coupled with their biodiversity potential means that it is important to consider the impact such habitats could make to a network. It is, however, important to remember that such fields are often very dynamic, with management practised on any one area of farmland changing constantly with time. Such changes may be marked (in terms of the move from one landcover to another) or more subtle (in terms of differences in timing and/or intensity of grazing pressure). In addition, it is also important to bear in mind that although some individual improved grassland or arable fields can be important for biodiversity in their own right (such as feeding grounds for wintering geese), in most cases the biodiversity contribution such intensively managed fields make is heavily influenced by their spatial and temporal relationship to other crops, features and elements in the landscape. In a spatial context, biodiversity value is generally higher where there is a patchwork of habitats – eg meadows, grass pastures, crops, woodland, fallows. In a temporal context, biodiversity value is generally higher when not all fields or areas are managed in the same way at the same time; so neighbouring farms with essentially the same production systems will sow and harvest crops at different times. This produces a patchwork of the same crop at different stages of development; ploughed ground, seedbed, young crop, mature crop, cut crop and harvested crop. In a similar fashion adjacent pasture under different ownership will be grazed in different ways (eg with different animals and at different stock densities) and at different times of the year. This diversity provides much more favourable conditions for plants and animals (especially invertebrates) to find areas with suitable conditions for them to complete their live-cycles (Bignal and McCracken, 2000).

However, the challenge from a habitat network perspective is two-fold. There is a need to consider how best to incorporate the habitat network approach within the context of constantly changing contents of such fields. In addition, there is a need to consider how best to change the type and intensity of management practised in such fields.

9.3 Species that may benefit from habitat networks

Where there is a mixture and close proximity of arable and grass fields, it means that at any time of the year a farm will generally provide a number of different and varied habitats with the potential to support a wide range of plant, invertebrate and bird species. Some of these species are intimately linked with the annual farming cycle and dependent on the management of each field as whole, while others are associated with the maintenance and management of the surrounding boundary features. Managed and cultivated farmland therefore forms an important and distinct habitat complex in its own right (McCracken and Bain, 2000). However, assessing the biodiversity associated with agricultural landscapes is not straightforward. This is particularly true for the fauna, since their relationship with the landscape can be complex.

- Individual species may require different habitats at different stages in their lifecycle. For example, dragonfly larvae develop within freshwater whereas adults require suitable riparian vegetation on which to rest and use as hunting bases. Lapwings nest in short bare vegetation in cereal fields but as soon as the chicks hatch the adults take to the neighbouring grassland fields in order to forage. Badgers prefer mosaic landscapes, where forest patches provide suitable den sites and surrounding agricultural fields are used for foraging

- Individual species may require a range of habitats at the same stage in their lifecycle. For example, brown hare utilise a mosaic of farmland habitats throughout the year. Many birds nest in cover (such as hedgerows and woodlands) but need open habitats in close proximity in which to feed

- Individual species may only be present at particular times of the year. For example, many breeding birds are only present in the spring and summer, many overwintering birds (eg geese) only occur in the winter while other species may only pass through on migration in the spring and winter

- Even within the same habitat, many species have exacting requirements. For example, bees require bare soil to allow them access to burrows in close proximity to flowering plants as source of foodstuffs

In addition, for many species we have knowledge of their broad habitat associations and needs but other factors which need to be taken into account (such as their mobility and dispersal ability in the landscape) are less well known. For example, many insects are strong fliers or (like spiders) drift with the wind and hence could be thought to not need physical connections in the landscape. However, on cereal farmland some species have been shown to move along hedgelines and the edges of patches of scrub and wood rather than crossing open ground. In addition, many invertebrates have precise microclimate requirements and those which are slow -moving, in particular, may not be able to traverse extensive areas where temperature and humidity are inappropriate owing to extensive shading or exposure (Andrews, 1993).

It is also important to bear in mind that the impact of fragmentation on species will vary between different types of farmland habitats (Opdam and Wascher, 2004). Consequently, species which are adapted to unpredictable habitat availability in space and time generally exhibit high mobility and therefore are less susceptible to fragmentation (irrespective as to whether this occurs due to natural or human-influenced disturbance processes). Hence, species associated with arable habitats (which are generally very dynamic) would be anticipated to be less vulnerable than those associated with less dynamic habitats, such as woodland, marshland and wetlands and unimproved grasslands, occurring within lowland agricultural landscapes. However, the context in which these different types of habitat sit will also be important in influencing the scale of the impact of fragmentation upon them.

Habitat networks would therefore be expected to have more of a potential impact on smaller, less mobile and more habitat specific species, especially where the network introduces more spatial cohesion into the landscape (Opdam *et al.*, 2003). It is also important to keep in mind that habitat networks do not necessarily need to result in similar habitats being contiguous. More mobile species can benefit by presence elsewhere in the landscape (provided they can get to these other areas) and one additional advantage is that an increased range of locations in the landscape increases variation in conditions available to these species and hence can serve to buffer the impacts of any site specific land use changes. In addition, the surrounding landscape matrix can be made less hostile to organisms and thus provide greater conductivity.

In England, out of 326 BAP priority species, 164 are considered to need action at the landscape scale to complement site protection strategies (Porter and Wright, 2003). This includes species and species groups such as; rare arable plants; greater horseshoe bat (*Rhinolophus ferrumequinum*), bumblebees, skylark and lapwing. A sub-set of these (98 species) were thought to require measures to specifically "reconnect" habitat and establish networks within the local landscape. Although a similar analysis has not been undertaken in Scotland, it seems likely that a number of priority species and species groups are likely to benefit from the creation of habitat networks. Examples of species that could benefit from the development of habitat networks are shown in Table 6.

Table 6 Examples of species which would be expected to benefit from better habitat connectivity and spatial cohesion in habitat networks created in lowland agricultural landscapes. Species selection based on UK BAP and LBAP species relevant to Scottish lowland agricultural landscapes. Species groups based on knowledge of relationships

Species group	**Open habitat specialists**	**Woodland habitat specialists**	**Generalists Benefiting from general increase in habitat diversity**
Plants	Purple ramping-fumitory; cornflower	Creeping lady's tresses (*Goodyear repens*); three nerved sandwort (*Moehringia trinervia*); chickweed wintergreen (*Trientalis europaea*); Sanicle (*Sanicula europaea*); wood anemone (*Anemone nemorosa*); common cow-wheat (*Melampyrum pratense*)	Wood club-rush (*Scirpus sylvaticus*)
Invertebrates	Ringlet butterfly (*Aphantopus hyperantus*); ground beetles; grasshoppers	Pearl bordered fritillary; speckled wood (*Pararge aegeria*)	Scotch argus (*Erebia aethiops*); small pearl bordered fritillary (*Bolaria selene*) spiders; bees
Mammals	Water vole (*Arvicola terrestris*)		Brown hare (*Lepus europaeus)* pipistrelle bat (*Pipistrellus pipistrellus*)
Birds	Grey partridge		Corn bunting; reed bunting; song thrush

10 APPLICATION OF FOCAL-SPECIES MODELLING TO CONSTRUCTING HABITAT NETWORKS IN THE SCOTTISH LOWLANDS

10.1 Applying the BEETLE model

The review of approaches to constructing habitat networks highlighted the value of focal-species modelling as a method which combines practical applicability with robust theoretical assumptions about species ecology. Application of the BEETLE focal species model to the development of lowland habitat networks has been tested (in part) in Wales (Watts *et al.*, 2004) and in West Lothian (Ray *et al.*, 2004b). In both cases, the focus was on developing woodland networks for generalist and specialist species using generic focal species modelling. Such an approach would be applicable to developing agricultural habitat networks (AHNs) but there is also scope for using specific species which has the advantage of making the process more transparent to end users (examples in Table 6).

Before embarking on the modelling process it is necessary to decide whether a pro-active (ie design networks for particular focal species) or reactive (assume certain structural changes will take place and then evaluate the consequences for focal species) approach should be taken. In both cases the starting point will be to evaluate the existing landscape in terms of its network value. This will give an indication of the potential for change, and in what direction change should take place. The ideal way of exploring this process is to identify case-study areas, involving stakeholders in defining desired landscape-change scenarios to test (see example of Clashindarroch, section 11.7). This will allow integration of differing “political” priorities amongst stakeholders as well as evaluating the capacity for change within physical and ecological constraints (Ray *et al.*, 2004a).

10.2 Sources of land cover data

In order to begin the process of constructing habitat networks, habitat data layers need to be assembled and synthesised. The main sources of habitat data are listed and described in Table 7. Although there are difficulties in reconciling different datastets, many of the earlier problems have been addressed and integrated broad habitat layers now exist for the whole of Scotland (Ray *et al.*,2004a). The LCM2000 data set was used by Humphrey *et al.* (2004a) but found to be unreliable in differentiating between some habitat types such as heathland and mire vegetation. Its use in habitat suitability modelling is not currently recommended (Humphrey *et al.*, 2004b). If available, phase I habitat survey data is by far the most useful source of information for modelling purposes (Figure 16) given that it is derived from ground survey and there is good repeatability amongst surveyors (Stevens *et al.*, 2004).

NVC is often not particularly useful for broad-scale habitat suitability modelling in that the datasets are at a level of habitat detail in excess of that normally available for most species. In addition, NVC mapped datasets usually contain a considerable number of code combinations (eg over 20,000 for Mull) needed to deal with mosaics/composites. Before any analysis takes place, aggregate habitat codes have to be ascribed to the different combinations to reduce the number of composite types. This can be extremely time consuming unless done on a very small scale (eg <1km^2).

One of the key aspects of the network construction process is the need to predict where successful creation/restoration of semi-natural vegetation can take place in the landscape. Models exist to predict suitability of areas for native woodland creation (Pyatt *et al.*, 2001; Towers *et al.*, 2002) and semi-natural open habitat creation (Ray *et al.*, 1999). The Native Woodland Model (Towers *et al.*, 2002) was developed to provide strategic information about the potential for native woodland development at the 1:50000 scale and above. NWM projections are based on a linkage between existing site conditions (soil and land cover) and NVC woodland communities. The NWM provides a single output description which could be: woodland of a single NVC community; woodland of two or more NVC communities; areas of scattered trees and scrub, and mosaics of NVC and non-NVC woodland types.

Table 7 Spatial and other data sets available for network modelling

Data	Description	Value	Reference
SAC, SPA, NNR and SSSI boundaries	Boundaries of protected areas/ sites	Give indication of areas of high conservation value in general	www.snh.gov.uk
Phase 1 Habitat Survey	Broad scale field mapping approach giving information on the extent and distribution of natural and semi-natural habitats	Ideal source of good quality habitat information, but limited in coverage to specific regions	(NCC, 1990; Stevens *et al.*, 2004)
Land Cover Map 2000 (LCM)	Satellite derived remote-sensed datasets providing broad habitat definitions	Covers the whole of Scotland, but there are problems with accuracy in mapping some habitat types	(Fuller *et al.*, 2001)
Land Cover Scotland 1988 (LCS88)	Remote sensed dataset derived from aerial photography taken in 1988; provides broad habitat definitions at 1:25 000 scale	Covers the whole of Scotland focusing on semi-natural habitats, is out of date, but currently being updated ("New Image of Scotland")	(MLURI, 1993)
National Inventory of Woodlands and Trees (NIWT)	Derived from LCS88 dataset plus updated to 1995 from FC sources; provides information on broadleaved/conifer woodland > 2ha and small woods and trees (0.1–2ha)	Baseline data source on woodland for Scotland	(Forestry Commission, 2002)
Scottish Forestry Grant Scheme	Regularly updated records of new planting	Gives composition and extent of new woodland areas which can give indication of habitat value	www.forestry.gov.uk
Scottish Semi-Natural Woodland Inventory (SSNWI)	Constructed over the period 1995–2001 using interpretation of aerial photographs taken in 1988. Map of all woodlands > 0.1ha classified according to degree of semi-natural character	Identifies all semi-natural woodland, useful when combined with NIWT to locate sites of high conservation importance	http://www.scotlandswoods.org.uk
Ancient woodland Inventory (AWI)	Map of all ancient (existing since since 1750) woodlands over 2ha in size	Identifies areas of key importance for woodland biodiversity	(Roberts *et al.*, 1992)
Scottish National Digital Soil Map (MLURI)	Broad-scale mapping of soil series at 1:250000 scale (1:50000 and 1:25000) soil maps occur for some lowland areas)	Of limited value in predicting soil type unless combined with other information (eg Digital Elevation Model; LCS88)	(Pyatt *et al.*, 2001; Towers *et al.*, 2002)

Table 7 Spatial and other data sets available for network modelling *(continued)*

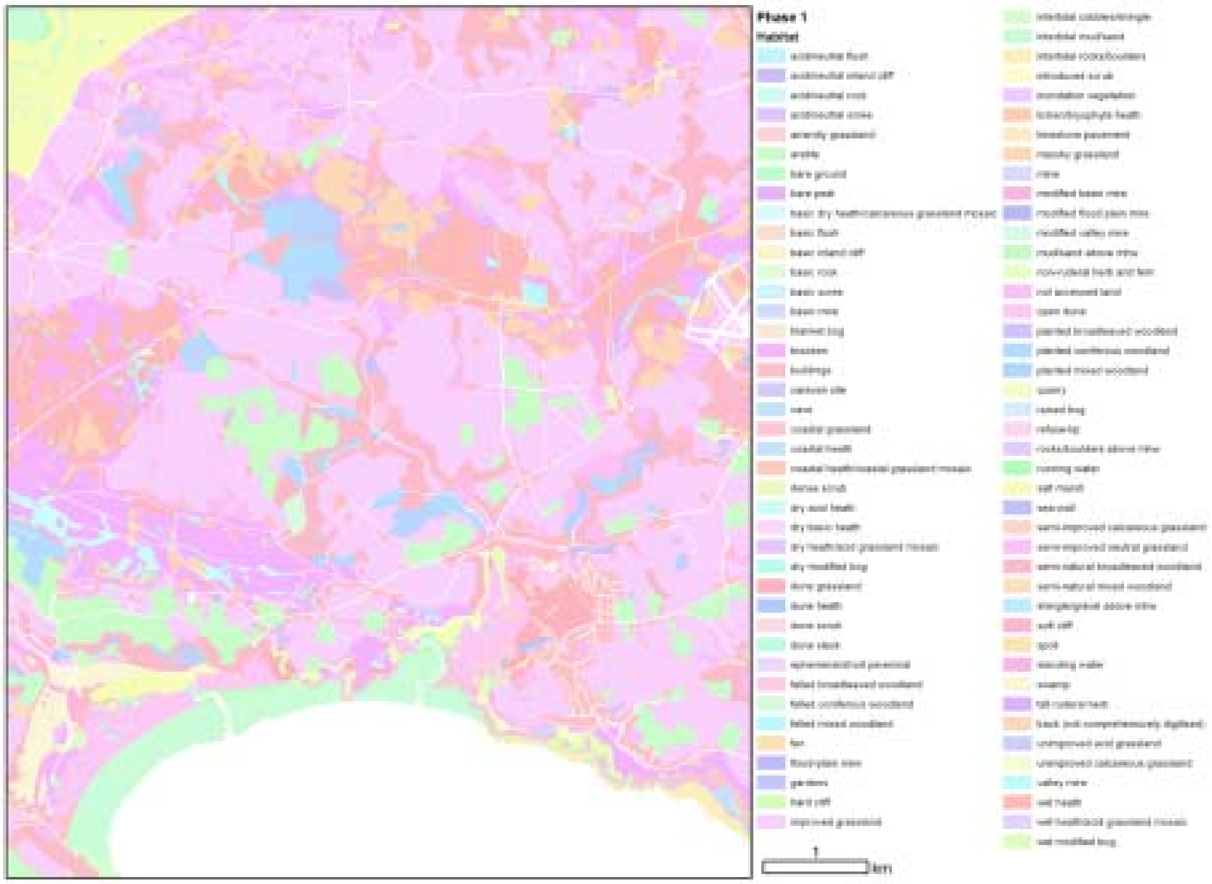
Data	Description	Value	Reference
Ordnance Survey Pan-Government product portfolio	Products include: 1) for large scale mapping – OS MasterMap; Land-Line; 1:10 000 Scale Raster; 2) for small scale mapping – 1:50 000 Scale Colour Raster; 1:50 000 Scale Gazetteer; 1:250 000 Scale Colour Raster; Strategi; Meridian 2	MasterMap is the new, more definitive, large-scale digital map of Great Britain, containing information on roads, tracks, paths etc. Gives accurate representation of woodland areas and boundaries and can identify linear features which can act as barriers to dispersal or as corridors	www.ordnance survey.co.uk
Ordnance Survey Digital Elevation Model (DEM)	Digital elevation data for whole of the country	Allows construction of elevation maps aiding in deriving ESC climatic and soil quality indices.	www.ordnance survey.co.uk
British Geological Survey 1:625 000 digital maps, (BGS)	Maps of geological series across Britain	Can help with predicting soil type and hence soil quality in ESC	http://www.bgs.ac.uk/geoindex/home.html
SNH BAP priority habitat report and maps	Maps and description of UK BAP priority habitats summary of all previous phase 1 and phase II survey information in Scotland	Provides information on location of key habitats in Scotland	(Ellis and Munro, 2004; Mackintosh *et al.*, 2004)
Ecological Site Classification	A tool for predicting suitability of areas for creating/restoring woodland and open-ground habitats based on climate and soil variables	Allows construction of suitability maps for different habitat types across the whole of Scotland	(Pyatt *et al.*, 2001; Ray, 2001)
National Vegetation Classification survey data	Various surveys covering SACs, SSSIs and other habitats of high conservation value in Scotland	Coverage is geographically limited and information is often too detailed to make meaningful links with species requirements	(Rodwell, 1991)
Scottish Integrated Agricultural Control System (SIACS)	Contains information on field sizes and crop types for very field in Scotland	Aggregated statistics available at parish level but data from individual land holdings are covered by the Data Protection Act	http://www.scotland.gov.uk/Topics/Agriculture/grants/18148/11836

Unfortunately the NMW has not been fully calibrated or evaluated in lowland environments, and the Ecological Site Classification (ESC) provides the only current alternative using both climate and soil data to predict vegetation type. Recent work by Corney *et al.* (2004) has confirmed the importance of both soil and climate in predicting vegetation community composition. A problem however, is the lack of soil information at the appropriate scale for many parts of Scotland, although methods have been piloted which allow prediction of soil type from soil series information and digital elevation models (Bailey *et al.*, 2003).

One important dataset which as far as we are aware has not been used in habitat network modelling before is the IACS information system. As part of the process of claiming agricultural subsidy payments, all farmers in Scotland are required to submit an annual IACS (Integrated Agricultural Control System) return to SEERAD each year which sets out the use each area of their farm is being put to in that year (see summary box). SEERAD hold a GIS Field Register of all IACS fields in Scotland which contains the gross area of each field and details on areas within each field which are excluded from subsidy payments (thereby providing an indication of the area within each field on which the farmer is eligible to claim arable and forage area payments). However, depending on the implications of decoupling, the IACS 2005 is likely to be different and may affect what is included in the form. Producers will no longer need to specify arable crops as they will be receiving payment decoupled from production.

This information is held electronically on the SIACS (Scottish Integrated Agricultural Control System) database, the main use of which is to validate the claims made by farmers each year with the information held about each field and to identify any claims of excess areas or duplicate claims. It should, therefore, be feasible to use extracts of the information collected to calculate broad summary indices of agricultural field diversity around any one point in the landscape on a year on year basis, although not on a spatially explicit basis. However, the SIACS information would have to be incorporated into a GIS together with other sources of information on location and extent of non-farmed habitats (eg woodlands, hedgerows, water margins, etc.) before it would be possible to obtain an indication of overall habitat diversity in the landscape and/or information on what (other than another agricultural field) habitats were contiguous to the fields themselves.

Figure 16 Extract from Watts *et al*. (2004) showing example of Phase 1 habitat data for a part of Wales



10.3 Sources of species data

The availability of autecological data is one of the major limitations in the species modelling process (Humphrey *et al.*, 2004a). However, the situation is improving all the time with new data being gathered as part of species action plan priorities. Information is available through species action plans, country agency reports (eg – Fowles, 2003) and published literature. Recently links have been established between LBAP regions and the National Biodiversity Network (www.nbn.org) which gives information on the occurrence and ecology of species for particular areas. The Biological Records Centre at CEH also provides species information. Expert opinion can also be consulted and inferences made from general observations. Table 8 gives an indication of the types of data needed for the modelling process, and an estimate of current availability. In general, good quality information on habitat preferences is available for most species using agricultural landscapes, whereas data on minimum patch sizes, dispersal abilities etc. is more patchy. The profiles illustrated in the summary boxes gives some examples for individual species where the availability of ecological data is quite good. Where information on specific species is lacking, the generic focal species approach can be used to model processes and structures not covered by specific species.

10.4 Importance of case study areas for testing approach

The process of constructing habitat networks using focal species has had only partial testing within the Scottish lowlands. Before the method can be applied more generally within a toolkit for landscape evaluation, there is a need for more extensive testing. This can achieved ideally through a set of case-studies covering contrasting lowland landscapes.

Table 8 Estimate of availability of ecological data for focal species in agricultural landscapes. Scores are: 1 = available for a few species; 2 available for some species; 3 = available for most species

Ecological profiles	Data availability
Minimum patch area	1
Maximum dispersal distance	2
Habitat preferences	3
Ability to move through non-habitat	2

Summary Box

Overview of information collected on each field* using standard IACS form

1. Basic Information section provides field information on the farm land and its location supplied to the farm pre-printed on the form based on information held in SIACS. The information recorded on each field includes:

- A code for the individual farm to which it belongs
- A unique 14-digit identifier (based on ordnance survey grid coordinates)
- The total area of the field (in hectares to 2 decimal places).
- The crop grown in the field in the year previous to the year for which information is being collected
- The area, to 2 decimal places, of each field that the farmers declared eligible for payments in the year previous to the year for which information is being collected.
- An indication as to whether the field in question is subject to an agri-environment or forestry measures.

2. Livestock Schemes – Forage section records both the type and area of forage being grown and claimed towards the farm's forage area calculation. A forage field is defined as a field available for maintaining animals and therefore includes different grassland categories, fodder root crops for stock feeding (eg turnips, swedes, kale and rape) and land that would otherwise be eligible for support but which has not been claimed under the arable crop support scheme (eg cereals, linseed, oilseeds, protein crops, etc., claimed as forage)

3. AAPS Arable Crops section records both the type and area of arable crop eligible for support payments and planned for harvest in the year to which the IACS form relates. Codes are used for both types and varieties of crop grown.

4. AAPS Set-Aside section records both the type and area of different types of set-aside present in each field

5. Other Land section. The IACS form must include all usable fields on the farm. This section therefore records the use and area of any fields on the farm not already entered on the form. This includes: land let out to others; land used for crops which are not eligible for support payments (eg potatoes, carrots, soft fruit, etc); grazeable fields that are not being claimed elsewhere on the form. However, the form does not record information on non-usable land (eg roads, woodlands, yards, buildings and ponds).

* A field is defined as an area of land with fixed boundaries which are sufficient obstructions to prevent the field, together with any of its neighbours, being worked as a single field. Fixed boundaries usually consist of physical features, for example trees, hedges, wide streams and so on. A parcel is a continuous area of land on which a single crop is grown by an individual farmer. If a field is being used for more than one purpose (for example part of it has been set aside), it contains two or more parcels. SEERAD do not allocate unique identifiers to parcels. Farmers account for each field on their farms in the IACS returns and show (where relevant) differences in parcel use within these fields.

Summary Box

Species profile for water vole (*Arvicola terrestris*) adapted from (Mathieson, 2001)

Habitat: Occurs along the banks of rivers, streams, canals, ditches, lochs and ponds. Each vole uses a series of burrows dug into the riverbank where the soil permits (ie not gravel bed or rock banks). These include residential burrows, comprising many entrances, inter-connecting tunnels, nest chambers, and bolt holes consisting of short tunnels ending in a single chamber. In the nest chamber, the nest consists of shredded grass. Permanent water is essential during periods of low flow in summer, while sites that suffer total submersion during protracted periods of winter flooding are untenable. Preferred shore type is predominantly earth with a bank profile that shows a stepped or steep incline into which the vole can burrow and create nest chambers above the water table. The amount of bankside and emergent vegetation cover is very important, with the best sites offering a continuous swathe of tall and luxuriant riparian plants (at least 60% ground cover). Sites excessively shaded by shrubs or trees are less favoured.

Minimum patch size: The best water vole populations may occur where the conditions favour a slow-moving watercourse, less than 3m wide, around 1m deep and which does not show extreme fluctuations in water level. River catchment headwaters, small backwaters and ditch systems, canals and pond habitats appear to be strongholds, as do riverine systems free from American mink colonisation. Increasing attention is also being focused on tiny upland streams in Scotland, which appear to be a more favourable habitat for water voles than previously realised.

Distribution: In Britain, surveys carried out by the Vincent Wildlife Trust have shown a major decline in the water vole population since 1900. Water voles are estimated to have disappeared from around 94% of their previously recorded sites. Prior to, and then concurrent with, the escape of American mink from fur farms, the destruction of the bankside habitats of water voles by river engineering works have led to much fragmentation and isolation of water vole populations. For example, a survey in 1993 by the Vincent Wildlife Trust reported that the River Forth catchment (including the Avon, Teith, Kelty, Duchray, Katrine, Allan and Devon rivers) had a highly fragmented distribution of water voles, with many sites negative. Much suitable riparian habitat was noted but the species was uncommon. Mink distribution was patchy indicating that fragmentation rather than predation was the most likely cause of the distribution.

Food: Water voles are herbivorous, feeding mainly on lush waterside vegetation of grasses, sedges, rushes and reeds. In the winter months, roots and barks of shrubs and trees form an important part of the diet, together with rhizomes, bulbs and roots of herbaceous plants.

Mobility: Survival of water vole populations, especially following local declines, often depends upon the movement and exchange of individuals between neighbouring colonies and habitats. However, although it is known that stretches of watercourse without bankside vegetation can act as a barrier to free movement, more information is required on the distances over which individuals will move and factors which will influence this.

Summary Box

Species profile for small pearl bordered fritillary (***Boloria selene***)

Habitat: Found along the edge of woodlands, coppiced woodlands, damp meadows and scrubby slopes in lowland areas. It thrives in transitory commercial woodlands where felling creates open patches. In Scotland it is also found in open wood pasture and woodland edges, usually where there is some grazing by deer or sheep (Asher *et al.*, 2001). The adult butterflies will only fly in full sunlight. Encroaching bracken and scrub are detrimental to the habitat of small pearl bordered fritillaries as it leads to loss of nectar plants.

Minimum patch size: No extensive data available along with no data on average colony sizes, although a recent report on this species at Clocaenog indicated that small populations were less than 100 individuals and medium populations were between 101 and 1000 individuals (Stewart *et al.*, 2004). In Clocaenog the size of the occupied habitat patches ranged from 0.05–8.5ha. The majority of the occupied patches were less than 4ha.

Distribution: Species is declining in England but remains locally abundant in Scotland and Wales. It breeds in damp grassy vegetation, where there are plenty of food plants in a lush sward.

Foodplant: The main food plants utilised are common birds foot trefoil (*Lotus corniculatus*), buttercups (*Ranunculus* spp.) as well as scabious (*Succisa pratensis*) and wild thyme (*Thymus praecox*). The females are often found in locally sunny patches where abundant violets are found and they use these locations for laying their eggs. The species prefers larger violets than the Pearl Bordered fritillary and are much more tolerant of damper conditions. However, they require violets growing in open sunny locations, and avoid those areas that do not receive at least 50% of direct sunlight during the day (Thomas *et al.*, in press).

Mobility: Very little is known about the mobility of the species. Studies of coppiced woodland in lowland England indicate that the species is highly sedentary. Four coppice panels were cut at varying distances from the existing population, with 20 adults transferred to each panel. Subsequent mark recapture studies indicated that the adults survived and bred in the new panels but there was very limited movement between panels, the greatest movement occurring between the original population and the panel cut immediately adjacent to it. They did not move into new clearings created a few hundred metres away (Thomas and Snazell, 1989). It is thought to be slightly more mobile in different habitats. Where habitat is extensive the species may exist in a series of metapopulations.

Studies in Clocaenog of marked individuals has shown individuals moving between 0.8 and 3.4km, with colonisation movement of between 350m and 100m (Stewart *et al.*, 2002). However in 2001, the mean distance travelled was 90m with 65% of movements less than 50m. In 2002, the mean distance moved was 48m with 76% of movements less than 50m.

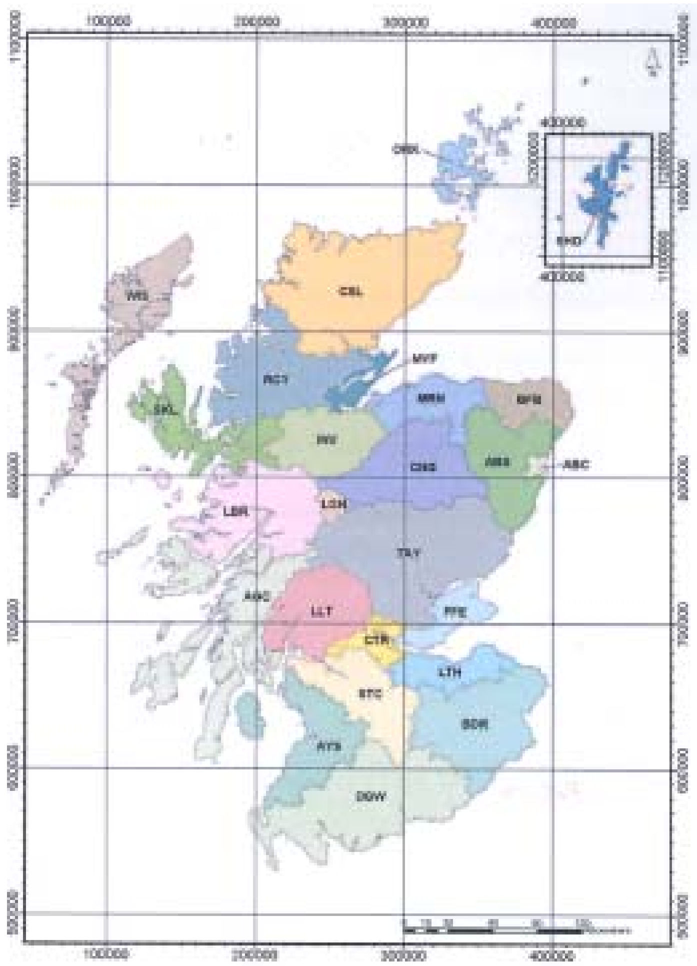
11 ASSESSING CAPACITY OF LANDSCAPES TO ACCEPT CHANGE FROM VISUAL/CULTURAL PERSPECTIVE

11.1 Landscape Character

In 1994 Scottish Natural Heritage initiated a National programme of Landscape Character assessments (LCA), (Figure 17). During the period 1994–1999, 30 LCA reports were completed, providing complete coverage of Scotland. Although the brief and methodology of the studies were influenced by the local requirements of the steering groups managing individual LCAs, the consultants were generally all working towards the achievement of six primary objectives for the programme (Martin and Swanwick, 2004):

- to establish an inventory of all the landscapes of Scotland;
- to raise awareness of Scotland's landscapes;
- to identify the main forces for change in Scotland's landscapes;
- to provide information to support various kinds of casework, including development control and other proposals for land use change;
- to provide information to help SNH, local authorities and others to input into development plans and other land use strategies; and
- to help inform national policy on issues relating to landscape interests.

Figure 17 LCA areas (Martin and Swanwick, 2004)



Code	LCA area
ABC	City of Aberdeen
ABS	Aberdeenshire
AGC	Argyll and Clyde
AYS	Ayrshire
BDR	Borders
BFB	Banff and Buchan
CNG	Cairngorms
CSL	Caithness and Sutherland
CTR	Central
DGW	Dumfries and Galloway
FFE	Fife
INV	Inverness
LBR	Lochaber
LGN	Lagan
LLT	Loch Lomond & Trossachs
LTH	Lothians
MRN	Moray and Nairn
MYF	Inner Moray Firth
ORK	Orkney
RCY	Ross and Cromarty
SHD	Shetland Islands
SKL	Skye and Lochalsh
STC	Glasgow and Clyde Valley
TAY	Tayside
WIS	Western Isles

It was envisaged that the LCA suite would help to inform a wide range of environmental decision making. Those practical applications broadly fell into two categories; planning and landscape conservation/ management (Martin and Swanwick, 2004). In the sphere of landscape conservation/management, applications of the LCA programme have been used to varying degrees in relation to forestry, agriculture and other land use change studies (Martin and Swanwick, 2004).

11.2 LCA applications in forestry

For forestry the theme of landscape character threads its way through all major policy documents:

> *'Forestry expansion must be balanced with the needs of other land uses and must respect the character of the landscape.' (Anon, 2004b – page 4)*
>
> *'The acceptability of future new planting will depend on the nature of the proposal, its scale and location, and the resulting impact on existing landscapes. SNH's LCA…provide valuable descriptive information about the landscape and these can be used to assess the impact of planting proposals.' (Anon, 2000 – page 32)*

The LCA documents have and are being used by local authorities in the preparation of the next generation of Indicative Forestry Strategies (IFS) to the requirements of Circular 9/1999 IFS (Scottish Executive, 1999).

The individual landscape units (Ayrshire Landscape Character Assessment, March 1998) have been used in the development of the Ayrshire Woodland Strategy to assess the sensitivity of different parts of the Ayrshire Landscape to woodland planting (www.south-ayrshire.gov.uk).

Summary Box

Landscape Character Assessment in Dumfries and Galloway

At the local level the Landscape Design Guidance for Forests and Woodlands in Dumfries and Galloway (Environmental Resources Management, 1998) demonstrated how the summarised advice on forests and woodlands in the Dumfries and Galloway LCA (Lande, 1996) and general advice in the Forestry Commission's suite of Forest Landscape Design Guidelines (Forestry Commission, 1994; Bell, 1998) could be integrated. For each landscape character type a guidance sheet was developed, describing how the distinctive local character of the different landscape types could be conserved and enhanced with respect to woodland cover, expansion and management.

Although the generated guidance was design led, meaning forest and woodland structure was looked at primarily from a visual perspective, and the design principles of shape, scale, diversity, unity and spirit of place (Forestry Commission, 1994), the guidance sheets provided essential contextual local ecology information and relevant commentary on perceived opportunities and constraints to forest and woodland management and expansion.

The Landscape Design Guidance for Forests and Woodlands (Forestry Commission, 1994) provided forest and woodland managers with practical advice on the successful integration of new woodlands and forests within the landscape, and to a degree the relative capacity of each individual landscape character type to accept such change. The landscape and potential scenic effect of forest and woodland expansion was illustrated by plan and perspective sketches of typical landscape type views. What it did not consider, however, was the wider biological implications on either the forest and woodland structure of the region or the potential impact on other land uses or sensitivities.

In contrast, the pair of Local Forestry Frameworks (LFF) in Dumfries and Galloway (Environmental Resources Management, 1998) and Langholm/Lockerbie (Environmental Resources Management, 2000) did consider other land uses and sensitivities.

Summary Box

Forest and Woodland Frameworks

Forest and Woodland Frameworks are advocated in the Scottish Office Circular 9/1999 for sensitive areas (Scottish Executive, 1999). They consist of product and process.

The product comprises map and text based documents that inform and guide those involved in the establishment and management of forests and woodlands. They explore the potential for integrating new forests and woodlands in empathy with the sensitivities of the area and also encourage good management of the existing tree resource.

The process consists of an inclusive approach involving public workshops and consultation. Frameworks provide a participatory forum for local communities, with the objective of avoiding conflict and developing consensus.

Forest and Woodland Frameworks can play an important role in helping to achieve a balance between the expansion and management of forestry, agriculture and other land uses, the conservation, enhancement and enjoyment of natural heritage, and opportunities for local employment. They help to inform land use strategy at a local or regional level. They also provide guidance to Forestry Commission staff on the determination of Scottish Forestry Grant Scheme (SFGS) applications, and they provide information to SFGS consultees to assist them in decision making.

The issues that were considered in the development of the LFFs were:

- agriculture
- archaeology and cultural heritage
- community aspirations and quality of life
- economics and employment
- education
- forest/woodland issues
- land use balance
- landscape character

- nature conservation and biodiversity
- recreation and tourism
- transport
- water quality

For nature conservation and biodiversity, the LFFs recognised that the maintenance and enhancement of biodiversity could not be achieved by the protection of designated nature conservation sites alone. They acknowledged the importance of the Local Biodiversity Action Plan (LBAP) and the contribution made by relevant Habitat Action Plans (HAPs) and Species Action Plans (SAPs), either prepared or in preparation for the area, and the ongoing or potential adverse affects of afforestation on other species and habitats. Future decisions on forestry expansion and management should incorporate measures to ensure maintenance and enhancement of biodiversity.

Although landscape character was a consideration (reference was made to both the aforementioned Dumfries and Galloway LCA and Landscape Design Guidance for Forests and Woodlands in Dumfries and Galloway), landscape character type areas were not selected as the sub-area unit of study for the LFFs. It was decided that 'in order to present the data in a meaningful way…' the LFF area would be divided into water catchments (as described in section 4.4).

Each catchment was sub-divided into tracts of land, which are then described and provided with guidance for forestry/woodland planting and management. The tracts were also categorised in terms of their potential for forestry expansion (the key issue behind the two LFFs). The categorisation was based on:

- existing land use;
- the number and relative significance of sensitivities identified following analysis of baseline data; and
- a review of public opinion as expressed through workshops and written responses.

The catchments ranged in categorisation from 'very high sensitivity', implying very limited opportunities for new planting, to 'few sensitivities', implying significant scope for forest expansion and 'existing forestry/ woodland' with associated opportunities for restructuring.

The main tool and output from the two LFFs was a map (reduced 1:50,000) of the entire LFF areas, illustrating the catchment area tracts and their category of sensitivity. This map is intended to be essential reference to all forest and woodland managers, especially with regards any proposals for forest and woodland expansion. In essence, any proposal within a high sensitive area would probably not receive support from either FC or consultees.

What was not covered by the LFFs were the potential landscape and visual effect of the advice being followed over a period of time, and guidance on the potential for habitat network development (even though the nature conservation and biodiversity section acknowledged that BAPs promote the principle of creating Forest Habitat Networks).

11.3 LCA applications in agriculture

By contrast to forestry, there is relatively little evidence of the use of the LCA programme outputs in agri-environment scheme targeting and evaluation in Scotland (Martin and Swanwick, 2004).

> *'This lack of focus on landscape issues within agri-environment initiatives in Scotland has been highlighted in a recent report to the Scottish Executive (Agriculture and Environment Working Group, 2002). This notes the importance of agricultural landscapes to quality of life and the tourist economy in Scotland. It points to the recognition given to landscape character issues in national planning policy and guidance in the forestry sector and suggests there is a need to develop similar practical advice on agricultural landscape design. It recommends that the Executive and its agencies should develop policies and guidance for this purpose, and that Rural Stewardship Scheme should be amended to respond more effectively to environmental (including landscape) concerns. The LCA programme could provide a starting point for this work. (page 31, (Martin and Swanwick, 2004).*

11.4 LCA and Historic Land-use Assessment

An essential element of LCA is an understanding of the historic dimension (Swanwick and Land Use Consultants, 2002). During the information gathering phase of LCA, archaeological and cultural heritage data will be recorded and incorporated into the 'sensitivities' database.

However, a Historic Land-use Assessment (HLA) developed by Historic Scotland (HS) and Royal Commission on the Ancient and Historical Monuments of Scotland (RCAHMS) will provide much greater understanding of the historic dimension of landscape character (over a third of Scotland is now covered by an HLA).

An HLA is a complementary study to LCA, interpreting the material remains of the past and providing perceptions and interpretations that allow us to understand the present day landscape. It focuses on the effect of human activity on the landscape. Its purpose is to both inform and facilitate the management of change to the historic environment, primarily at the landscape scale (Fairclough and Macinnes, undated).

The developed methods and approaches are similar to LCA, particularly the spatial map based use of information in a GIS environment. This facilitates the incorporation of HLA into LCA.

The HLA contribution to landscape understanding lies in the following:

- a concern with successive layers in the land – "time depth";
- an interpretation of the whole modern landscape and its predominant historic character;
- a particular concern for defining and explaining landscape character in historic terms;
- the ability to identify the patterns and historic significance of major land use such as woodland, moorland, designed landscapes etc;
- the ability to describe some of the character of previous episodes of landscape, and in other ways to define time depth;
- the ability to measure more recent change in landscape character.

11.5 Applications of HLA

HLA can play a role in land management, informing, for example, agri-environment and forestry schemes, both strategically and at the level of land-units. When achieving wider coverage, it will be able to provide national or regional overviews, and help to define local characteristics as a basis for prioritising actions from national to local level.

HLA provides an overview of cultural sites and landscapes, and can combine with LCA to define key landscape characteristics for protection, management and interpretation. It can assist in monitoring landscape change by providing baseline information against which change can be measured. Alongside LCA, it can also facilitate an integrated approach to countryside management, relating land-use change to existing character in a way which is better informed about the origins of that character (Fairclough and Macinnes, undated).

11.6 Landscape sensitivity and landscape capacity

LCA is being widely employed as a tool to help guide decisions about the allocation and management of land for different types of development. Work of this type inevitably involves consideration of the sensitivity of different types and areas of landscape and of their capacity to accommodate change and particular types of development (Swanwick, undated). Swanwick (undated) acknowledge that the topic of landscape sensitivity and capacity proved to be one of the most difficult to deal with during the development of the latest LCA guidance (Swanwick and Land Use Consultants, 2002) and offered three terms; two for sensitivity, one for capacity:

1. **Overall landscape sensitivity** This term should be used to refer primarily to the inherent sensitivity of the landscape itself, irrespective of the type of change that may be under consideration. It is likely to be most relevant in work at the strategic level, for example in preparation of regional and sub-regional spatial strategies. Landscape sensitivity can be defined as embracing a combination of:

   - the sensitivity of the landscape resource (in terms of both its character as a whole and the individual elements contributing to character);

   - the visual sensitivity of the landscape, assessed in terms of a combination of factors such as views, visibility, the number and nature of people perceiving the landscape, and the scope to mitigate visual impact.

2. **Landscape sensitivity to a specific type of change** This term should be used where it is necessary to assess the sensitivity of the landscape to a particular type of change or development. It should be defined in terms of the interactions between the landscape itself, the way that it is perceived and the particular nature of the type of change or development in question.

3. **Landscape capacity** This term should be used to describe the ability of a landscape to accommodate different amounts of change or development of a specific type. This should reflect:

   - the inherent sensitivity of the landscape itself, but more specifically its sensitivity to the particular type of development in question, as in 1 and 2. This means that capacity will reflect both the sensitivity of the landscape resource and its visual sensitivity;

   - the value attached to the landscape or to specific elements in it.

11.7 LCA and the visual landscape

The previous discussion on landscape sensitivity raised the subject of assessing a landscapes visual sensitivity to change. Only one of the aforementioned studies attempted to consider the visual aspects of potential forestry and woodland expansion; the Landscape Design Guidance for Forests and Woodlands in Dumfries and Galloway (Environmental Resources Management, 1998). This was achieved by a landscape architect preparing a composite sketch of a specific landscape character type and adding outline shapes of woodland types considered appropriate in their shape, scale and quantity, all to reflect the design guidance. Although this approach was considered appropriate at the time there is no denying they were both simplistic and limited in what they could communicate.

Today, however, it is possible to combine a wealth of information collected as a database on a GIS system, interpret that information in 3D model form and drape this information over a digital terrain model (DTM) to produce 'visualisations' of proposed landscape. An example of this approach is illustrated in the EU 5th framework Research & Technical Development (RTD) project "Visulands" (www.esac.pt/visulands_esac) but only in Portuguese.

VisuLands is a pan-European project whose main aims are to:

- develop visualisation tools to support public involvement in the assessment of landscape change;
- facilitate improved public participation in landscape policy and planning issues;
- look to the relationship between visual qualities and other landscape functions such as biodiversity, cultural heritage, amenity and sustainable production.

In Scotland the FE forest of Clashindarroch, and surrounding primarily agricultural landscape have been selected as a case study site.

The main case study objective is:

- to develop visualisation and planning tools for land use change, and engage end users (public and professional) in their assessment.

Subsidiary milestones/objectives are:

- representation of current status and alternative land use/forest management scenarios (and associated predictions on impacts of other values) against agreed criteria for the site;
- exploration of drivers, agreement on scenarios, collation of necessary data and development of visualisations using different systems;
- test visualisations with end-users to address specific queries
- assess baseline status, and implications of alternative land cover/land use/land management expressed in the chosen scenarios, demonstrating use of visual and non-visual indicators, and trade-offs between land use functions;
- assess effectiveness of selected mechanisms for demonstrating change;
- development of suitable educational materials for communicating the outcomes of choices of scenarios on landscape evolution, and engaging people in associated participatory activities.

These objectives are to be considered in three proposed landscape change scenarios for Clashindarroch forest, each representing alternative, radically different but not likely land use options (Table 9). In essence, the development of three 'parallel universes' for Clashindarroch; three distinctive scenarios that explore the impacts of different decisions about tree-planting made at the time when Clashindarroch forest was created. These scenarios will be used as the basis for visualisations and calculation of associated indicator values (eg measures of biodiversity, productivity etc.).

The data and visualisation tools to be utilised are:

- ArcView GIS 3-D Forester (FR)
- ERDAS VGIS (MLURI)
- Virtual Natural Studio (MLURI)
- Digital Terrain Models, using nationally available, standard datasets;
- Land cover data from nationally available, standard datasets, plus local management data;
- Aerial imagery, collected by a standard means, for landscape textures;
- Billboards and 3D models of individual features (eg trees).

The landscape change scenarios will be developed by Forest Research, utilising the ArcView GIS 3-D Forester extension to generate the visualisations. This is an FC in-house system that has the unique capability of generating images by interpreting the recorded database, including visualising projected changes to the forest over time.

More realistic images will be generated by the Macaulay Institute utilising their in-house ERDAS VGIS system and, if available, Virtual Natural Studio. These facilities should be capable of modelling the more detailed features in the landscape, illustrating the subtle textural and colour transitions of vegetation mosaics. The development and visualisation of the scenarios is anticipated to be completed by September 2005.

11.8 Landscape and visual assessment

The development of a habitat network will have an effect on the landscape of the candidate area. The significance of that effect on the character of that landscape, and peoples' visual perception of the area's scenic qualities will depend on what is proposed and how that proposal will interact with what is already there.

To help determine those effects and to inform changes to the proposal each topic in this section has an invaluable part to play. The common denominator between them is the collection of a comprehensive, accurate and relevant database on GIS.

LCA is an invaluable tool towards understanding the existing character of a local area, and informing the appreciation of the landscape's sensitivity and capacity for change. The value of the LCA would be significantly increased if there was an associated HLA covering the local area. In combination, these assessments will help to confirm the feasibility of the proposal, identify any constraints and help to inform any design considerations that will ensure both the conservation and enhancement of landscape character.

The generation of associated visualisations will not only help towards the evaluation of the proposals but also facilitate stakeholder and the general public understanding of what is being considered. Further, the utility of computer generated visualisation systems means that those consulted will be capable of interrogating both viewpoints and content.

Table 9 Scenarios for Clashindarroch

Scenario	What it would provide and demonstrate	How the scenario would be constructed	How the scenario would be progressed in time	Assumptions
A. No new trees	Visualisation of landscape without afforestation	Old OS maps and 1940's aerial photographs to capture remnant tree cover and broad vegetation type	No changes proposed to vegetation mosaic;	Assume land use equivalent to adjacent agricultural land
B. Trees for timber production	Planting of the extent of Clashindarroch forest with the optimal mix of timber trees	Use Ecological Site Classification to predict the best option given the site /climate combination	Growth rate as predicted by ESC, and felling 'rules' from investment appraisal guides; replanting with same species in year after felling.	Assume that volume yield equates to economic yield
C. Native woodland trees	Productivity and appearance of forest if a native tree cover had been restored	Use of digital soils and ESC-GIS to predict native woodland types	No felling; trees progressed at growth rate predicted by ESC; then held constant.	Assumes no stand break – up once maturity is reached
D. 'Current design plan'	Baseline of existing management, representing existing compromise between 3 end-points (A-C) above	Digitising current felling and restocking plans; 2000–2005		

11.9 Landscape change and public opinion

Stakeholder engagement is recognised as an essential element of land use and landscape planning. For example, for the Local Forestry Frameworks in Dumfries and Galloway it was the key aspect of the projects. They were developed from the bottom up, putting all those with an interest in the consequences of the continued expansion of forests and woodlands in the prescribed areas at the heart of the project.

Understanding the consequences of landscape change and public attitudes to those changes is the subject of a project recently initiated by Scottish Natural Heritage; Landscape Change Scenarios (C. Rowse *pers. com.*).

The methodology for a pilot study is currently in preparation. However, some of the criteria that will have to be determined and tools being considered for application in the study include those that have been applied in previous studies:

- scale of analysis
- landscape Character Assessment
- natural Heritage Futures
- visualisations
- public attitude assessment

12 APPROACH TO ASSESSING RECREATION AND ACCESS OPPORTUNITIES

12.1 Context

It is acknowledged that habitat networks are set within the wider context of sustainable development. In addition to ecological benefits, these networks can potentially yield many wider economic and social benefits. There is a belief that many features of ecological networks, such as riparian corridors, will assist key environmental functions (eg species dispersal, hydrological processes such as water quality and flooding etc.) and also impact on associated factors (eg recreation, countryside character – Smith and Helmund, 1993).

These wider non-ecological benefits considerably strengthen the case for the development of habitat networks (Dover, 2000). For example, the recommended establishment of greenways in the urban environment is intended to provide both recreational and wildlife benefits (Barker, 1997). A corner stone of the new Scottish Biodiversity Strategy is to link the conservation of biodiversity much more closely to benefits for people and communities (Anon, 2004a); hence the increasing emphasis on establishing new woodland in urban areas and diversifying the farmed landscape (Anon, 2000).

One of the important aspects to consider as a consequence of network development is the potential impact on recreational value of the landscape. There are general concerns such as visual effects of increasing cover of a particular land type (eg forestry) and the impact on landscape quality and hence on recreation and access value. These issues can be dealt with through the LCA process (see section 11) which assesses visual impact on landscape character. However, there are also practical, on the ground issues to consider relating to habitat networks and their impact on recreation and access.

12.2 Impact of disturbance on wildlife

One potential problem is that the expansion of networks in the lowlands may encourage the development of additional habitat in areas currently used for recreation. From an ecological perspective there may be pressure to remove or adjust path routes, picnic sites, parking etc. to minimise disturbance to wildlife. In addition, there may be pressure from the ecological perspective to improve the wildlife value of existing networks.

As a result of recent changes in legislation (Land Reform (Scotland) Act 2003 – www.scotland-legislation.hmso.gov.uk) it is expected that people pressure on lowland agricultural land will increase and is likely to be focused on the more wildlife-rich field margins (as crops are excluded from rights of access at certain stages of growth). The effects of recreational disturbance on wildlife has been the subject of a number of recent reviews (Just Ecology, 2005) and research (Finney *et al.*, 2005). Generally there appears to be little effect of disturbance to wildlife in woodlands as in the vast majority of cases people keep to paths and do not venture too far from formal facilities. This also appears to apply to upland breeding birds. In a study of the effects of disturbance on golden plover in the Pennines, Finney *et al*. (2005) found implementation of simple people behaviour management methods (ie construction of a new path) significantly reduced the impact of recreational disturbance.

The bulk of research into wildlife disturbance focuses on water birds (Just Ecology, 2005) such as wintering wild fowl (ducks geese and swans)and waders, although there is some data on terrestrial birds. There appears to be very little information on the effects of disturbance on other species-groups. Just Ecology (2005) recommend a number of key principles to adopt when attempting to minimise effects of disturbance.

These are listed in Table 10 together with some of the related rules that could be adopted when a network analysis is undertaken. The idea would be to try and optimise provision of recreation facilities without compromising the integrity and hence the rationale for having a habitat network, nor the public's right to have reasonable access to enjoy wildlife (Anon, 2004a).

Table 10 Principles for minimising disturbance on wildlife translated into network analysis rules for use in a GIS from Just Ecology (2005)

Principle	Network analysis rule
Tailor management to the most sensitive species present	Identify and obtain ecological data for most sensitive species using network
Physically separate areas to be used by wildlife (either spatially or temporally eg time of year) from areas to be used for recreation	Establish tolerance distance measures[1] (exclusion zones) between/around habitat and facilities to minimise disturbance – calibrated by needs of most sensitive species
Ensure that sufficient habitat is available elsewhere in the landscape for displaced wildlife to re-congregate	Where it is not possible to ensure adequate separation between habitat and facility then test availability of functionally connected habitat – if insufficient then feedback to location of facility
Generally improve quality of recreational facilities	Calibrate exclusion zone by quality of facility – eg greater distance required if path in poor state of repair
Account should be taken of likely frequency duration, and intensity of disturbance	Calibrate exclusion zones on basis of disturbance type (eg main road; car park, etc.) – long duration low impact disturbance likely to have more impact then short duration, high impact disturbance, therefore will have wider exclusion zone

1 A set of potential tolerance distances are given in Just Ecology (2005)

The principles outlined in Table 10 are largely concerned with exclusion zone analysis, but there is also potential in the methodology for including the effects of amelioration measures such as provision of "hides" instead of paths, which would not reduce the recreational value of the network, but would increase its ecological robustness. Amelioration methods could be given separate scores.

There are sources of recreation data available in GIS. For example the Scottish Path Record (SPR) is a large database including all mapped lines obtained from the OS product OSCAR (www.ordnancesurvey.gov.uk). OSCAR provides details of all roads and paths updated by Local Authorities (LA) as and when they obtain new information. However, very little of the original linework came from local knowledge and subsequently it would seem that a fair number of paths exist on the ground that were not in any of the source datasets (L. Renwick *pers. comm.*). Unfortunately the SPR is not a single dataset. Each LA, plus the two National Park Authorities are responsible for maintenance of their part of the SPR. Some LAs have updated the data, some have not. Some plan to alter the data structure, often to fit existing systems or ways of working.

In addition to the SPR there are also other recreation- related datasets which would be useful in any network analysis such as: The National Monument Record for Scotland; The Record of Scheduled Ancient Monuments, and The Historic Gardens and Designed Landscapes Record. These are likely to indicate constraints on network expansion.

The potential for complexity in the analyses of recreational analysis and the approach suggested here would have to be tested in case-study areas in order to evaluate the iterative process between ecological and recreational design requirements.

13 CONCLUSIONS AND RECOMMENDATIONS FOR DEVELOPING LANDSCAPE EVALUATION TOOLS

13.1 Introduction

In this section we draw together the recommendations and conclusions from preceding sections and outline how tools might be developed and integrated to aid in the implementation of lowland habitat networks. Clearly, three different types of evaluation tool can be envisaged: ecological, landscape character and recreation. If these could be integrated within a GIS framework, then there is an opportunity to develop a powerful decision support tool with good end-user functionality. Ideally tools which evaluate socio-economic impacts and other environmental impacts (eg hydrological cycles) should also be included in this suite (eg – Thompson *et al.*, 2003), but are beyond the scope of this present study. There is scope for post-hoc evaluation of model outputs but this would best be undertaken as part of a case-study assessment.

13.2 End-users and spatial scale

In section four we discussed the importance of defining scale in constructing habitat networks. The key is to achieve a compromise between the requirements of species and end-user objectives/questions and also to account for data availability and quality. The principle end-users identified were strategic planners rather than individual farmers, with the focus on evaluating how groups of farms can work together to achieve landscape-scale objectives. As a pragmatic rule of thumb, it is suggested that areas of approximately $200km^2$ (eg sub-catchment scale) might be the most appropriate for case-studies of landscape change.

13.3 Focal species modelling

Focal-species modelling using either generic or specific species profiles is recommended as a tool for the ecological evaluation of landscape change as it emphasises functional connectivity, as opposed to physical connectivity, as the key to assessing network quality. Focal species modelling sits mid way on the modelling continuum between simple structure-based models and detailed species-based models. In this respect it offers a practical approach applicable at a range of scales based on robust theoretical assumptions. The BEETLE modelling tool allows for the execution of focal species modelling within a GIS environment. However, there is a need to test the model in lowland case-study areas to test performance with respect to end-user requirements and data availability. In particular the potential value of the SIACS dataset (see section 10.2) as a provider of land cover (and hence habitat) data needs to investigated.

13.4 Landscape character assessment

LCA appears to be a robust and usable method of assessing the landscape impact of network development. The inclusion of Historic Landscape Assessment in the evaluation is also recommended. There is considerable scope to include visualisations of landscape change within the evaluation package using some of the techniques used in the VISULANDS project. The outputs from the character assessment could feedback to the design process to modify design prescriptions if these are seen to overly compromise the visual landscape and overall character of the area.

The suite of LCA does, however, have recognised deficiencies (Martin and Swanwick, 2004) when assessed against the current guidance for the preparation of LCA (Swanwick and Land Use Consultants, 2002). The use of LCA in future work would benefit from the identified shortcomings and weaknesses being resolved and the entire suite brought up to contemporary guidance level. **Also, if the suite of information could be restructured to follow the natural heritage framework of the suite of NHFs it would encourage the use and integration of both datasets**.

13.5 Recreation and access assessment

In section 12 a rule-based approach is proposed for assessing the impacts of recreation on wildlife disturbance. This has important implications for network design as there is no merit in designing habitat networks if they will never be used by the target species on account of disturbance caused by recreation. The approach is to develop rules for exclusion zones around habitats, but allowing options to be tested, changed and re-tested to achieve optimal balance.

13.6 Prospects for development of integrated PC decision support tools

At a recent workshop convened to evaluate the focal species approach (Humphrey, 2004) there was general end-user support for developing BEETLE as a GIS-based decision – support tool, targeted at the informed end-user (eg strategic planners with ecological knowledge). There is considerable scope for integrating focal species-modelling with landscape character and recreational impact modelling within a GIS. ArcView 9 is recommended as the GIS package best suited for the types of integrative analysis required. It is recommended that the network design should be primarily driven by ecological objectives, but that the landscape and recreation modules designed to provide feedback to the ecological modelling process with network designs being further modified until the desired set of landscape-change scenarios are obtained. Three-D visualisations will form an important tool for making scenario-analysis more understandable by end-users. **It is important to note that this vision of an integrated network development and analysis tool is still some way off. The first stage of the process of development is to test whether it is feasible and practical to integrate Landscape Character Assessment, Historical land-use Assessment, ecological and recreational models into one comprehensive analysis package.** Currently the timetable for implementing BEETLE as a GIS-based decision support tool is end 2005.

13.7 Need for case studies to illustrate and test the network analysis tools

Case-study sites are needed to test the network construction and analysis tools described in outline in this report. This review has not demonstrated whether it is possible to have a positive interplay on the ground between FHNs and non-wooded AHNs. In addition, there is a need to develop and pilot the integrated network development and analysis tool kit. The BEETLE model has to be fully tested within lowland agricultural landscapes to evaluate its applicability to end-users and its ability to deal with the habitat and species datasets available. Model testing and evaluation can only be done in the context of real landscapes and data. Ideally 3–4 different case-study landscapes would be selected representing contrasting land-use mixes; for example western pasture-based farmed landscapes v eastern arable v marginal Upland/lowland. The latter case study area will be particularly interesting in that it will highlight issues associated with upland/lowland boundaries such as consequences of overlaps of semi-natural habitats and need for an integrative approach.

14 CREATING AND MAINTAINING HABITAT NETWORKS – OPPORTUNITIES AND CONSTRAINTS

14.1 Introduction

Having discussed the prospects for developing and testing the network analysis tools through case-study analysis (section 13), there is a need to consider the likelihood of networks being established in reality. The aim of this section is to consider some of the opportunities and constraints on network creation and maintenance. This is by no means an exhaustive analysis, but highlights some of the key issues. Further consideration of these will be needed after the case-study phase of the project. The issues are organised into four categories: (1) ecological; (2) practical; (3) economic.

14.2 Ecological issues

Landscape dynamics and successional processes. One of the shortcomings of BEETLE is that, although it is a spatially explicit modelling approach, it is not temporally explicit and takes no account of population or landscape dynamics. The landscape dynamics element is included in the form of different scenarios as defined by the end user. Usually this entails defining different combinations of land-cover types to be achieved within a certain timeframe. In Wales, Watts *et al.* (2004) suggested a time-scale of 50 years for the establishment of the proposed woodland habitat network plan; semi-natural non-wooded habitats could be re-established in 5–10 years depending on site conditions. Nevertheless landscapes are very dynamic over time and management intention could be diluted by natural succession thus compromising the original intention.

1. **Climate change** Climate change is an important driver of landscape change (Opdam and Wascher, 2004) and the implications for habitats and species are only beginning to be analysed (Hossell *et al.*, 2000). Climate change is anticipated to have a proportionately greater impact on short-term agricultural habitats than more “permanent” semi-natural habitats which may not change so quickly. It is likely that network designs which include crop habitats as a dominant element may go out of date rather rapidly. The suite of integrative network development and analysis tools proposed in section 13 would have the potential to track these rapid changes in habitat type and configuration by allowing regular updates of the land-cover data. There is also the opportunity of using BEETLE to model the future effects of climate change since anticipated changes in accumulated temperature, moisture deficit etc., across the UK have been used to predict changes in woodland composition and distribution of major tree species (Broadmeadow, 2002). This work has been further extended to predict the effects of climate change on Natural Heritage Future scenarios for a range of land use types (eg – Hossell *et al.*, In prep). **The practical implications of these predictions could be modelled within the case study areas.**

2. **Biophysical constraints** While climate is a clear constraint (or opportunity depending on perspective) on network development, so too are soils. Where soils have undergone decades of agricultural improvement and there are residual fertility problems, there may be little prospect of restoring semi-natural habitats (be they woodland or open ground) in the short to medium term. In addition, it may be impossible to establish habitats in the places they are needed simply because of biophysical constraints such as topography, terrain and hydrology.

14.3 Practical issues

A range of practical issues which might constrain the development of habitat networks on the ground can be envisaged,

1. **Lack of funding** Agricultural incentives may not be geared specially towards creating networks and therefore there could be lack of incentive for farmers to work together at wider spatial scales.

2. **Land use policy and planning** Watts and Selman (2004) indicate that although there is recognition of the need to deliver action on a 'wider countryside', rather than on a purely site-centred, basis (Anon, 2004a), there are few statutory powers to enforce compliance with spatial rural land-use strategies and biodiversity plans at the landscape scale. The Nature Conservation (Scotland) Act 2004 does confer some additional powers on SNH to make land management orders to ensure protection of SSSIs. However, this may only be applicable in a minority of cases. In most instances, goodwill, persuasion and financial incentives will be the main planning "tools" available to help with getting networks established.

3. **Cultural resistance to change** Even if all planning conditions and incentives are met there may be cultural resistance to change. Local communities may be unwilling to accept changes such as, for example, the addition of more woodland and scrub to the landscape. Farmers are often reluctant to reduce productivity, or accept downgrading of the agricultural value of fields etc. in order to convert to semi-natural habitat. However, the decoupling of production from subsidies may provide increased opportunities for land use change.

4. **Changes in ownership** Ownership may be too transient to allow long-term planning.

5. **Lack of knowledge of restoration methods** Although there is guidance available on how to create and restore semi-natural habitats (eg – Anderson, 2001; Thompson *et al.*, 2003; Pywell *et al.*, 2002) there is little experience of the types of large-scale restoration needed to create habitat networks.

6. **Data availability** In section 10 we reviewed the types of habitat and species data available for constructing habitat networks. These are variable in quality and coverage. Remote sensed land cover data may be out of date or of insufficient accuracy requiring extensive ground truthing before networks can be implemented. This may impact on the time and resources available making it more difficult to establish networks in the desired time-frame. **It is recommended that in the case study phase of this project, some time is invested in ground-truthing remote sensed data before modelling commences.**

14.4 Economic issues

1. **CAP support mechanisms** Scottish agriculture is entering a period of great uncertainty, since it is unclear exactly what impacts the changes to the CAP support mechanisms will have on farming practices, land-use, agricultural landscapes and farmland biodiversity. There appears to be some scope for biodiversity gains to occur on what was previously intensively-managed farmland. However, any such reversal of biodiversity fortunes is not anticipated to be uniform across all agricultural sectors. Indeed, it is likely that dairy farms in particular will continue to have an adverse impact as economic pressures

drive those farmers who continue in this sector to increase herd sizes and the associated area of land that they farm. It is, therefore, important to appreciate that for parts of the lowland landscape, some beneficial land-use change may happen on the ground which either increases the amount and variety of different habitats in the landscape or makes application of the habitat network approach easier (eg through freeing up more land for management specifically with conservation objectives in mind). Conversely, some farm types may put an increasing focus on achieving greater productivity (especially by minimising costs in order to maximise their income from marginal enterprises) and may continue to produce landscapes in need of a greater habitat network focus. Such factors are important to bear in mind when considering where best the habitat network approach should be targeted.

2. **Greater need for collaboration between farmers** What is, however, clear is that to obtain impact at the landscape scale required, there needs to be a greater collaboration between farmers in any one area with regard to setting actions and priorities. The experience gained by SEERAD in allowing collaborative bids into the Rural Stewardship Scheme and the impending roll-out of Land Management Contracts in Scotland should provide a good baseline on which to base the development of potential schemes and approaches to encourage greater consideration of these scales in the decision-making process.

3. **Reform of Rural Development Measures** It is also important to bear in mind that it is likely that changes may be necessary by 2007 to the wider package of Rural Development measures available in Scotland. While this may provide opportunities for an increased focus on agri-environment (and hence provide even more of a supporting framework on which to base the habitat network approach), the need to achieve a balance in spend between this and other priorities (measures to support restructuring and economic competitiveness; measures to support rural diversification) may possibly lead to some restrictions on the amount of funds which are available specifically for agri-environment approaches. In addition, focus on the implementation of the Water Framework Directive in Scotland will increase markedly in the coming years. Scotland has been at the forefront on the development on thinking and action on this issues in recent years. It will be important to ensure that consideration of biodiversity issues (and the role for the habitat network approach in delivering these) is included in this process.

4. **Conclusion** All these factors will not only influence the scale and type of land use changes happening on the ground in different areas of Scotland, but just as importantly will have a big impact (through the impacts on farm economics and viability) on the willingness or otherwise of farmers to participate in schemes. It is therefore important that these factors and the wider ecological and economic context in which they are operating are borne in mind during the further development of the habitat network approach.

15 REFERENCES

Adriaensen, F., Chardon, J.P., De Blust, G., Swinnen, E., Villalba, S., Gulinck, H. and Matthysen, E. (2003). The application of 'least-cost' modelling as a functional landscape model. *Landscape and urban planning*, **64**, 233–247.

Ahern, J. (1995). Greenways as a planning strategy. *Landscape and urban planning*, **33**, 131–155.

Akçakaya, H.R. (1994). *RAMAS/GIS: linking landscape data with Population Viability Analysis (version 1.0).* Applied Biomathematics, Setauket, New York.

Akçakaya, H.R. and Atwood, J.L. (1997). A habitat-based metapopulation model of the California Gnatcatcher. *Conservation Biology*, **11**, 422–34.

Anderson, A.R. (2001). Deforesting and restoring peat bogs: a review. *Forestry Commission Technical Paper* No. 32. Forestry Commission, Edinburgh.

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

Andrews, J. (1993). The reality and management of wildlife corridors. *British Wildlife*, **5**, 1–7.

Angelstam, P. and Andersson, L. (2001). Estimates of the needs for forest reserves in Sweden. *Scandinavian Journal of Forest Research (Suppl.)*, **3**, 38–51.

Anon (1995). *Biodiversity: the UK steering group report. Vol 2. Action Plans*. HMSO, London.

Anon (2000). *Forests for Scotland: The Scottish Forestry Strategy*. Forestry Commission, Scottish Executive, Edinburgh.

Anon (2002). *Custodians of Change. Report of Agriculture and Environment Working Group*. Scottish Executive, Edinburgh.

Anon (2003). *Scottish Forestry Grants Scheme*. Scottish Executive, Forestry Commission, Edinburgh.

Anon (2004a). *Scotland's Biodiversity: It's in Your Hands: A strategy for the conservation and enhancement of biodiversity in Scotland.* Scottish Executive, Edinburgh.

Anon (2004b). *The UK Forestry Standard: the Government's approach to sustainable forestry*. Forestry Commission, Edinburgh.

Arrhenius, O. (1921). Species and area. *Journal of Ecology*, 95–99.

Asher, J., Warren, M., Fox, R., Harding, P., Jeffcoate, G. and Jeffcoate, S. (2001). *The Millennium Atlas of Butterflies in Britain and Ireland.* Oxford University Press, Oxford.

Bailey, N., Clements, T., Lee, J.T. and Thompson, S. (2003). Modelling soil series data to facilitate targeted habitat restoration: a polytomous logistic regression approach. *Journal of Environmental Management*, **67**, 395–407.

Barker, A.M. (2004). Insects as food for farmland birds: is there a problem? In: *Insects and Bird Interactions*. H.F. van Emden and M. Rothschild (eds). Intercept, Andover. pp 37–50.

Barker, G. (1997). A framework for the future: green networks with multiple uses in and around towns and cities. *English Nature Research report*. No. 256. English Nature, Peterborough.

Bell, S. (1998). Forest design planning: a guide to good practice. *Forestry Commission Practice Guide*. No. 012. Forestry Commission, Edinburgh.

Bender, L.C., Roloff, G.J., Haufler, J.B. (1996). Evaluating confidence intervals for habitat suitability models. *Wildlife Society Bulletin*, **24**, 347–352.

Bennett, A.F. (2003). *Linkages in the landscape: the role of corridors and connectivity in wildlife conservation*. IUCN, Gland, Switzerland.

Benton, T.G., Vickery, J.A. and Wilson, J.D. (2003). Farmland biodiversity: is habitat heterogeneity the key? *Trends in Ecology and Evolution*, **18**, 182–188.

Bignal, E.M. and McCracken, D.I. (2000). The nature conservation value of European traditional farming systems. *Environmental Reviews*, **8**, 149–171.

Bolck, M., de Togni, G., van der Sluis, T. and Jongman, R.H.G. (2004). From models to reality: design and implementation process. In: *Ecological Networks and Greenways: Concepts, Design, Implementation*. R.H.G. Jongman and G. Pungetti (eds.). Cambridge University Press, Cambridge. pp 128–149.

Bouwma, I.M., Jongman, R.G.H. and Butovsky, R.O. (2002). *Indicative map of the Pan-European Ecological Network for Central and Eastern Europe*. European Centre for Nature Conservation, Tilburg, Netherlands.

British Geological Survey (1979).

Broadmeadow, M. (2002). *Climate change: impacts on UK forests*. Forestry Commission Bulletin. No. 125. Forestry Commission, Edinburgh.

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

Bruinderink, G.G., Van Der Sluis, T., Lammertsma, D., Opdam, P. and Pouwels, R. (2003). Designing a coherent ecological network for large Mammals in Northwestern Europe. *Conservation Biology*, **17**, 549–557.

Buckley, G.P. and Fraser, S. (1998). Locating new lowland woods. *English Nature Research Report*. No. 283. English Nature, Peterborough.

Cajander, A.K. (1926). The theory of forest types. *Acta Forestalia Fennica*, **29**, 1–108.

Chamberlain, D.E. and Fuller, R.J. (2001). Contrasting patterns of change in the distribution and abundance of farmland birds in relation to farming system in lowland Britain. *Global Ecology & Biogeography*, **10**, 399–409.

Chardon, J.P., Adriaensen, F. and Matthysen, E. (2003). Incorporating landscape elements into a connectivity measure: a case study for the Speckled wood butterfly (*Pararge aegeria* L.). *Landscape Ecology*, **18**, 561–573.

Cole, L.J., McCracken, D.I., Downie, I.S., Dennis, P., Foster, G.N., Waterhouse, A., Murphy, K.J., Griffen, A.L. and Kennedy, M.P. (In press). Comparing the effects of farming practices on ground beetle (Coleoptera: Carabidae) and spider (Araneae) assemblages on Scottish farmland. *Biodiversity & Conservation*.

Connor, E.F. and McCoy, E.D. (1979). The statistics and biology of the species-area relationship. *American Naturalist*, **113**, 791–833.

Corney, P.M., Le Duc, M.G., Smart, S.M., Kirby, K.J., Bunce, R.G.H. and Marrs, R.H. (2004). The effect of landscape-scale environmental drivers on the vegetation composition of British woodlands. *Biological Conservation*, **120**, 491–505.

Cowley, M.J.R., Wilson, R.J., León-cortés, J.L., Gutiérrez, D., Bulman, C.R. and Thomas, C.D. (2000). Habitat-based statistical models for predicting the spatial distribution of butterflies and day-flying moths in a fragmented landscape. *Journal of Applied Ecology*, **37**, 60–72.

Dennis, P., Cole, L.J., McCracken, D.I., Griffen, A.L., Downie, I.S., Murphy, K.J., Furness, R.W., Foster, G.N. and Waterhouse, A. (in press). Physiographic and land use factors affecting the distribution of plant and animal diversity in a Scottish river catchment. *Journal of Environmental Management.*

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

Dolman, P.M. and Fuller, R. (2003). The process of species colonisation in wooded landscapes: a review of principles. In: *The Restoration of Wooded Landscapes – Proceedings of a conference held at Heriot Watt University, Edinburgh, 14–15 September 2000*. J. Humphrey, A. Newton, J. Latham, H. Gray, K.H. Kirby, E. Poulsom and C. Quine (eds). Forestry Commission, Edinburgh. pp 25–36.

Donald, P.F., Green, R.E. and Heath, M.F. (2000). Agricultural intensification and the collapse of Europe's farmland bird populations. *Proceedings of the Royal Society of London Series B*, **268**, 25–29.

Donovan, M.L., Rabe, D.L. and Olson, C.E. (1987). Use of Geographic Information Systems to develop habitat suitability models. *Wildlife Society Bulletin*, **15**, 574–79.

Dover, J.W. (2000). Human, environmental and wildlife aspects of corridors with specific reference to UK planning practice. *Landscape Research*, **25**, 333–344.

Dover, J.W. and Bunce, R.G.H. (eds) (1998). *Key concepts in landscape ecology*. Proceedings of the 1998 IALE European Congress, held at Myerscough College, Preston, 3rd-5th September 1998. IALE (UK).

Dunning, J.B. and Stewart, D.J. (1995). Spatially explicit population models: current forms and future uses. *Ecological Applications*, **5**, 3–11.

Ellis, N.E. and Munro, K. (2004). A preliminary review of the distribution and extent of BAP priority habitats across Scotland. *Scottish Natural Heritage Commissioned Report*. No.044 (ROAME No. F00NA02). Scottish Natural Heritage, Edinburgh.

Engler, R., Guisan, A. and Rechsteiner, L. (2004). An improved approach for predicting the distribution of rare and endangered species from occurrence and pseudo-absence data. *Journal of Applied Ecology*, **41**, 263–274.

English Nature (2002). *Old meadows and pastures: ancient and threatened habitats.* English Nature Booklet. English Nature, Peterborough.

Environmental Resources Management (1998). *Landscape design guidance for forests and woodlands in Dumfries and Galloway.* Forestry Commission, Scotland.

Environmental Resources Management (2000). *Langholm/Lockerbie and Galloway Local Forestry Frameworks.* Dumfries and Galloway Council, Dumfries.

European Commission (2000). From land cover to landscape diversity in the European Union. Europa, Brussels.

Eyre, M.D., Rushton, S.P., Luffa, M.L. and Telfer, M.G. (2004). Predicting the distribution of ground beetle species (Coleoptera,Carabidae) in Britain using land cover variables. *Journal of Environmental Management*, **72**, 163–174.

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

Fairclough, G. and Macinnes, L. (undated). *Landscape Character Assessment guidance for England and Scotland. Topic Paper 5: Understanding Historic Landscape Character.* Scottish Natural Heritage and The Countryside Agency, Edinburgh.

Farina, A. (1998). *Principles and methods in landscape ecology.* Chapman and Hall, London.

Ferris, R., Purdy, K., Humphrey, J. and Quine, C. (2000). An introduction to new landscape ecology research to enhance biodiversity in British forests. *Forestry Commission Information Note* No. 34. Forestry Commission, Edinburgh.

Finney, S.K., Pearce-Higgins, J.W. and Yalden, D.W. (2005). The effect of recreational disturbance on an upland breeding bird, the golden plover *Pluvialis apricaria*. *Biological Conservation*, **121**, 53–63.

Foppen, R.P.B., Bouwma, I.M., Kalkhoven, J.T.R., Dirksen, J. and van Opstal, S. (2000). Corridors of the Pan-European Ecological Network: concepts and examples for terrestrial and freshwater vertebrates. *ECNC Technical report series.* European Centre for Nature Conservation, Tilburg, The Netherlands.

Forestry Commission (1994). *Forest landscape design guidelines.* Forestry Commission, Edinburgh.

Forestry Commission (2001). Completing the "Jigsaw". *GLEAN – Forestry Commission Grants and Licences News.* Issue 10 (January, 2001). Forestry Commission, Edinburgh.

Forestry Commission (2002). *National inventory of woodland and trees*: Scotland. Forestry Commission, Edinburgh.

Forestry Commission (2004). *Forestry facts & figures 2004.* National Statistics Publication. Economics & Statistics, Forestry Commission, Edinburgh.

Forman, R.T.T. and Godron, M. (1986). *Landscape ecology.* Wiley and Sons, New York.

Fowler, J. and Stiven, R. (2003). *Habitat networks for wildlife and people: the creation of sustainable forest habitats.* Forestry Commission Scotland, Scottish Natural Heritage, Edinburgh.

Fowles, A.P. (2003). Guidance notes for the definition and mapping of habitat quality for marsh fritillaries. *Countryside Council for Wales Natural Science Report*. No. 03/5/01. Countryside Council for Wales, Bangor.

Frank, K. (2004). Ecologically differentiated rules of thumb for habitat network design – lessons from a formula. *Biodiversity and Conservation*, **13**, 189–206.

Franklin, J.F. (1989). Toward a new forestry. *American Forests*, **December 1989**, 37–44.

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

Freckleton, R.P. and Watkinson, A.R. (2003). Are all plant populations metapopulations? *Journal of Ecology*, **91**, 321–324.

Freudenberger, D. and Brooker, L. (2004). Development of the Focal Species Approach for Biodiversity Conservation in the Temperate Agricultural Zones of Australia. *Biodiversity and Conservation*, **13**, 253–274.

Fuller, R.M., Smith, G.M., Sanderson, J.M., Hill, R.A. and Thomson, A.G. (2001). *The UK Land Cover Map 2000: construction of a parcel-based vector map from satellite images*. Centre for Ecology & Hydrology, Cambridge.

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

Gardner, R.H. and O'Neill, R.V. (1991). Pattern, process and predictability: the use of neutral models for landscape analysis. In: *Quantitative Methods in Landscape Ecology: The Analysis and Interpretation of Landscape Heterogeneity*. M.G. Turner and R.H. Gardner (Eds.). Springer-Verlag, New York. pp 289–307.

Gaston, K.J., Pressey, R.L. and Margules, C.R. (2002). Persistence and vulnerability: retaining biodiversity in the landscape and in protected areas. *Journal of Biosciences*, **27**, 361–384.

Gergel, S.E. and Turner, M.G. (eds) (2002). *Learning landscape ecology: a practical guide to concepts and techniques*. Springer-Verlag, New York.

Gibson, L.A., Wilson, B.A., Cahill, D.M. and Hill, J. (2004). Spatial prediction of rufous bristlebird habitat in a coastal heathland: a GIS-based approach. *Journal of Applied Ecology*, **41**, 213–223.

Gilbert, F.S. (1980). The equilibrium theory of island biogeography: fact or fiction? *Journal of Biogeography*, **7**, 209–235.

Gkaraveli, A., Good, J.E.G. and Williams, J.H. (2004). Determining priority areas for native woodland expansion and restoration in Snowdonia National Park, Wales. *Biological Conservation*, **115**, 395–402.

Goodman, D. (1987). The demography of chance extinction. In: *Viable Populations for Conservation*. M.E. Soulé (Ed.). Cambridge University Press, Cambridge.

Gray, H. and Stone, D. (2003). Applications of spatial data in strategic woodland decisions: an example from the Isle of Mull. In: *The Restoration of Wooded Landscapes*. Proceedings of a conference held at Heriot Watt University, Edinburgh, 14–15 September 2000, J.W. Humphrey, A.C. Newton, J. Latham, H. Gray, K.J. Kirby, C.P. Quine and E. Poulsom (Eds.). Forestry Commission, Edinburgh. pp. 69–76.

Greene, D.F., Canham, C.D., Coates, K.D. and Lepage, P.T. (2004). An evaluation of alternative dispersal functions for trees. *Journal of Ecology*, **92**, 758–766.

Gustafson, E.J. (1998). Quantifying landscape spatial pattern: what is the state of the art? *Ecosystems*, **1**, 143–156.

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

Hanski, I. (1999). *Metapopulation ecology*. Oxford University Press, Oxford.

Hanski, I. and Gilpin, M. (1991). Metapopulation dynamics: brief history and conceptual domain. *Biological Journal of the Linnean Society*, **42**, 3–16.

Hanski, I.E. and Gilpin, M.E. (1997). *Metapopulation biology: ecology, genetics and evolution*. Academic Press, New York.

Harmer, R. (1999). Using natural colonisation to create or expand new woodlands. *Forestry Commission Information Note*. No. 23. Forestry Commission, Edinburgh.

Hawkins, V. and Selman, P. (2002). Landscape scale planning: exploring alternative land use scenarios. *Landscape and urban planning*, **60**, 211–224.

Heikkinen, R.K., Luoto, M., Virkkala, R. and Rainio, K. (2004). Effects of habitat cover, landscape structure and spatial variables on the abundance of birds in an agricultural-forest mosaic. *Journal of Applied Ecology*, **41**, 824–835.

Henle, K., Davies, K.F., Kleyer, M., Margules, C. and Settele, J. (2004a). Predictors of species sensitivity to fragmentation. *Biodiversity and Conservation*, **13**, 207–251.

Henle, K., Lindenmayer, D.B., Margules, C.R., Saunders, D.A. and Wissel, C. (2004b). Species survival in fragmented landscapes: where are we now? *Biodiversity and Conservation*, **13**, 1–8.

Hepburn, L.V. (2000). Establishment of a register of species-rich hedgerows in Scotland 1998–1999. *Scottish Natural Heritage Commissioned Report*. F99AA104. Scottish Natural Heritage, Edinburgh.

Hodcroft, D. and Alexander, D. (2004). Ecological frameworks in north west England. *Town and Country Planning*, **October 2004**, 282–284.

Hofmann, H. and Marktanner, T. (1995). *Butterflies and moths of Britain and Europe*. Harper Collins, London.

Holland, J.M. (2004). The impact of agriculture and some solutions for arthropods and birds. In: *Insects and Bird Interactions*. H.F. van Emden and M. Rothschild (Eds.). Intercept, Andover. pp 51–73.

Hope, J.C.E. (2003). *Modelling forest landscape dynamics in Glen Affric, northern Scotland*. Unpublished PhD thesis. Department of Environmental Sciences, University of Stirling, Stirling.

Hope, J.C.E. and Humphrey, J.W. (2004). Ecological continuity and spatio-temporal connectivity: modelled response of the lichen *Bryoria furcellata* to landscape dynamics. In: *Landscape Ecology of Trees and Forests*. Proceedings of the twelfth annual IALE(UK) conference, held at the Royal Agricultural College, Cirencester, 21–24 June 2004, R. Smithers (Ed.). IALE, Stoke on Trent. pp 35–42.

Hossell, J.E., Briggs, B. and Hepburn, I.R. (2000). *Climate change and UK nature conservation: a review of the impact of climate change on UK species and habitat conservation Policy.* HMSO, London

Hossell, J.E., Clemence, B. and Scanlon, S. (In prep). Scenarios of the future – Scotland under climate change: farmland. *Scottish Natural Heritage Commissioned Research report.* Scottish Natural Heritage, Edinburgh.

Hudgens, B.R. and Haddad, N.M. (2003). Predicting which species will benefit from corridors in fragmented landscapes from population growth models. *The American Naturalist*, **161**, 808–820.

Humphrey, J.W. (2004). *Balancing upland and woodland strategic priorities phase 2: application of ecological landscape modelling.* Report on end-user workshop, Fisher's Hotel, Pitlochry, 24 March 2004. Unpublished contract report to Scottish Natural Heritage and Forestry Commission Scotland. Forest Research, Roslin, Midlothian.

Humphrey, J.W., Ferris, R. and Quine, C.P. (eds) (2003a). *Biodiversity in Britain's planted forests: results from the Forestry Commission's biodiversity assessment project.* Forestry Commission, Edinburgh.

Humphrey, J.W., Hope, J.C.E. and Poulsom, eg (2004a). *Balancing upland and woodland strategic priorities – phase 2.* Unpublished contract report to Forestry Commission Scotland and Scottish Natural Heritage. Forest Research, Roslin, Midlothian.

Humphrey, J.W., Newton, A.C., Latham, J., Gray, H., Kirby, K.J., Quine, C.P. and Poulsom, eg (eds) (2003b). *The restoration of wooded landscapes.* Proceedings of the conference 'Restoration of Wooded Landscapes' Heriot Watt University, Edinburgh, 12–14 September 2000. Forestry Commission, Edinburgh.

Humphrey, J.W., Ray, D., Watts, K., Brown, C., Poulsom, E.G., Griffiths, M. and Broome, A.C. (2004b). Balancing upland and woodland strategic priorities. *Scottish Natural Heritage Commissioned Report.* No. 037 (ROAME No. F02AA101). Scottish Natural Heritage, Edinburgh.

Humphrey, J.W. and Watts, K. (2005). Biodiversity indicators for UK managed forests: development and implementation at different spatial scales. In: *Monitoring and indicators of forest biodiversity in Europe – from ideas to operationality.* Proceedings of the EFI/UFRO Conference 12–15 November 2003. M. Marchetti and T.-B. Larsson (Eds.). European Forest Institute, Joennsu.

Hutton, S.A. and Giller, P.S. (2003). The effects of the intensification of agriculture on northern temperate dung beetle communities. *Journal of Applied Ecology*, **40**, 994–1007.

Jongman, R.H.G. (1995). Nature conservation planning in Europe: developing ecological networks. *Landscape and urban planning*, **32**, 169–183.

Jongman, R.H.G. and Pungetti, G. (2004a). Introduction: ecological networks and greenways. In: *Ecological Networks and Greenways: Concepts, Design, Implementation.* R.H.G. Jongman and G. Pungetti (Eds.). Cambridge University Press, Cambridge. pp 1–4.

Jongman, R.H.G. and Pungetti, G. (eds) (2004b). *Ecological networks and greenways: concepts, design, implementation.* Cambridge University Press, Cambridge.

Joyce, K.A., Holland, J.M. and Doncaster, C.P. (1999). Influences of hedgerow intersections and gaps on the movement of carabid beetles. *Bulletin of Entomological Research*, **89**, 523–531.

Just Ecology (2005). Review of recreational disturbance research on selected wildlife in Scotland. *Scottish Natural Heritage Commissioned Report.* AB(03AA612)030453. Scottish Natural Heritage, Edinburgh.

Kintsch, J.A. and Urban, D.L. (2002). Focal species, community representation and physical proxies as conservation strategies: a case study in the Amphibolite Mountains, North Carolina, USA. *Conservation Biology*, **16**, 936–947.

Kirby, K.J. (1995). Rebuilding the English countryside: habitat fragmentation and wildlife corridors as issues in practical conservation. *English Nature Science Report* No.10. English Nature, Peterborough.

Kleijn, D. and Sutherland, W.J. (2003). How effective are European agri-environment schemes in conserving and promoting biodiversity? *Journal of Applied Ecology*, **40**, 947–969.

Klinka, K., Karajina, V.J., Ceska, A. and Scagel, A.M. (1989). *Indicator plants of coastal British Columbia*. UBC Press, Vancouver.

Kortland, K. (2003). Multi-scale forest habitat management for Capercaillie. *Scottish Forestry*, **57**, 91–95.

Kreyer, D., Oed, A., Walther-Hellwig, K. and Frankl, R. (2004). Are forests potential landscape barriers for foraging bumblebees? Landscape scale experiments with *Bombus terrestris* agg. and *Bombus pascuorum* (Hymenoptera, Apidae). *Biological Conservation*, **116**, 111–118.

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

Lambeck, R.J. (2002). Focal species and restoration ecology: response to Lindenmayer et al. *Conservation Biology*, **16**, 549–551.

Lande, R. (1996). The meaning of quantitative genetic variation in evolution and conservation. In: *Biodiversity in Managed Landscapes: Theory and Practice*. R.C. Szaro and D.W. Johnston (Eds.). Oxford University Press, New York. pp 27–40.

Lee, J.T., Bailey, N. and Thompson, S. (2002). Using geographical information systems to identify and target sites for creation and restoration of native woodlands: a case study of the Chiltern Hills, UK. *Journal of Environmental Management*, **64**, 25–34.

Lee, J.T., Woody, S.J. and Thompson, S. (2001). Targeting sites for conservation: Using a patch-based ranking scheme to assess conservation potential. *Journal of Environmental Management*, **61**, 367–380.

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

Levins, R. (1970). Extinction. In: *Some Mathematical Questions in Biology – Lectures in Mathematics in the Life Sciences*. M. Gertenshaubert (Ed.). American Mathematical Society, Providence, Rhode Island. pp 77–107.

Li, H. and Wu, J. (2004). Use and misuse of landscape indices. *Landscape Ecology*, **19**, 389–399.

Lindenmayer, D.B., Cunningham, R.B., Donnelly, C.F. and Leslie, R. (2002). On the use of landscape surrogates as ecological indicators in fragmented forests. *Forest Ecology and Management*, **159**, 203–216.

Lindenmayer, D.B. and Fischer, J. (2002). Sound science or social hook – a response to Brooker's application of the focal species approach. *Landscape and urban planning*, **62**, 149–158.

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

MacDonald, M.A. (2003). The role of corridors in biodiversity conservation in production forest landscapes. *Tasforests*, **14**, 41–52.

Mackintosh, J., Hawker, D., Munro, K. and Smith, M. (2004). The distribution and extent of lowland grassland National Vegetation Classification (NVC) types and Biodiversity Action Plan (BAP) habitats in Scotland. *Scottish Natural Heritage Commissioned Report*. No. 009 (ROAME No. F97AA100). Scottish Natural Heritage, Edinburgh.

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

Marren, P. (1992). *The wild woods: a regional guide to Britain's ancient woodland*. David and Charles plc, Newton Abbot, Devon.

Martin, J. and Swanwick, C. (2004). Overview of Scotland's national programme of Landscape Character Assessment. *Scottish Natural Heritage Commissioned Report*. No. 029 (ROAME No. F03AA307). Scottish Natural Heritage, Edinburgh.

Mathieson, S. (2001). Water vole species action plan. *Species Action Plan. Stirling Local Biodiversity Action Plan*. Stirling.

Maudsley, M.J. (2000). A review of the ecology and conservation of hedgerow invertebrates in Britain. *Journal of Environmental Management*, **60**, 65–76.

McCollin, D., Jackson, J.I., Bunce, R.G.H., Barr, C.J. and Stuart, R. (2000). Hedgerows as habitat for woodland plants. *Journal of Environmental Management*, **60**, 77–90.

McCracken, D.I. (2004). *Farmland and high nature value areas*. Contribution prepared for ECN/ETB/NPB input to EEA Sub-report Halting Biodiversity Loss. Scottish Agricultural College, Auchencruive, Ayr.

McCracken, D.I. and Bain, C. (2000). Farmlands and lowland grasslands. In: *Action for Scotland's Biodiversity*. M.B. Usher, C. Bain and S. Kerr (Eds.). Scottish Biodiversity Group, Edinburgh. pp 49–61.

McCracken, D.I. and Tallowin, J.R. (In press). Swards and structure: the interactions between farming practices and bird food resources in lowland grasslands. *Ibis*.

McGowan, G.M., Palmer, S.C.F., French, D.D., Barr, C.J., Howard, D.C., Smart, S.M., Mackey, E.C. and Shewry, M.C. (2002). Trends in broad habitats: Scotland 1990–1998. *Scottish Natural Heritage Commissioned Report*. F00NB03. Scottish Natural Heritage, Edinburgh.

Melbourne, B.A., Davies, K.F., Margules, C.R., Lindenmayer, D.B., Saunders, D.A., Wissel, C. and Henle, K. (2004). Species survival in fragmented landscapes: where to from here? *Biodiversity and Conservation*, **13**, 275–284.

MLURI (1993). *The Land cover of Scotland 1988. LCS88 Final Report*. The Macaulay Land Use Research Institute, Aberdeen.

Moser, D., Zechmeister, H.G., Plutzar, C., Sauberer, N., Wrbka, T. and Grabherr, G. (2002). Landscape patch shape complexity as an effective measure for plant species richness in rural landscapes. *Landscape Ecology*, **17**, 657–669.

Murrell, D.J. and Law, R. (2000). Beetles in fragmented woodlands: a formal framework for dynamics of movement in ecological landscapes. *Journal of Animal Ecology*, **69**, 471–483.

NCC (1990). *Handbook for Phase 1 Habitat Survey: a technique for environmental audit.* Reprinted by the Joint Nature Conservation Committee in 1993 and 2003 with minor corrections. Nature Conservancy Council, Peterborough.

Noss, R.F., Carroll, C., Vance-Borland, K. and Wuerther, G. (2002). A multi-criteria assessment of the irreplaceability and vulnerability of sites in the Greater Yellowstone Ecosystem. *Conservation Biology*, **16**, 895–908.

Nowicki, P. (in prep). *Ecosystem approach applied to spatial planning*. A report of the BIOFORUM project.

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

Opdam, P. (2002). Assessing the conservation potential of habitat networks. In: *Applying Landscape Ecology in Biological Conservation*. K.J. Gutzwiller (Ed.) Springer-Verlag, New York. pp 381–404.

Opdam, P., Foppen, R. and Vos, C. (2002). Bridging the gap between ecology and spatial planning in landscape ecology. *Landscape Ecology*, **16**, 767–779.

Opdam, P., Verboom, J. and Pouwels, R. (2003). Landscape cohesion: an index for the conservation potential of landscapes for biodiversity. *Landscape Ecology*, **18**, 113–126.

Opdam, P. and Wascher, D. (2004). Climate change meets habitat fragmentation: linking landscape and biogeographical scale levels in research and conservation. *Biological Conservation*, **117**, 285–297.

Ostermann, O.P. (1998). The need for management of nature conservation sites designated under Natura 2000. *Journal of Applied Ecology*, **35**, 968–973.

Ovaskainen, O. and Hanski, I. (2004). From individual behavior to metapopulation dynamics: unifying the patchy population and classic metapopulation models. *The American Naturalist*, **164**, 364–377.

Ozanne, C.M.P., Speight, M.R., Hambler, C. and Evans, H.F. (2000). Isolated trees and forest patches: patterns in canopy arthropod abundance and diversity in *Pinus sylvestris* (Scots pine). *Forest Ecology and Management*, **137**, 53–63.

Pannell, J.P. and Obbard, D.J. (2003). Probing the primacy of the patch: what makes a metapopulation? *Journal of Ecology*, **91**, 485–488.

Parr, R., Moss, R., Marquiss, M., Redpath, S., Newton, I. and Picozzi, N. (1995). *Optimising upland habitat mosaics*. Unpublished Report Contract Report to Scottish Natural Heritage. Institute of Terrestrial Ecology, Banchory.

Patterson, G.S. and Anderson, A.R. (2000). Forests and peatland habitats. *Forestry Commission Guideline Note*. No. 1. Forestry Commission, Edinburgh.

Peach, W.J., Denny, M., Cotton, P.A., Hill, I.F., Gruar, D., Barritt, D., Impey, A. and Mallord, J. (2004). Habitat selection by song thrushes in stable and declining farmland populations. *Journal of Applied Ecology*, **41**, 275–293.

Peck, S. (1998). *Planning for biodiversity: issues and examples*. Island Press, Washington.

Peterken, G.F. (1999). Clyde Valley forest habitat network. *Scottish Natural Heritage Commissioned Report*. F99LI09. Scottish Natural Heritage, Edinburgh.

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

Peterken, G.F. (2002). *Reversing the habitat fragmentation of British woodlands*. WWF-UK, Godalming, Surrey.

Peterken, G.F. (2003). Developing forest habitat networks in Scotland. In: *The Restoration of Wooded Landscapes. Proceedings of the conference 'Restoration of Wooded Landscapes' Heriot Watt University, 12–14 September 2000.*, J.W. Humphrey, A.C. Newton, J. Latham, H. Gray, K.J. Kirby, E. Poulsom and C.P. Quine (Eds). Forestry Commission, Edinburgh. pp 85–92.

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

Petersen, I., Masters, Z., Hildrew, A.G. and Ormerod, S.J. (2004). Dispersal of adult aquatic insects in catchments of differing land use. *Journal of Applied Ecology*, **41**, 934–950.

Petit, S., Griffiths, L., Smart, S.S., Smith, G.M., Stuart, R.C. and Wright, S.M. (2004). Effects of area and isolation of woodland patches on herbaceous plant species richness across Great Britain. *Landscape Ecology*, **19**, 463–472.

Piessens, K., Honnay, O. and Hermy, M. (2005). The role of fragment area and isolation in the conservation of heathland species. *Biological Conservation*, **122**, 61–69.

Porter, K. and Preston, S. (2001). *Lifescapes progress report*. Unpublished report to General Committee of Council. English Nature, Peterborough.

Porter, K. and Wright, R. (2003). *Working at a "landscape" scale: the role of English Nature*. Unpublished paper to General Committee of Council. GC P03 32. English Nature, Peterborough.

Pyatt, D.G., Ray, D. and Fletcher, J. (2001). An ecological site classification for forestry in Great Britain. *Forestry Commission Bulletin* No.124. Forestry Commission, Edinburgh.

Pyatt, D.G. and Suárez, J.C. (1997). An ecological site classification for forestry in Great Britain with special reference to Grampian, Scotland. *Forestry Commission Bulletin* No.20. Forestry Commission, Edinburgh.

Pywell, R.F., Pakeman, R.J., Allchin, E.A., Bourn, N.A.D., Warman, E.A. and Walker, K.J. (2002). The potential for lowland heath regeneration following plantation removal. *Biological Conservation*, **108**, 247–258.

Quadrat Scotland (2002). The network of wildlife corridors and stepping stones of importance to the biodiversity of East Dunbartonshire. *Scottish Natural Heritage Commissioned Report*. F01LI04b. Scottish Natural Heritage, Edinburgh.

Quelch, P.R. (2000). Upland pasture woodlands in Scotland. Part 1. *Scottish Forestry*, **54**, 209–214.

Quelch, P.R. (2001). Upland pasture woodlands in Scotland. Part II: possible origins and examples. *Scottish Forestry*, **55**, 85–92.

Quine, C.P. and White, I.M.S. (1993). Revised windiness scores for the windthrow hazard classification: the revised scoring method. *Forestry Commission Research Information Note*. 230. Forestry Commission, Edinburgh.

Ratcliffe, P.R., Peterken, G.F. and Hampson, A. (1998). A forest habitat network for the Cairngorms. *Scottish Natural Heritage Research, Survey and Monitoring Report*. No. 114. Scottish Natural Heritage, Battleby, Redgorton, Perth.

Ray, D. (2001). *Ecological site classification decision support system (ESC-DSS)*. Forestry Commission, Edinburgh.

Ray, D., Clare, J. and Purdy, K. (2003). Applying an ecological site classification to woodland design at varying spatial scales. In: *The Restoration of Wooded Landscapes – Proceedings of a conference held at Heriot Watt University, Edinburgh, 14–15 September 2000*. J. Humphrey, A. Newton, J. Latham, H. Gray, K.H. Kirby, E. Poulsom and C. Quine (Eds.). Forestry Commission, Edinburgh. pp 59–68.

Ray, D., Hope, J.C.E. and Watts, K. (2004a). *Development of a forest habitat network strategy in West Lothian*. Unpublished Contract Report to Scottish Natural Heritage. Forest Research, Edinburgh.

Ray, D., Hope, J.C.E. and Watts, K. (2004b). Development of a Forest Habitat Network Strategy in West Lothian. *Scottish Natural Heritage Commissioned Report*. No. 049 (ROAME No. F03L J15). Scottish Natural Heritage, Edinburgh.

Ray, D., Sterling, M., Broome, A.C., Fletcher, J., Humphrey, J.W. and Pyatt, D.G. (1999). *Potential for new native woodlands in the Ochils*. Unpublished Contract report to Friends of the Ochils. Forest Research, Roslin, Midlothian.

Ray, D., Watts, K., Hope, J.C.E. and Humphrey, J.W. (2004c). Developing forest habitat networks in southern Scotland. In: *Landscape Ecology of Trees and Forests*. Proceedings of the twelfth annual IALE(UK) conference, held at the Royal Agricultural College, Cirencester, 21–24 June 2004, R. Smithers (Ed.). IALE UK, Stoke on Trent. pp 216–223.

Reed, T.M. (1983). The role of species-area relationships in reserve choice: a British example. *Biological Conservation*, **25**, 263–271.

Ricketts, T.H. (2001). The matrix matters: effective isolation in fragmented landscapes. *The American Naturalist*, **158**, 87–99.

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

Robinson, R.A. and Sutherland, W.J. (2002). Post-war changes in arable farming and biodiversity in Great Britain. *Journal of Applied Ecology*, **39**, 157–176.

Robinson, R.A., Wilson, J.D. and Crick, H.Q.P. (2001). The importance of arable habitat for farmland birds in grassland landscapes. *Journal of Applied Ecology*, **38**, 1059–1069.

Rodwell, J.S.(Ed.) (1991). *British plant communities 1: woodlands and scrub*. Cambridge University Press, Cambridge.

Rushton, S.P., Lurz, P.W.W. and Garson, P.J. (1997). Modelling the distribution of the red and grey squirrel at the landscape scale: a combined GIS and population dynamics approach. *The Journal of Applied Ecology*, **34**, 1137–1154.

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

Scottish Executive (1999). Indicative Forestry Strategies. *Scottish Executive Circular*. Circular 9/1999. Scottish Executive, Edinburgh.

Scottish Executive (2004). *Rural Stewardship Scheme*. SEERAD, Scottish Executive, Edinburgh.

Scottish Natural Heritage (2001). *Natural Care Strategy*. Scottish Natural Heritage, Edinburgh.

Scottish Natural Heritage (2002a). *Natural Heritage Futures and Overview*. Scottish Natural Heritage, Edinburgh.

Scottish Natural Heritage (2002b). *Natural Heritage Futures: farmland*. Scottish Natural Heritage, Edinburgh.

Selman, P. (2000). *Environmental planning – second edition*. Sage Publications, London.

Settele, J., Margules, C., Poschlod, P. and Henle, K. (1996). *Species survival in fragmented landscapes*. Kluwer Academic Publishers, Dordrecht.

Shafer, C.L. (1994). *Nature reserves: island theory and conservation practice*. Smithsonian Institution Press, Washington.

Simberloff, D. (1998). Flagships, umbrellas and keystones: is single-species management passe in the landscape era? *Biological Conservation*, **83**, 247–257.

Smith, D.S. and Helmund, P.C. (1993). *Ecology of greenways*. University of Minnesota Press, Minesota.

Stephens, P.A., Freckleton, R.P., Watkinson, A.R. and Sutherland, W.J. (2003). Predicting the response of farmland bird populations to changing food supplies. *Journal of Applied Ecology*, **40**, 970–983.

Stevens, J.P., Blackstock, T.H., Howe, E.A. and Stevens, D.P. (2004). Repeatability of Phase 1 habitat survey. *Journal of Environmental Management*, **73**, 53–59.

Stewart, K., Bourn, N., Watts, K. and Griffiths, M. (2004). The persistence of a key butterfly species *Boloria selene* (small pearl-bordered fritillary) in Clocaenog Forest and the impact of landscape-scale processes. In: *Landscape Ecology of Trees and Forests – Proceedings of the 2004 IALE (UK) Conference, held at Cirencester Agricultural College, Gloucestershire, 21st–24th June 2004*. R.J. Smithers (Ed.). IALE(UK).

Stewart, K.E., Bourn, N.A.D., Jenny, J. and Roberts, R.E. (2002). *The status, mobility and habitat requirements of the small pearl-bordered fritillary (Boloria selene) in Clocaenog forest: an overview of three year's work*. Unpublished Contract Report to Forest Research No. S02–17. Butterfly Conservation, Dorset.

Sutcliffe, O.L., Bakkestuen, V., Fry, G. and Stabbetorp, O.E. (2003). Modelling the benefits of farmland restoration: methodology and application to butterfly movement. *Landscape and urban planning*, **63**, 15–31.

Swanwick, C. (undated). *Landscape Character Assessment guidance for England and Scotland, Topic Paper 6: techniques and criteria for judging capacity and sensitivity.* Scottish Natural Heritage and The Countryside Agency., Edinburgh.

Swanwick, C. and Land Use Consultants (2002). *Landscape Character Assessment guidance for England and Scotland*. Scottish Natural Heritage, Edinburgh.

Tamis, W.L.M. and VantZelfde, M. (1998). An expert habitat suitability model for the disaggregation of bird survey data. Bird counts in the Netherlands downscaled from atlas block to kilometre cell. *Landscape and urban planning*, **40**, 269.

Tattersall, F.H., Macdonald, D.W., Hart, B.J., Johnson, P., Manley, W. and Feber, R. (2002). Is habitat linearity important for small mammal communities on farmland. *Journal of Applied Ecology*, **39**, 643.

Theobald, D.M. (2003). Targeting conservation action through assessment of protection and exurban threats. *Conservation Biology*, **17**, 1624–1637.

Thomas, J.A. and Snazell, R.G. (1989). Declining fritillaries: the next challenge in the conservation of Britain's butterflies. *Institute of Terrestrial Ecology Annual Report 1988–1989*. Institute of Terrestrial Ecology, Cambridge. pp. 54–56

Thomas, J.A., Snazell, R.G. and Moy, I.L. (in press). *The conservation of violet feeding fritillaries in Southern Britain*. English Nature, Peterborough.

Thomas, R. (2000). English Nature habitat restoration project. *English Nature Research Report*. No. 377. English Nature, Peterborough.

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

Thompson, S., Larcom, A. and Lee, J.T. (1999). Restoring and enhancing rare and threatened habitat under agri-environment agreements: a case study of the Chiltern Hills Area of Outstanding Natural Beauty, UK. *Land Use Policy*, **16**, 93–105.

Thompson, S., McElwee, K. and Lee, J.T. (2001). Using landscape characteristics for targeting habitat conservation and restoration: a case study of ancient semi-natural woodland in the Chiltern Hills Area of Outstanding Natural Beauty, UK. *Landscape Research*, **26**, 203–223.

Thorne, J.F. (1993). Landscape ecology: a foundation for greenway design. In: *Ecology of Greenways: Design and Function of Linear Conservation Areas.* D.S. Smith and P.C. Hellmund (Ed.). University of Minnesota Press, Minneapolis. pp 23–42.

Tischendorf, L., Bender, D.J. and Fahrig, L. (2003). Evaluation of patch isolation metrics in mosaic landscapes for specialist vs. generalist dispersers. *Landscape Ecology*, **18**, 41–50.

Towers, W., Hester, A.J. and Malcolm, A. (2002). The use of the native woodland model for the calculation of local targets for habitat action plans. *Scottish Forestry*, **56**, 196–198.

Townshend, D., Stace, H. and Radley, D. (2004). *State of nature: lowlands – future landscapes for wildlife*. English Nature, Peterborough.

Tudor, O., Dennis, R.L.H., Greatorex-Davies, J.N. and Sparks, T.H. (2004). Flower preferences of woodland butterflies in the UK: nectaring specialists are species of conservation concern. *Biological Conservation*, **119**, 397–403.

Turner, B.G. and Gardner, R.H. (1991). *Quantitative methods in landscape ecology: the analysis and interpretation of landscape heterogeneity*. Springer-Verlag, New York.

UK Biodiversity Group (1998). *Tranche 2 Action Plans. Volume II – terrestrial and freshwater habitats*. English Nature, Peterborough.

UNCED (1992). *Earth Summit. United Nations Conference of Environment and Development. Agenda 21 – action plan for the next century*. The Regency Press, London, Rio de Janeiro.

UNEP (2003). *Item 4.1 of 9th meeting of the subsidiary body on scientific, technical and technological advice*. United Nations Environment Programme, Montreal, Canada.

Usher, M.B., Bain, C. and Kerr, A. (2000). *Action for Scotland's biodiversity. Scottish Biodiversity Group*. The Stationery Office, Edinburgh.

Van Rooij, S., van der Sluis, T., Steingröver, eg and Clarke, S. (2004). Applying landscape ecological methods to analyse and design ecological networks. In: *Landscape ecology of trees and forests*. R. Smithers (Ed.). IALE (UK), Woodland Trust, Grantham. pp 208–215.

Van Rooij, S.A.M., Steingrover, E.G. and Opdam, P.F.M. (2001). *Corridors for life: scenario development of an ecological network in Cheshire County*. Alterra, Wageningen.

Verboom, J., Foppen, R., Chardon, P., Opdam, P. and Luttikhuizen, P. (2001). Introducing the key patch approach for habitat networks with persistent populations: an example for marshland birds. *Biological Conservation*, **100**, 89–101.

Verboom, J. and Pouwels, R. (2004). Ecological functioning of ecological networks: a species perspective. In: *Ecological Networks and Greenways: Concepts, Design, Implementation*. R.H.G. Jongman and G. Pungetti (Eds.). Cambridge University Press, Cambridge. pp 56–70.

Verheyen, K., Velland, M., van Calster, H., Peterken, G.F. and Hermy, M. (2004). Metapopulation dynamics in changing landscapes: a new spatially-realistic model applied to forest plants in central Lincolnshire, United Kingdom. In: *Landscape ecology of trees and forests*. R. Smithers (Ed.). IALE (UK), Woodland Trust, Grantham. pp 19–26.

Vickery, J.A., Tallowin, J.R.B., Feber, R.E., Atkinson, P.W., Asteraki, E.J., Fuller, R.J. and Brown, V.K. (2001). Changes in lowland grassland management: implications for invertebrates and birds. *Journal of Applied Ecology*, **38**, 647–664.

Vos, C., Baveco, H. and Grashof-Bokdam, C.J. (2002). Corridors and species dispersal. In: *Applying Landscape Ecology in Biological Conservation*. K.J. Gutzwiller (Ed.). Springer-Verlag, New York. pp 84–104.

Vos, C.C., Verboom, J., Opdam, P.F.M. and Ter Braak, C.J.F. (2001). Toward ecologically scaled landscape indices. *The American Naturalist*, **183**, 24–41.

Watts, K. (2003). *Landscape ecology project: phase 4 project plan – development of prototype landscape ecology evaluation tools to assist large-scale forest planning*. Forest Research, Woodland Ecology Branch, Alice Holt, Farnham.

Watts, K. and Griffiths, M. (2004). Exploring structural connectivity in Welsh woodlands using neutral landscape models. In: *Landscape Ecology of Trees and Forests – Proceedings of the 2004 IALE (UK) Conference, held at Cirencester Agricultural College, Gloucestershire, 21st–24th June 2004*. R.J. Smithers (Ed.). IALE(UK), Grantham.

Watts, K., Griffiths, M., Quine, C.P., Ray, D. and Humphrey, J.W. (2004). Woodland habitat network strategy for Wales. *Unpublished contract report to Countryside Council for Wales CCW/FCW Tender No.: CT-W.H.N.S.W – 0303*. Forest Research, Roslin, Midlothian.

Watts, K. and Selman, P. (2004). Forcing the pace of biodiversity action: a force-field analysis of conservation effort at the 'landscape scale'. *Local Environment*, **9**, 5–20.

Webster, S. and Felton, M. (1993). Targeting for nature conservation in agricultural policy. *Land Use Policy*, **10**, 67–82.

Weyrauch, S.L. and Grubb, T.C. (2004). Patch and landscape characteristics associated with the distribution of woodland amphibians in an agricultural fragmented landscape: an information-theoretic approach. *Biological Conservation*, **115**, 443–450.

Wickramasinghe, L.P., Harris, S., Jones, G. and Vaughan, N. (2003). Bat activity and species richness on organic and conventional farms: impact of agricultural intensification. *Journal of Applied Ecology*, **40**, 984–993.

Wilson, G. (1997). Selective targeting in environmentally sensitive areas: implications for farmers and the environment. *Journal of Environmental Planning and Management*, **40**, 199–215.

Worrell, R. and Francis, J.L. (2003). Establishing woodland flora in broadleaf and conifer woodlands. *Scottish Forestry*, **57**, 203–210.

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

Wu, J. and Hobbs, R. (2002). Key issues and research priorities in landscape ecology: An idiosyncratic synthesis. *Landscape Ecology*, **17**, 355–365.

Young, J., Nowicki, P., Alard, D., Henle, K., Johnson, R., Matouch, S.J.N. and Watt, A. (2003). *Conflicts between human activities and the conservation of biodiversity in agricultural landscapes, grasslands, forests, wetlands and uplands in Europe*. A report of the BIOFORUM project.

Young, J., Watt, A., Nowicki, P., Alard, D., Clitherow, J., Henle, K., Johnson, R., Laczko, E., McCracken, D., Matouch, S. and Niemela, J. (In press). Towards sustainable land use: identifying and managing the conflicts between human activities and biodiversity conservation in Europe. *Biodiversity & Conservation*.

Appendix 1

Macaulay Institute has recently initiated two new landscape modelling projects relevant to species modelling in lowland agricultural landscapes. The first entitiled *Spatially explicit models of biodiversity at the landscape and macro-scale* is aimed at improving the understanding of how biodiversity patterns are generated at local, landscape and national scale. It is envisaged that the simulations conducted will draw largely on data obtained in other projects ongoing within Macaulay (especially focused on impacts on browsing in woodlands). The intention is to provide strategic scientific underpinning regarding some of the principles that need to be taken into account when managing landscapes to protect species diversity. The project is focused on developing methods that allow the disentangling of chance effects in species composition from effects of environmental and biological interactions, reconciling dispersal-based and niche-based explanation for community structure. A particular focus will be coexistence, patterns of relative abundance and the role and origin of autocorrelation of spatial distribution in structuring assemblages. Biodiversity patterns (such as how species richness varies with the extent of the investigated area, detection of biogeographic boundaries for species assemblages, patterns in life history traits and distribution) of both plant and animal communities are under investigation. It is expected that the results will provide an increased understanding of how species diversity patterns are generated at different scales and how processes at different scales interact.

In the second project *Species distributions, biodiversity and ecosystem functioning in changing environments* the aim is to increase understanding of how spatial variation in biodiversity arises through the impact of different, individual species in machair habitats in the western isles. In particular, it is investigating the role of individual species' spatial and temporal distribution patterns in creating emergent biodiversity patterns (variation in both numbers and kinds of species present from place to place). This is based on the premise that some individual species can contribute to biodiversity patterns more than others. However, it is not clear whether or not species with particular characteristics dominate biodiversity patterns, yet this is very important in understanding the processes controlling the latter. It is anticipated that identifying species within different functional groups and with different characteristics, such as life-history traits, that are important for biodiversity patterns, will help assess the sensitivity of the latter to climate and land-use change. Field experiments are being used to investigate the role of the potential key drivers of climatic and anthropogenic (mainly agricultural) disturbance in controlling biodiversity patterns. It is intended that this will allow assessment of the impact of the spatial scale of different kinds of disturbances (large areas such as fields versus small patches) on the interaction between individual species and the resultant biodiversity patterns. The focus is therefore on how individual species characteristics affect how much they control biodiversity patterns, and how affected in turn these species are by climatic and anthropogenic disturbance. It is expected that the outcome of this research will be an understanding of how individual species contribute to biodiversity patterns against a background of natural and anthropogenic environmental change.