The Restoration of Wooded Landscapes

Proceedings of a conference held at Heriot Watt University, Edinburgh, 14–15 September 2000

Edited by Jonathan Humphrey, Adrian Newton, Jim Latham, Helen Gray, Keith Kirby, Elizabeth Poulsom and Chris Quine

Forestry Commission

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Preface

This publication comprises the proceedings of the conference 'The Restoration of Wooded Landscapes', held at Heriot Watt University in September 2000. The principle aim of the conference was to bring together researchers, practitioners and policymakers to allow a full and free exchange of views, information and ideas on the theme of native woodland restoration at the landscape scale (areas in excess of 1 km^2). This includes creating new native woodland, restoring planted ancient woodland, and expanding existing native woodlands.

There are currently a large number of initiatives throughout Britain concerned with the restoration of wooded landscapes. Increasing resources are being channelled into woodland restoration, and partnerships of private and public organisations and individuals are becoming larger and ever more complex. The recently published Habitat Action Plans for native woodlands require a strategic approach to the design and implementation of restoration schemes at the landscape scale and, in particular, the creation of woodland habitat networks. There are a number of gaps in the information required to undertake such a strategic approach, and indeed the approach itself has been questioned. In many respects the pace of restoration has been such that the science has often been struggling to keep up with practice, and there has been a lack of specific guidance and synthesis of current knowledge. Questions have arisen such as:

- What do we want, ecologically or socially driven restoration, or both?
- What is favourable ecological condition, i.e. how do we monitor progress, and how do we measure success?
- What should be the balance between woodland restoration and expansion?
- Do we have the appropriate tools for designing new landscapes?

Given these uncertainties, landscape scale restoration has been approached from a variety of perspectives including not only ecological aspects, but also economic and social issues. The strengths and weaknesses of these different approaches need to be evaluated in order to improve guidance available to planners and managers.

This publication aims to:

- Synthesise current knowledge relating to the ecology of wooded landscapes and design of new woodland areas.
- Illustrate, through case studies, how ecological knowledge has been applied to the design and management of native woodland restoration schemes, the identification of opportunities and constraints, and the evaluation of ecological and economic costs and benefits.

The publication is aimed at woodland managers, planners and policymakers concerned with the restoration of native woodland at the landscape scale. A number of the chapters reviewing the main themes in the field of landscape ecology will be of interest to applied ecologists and researchers.

There are three main sections. Section One: *Introduction and context* reviews current international and UK forestry policies and incentives and how they relate to the delivery of woodland habitat restoration targets (Chapter 1). Restoration activities within the UK are placed within the international context and ways of prioritising restoration at the regional and country levels are discussed (Chapter 2).

In Section Two: *Research and modelling tools* there is a review (Chapter 3) of the latest research on species colonisation processes within wooded landscapes, looking critically at the concept of habitat networks and landscape linkages. Chapter 4 describes research into natural regeneration processes on restored woodland sites, whilst Chapters 5, 6 and 7 demonstrate a range of new GIS-based tools of use in the design and planning of landscape scale restoration

Section Three: *National and regional planning* focuses on strategic planning of restoration at the regional scale either by large owners (Chapters 11 and 12) or at the country level (Chapters 8, 9 and 10). Chapter 13 puts forward an argument for focusing restoration effort on degraded ancient woodland sites rather than the development of habitat networks and linkages at the landscape scale.

Section Four: *Local case studies* are all about practical implementation, with Chapters 14–17 showing how science and policy are translated into action on the ground. The problems associated with implementing large-scale restoration schemes are described together with some of the valuable lessons learned.

Section 5: *Conclusions* comprises of Chapter 18 which attempts to synthesise the main problems and issues relating to woodland restoration at the landscape scale as highlighted in the preceding chapters.

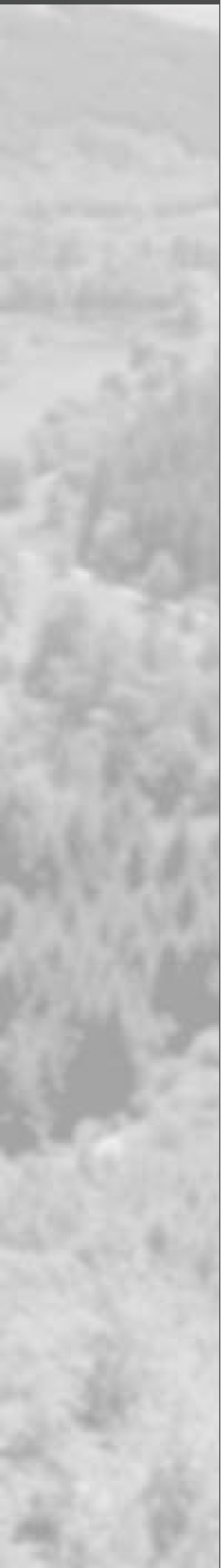
Jonathan Humphrey
Adrian Newton

December 2001

SECTION ONE

Introduction and context

CHAPTER 1



The UK policy context

Tim Rollinson

Introduction

Over many thousands of years, we in the UK cleared almost all of our natural woodland cover. Our forests helped to fuel our economic development and satisfy the demands of an increasing population for timber, fuel and farm land. But we paid a price; at the beginning of the 20th century woodlands in the UK covered just 5% of the land area, and little of this resembled the natural woodland cover. In the past century a million hectares of land was reforested, increasing our forest cover to over 10%. This was a substantial achievement. Throughout this period, we have had to address the challenges of rehabilitating and restoring our woodlands and forests. Our new forests are very different from what we know of our lost natural woodlands, but they have put woodland back on the map. We are improving them and, at the beginning of the 21st century, we can hand on a *bigger* woodland legacy to the next generation. A further challenge is to make sure that it is also a *better*, and truly sustainable, legacy.

The global background

In 1992 the world's leaders committed themselves to sustainable development at the United Nations Conference on Environment and Development known as the Earth Summit (UNCED, 1992). The conference produced the first global agreement on how the world's forests should be managed in the *Statement of Forest Principles*. Since the Earth Summit, the UK and other European governments have built on the Rio Forest Principles and are committed to implementing:

- *The guidelines for the sustainable management of forests in Europe* – agreed at Helsinki in 1993 (Ministerial Conference on the Protection of Forests in Europe, 1993).
- *The guidelines for the conservation of the biodiversity of European forests* – also agreed at Helsinki (Ministerial Conference on the Protection of Forests in Europe, 1993).
- *The declaration and resolutions of the pan european ministerial conference on the protection of forests in Europe* – agreed at Lisbon in 1998 (Ministerial Conference on the Protection of Forests in Europe, 1998).

The Helsinki Guidelines interpreted the Rio Principles for European conditions and articulated the common concern of European countries to manage their forests sustainably. Through the Lisbon declaration, countries gave further recognition to the social and cultural importance of forestry in Europe. These international agreements are an expression of world-wide interest in sustainable forestry. Following their adoption, European countries, including the UK, have agreed a range of criteria for defining sustainable forest management and indicators for measuring progress towards it.

Sustainable forestry in the UK

Sustainable forestry is one component of the UK Government's wider commitment to sustainable development. In 1999, the Government published *A better quality of life: a strategy for sustainable development in the UK* (Anon., 1999). The Strategy confirmed that the Government's approach to sustainable forestry is based on: better management of existing forests; the continuing expansion of the woodland area; and conservation of natural capital – biodiversity, air, soil and water. Defining sustainable forest management is complex. It results from the interaction of the three functions of forests – economic, social and environmental, as represented in Figure 1.1.

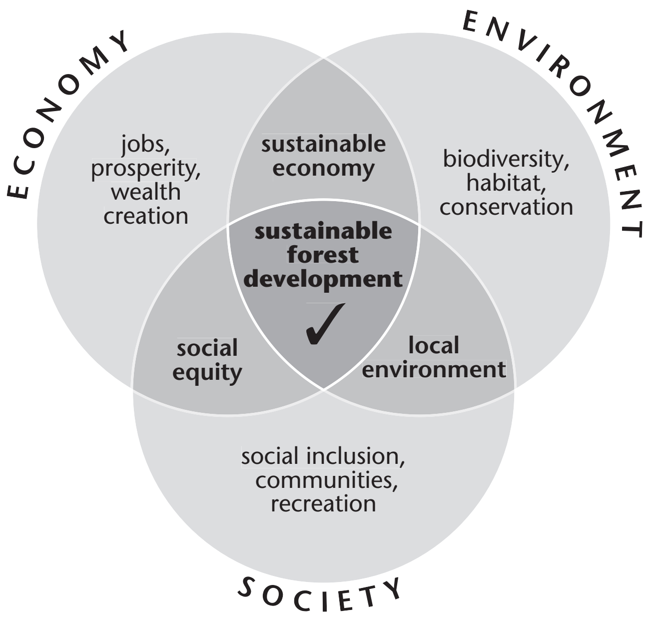


Figure 1.1

A conceptual model of sustainable forest management.

Biodiversity and the other environmental values of forests must be balanced with economic and social values in decisions about sustainable forestry. While the remnant ancient and semi-natural woodlands are the best overall for biodiversity, our maturing and restructured planted forests have an increasingly important role to play in the UK. Indeed, over the past 20 years, there has been unprecedented interest in the management and restoration of all types of woodlands and their biodiversity. Table 1.1 gives examples of some of the policy and practice initiatives that have been introduced.

Table 1.1

Some policy and practice initiatives relating to woodlands.

Year	Initiative
1985	Government broadleaves policy introduced
1989	Native pinewood scheme launched
1994	Guides for management of semi-natural woodlands published
1996–8	Habitat and species action plans published
1998	UK forestry standard published
1999	Scheme to create new native woodlands in national parks launched

The UK forestry standard

A cornerstone of the Government's commitment to sustainable forest management is the *UK forestry standard*, published in 1998 (Forestry Commission., 1998). The standard provides a single, comprehensive statement of the Government's approach to sustainable forestry in the UK. It explains how the principles of sustainability will be delivered in practice and lists the criteria and indicators for the sustainable management of all forests in the UK. The standard includes guidance on a range of forest management practices including new woodland creation, new native woodland creation, felling and restocking planted woodland, managing semi-natural woodland, and planting and managing small woods.

The UK Biodiversity Action Plan and forestry

The Government published *Biodiversity: the UK action plan* (UKBAP) in 1994 (Anon., 1994). The overall goal is to 'conserve and enhance biological diversity within the UK and to contribute to the conservation of global biodiversity through all appropriate mechanisms'. The emphasis is on partnership between public and private sector and NGOs at local, regional and national levels, and across sectors. The UKBAP lays emphasis on integrating biodiversity conservation measures into all sectors of economic activity so that it becomes part of sustainable development. In addition priority species and habitats have been defined and are subject to multi-agency and cross-sectoral action plans. The focus of attention has now shifted to implementation of the Biodiversity Action Plan targets throughout the UK (Anon., 1995).

Delivery

The development and publication of the *UK forestry standard* and *Biodiversity: the UK action plan*, together with the introduction of a range of schemes and incentives to encourage delivery on the ground, has resulted in a very substantial increase in the creation of new native woodlands in the UK. The increased planting of new broadleaved woodland during 1978–2000, most of which has been with native species, is shown in Figure 1.2 while Figure 1.3 shows the increase in the creation of new native pinewoods in the Scottish Highlands since 1990.

While the focus of action has often been at the individual forest level, the emphasis now is shifting to how woodlands can be linked to form habitat networks. The key delivery mechanisms are set out in Table 1.2.

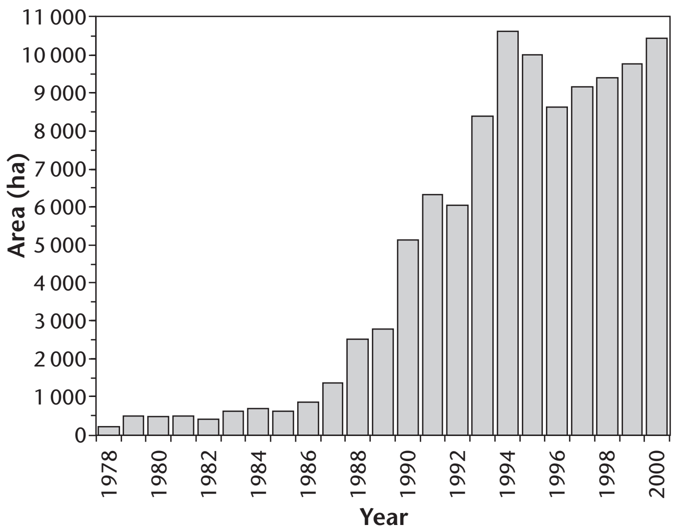


Figure 1.2

Planting of new broadleaved woodlands.

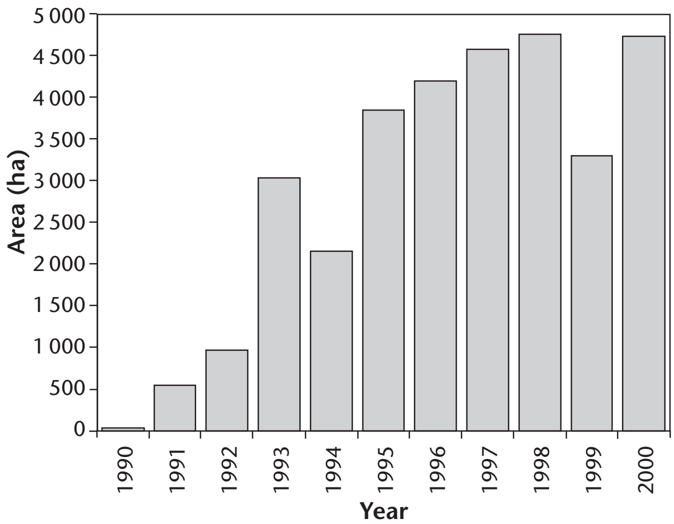


Figure 1.3

Planting of new native pinewoods in the Scottish Highlands, 1990–2000.

Table 1.2

Key delivery mechanisms for habitat networks.

Mechanism	Initiative
Research & Inventory	To provide an informed basis for delivery
Standards	To set the requirements for good forestry practice
Guidance	To encourage adoption of best practice
Regulations	To protect the environment and control potentially damaging operations
Incentives	To encourage adoption of new programmes

Working together

Real progress has been made in recent years in delivering a range of policies for woodland restoration. We have learned that the greatest progress will be made where research, policy, regulation, incentives, and published guidance are made to work together effectively. This requires: a shared understanding of the issues and barriers to progress, flexibility of approach to accommodate the needs of many stakeholders, resources to deliver desirable programmes and a more 'joined-up' approach with stakeholders working together and not solely to their own agendas.

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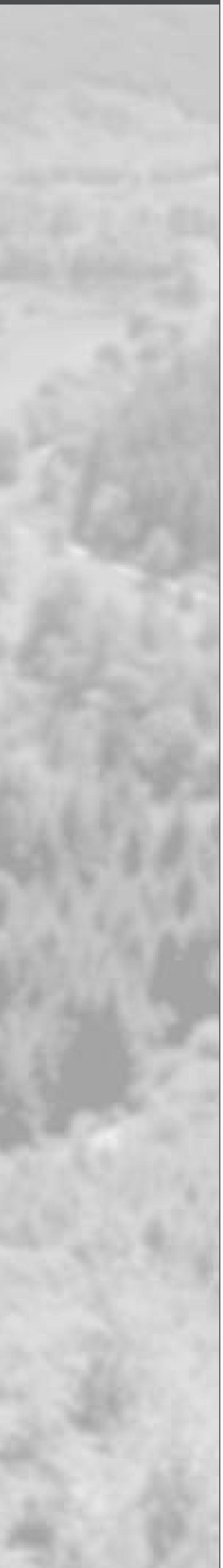
CHAPTER 2

Restoration of wooded landscapes: placing UK initiatives in a global context

Adrian Newton and Valerie Kapos



Summary

Current initiatives in the UK focusing on the restoration of wooded landscapes can be viewed as part of a global effort for improving the conservation status and habitat value of forest ecosystems. In response to the high rates of forest loss and degradation experienced in many parts of the world, a large number of forest restoration projects have been initiated, supported by international policy commitments such as the Convention on Biological Diversity (CBD). In order to ensure that the resources available for restoration are being focused effectively, methods are needed for setting priorities at global and regional as well as local scales. Here we describe how such priorities might be defined, with reference to available information on global forest cover and through a case study undertaken for the Mediterranean region. We highlight the need for information about the current distribution of restoration projects in order to assess whether current activities are adequately addressing restoration priorities. Although woodland restoration efforts in the UK may make only a minor contribution to conservation of global biodiversity, the techniques and approaches being developed will be applicable in many other areas. Experience gained in the UK could therefore be of high value in demonstrating how restoration of wooded landscapes can best be achieved in practice, particularly in areas characterised by a high degree of forest loss and intensive patterns of land-use.

Introduction

The latest estimates of global deforestation produced by the Forestry and Agriculture Organisation (FAO) suggest that approximately 13.5 million hectares of forest are being cleared each year, with highest rates of loss being recorded in Africa and South America (FAO, 2001). These analyses suggest that the main factor responsible for causing forest loss is conversion to agricultural land, including pastures and shifting cultivation. Although forest cover in industrial countries is considered to be roughly stable (FAO, 2001), this reflects the fact that losses of native forest cover in such countries are being offset by establishment of commercial plantations, often of exotic species. In general, plantation forests are considered to be of lower value as a habitat for native biodiversity than natural forests (Groombridge and Jenkins, 2000) and, therefore, even in industrialised countries, the availability of habitat for forest-dwelling organisms is declining.

Increasing awareness of the high rates of forest loss and widespread environmental degradation has led to a growth of interest in both the science and practice of ecological restoration (Niering, 1997; Dobson *et al.*, 1997). The main aim of ecological restoration is to re-establish the key characteristics of an ecosystem, such as composition, structure and function, which were present prior to degradation (Jordan *et al.*, 1987; Hobbs and Norton, 1996; Dobson *et al.*, 1997; Higgs, 1997). It has been suggested that ecological restoration is a crucial complement to the establishment of protected areas for safeguarding biodiversity (Dobson *et al.*, 1997), and it is anticipated that restoration will become an increasingly central activity in environmental management in the future (Niering, 1997).

This aim of this chapter is to provide a global overview of forest restoration, to enable the many woodland restoration initiatives currently being developed in the UK to be placed in a broader context. Firstly, the global priorities for forest restoration are considered, with reference to relevant international policy initiatives and the current status of forest resources. The contribution being made by restoration activities in different parts of the world towards meeting these global priorities is then discussed. The methods by which restoration priorities may be developed at the regional scale are subsequently illustrated by reference to an example from the Mediterranean region. Consideration is then given to how such approaches might be applied to the UK to identify suitable sites for woodland restoration. In addition, an assessment is presented of the contribution being made by projects in the UK towards meeting global priorities for forest restoration.

Forest restoration: definitions and approaches

Efforts to develop a precise and workable definition of the term 'restoration' have been the cause of some debate and controversy (Wyant *et al*., 1995), most notably among the membership of the Society for Ecological Restoration (SER) (Higgs, 1997). Much of this debate has centred on whether or not ecological restoration should seek to re-establish an ecosystem equivalent to some historical state (Atkinson, 1994; Jordan, 1994; Higgs, 1997). The problem of defining restoration objectives in this way results from the widespread difficulty in defining such a state with precision (Hobbs and Norton, 1996; Palmer *et al.,* 1997), because of the lack of historical information or appropriate modern analogues. In addition, cultural values of nature and the views of different stakeholders have increasingly been incorporated into restoration objectives (Adams, 1996; Higgs, 1997). Further debate has focused on precisely what should be restored, and therefore how the success of a restoration project might be measured. Apart from 'emulating the structure, function, diversity and dynamics of the specified ecosystem' (Higgs, 1997), it has been proposed that the objective of restoration should be 'renewal of ecosystem health' (Higgs, 1997) or 'assisting the recovery and management of ecological integrity' (Society for Ecological Restoration, 1997).

A detailed discussion of the relative merits of different definitions of ecological restoration is beyond the scope of this chapter. However, in a review of current woodland restoration projects in Scotland, Newton *et al*. (2001) suggested that because of the long history of human impact on Scottish forests, restoration of a historical state will rarely if ever be achievable, and therefore the explicit aim of restoration projects should simply be to provide an improved habitat for wildlife. This pragmatic objective has the advantage of enabling progress to be evaluated relatively easily. Given such an objective, ecological restoration can be most readily achieved by recreating native woodland cover in areas that have been completely deforested, or by rehabilitating degraded forest ecosystems. Although it may be useful to differentiate between these two approaches (Housden, 1997), in practice, many projects include elements of both (Newton *et al*., 2001), and therefore both are included in the concept of ecological restoration adopted here. It should be noted, however, that the terms 'restoration', 'recreation' and 'rehabilitation' have previously been used in different ways by different authors.

One issue that has been raised in the literature is whether conventional (i.e. monospecific) plantation forestry constitutes a form of ecological restoration. The current global rate of plantation establishment is around 4.5 million ha per year, with some 30 million ha of plantations successfully established during the 1990s, half of which constitute reforestation of previously forested lands (FAO, 2001). Although, as noted earlier, plantation forests are generally considered to be of lower value than natural forests as wildlife habitat (Groombridge and Jenkins, 2000), particularly where they are composed of exotic rather than native tree species (Newton and Humphrey, 1997), plantations can at least in some situations be of significant habitat value (Humphrey *et al*., 2000). For example, since the late 1980s, a major programme of forest restoration has been undertaken in the Lower Mississippi Alluvial Valley, with the aim of restoring 'bottomland hardwood' forests. This area once supported the largest expanse of forested wetlands in the USA and it is estimated that over 200 000 ha of forest plantations will have been established by the end of the current decade (Stanturf *et al*., 2000). The explicit aim of the programme is to improve wildlife habitat. However, afforestation in this area generally involves establishment of native species in single-species plantations; it is anticipated that other tree species will colonise naturally with time (Stanturf *et al*., 1998). It is recognised that many of the area's threatened species require diverse forests of complex structure as habitat (Stanturf *et al*., 2000), and therefore the habitat value of these plantation forests would appear to be limited, at least in the short term. The issue of whether plantations should be established at a density suitable for commercial harvesting has also attracted debate; such financial benefits might make restoration cost-effective for private landowners (Stanturf *et al*., 1998; 2000). Analysis of the extent to which the expanding global resource of plantation forests may provide a habitat for native species, and how such forests might be managed to optimise their value as wildlife habitat, are areas worthy of research attention.

Recent FAO analyses also highlight the fact that natural recolonisation of forest is occurring in many industrialised countries following abandonment of agricultural land. This is particularly noticeable in some countries within the Commonwealth of Independent States (CIS), including the Russian Federation. In addition, analyses of remote sensing data indicate that some 1 million ha of agricultural land revert to forest each year in tropical areas (FAO, 2001). Such figures suggest that forest restoration is occurring naturally in at least some parts of the world, as a result of shifting patterns of land-use. However, the overall global trend is a continuing decline in the area and quality of forest habitat, and for this reason ecological restoration is an increasingly high priority.

Setting global priorities for forest restoration

The UN Convention on Biological Diversity (CBD), Article 8f, states that parties should 'rehabilitate and restore degraded ecosystems and promote the recovery of threatened species, through the development and implementation of plans or other management strategies'. The Convention has now been ratified by more than 180 countries, and therefore provides a global policy commitment in support of ecological restoration activities. At the global level, the implementation of Article 8f is being addressed by the CBD through the development of a thematic work programme, focusing on identifying research and technological priorities and practical actions. At the national level, the mechanisms by which the Convention might be implemented vary between different countries; in the UK, this has involved the development of Biodiversity Action Plans which incorporate a number of different restoration activities focusing on woodland habitats (Housden, 1997; Anon., 1995).

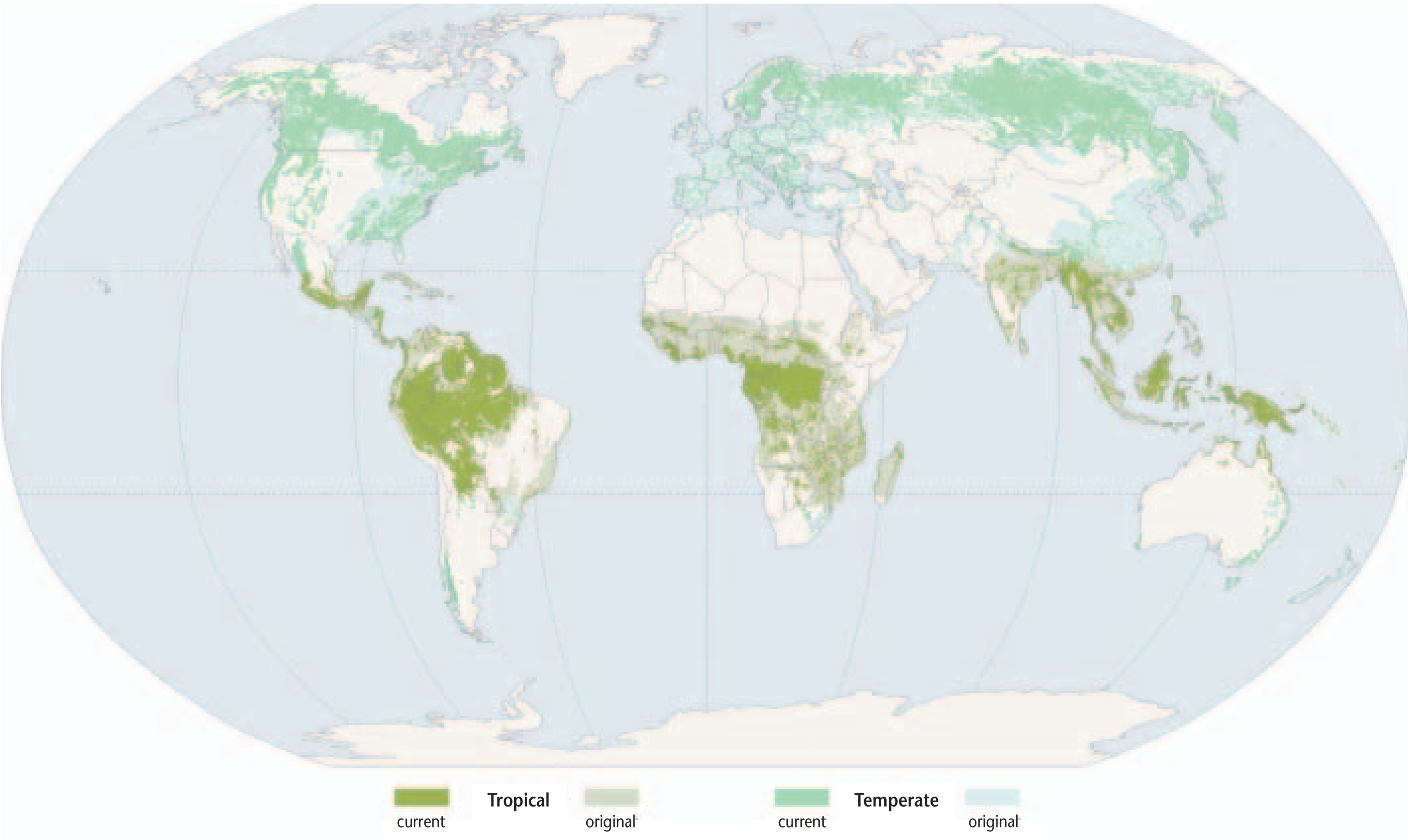
In order to develop a global strategy for restoration in support of the CBD, it is relevant to consider how priorities for forest restoration might be identified most appropriately at the global scale. Given that no such assessment has been undertaken previously, we present here a brief overview of how such priorities might be defined, with particular reference to the criteria that could be used for their selection.

Potential of a given area to support forest cover

Logically, the most important criterion is whether a given area has the potential to support forest cover. A comparison of current forest distribution with the earth's original forest cover could conceivably be used to identify those areas that have suffered the highest degree of forest loss, and which could therefore be considered as priorities for restoration. In fact, it is difficult to estimate with much precision the extent to which forest cover has been removed by human activity. The concept of 'original' forest cover is itself imprecise; forest cover has been highly dynamic in many areas during the Quaternary period and the effects of human impact are often very difficult to separate from other factors such as climate change (Matthews *et al*., 2000). However, a number of attempts have been made to describe the extent of potential forest cover in the absence of human intervention. For example, in 1999, WWF-US produced a global map of major habitat types on the basis of their ecological characteristics and associated climate (Matthews *et al*., 2000).

A map of potential forest cover has also been developed by UNEP-WCMC (Iremonger *et al.,* 1997), to illustrate the probable distribution of closed forest world-wide prior to the impact of agricultural activity and subsequent to climatic recovery from the last ice age about 6 000–8 000 years ago (see Figure 2.1). The map was compiled from seven potential vegetation datasets (Bohn and Katenina, 1994; Carnahan, 1989; Dinerstein *et al*., 1995; MacKinnon, 1996; Milanova and Kushlin, 1993; Olson and Dinerstein 1998; White, 1983), which between them cover the globe. The forest vegetation classes were selected from these maps, as a first approximation to global potential forest cover. The resulting composite global map was generally similar to that produced by WWF-US, but with some notable differences (for example, the area of potential forest in Africa suggested by the UNEP-WCMC analysis was substantially higher than that estimated by WWF-US).

Figure 2.1 *Global potential, original and current forest cover (pixel size 1 km²). The lightest green areas are those that were potentially once forested, where forest cover no longer remains, (from Iremonger* et al.*, 1997).*



UNEP-WCMC have also compiled a global map of current forest cover, at a nominal scale of 1:1 000 000 from a variety of national and regional sources, including remote sensing and other data types (Iremonger *et al.*, 1997). The potential forest cover may be usefully compared with current forest cover, to identify those areas where deforestation has been most pronounced (Figure 2.1). Such a comparison highlights the enormous forest losses that have occurred in north temperate areas, including much of Europe and China, and also eastern USA. In the tropics, substantial areas of forest have been cleared from SE Asia, sub-Saharan Africa, parts of Central America and northern South America, and the Atlantic region of Brazil. These deforested areas could nominally support forest again at some time in the future, and are therefore candidates for restoration activities.

Feasibility of restoration: regional variations

It is important to recognise that different regions vary with respect to the feasibility of restoration; this could be considered as an additional criterion for identifying priorities. For example, areas under intensive cultivation for food production are unlikely to be available for restoration of forest cover. Thus a crude revision of the potential area for forest restoration (Figure 2.1) can be made by subtracting the areas of the globe determined by Matthews (1983) to be intensively cultivated (Figure 2.2). The resulting total area of more than 20 million km^2 is still far higher than is actually available for increasing forest cover. The estimate could be improved by using higher resolution land-use data that are now becoming available. In addition, the effects of soil degradation and climate change need to be taken into account, together with urban development and other forms of intensive land-use, to define with more precision those areas where forest restoration might be feasible.

Extent and rate of loss of forest types

A more thorough analysis of forest loss and priorities for restoration would consider the extent and rate of loss of different forest types. Although such analyses are complicated by the great variety of forest classification systems in use world-wide, available data suggest that tropical montane forests are undergoing the highest rates of clearance of any major forest type (Matthews *et al.*, 2000). Moist and dry tropical and sub-tropical forests are also undergoing high rates of loss and could therefore also be considered as high priorities for restoration efforts.

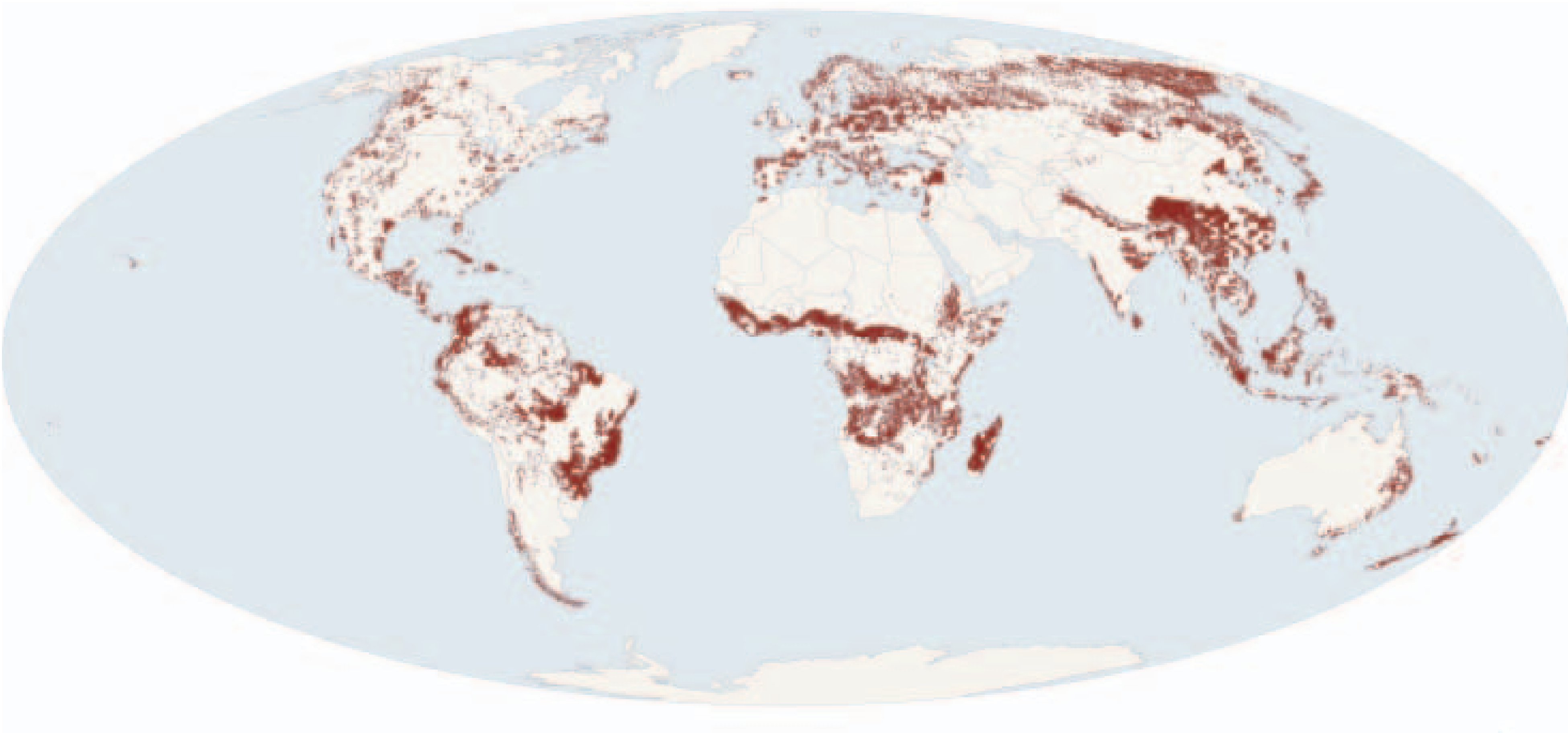
Forest condition

However, it is not simply a decline in forest area that poses a threat to biodiversity, but a decline in the ecological condition of forest ecosystems. The structure and composition of forest communities may be modified by a variety of human impacts, such as road construction, timber extraction, fires, and browsing by domestic animals. Forest condition could therefore be considered as an additional criterion with respect to developing priorities for restoration; those forests that have suffered greater degradation might be accorded higher priority. However, ecological condition is not necessarily easy to define precisely or to measure. For example, attempts to assess forest 'naturalness', or the degree of resemblance to conditions that would prevail in the absence of human intervention, have proved very difficult to implement in practice (Matthews *et al.*, 2000). Keddy and Drummond (1996) provide a useful list of ecological properties that could be used to monitor the condition of deciduous forests, including tree size and canopy composition, quantity and quality of coarse woody debris, and the composition of the herbaceous ground flora, avian, fungal and carnivore communities. Other measures of forest condition are perhaps more amenable to assessment at regional and global scales; for example the extent to which forest habitats have been fragmented may be readily measured using aerial photographs or remotely sensed data, analysed with GIS. Spatial analyses have also been used to develop a 'Wilderness Index' based on remoteness from roads, settlement and intense land-use (Lesslie and Maslen, 1995). This method highlights areas exposed to human activity, which by implication, could be considered as being of relatively high priority for restoration.

Conservation importance and contribution to global diversity

Another key issue is that forest areas differ in terms of their conservation importance, or their contribution to global biodiversity. A number of different approaches to identifying priority areas for conservation have been developed, and these could similarly be applied to identifying priorities for restoration. For example, WWF-US developed a 'Global 200' categorisation of the world's most important ecoregions from a conservation perspective (Olson and Dinerstein, 1998). Some two-thirds

Figure 2.2 *A first approximation of the global area available for forest restoration based on the potentially forested area no longer supporting forest cover (shown in Figure 2.1) and excluding areas under intensive cultivation (according to Matthews, 1983) (pixel size 1 km²). The resulting areas shown in brown total over 20 million km².*



of the ecoregions identified are classified as forest. BirdLife International has identified 218 endemic bird areas (EBAs) worldwide; these are areas of particular importance for bird species with restricted ranges (Stattersfield *et al*., 1998). Some 83% of EBAs occur in forested areas. Similarly, WWF and IUCN have identified 234 centres of plant diversity worldwide (WWF and IUCN, 1994); about 80% of these are found in forests. Each of these measures highlights the conservation importance of different forest areas; they could be used either individually or in combination as selection criteria for restoration.

Furthermore, the ecological functions that restored forests might serve, such as biodiversity preservation or water catchment protection, can provide additional criteria for prioritising target areas for restoration programmes at many spatial scales.

Global distribution of forest restoration projects

Given the variation in the extent of forest loss and degradation in different parts of the world, and the variation in the conservation importance of different forest types, it is relevant to consider the global distribution of current forest restoration activities. Ideally, such activities would be focused on those forest areas most in need of restoration but, as noted above, no comprehensive analysis of restoration priorities has yet been undertaken.

Information on the current distribution of forest restoration initiatives is generally lacking. Although there are now a number of scientific journals (such as *Restoration Ecology* and *Ecological Restoration*) which disseminate information about different restoration projects, there is no central repository of information or listing of restoration activities. It is therefore difficult to assess the geographic distribution of current projects. To provide a preliminary overview of current activity, we undertook an internet search (using a search engine, www.yahoo.com) using the keywords ‘forest restoration’, and then visited the websites of the first 100 ‘hits’, restricting our assessment to those websites which described an actual forest restoration programme currently in progress. This approach to sampling is obviously biased: only a subset of restoration projects possess a website, and the use of these keywords will inevitably restrict the sample to those projects which include these words on their website. Unsurprisingly, most of the projects which were located using this method are being undertaken in countries where the main language spoken is English (Figure 2.3).

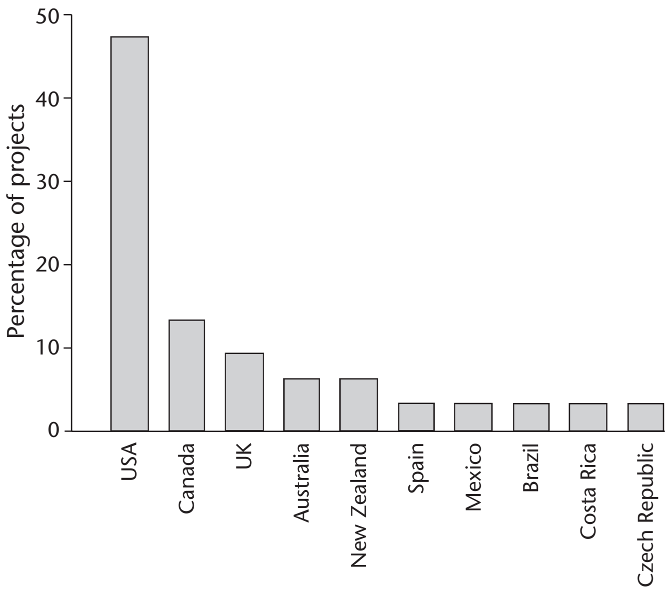


Figure 2.3

The geographical distribution of 100 forest restoration projects identified by country during an internet survey.

The NGO most active in forest restoration at the global scale is the World Wide Fund for Nature (WWF), working in partnership with the International Union for the Conservation of Nature (IUCN – www.iucn.org). The WWF/IUCN *Forests for Life* programme aims to ‘halt and reverse the loss and degradation of forest and all kinds of woodlands – particularly old growth forest – worldwide' (WWF/IUCN, 1996). To achieve this, the programme has defined five key objectives, of which the third relates explicitly to restoration, stating that WWF/IUCN will seek 'to develop and implement environmentally sound and socially equitable forest restoration programmes'. The strategy also notes: the need for identification of areas where forest has disappeared and ecological restoration is both needed and feasible; technical options for forest restoration; human needs dependent on forests and their implication for restoration programmes; and calls for the introduction of appropriate forest restoration projects (WWF/IUCN, 1996). Such projects have been initiated in the Lower Mekong (China), Scotland, New Caledonia, the Mediterranean, Danube/Carpathians, Jhabua (India), and throughout Central America. Most importantly, given the focus of this volume, WWF/IUCN are increasingly recognising the critical importance of developing plans for forest restoration at the landscape scale (M. Aldrich, personal communication).

Setting restoration priorities at the regional scale

More detailed approaches to prioritising restoration efforts can be developed at regional scales. Enhanced criteria are needed for selecting high priority sites. As an illustration of how such criteria may be developed, we describe here a case study of the Mediterranean region undertaken for WWF International (WCMC, 2000). To our knowledge, this is the first such study that has been carried out. It is described here to illustrate the methods by which similar analyses might be applied in other regions, such as northern Europe.

The objective of this case study was to define broad areas or landscapes that could form a focus for forest restoration activities. The aim was therefore to develop a decision-making tool for prioritising restoration activities at the regional level, rather than guiding activities at the site level. Selecting from the many ecological and social criteria that could potentially be used to identify priority areas for restoration depends on the precise objectives of the restoration programme. For example, criteria used for locating projects with objectives focusing on biodiversity conservation may be very different from those for projects with timber production, protection of water catchments or prevention of desertification as the main objectives. In addition, the selection of criteria is constrained by the availability, resolution and quality of data.

For this study, which assumed a principal focus on biodiversity conservation, five criteria were selected for defining priority areas for restoration at the regional level (Table 2.1). These criteria focus on locating areas which were once forested but are now largely unforested (Figure 2.4), and characterised by low population density. Additional criteria focus on the assumption that restoration projects are most likely to be successful, at least in conservation terms, if they are located near to remaining woodland areas, particularly those rich in biodiversity (Table 2.1). A Geographical Information System (GIS) was used to define priority areas according to these criteria, by applying each criterion consecutively and by subtracting areas not considered to have potential for restoration at each step. The composite map produced as a result of this process (Figure 2.5) represents a first attempt to identify priority areas for forest restoration for the Mediterranean region. Such an analysis could usefully be compared with the location of existing projects to identify priority areas for future restoration initiatives.

Of the area included in the Mediterranean analysis, approximately 15 million km^2 may once have supported forest but no longer do so. Although this area might therefore be appropriate for forest restoration, conflicting demands on the land and biological and climatic constraints limit the feasibility of restoring forest in much of this area. Applying the criteria used in the GIS analysis has made it possible to identify a relatively small fraction (just over 670 000 km^2) of this enormous area as being of the highest potential for restoration. More detailed investigations of priority and feasibility can realistically be conducted for this smaller area and the locations of proposed projects can be evaluated against this criterion, among many others.

This example illustrates the value of GIS as a visual tool for prioritising areas for restoration efforts. However, the selection of criteria for defining such sites is clearly of key importance. For example:

Table 2.1 *Selected candidate criteria for defining priority areas for forest restoration at a regional scale (from WCMC, 2000)*

Criteria	Critical assumptions	Justification for assumptions
1. Original forest areas which are currently unforested	Current forest areas are not a high priority for restoration; all areas which once supported forest but are now deforested have potential for forest restoration.	Where not already under active management, many forest areas (defined as those with >30% canopy cover and transitional forests with >10% canopy cover) contain sufficient numbers of native species to be able to revert to their natural state. In fact, many degraded forests may be priorities for restoration, but there are insufficient data on forest condition to be able to distinguish them from, natural forests and forests managed for other purposes.
2. Areas containing woodland which are currently unforested	Areas outside continuous forest, but containing patches of woodland in arable land and/or grassland are suitable for forest restoration.	Forest restoration is likely to be most feasible and effective in areas where native woodland species are still found, even if their distribution is highly fragmented. Arable land and grassland with no residual woodlands represent more intensive land-use, but could still be restored.
3. Areas of low population density	Areas of high population density (>800 persons per km^2 for the Mediterranean region) are unsuitable for forest restoration.	In many countries there is a close correlation between population density and pressure on natural resources (e.g. for non-timber forest products and firewood), reducing the possibilities for successful restoration in areas of high population density. In the developed countries of the Mediterranean region, this correlation is weakened by lack of dependence on local forest resources. However the threshold population density has been raised to limit the extent of excluded areas.
4. Areas in close proximity to forests	Areas immediately adjacent to existing forests and <1 km from the forest boundary are priorities for restoration.	In addition to the importance of proximity to reservoirs of native species, restoration efforts are likely to be focused around existing forests due to site suitability, and planning constraints on other lands.
5. Areas rich in biodiversity	The principal purpose of forest restoration is to conserve biodiversity; areas rich in forest biodiversity are therefore priorities for forest restoration.	Restored forests may have other benefits, such as protection of watersheds or even (limited) production of timber. However, other managed forests can offer these benefits more cost-effectively, while being less suited to biodiversity conservation, so restoration efforts should be focused on areas of high biodiversity importance.

1. Rather than focus on areas close to existing forest areas, could it be argued that restoration is most needed in areas where little native woodland remains (Newton and Ashmole, 1998)?
2. Should restoration focus more on areas most threatened by human activities, or areas where such pressures are less intense?
3. Should projects focus on rehabilitation or recreation of forest areas?
4. Should efforts be concentrated on linking fragmented forests or increasing the core area of remaining fragments?

The answers to such questions will obviously differ, depending on the precise objectives of the restoration programme in question, and the resources available. Additional criteria that may also need to be considered in setting restoration priorities include the legal status of forest areas, forest type and ecological characteristics, land tenure, historical land-use patterns, the extent of soil degradation, the condition of remaining forest, and socio-economic factors (such as the value of alternative land-uses). Potentially, data on such aspects could be incorporated into a GIS to assist the decision-making process, enabling the outcomes of selecting different criteria to be compared. Additional tools, such as decision trees or optimisation methods, could be linked to the GIS to assist the decision-making process (Kersten *et al.*, 2000).

Figure 2.4

Map of current and potential forest cover in the Mediterranean region (from WCMC, 2000) (pixel size 1 km²).

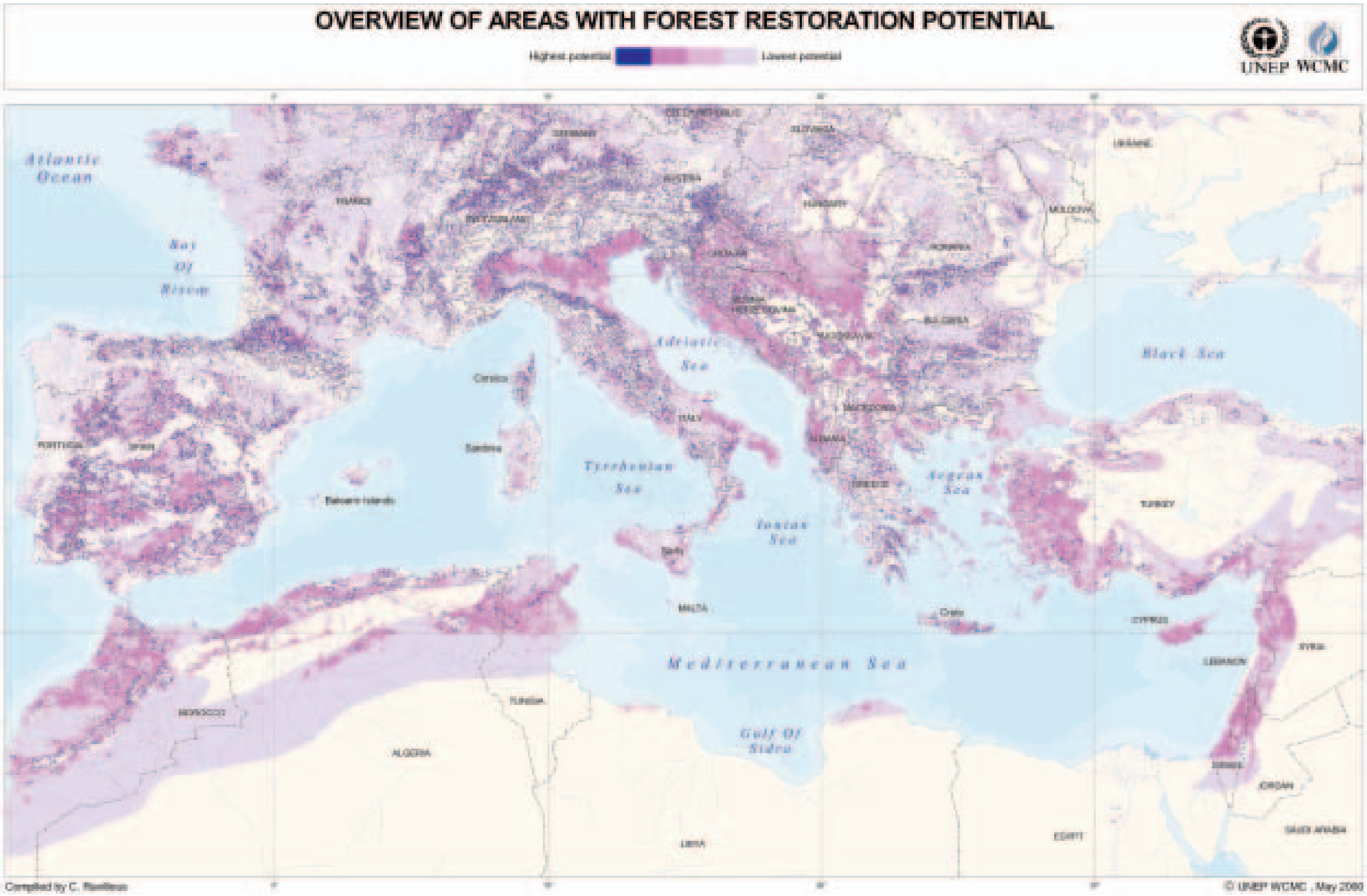


Figure 2.5

Results of a GIS analysis defining priority areas for forest restoration according to the criteria listed in Table 2.1 (from WCMC, 2000) (pixel size 0.25 km²).

The approach described above for setting regional priorities could also be used for identifying suitable locations for restoration projects at the national or sub-national level. To date, no such analysis has been undertaken in the UK (Ferris and Purdy, 1999; Towers, 1999), although it would be of value for developing a coherent national strategy for woodland restoration. At present, woodland restoration projects in the UK are largely undertaken on an opportunistic basis, with little reference to any overall strategy (Newton *et al.*, 2001). Consideration of restoration of wooded landscapes in a regional or even global context would assist the development of such a strategy.

The contribution of the UK to global restoration efforts

The forests of the UK could be considered to be of minor importance in terms of their contribution to global biodiversity. The UK is characterised by a high degree of forest loss that has occurred over many millennia, in sharp contrast to the rapid deforestation that has occurred in many tropical countries in recent decades. The diversity of species associated with woodlands in the UK is relatively low compared to those of neighbouring continental areas, as a result of its distinctive post-glacial history. However, the UK does possess woodland types of at least regional significance in terms of their conservation value for example the native pinewoods of Scotland and the oakwoods of the Atlantic seaboard (Newton and Humphrey, 1997; Peterken, 1996).

Woodland restoration initiatives in the UK face a number of particular challenges. Firstly, as noted above, the remaining extent of forest cover is very low, which can act as a significant constraint to restoration efforts, for example by limiting the availability of locally adapted germplasm or sources of potential colonists (Newton and Ashmole, 1998; Ennos, 1998; Ennos *et al.*, 1998). Secondly, those native woodlands that do remain have been substantially affected by the impacts of human activity over a prolonged period, extending over centuries if not millennia (Peterken, 1996). This greatly complicates the definition of restoration objectives, and also limits the understanding of ecological processes in native woodland communities. For example, it is impossible to define with precision the characteristics of disturbance regimes prevalent in UK forests prior to human impact, and their relationship to the composition, structure and function of woodland communities (Newton *et al.*, 2001; Peterken, 1996). Thirdly, the UK is a populous country, particularly in the south-east of England. This has resulted in intensive patterns of land-use, leaving little scope for the expansion of native woodland areas without impinging on alternative land-uses and economic activities. These characteristics contrast markedly with areas such as the eastern United States, where forest has recovered naturally over extensive areas following abandonment of farmland in the late 19th century (Peterken, 1996).

Considered from a global perspective, woodland restoration projects in the UK play an important role in demonstrating the challenges of undertaking restoration in disturbed and degraded habitat in areas of intensive land-use. Potentially, experience in the UK could also provide a valuable demonstration of how such challenges can be overcome. Woodland restoration projects in the UK tend to be characterised by a high degree of collaboration between both governmental and non-governmental organisations (NGOs). A supportive environment policy and the availability of financial support (for example through the National lottery) have also been crucial factors in encouraging the rapid growth of interest in native woodland restoration in recent years (Newton and Humphrey, 1997; Newton *et al.*, 2001). Restoration initiatives in the UK could also be of value in demonstrating how best practice might be achieved, given the relatively high level of technical skills and access to resources that exist in this country, as illustrated by other contributions to this Technical Paper.

Conversely, the experiences of restoration projects in the UK also demonstrate how difficult it can be to overcome some of these challenges, and the high expense involved. The UK is host to an increasing number of overseas visitors interested in learning how woodland restoration can be achieved in practice. For example, during 2000 the Society for Ecological Restoration held its first international conference in this country, and a delegation of international visitors took the opportunity to visit a number of different restoration projects in progress. One of the most potent lessons for visitors is how difficult it can be to replace forest ecosystems once they have been lost. Hopefully, such experiences will encourage the effective conservation of the forests that remain, reducing the need for restoration in the future.

Conclusions

The need for forest restoration has been identified by a number of global initiatives, as a key activity for reversing the trend of forest loss and improving biodiversity conservation. These initiatives include the Convention on Biological Diversity (CBD) and the international forest processes (IPF and IFF). A large number of NGOs, government departments, research institutes and private sector companies are involved in forest restoration initiatives throughout the world. At present, such activities are largely undertaken in an opportunistic manner, rather than as a contribution to an overall strategy. Priorities for forest restoration therefore need to be developed at both global and regional scales in order to ensure that resources are being focused effectively on those forest areas most in need of restoration.

Although a number of preliminary suggestions are presented here, further research is clearly required to define how global and regional priorities for forest restoration might be defined most appropriately. In addition, there is a need to assess the distribution of restoration projects, in order to determine whether current activities are adequately addressing restoration priorities. At present, there is no central repository of information on the distribution and characteristics of forest restoration projects to provide guidance for focusing future restoration efforts. The Forest and Drylands Programme of UNEP-WCMC is therefore developing an open access, on-line database of forest restoration projects that will facilitate exchange of knowledge and experience among projects, provide a basis for analysis of existing projects, and thereby improve prioritisation, design and execution of future restoration efforts.

Current restoration activities in the UK can therefore be viewed as part of a global effort for improving the conservation status and habitat value of forest ecosystems. It is probable that woodland restoration efforts in the UK will only ever have a minor impact on conservation of biodiversity worldwide. However, the approaches and techniques for forest restoration being developed in the UK could be readily applicable in other parts of the world, and experience gained in the UK is already proving valuable for demonstrating how the challenges facing restoration efforts world-wide can be overcome.

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SECTION TWO

Research and Modelling tools

CHAPTER 3

The processes of species colonisation in wooded landscapes: a review of principles

Paul M. Dolman and Robert J. Fuller

Summary

We review theoretical and empirical aspects of species dispersal and colonisation in relation to the creation of new native woodlands in Britain. The relevance of the 'meta-population' concept and the usefulness of linkage and movement corridors are questioned. Empirical evidence suggests that detrimental edge effects may have major consequences for the long-term conservation potential of re-created woodland. We argue that maximising woodland size and contiguity, in order to increase the core area and decrease the proportion of habitat close to external edges, should be a higher priority within restoration plans than the creation of corridors between isolated woods. Nonetheless, the ecological importance of habitat mosaics and long-established woodland edges needs to be recognised. Many specialist woodland species will take a very long time to colonise newly created woodlands. This is due to specialist habitat requirements, slow development of suitable habitat within recipient sites, and dispersal ability that may be intrinsically limited. Because of this, core areas should be concentrated around existing remnants of ancient and semi-natural woodland. Key areas for future research relevant to the colonisation and persistence of populations in restored woodlands include responses of different taxa to edges and the processes of species-specific dispersal.

Introduction

Man has depleted Britain of much of its native woodland over a long historical period. The recreation of native woodland, which we define simply as woodland composed of native tree species, is therefore a laudable conservation aim. However, the re-creation of woodland ecosystems, similar to those in extant patches of native woodland, is likely to be an extremely slow process. In order for diverse communities of woodland plants and animals to become established, species must be able to colonise the wood and this requires that multiple conditions are fulfilled. We define 'colonisation' as a two-fold process involving initial dispersal from a source population to the new wood and also the subsequent development of a persistent and viable population. Many factors potentially influence the probability of colonisation, including those operating at the scale of the surrounding landscape, others at the scale of the wood itself and others that are intrinsic to the species (Fuller and Warren, 1991). This chapter: (a) considers the empirical support for key aspects of theory concerning processes of species colonisation and draws conclusions about the relevance of this theory to the re-creation of temperate wooded landscapes, especially in Britain and (b) briefly reviews the dispersal ability of different woodland taxa.

Defining landscape structure

The nature of the regional landscape within which the new wood is located influences the availability of a viable regional source population – a prerequisite for colonisation. Aspects of landscape structure also have major consequences for the process of dispersal from the source population into the new woodland, though this also depends on dispersal characteristics of the species itself. Clearly, for successful establishment of dispersing individuals, a range of suitable macro- and microhabitats must be available within the 'new' woodland; for some species associated with mature woodland this may take many hundreds of years to develop. (A review of the significance and processes of habitat development is beyond the scope of this chapter.) Population establishment may also depend on spatial structure within the woodland, as well as the context of the woodland within the surrounding landscape. Finally, it is important to recognise that sources for dispersal may occur in pre-existing woodland fragments within the matrix of the re-created forest. For these reasons, we consider the landscape structure *within* the new woodland, as well as that of the surrounding regional landscape.

Landscape structure is a critical factor affecting species colonisation, therefore we start by defining its components and outlining their ecological significance. In doing so we broadly follow the framework of spatial processes provided by Dunning *et al.* (1992).

Landscape composition

This refers to the relative amounts of each habitat in the landscape. An important point to emerge from modelling studies of landscape composition is the strong non-linearity of habitat area effects. For example, simulations (e.g. Andrén, 1994; Bascompte and Solé, 1994) show that as the proportion of suitable habitat decreases from initially high levels, patch size and contagion/isolation show little change at first. However, once total habitat declines below a critical value, area and isolation effects compound the effects of habitat loss. Similar 'threshold' responses have been found in a number of species-specific studies. Models of capercaillie *Tetrao urogallus* populations in Sweden suggest a strongly non-linear response of population density to the proportion of old boreal forest retained in the landscape (Anglestam, 1992), while densities of American martens *Martes americana* show a non-linear response to the proportion of non-forested landscape, being absent in landscapes with >25% open ground (Hargis *et al.,* 1999).

Patch size and contagion

The 'grain size' of habitat patches within complex mosaics may vary enormously. In the case of gap dynamics, openings range from individual fallen trees within closed woodland to patches of many hectares (Peterken, 1996). Species differ in how they 'perceive' or respond to a given landscape grain, depending on the relative scale of their population structure and dispersal ability. For example, a habitat 'patch' that supports an isolated population of a poorly dispersing invertebrate may represent an individual territory within a local bird population, or a transient feeding patch within the home range of an individual deer. Species with minimum area requirements may not persist in a landscape in which suitable habitat is dispersed in patches that are too small; examples of area thresholds for woodland birds are given by Hinsley *et al*. (1995). The degree of *contagion* of patches of similar habitat (or conversely their isolation) may be important to 'core' species with limited dispersal ability. In contrast, more mobile species may be able to use resources distributed across isolated patches; for example the great spotted woodpecker *Dendrocopos major* readily flies over open areas to exploit small scattered patches of deciduous trees. Decreasing patch size may also reduce the quality of habitat through edge effects.

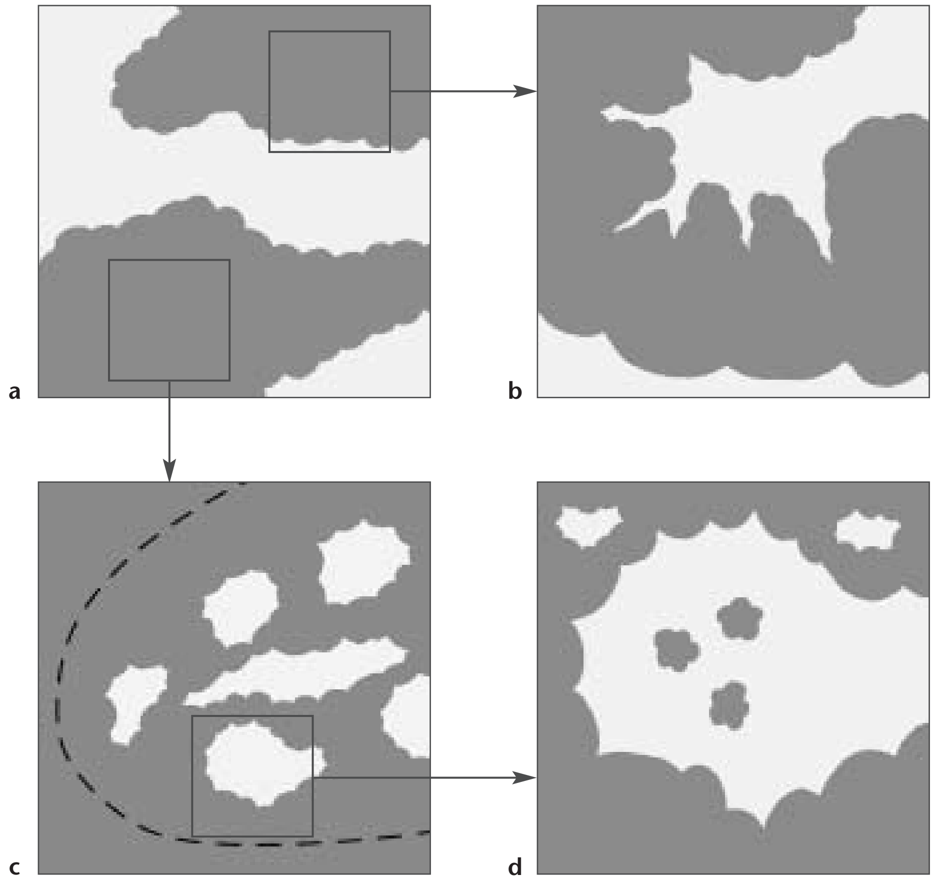
Landscape complementation through habitat juxtaposition

Species often derive differing non-substitutable resources from more than one habitat (Dunning *et al*., 1992). For example, many saproxylic invertebrates that depend on deadwood also require nectar resources during the adult stage (Kirby and Drake, 1993), while green woodpeckers *Picus viridis* require mature trees for nesting and open ground for foraging. A number of bat species require suitable nursery roosts located adjacent to a variety of foraging habitats that can provide food resources throughout the breeding season and in different weather conditions (Duverge and Jones, 1994). In such cases, both the relative abundance and proximity of differing habitats will influence habitat occupancy and population density. This may have consequences for regional persistence and subsequent colonisation of new woods, but also for the design of the 'core' woodland.

Fractal landscapes

Theoretical and empirical treatments of landscape composition generally consider habitat patches as homogeneous and mutually exclusive. In reality, landscape structure is nested in a hierarchy of scales and may be 'fractal', i.e. patches and networks exist within patches. What we recognise depends on the scale at which we look – differing patterns of patch boundaries, grain size, fractal dimension and shape metrics emerge when a single habitat is analysed at coarser or finer resolution (Wiens, 1995; Forman, 1995a). An example is developed in Figure 3.1. Such fractal complexity is likely to be of critical importance to the establishment, persistence and dispersal of specialists within woodland.

Figure 3.1 *Illustration of how fractal landscapes may apply in a woodland context.*
a. Forest blocks within landscape dark grey.
b. Open habitat within forest (e.g. grazed areas).
c. Large-scale patches with a higher frequency of windthrow due to topographic exposure.
d. Individual old trees surviving within regenerating windblown patch, and individual tree fall gaps within nearby old growth.



Relevant theory of spatial population processes

Source-sink dynamics

Pulliam (1988) postulated that sub-populations occupying patches of habitat of differing quality may give rise to source-sink dynamics. ‘Sinks’ are habitat patches where population productivity is insufficient to balance mortality and their persistence depends on reinforcement by immigration of individuals from nearby productive ‘source’ populations. Despite the intuitive appeal of this idea and its frequent adoption as a guiding principle (e.g. Forman, 1995a), few convincing examples exist. In particular, the definition of populations as ‘sinks’ implicitly assumes that they would become extinct if cut off from the ‘source’ of immigrants. However, this ignores the potential for stabilisation at a lower equilibrium density, as may occur for example with a compensatory density-dependent increase in productivity (Watkinson and Sutherland,1995). Nonetheless, there are many examples of species showing variation in birth and death rates in different habitats. A more generalised approach, that is particularly relevant to animal populations, is given by Brown (1969). This emphasises the importance of ‘sub-optimal’ habitats in increasing total population productivity and ‘buffering’ the core population in the optimal habitat, thereby increasing both the total size and resilience of the population.

Meta-population theory

A meta-population may be defined as consisting of a number of extinction-prone local populations, occurring in discrete patches of suitable habitat. Regional persistence results from turnover of these semi-isolated local populations, through area-dependent local extinction balanced by recolonisation (Hanski, 1999). Meta-population theory has been canonised as a new paradigm in conservation biology, replacing controversial principles from island biogeography in guiding reserve design (Hoopes and Harrison, 1998; Harrison and Bruna, 1999). But it is very important to realise that even where the distribution of populations of a species, viewed at one snapshot in time, reveals a pattern of occupied patches and unoccupied but otherwise apparently suitable patches (e.g. Opdam *et al.*, 1994), the regional population system is *not* necessarily a meta-population (e.g. Hinsley *et al.*, 1994). Applied uncritically, meta-population theory could lead to the conclusion that single isolated populations are always doomed and costly strategies involving multiple connected reserves are always necessary (Harrison, 1994). So what is the evidence for meta-population processes being important in regional persistence, particularly in woodlands?

A review of empirical data suggests that few animal species persist via 'patch turnover' – the precarious balance of extinction / recolonisation of discrete local populations. Those that do, tend to be species that depend on ephemeral early-successional habitat, but have limited dispersal ability (Harrison, 1994). Examples include the pool frog *Rana lessonae* and a few butterflies (Harrison and Fahrig, 1995; Hanski, 1999; Harrison and Bruna, 1999). In contrast, most species probably exist as either:

- 'Patchy' populations with local aggregations linked by continuous dispersal.
- Virtually or completely isolated populations showing long-term persistence; here, local extinction is part of regional population decline, not a balanced meta-population process.
- A 'mainland-island' system with small ephemeral populations repeatedly re-established by colonisation from nearby persistent large populations.

Consequently Harrison (1994) concluded that, to be useful, the meta-population concept should be adopted in a broader and vaguer sense, as a regional set of spatially distributed populations between which dispersal may or may not occur. This broader interpretation precludes simple generalities about 'optimal' conservation design.

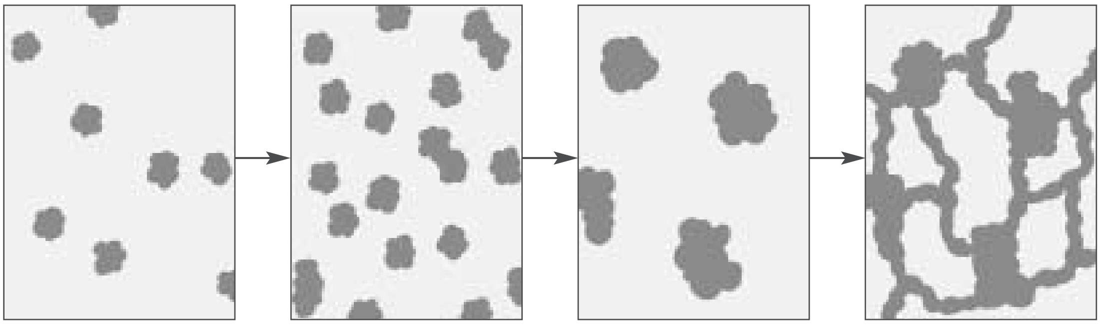
Eriksson (1996) argues that an animal-oriented view has dominated attempts to formulate spatial population models and provides a valuable review of regional-scale plant population dynamics. This emphasises the importance of persistent local populations that survive long enough to bridge periods of unfavourable habitat quality (such as may occur through successional development). Eriksson defines this as 'remnant population dynamics', in contrast to examples of meta-population, or putative source-sink, dynamics. Remnant population dynamics are particularly frequent among long-lived plants with clonal propagation, such as aspen *Populus tremula*, bracken *Pteridium aquilinum*, bilberry *Vaccinium myrtillus* and other ericaceous shrubs, and also among plants with extensive seedbanks. Eriksson (1996) suggests that remnant population dynamics may be common in temperate and boreal landscapes, as evidenced by the proportions of regional flora exhibiting clonal propagation and persistent seedbanks. The finding by Honnay *et al.* (1999a; 1999b) that 'species-relaxation' (i.e. area-dependent stochastic extinction) was of minor importance in explaining nested patterns of species distributions of forest-core plant species in Flemish woodland fragments, supports the conclusion that many such species have highly persistent populations, even where population size is small. Similarly, Harding and Rose (1986) and Peterken (1993) provide evidence that some plant species with very limited dispersal abilities will persist almost indefinitely in stable woodland environments.

In contrast, meta-population dynamics (both classic and mainland variants, as well as putative source-sink systems) tend to be found in short-lived plants colonising small-scale disturbances (Eriksson, 1996). Similar dynamics may also occur in some other species that are dependent on within-woodland dispersal between ephemeral habitats, such as basidiomycetes or mosses colonising decaying logs and also in tree parasites such as mistletoe *Loranthus europaeus* (Herben *et al.*, 1991; Eriksson, 1996; Marren and Dickson, 2000).

Studies of habitat fragmentation

Many empirical and theoretical studies have addressed issues of habitat fragmentation. A number of guiding principles have emerged for landscape design (Figure 3.2).

Figure 3.2 *Effects of landscape structure on population persistence (based on Harrison and Fahrig, 1995). The probability of population persistence increases from left to right.*



More habitat *Larger patches* *Greater connectivity*

In summary, more habitat is better than less, larger patches are better than smaller patches, and greater connectivity is believed to further increase the conservation value of the landscape, facilitating dispersal and reducing the probability of local extinction (e.g. Forman, 1995b; Harrison and Fahrig, 1995). These principles can be applied at varying scales. In terms of patches of forest within the landscape they can be used to guide actions designed to favour regional persistence of populations to act as a source for colonisation and to facilitate dispersal of individuals to the newly created 'core'. Within a forest they can be used to identify spatial structures and habitats likely to affect colonisation from pre-existing internal refugia and the chance of colonists establishing persistent populations.

A caution is required in the context of habitat creation. Principles derived from studies focused on mitigating species losses from habitats undergoing fragmentation may not be directly applicable in reverse, i.e. to facilitating the colonisation of restored or recreated habitat.

Patch size and isolation

Studies of fragmentation have led to two major generalisations. Firstly, that small isolated fragments retain fewer species than larger, less isolated fragments; secondly, that the species retained will not be a random subset of the original species pool, but instead will have been filtered by ecological, life history and functional attributes. A number of elegant studies have manipulated artificial patch networks at micro- and mesocosm scales (e.g. experimentally fragmented patches of moss on rocks and experimental patches of grassland), but effects of patchiness in such studies are largely behavioural and would not necessarily translate to larger spatial scales (Harrison and Bruna, 1999). Most of the rigorous studies of large-scale fragmentation have been conducted in neotropical forest (e.g. review by Harrison and Bruna, 1999). These studies show that fragments tend to become biologically impoverished, with a loss of specialists and retention or gain of generalists. Such community modification can potentially alter key ecological processes such as pollination, decomposition and nutrient cycling, seed dispersal and predation (Harrison and Bruna, 1999). However, it is not clear to what extent conclusions from these studies hold for temperate and boreal systems. Although relatively little work has been carried out on landscape-scale fragmentation in temperate regions, both patch size and isolation have been shown to affect species richness of a variety of animal taxa. Examples include birds in farmland woodlots in eastern England (Hinsley *et al.*, 1995), pine plantations in Spain (Diaz *et al.*, 1998) and Dutch marshlands (Opdam *et al.,* 1994), habitat specialist butterflies in montane fens in Switzerland (Wettstein and Schmid, 1999) and specialist heathland invertebrates in southern England (Hopkins and Webb, 1984; Webb, 1989).

In contrast, for temperate forest plants, landscape-scale studies suggest that many species are highly persistent even in small fragments (see above) and species-area effects owe more to habitat heterogeneity than differences in species density per unit area *per se* (e.g. Honnay *et al.,* 1999a; 1999b). Consequently, in temperate regions even small, isolated forest remnants may retain significant conservation value, while the cumulative species richness of small, geographically separated woodlands may exceed that of a single large woodland of similar area.

Edge effects

Fragmentation alters physical conditions and creates edges that differ in habitat quality to interiors. There is much empirical evidence to support the conclusion that edge effects are key mechanisms in both tropical and temperate forests (e.g. Murcia, 1995; Didham *et al.*, 1996; Paton, 1994; McCollin, 1998). This leads to the 'core area concept' (Laurance and Yensen, 1991) and an emphasis on the creation of large contiguous blocks of woodland rather than scattered woodland patches. The mechanisms underlying deleterious edge effects may include increased predation and nest parasitism, altered microclimates, altered habitat structures and in agricultural landscapes the penetration of fertiliser and biocidal sprays (Andrén, 1995; McCollin, 1998; Honnay *et al.*, 1999a). However, edge effects are not always negative. In 'cultural landscapes' such as those of lowland England, many edges at the interface of woodland and farmland and between semi-natural habitats are long established and support rich communities of shrubs, trees, invertebrates and birds (Fuller and Warren,1991).

To date there has been insufficient research to evaluate the magnitude of edge effects for different taxa in varying types of temperate forests. A review by McCollin (1998) shows that in temperate forests microclimate modification at forest edges may penetrate 20–60 metres, with effects on air temperature, light intensity, relative humidity and vapour pressure deficit, litter moisture, and the moisture and temperature of soils. Some preliminary conclusions regarding the consequence of edge effects for woodland plants can be drawn from distributional patterns between woodlands. The results of such studies from woodland fragments in Flanders (Honnay *et al.*, 1999a; 1999b) and southern Poland (Dzwonko and Loster, 1988; 1992) are scale-dependent, with little evidence for any loss of 'core' vascular plants due to edge effects except in very small fragments. This suggests that, for most woodland plants, deleterious influence of edges do not extend particularly far into woodlands, and that fragments of only a few hectares may retain 'core' habitat. However, there are indications that some species may be more sensitive to edge effects, particularly ferns that may be detrimentally affected by reduced air humidity (Honnay *et al.,* 1999b). In contrast, beneficial effects of increased perimeter length (controlling for area) have been found for the species richness of some plant groups, including woody species, lianas and others previously classified as being dependent on 'edges and clearings' (Honnay *et al.*, 1999a). These conclusions remain tentative and should be further explored by studies specifically designed to evaluate the consequences of edge effects in different types of woodland.

Didham *et al.* (1996; 1998) show that various invertebrate functional groups may be particularly susceptible to edge effects, at least in neotropical forest fragments. They found that both forest area and proximity to edge affected species richness and population density of beetles. There is a need for similar studies in temperate woodlands. In conclusion, it is likely that edge effects will reduce the quality of much temperate woodland for *specialist* species, such as molluscs and epiphytes associated with veteran trees growing in conditions of stable microclimate (Harding and Rose, 1986; Rose, 1993).

Movement corridors and connectivity

A major generalisation from studies of fragmentation is that between-patch movement can be a key factor maintaining population persistence in habitat fragments. This perception has greatly influenced management recommendations, leading to the 'ideal' of the connected landscape (e.g. Forman, 1995b; Potter, 1995). This can manifest itself in various ways. For instance , within farmland, scattered woods may be linked by hedgerow networks, while within woods rides can provide connectivity. Harrison and Bruna (1999) challenge this paradigm of 'linkage', pointing out the serious mismatch between theory and evidence; although theory portrays fragmentation as a problem of dispersal, empirical evidence suggests that edge effects and patch size are the overwhelming problems for many species.

The literature concerning corridors contains much confusion and apparently opposing views. For example, Harrison and Bruna (1999) conclude that 'no evidence supports the proposition that corridors can mitigate overall loss of habitat' while Beier and Noss (1998) state that 'it is safe to assume that a connected landscape is preferable to a fragmented landscape…. the evidence from well-designed studies suggests that corridors are valuable tools'. In addition to such conflicting views, there is often ambiguity and confusion between: (i) corridors that essentially allow movement of individuals dispersing from a population source in a 'core' area to other suitable patches of habitat, (ii) resident populations occupying linear habitats, that may allow range expansion by 'percolation', and (iii) the periodic use of linear habitats during foraging movements within an individual's home-range. For example, numerous studies have shown that the probability of some animal species occupying a wood may be increased by both wood size and connectivity. But where the species is resident in the 'corridor', for example dormouse (Eden and Eden, 1999) and many of the bird species considered by Hinsley *et al.* (1995), such effects may result from buffering population size through greater habitat area, rather than connectivity and inter-patch dispersal.

Cain *et al.* (1998) argue that 'because it would take forest herbs a hopelessly long period of time to move long distances via the standard processes of seed dispersal, our results suggest that at large spatial and temporal scales, corridors may be of little consequence for the direct (unaided) dispersal of woodland herbs.' Helliwell (1975) and Fritz and Merriam (1994) both found very few forest-understorey plants growing in hedgerows thought to act as potential dispersal corridors. The species most likely to filter along corridors are those for which the corridor provides suitable habitat capable of supporting a population, i.e. not specialist 'core' species (Kirby, 1995). Honnay *et al.* (1999a) for example, point out that forest 'core' plant species are generally poor at establishing in disturbed soil. In similar conclusions, Peterken (1993) has pointed out that '(secondary) woods rapidly acquire the catholic and gap-phase species from hedgerows, pastures and disturbed ground but other species take centuries to arrive, if they arrive at all'. The take home message appears to be, that corridor networks of hedgerows and other linear features will help rapid colonisation of new woodlands by widespread and common species that would get there anyway, but will be of little help in the dispersal and colonisation of specialist and 'core' species that are of greatest conservation concern.

Dispersal ability of different groups: preliminary review of empirical evidence

What do we know of the dispersal ability of different species and functional groups? One generality is that dispersal tends to be leptokurtic, i.e. the majority of individuals disperse only a relatively short distance, while a small number undertake considerable long-range dispersal. This is illustrated by studies of dispersal in British populations of birds (Paradis *et al.*, 1998). However, these same data also illustrate the 'Achilles heel' of most attempts to study dispersal. The results for blackbird *Turdus merula*, for instance, derive from analysis of many thousands of ringing recoveries, yet the leptokurtic tail of long-range dispersers is still only represented by a handful of individuals. The quality of data available to quantify dispersal data in birds is highly unusual; for most taxa there are severe practical difficulties in gathering equivalent data (Portnoy and Willson, 1993; Eriksson 1996; Harrison and Bruna, 1999). Consequently, we lack quantifiable knowledge about the all important leptokurtic tail, so that generally we cannot distinguish between alternative statistical models that describe very different frequency distributions of dispersal distance (e.g. negative exponential versus algebraic distributions). This is a serious shortcoming, as the model which is chosen to describe the frequency of maximal/long-range dispersal events, may have a strong effect on the outcome and predictions of attempts to model dispersal and rates of recolonisation (e.g. Fahrig, 1991; Akcakaya, 1992; Cary *et al.*, 1992; Portnoy and Willson, 1993; Bennett, 1998).

An alternative approach is to infer dispersal ability by observing patterns of population spread and species occurrence. For example, among birds, willow warbler *Phylloscopus trochilus* is clearly a good disperser being able to rapidly colonise remote new habitats (Gillings *et al.*, 2000), whereas nuthatch *Sitta europaea* is a poor disperser (Matthysen and Currie, 1996). Among mammals, many are readily able to cross landscapes to colonise new habitats, as shown by rapid range expansions of introduced deer in England (Arnold, 1993), bank vole *Clethrionomys glareolus* introduced into Ireland (Smal and Fairley, 1984) and, more recently, re-expansion of polecat *Mustela putorius* range in Britain (Branson, 1998). In contrast, for bats, low productivity of source populations and the social structure of colonial breeding may restrict the establishment of new breeding groups.

Much of our understanding comes from comparing fauna and flora in ancient and nearby secondary woodland. For example, Peterken's classic work on vascular plant indicators of ancient woodland in Lincolnshire shows that, at least in this region, species such as pendulous sedge *Carex pendula* and wood anemone *Anemone nemorosa* have limited ability to disperse and colonise (Peterken, 1974; Peterken and Game, 1984), while herb paris *Paris quadrifolia* appears to have poor dispersal ability throughout its European range (Peterken, 2000a). Forest core plant species are generally characterised by a lack of long-distance dispersal mechanism and are usually not able to colonise empty forest patches (e.g. Honnay *et al.*, 1999b). In Peterken's study, most secondary woodlands colonised by 'ancient indicator species' are contiguous with, or near to, ancient woods or hedge refugia (Peterken, 2000a). Similarly, in southern Sweden dispersal of woodland field layer plants across ecotones between ancient woodlands and adjacent recent deciduous woods (on former arable land) occurs at rates of only 0–1.25 metres per year (Brunet and von Oheimb, 1998). However, the probability of locating indicator species does not fall to zero with distance from source, suggesting that even for these poor dispersers occasional long-distance events may occur (Peterken, 2000b). Cain *et al.* (1998) show the critical role of accidental rare long-distance dispersal events in the colonisation of forest-understorey habitat by woodland herbs. Such events may also have been important in anomalous colonisations of secondary woodlands in the Netherlands by ant-dispersed species and other plants that apparently lack any mechanism for long-distance dispersal (Grashof-Bokdam and Geertsema, 1998).

In addition to vascular plants, many other species and assemblages are largely restricted to ancient woodland. Examples include epiphytic lichens (Rose, 1976), basidiomycetes such as chanterelles and tooth fungi (Marren and Dickson, 2000), bryophytes (Ratcliffe, 1968; but see also Rose, 1993, who argues that most habitat-specific mosses are mobile and not restricted to ancient woodland), molluscs (Boycott, 1934) and invertebrates (Harding and Rose, 1986; Warren and Key, 1991). In the New Forest, 150-year-old oak stands adjacent to ancient woodland generally show little colonisation by epiphytic lichens of the Lobarion assemblage and even the oldest plantations have only been penetrated to a distance of 200 metres in 300 years (Rose, 1993).

From the evidence available it seems reasonable to conclude that many *specialist* woodland species have severely limited dispersal ability, with the result that populations have only persisted at sites with a continuity of suitable conditions. While this general picture is probably sound, caution should be exercised in interpreting the restricted distribution of some ancient woodland specialists. In some cases relict populations are barely viable, or in the case of many epiphytic lichens, sterile (Harding and Rose, 1986). Thus they are not currently capable of functioning as source populations, presumably due to degraded habitat suitability, edge effects and air pollution (e.g. Rose, 1993). A number of other factors may impede successful colonisation through unsuitability of the receptor site; for example-absence of particular microhabitats or presence of competitors. Given the highly specific habitat requirements of many saproxylic invertebrates (Warren and Key, 1991), epiphytes (Rose, 1976) and fungi associated with deadwood, the apparent failure to disperse may reflect small and localised source populations and a lack of suitable recipient sites as much as an intrinsic lack of dispersal ability.

Conclusions and recommendations

The development of functioning ecosystems within re-created native woodland will be an extremely long process, extending over hundreds of years. We make the following points concerning the implementation of woodland restoration where the aim is to restore biological communities, including specialist woodland species:

- Core habitats take a very long time to develop in quality (e.g. through succession, the development of microhabitats and soil). Once core habitat quality has developed, dispersal may then occur to secondary sub-optimal habitats that initially lacked a species. But for some species the newly created 'core' may remain a sink for many years and the existence of a few dispersers in a recently created habitat should not be taken as evidence that a self-sufficient population has become established. This emphasises the importance of retaining existing habitats and refugia as a source of colonists into the future. In this respect the importance of small and isolated remnant patches of ancient woodland should not be underestimated.

- Emphasis should be placed on establishing large core areas and on reducing deleterious edge effects. It is argued that in the long-term, this strategy will enhance the probability of successful colonisation by many genuine woodland-specific species (rather than generalists). However, it should also be recognised that creation of single large blocks of forest may not always be the best solution in attempting to maximise ecological richness at a regional scale. Where this is an objective, it may be preferable to create discrete woodland blocks across a range of soil and environmental conditions.

- Creation of corridors, with the aim of facilitating dispersal, is of doubtful value for many woodland species.

- Spatial and structural heterogeneity within and between landscapes will be valuable – different approaches and outcomes will be desirable in different locations. Although large contiguous blocks of woodland generally may be recommended in order to provide core habitat for specialist species, varied mosaics of woodland and other semi-natural habitats may favour species that require complementary resources and can help to maintain biological communities associated with long-established edges. Such mosaics may be encouraged to develop with a modicum of planting followed by treatments such as variable grazing in space and time. Where sufficient space is available, substantial areas of old-growth can be accommodated within planned mosaics. A planned approach may be especially valuable in maintaining diverse edge communities in some areas but avoiding deleterious edge effects in others. Where very large tracts of minimum intervention forest can be developed, gap dynamics would gradually generate natural mosaics which are likely to be very different in structure to ‘planned mosaics’.

- Regarding future research, there is an ongoing need for theoretical and empirical studies on communities at a landscape scale, particularly focusing on species interactions and keystone processes structuring ecosystems. More specifically, far more attention needs to be given to understanding edge effects and species-specific dispersal. Better documentation is needed of the magnitude and range of edge effects in temperate woodlands, the relative susceptibility of differing taxa and functional groups to such impacts and the underlying mechanisms. Our understanding of the dispersal abilities of many specialist woodland taxa is severely limited and the mechanisms that contribute to low dispersal are poorly understood. Creation of new wooded landscapes provides a valuable opportunity to design experiments on dispersal and colonisation. Studies using molecular markers can provide evidence of phylogeographic intraspecific population structure arising from historic separations, past colonisation events, in situations of limited contemporary gene flow between populations (Avise, 1994; Nichols and Hewitt, 1994; Ibrahim *et al.*, 1996; Cain *et al.*, 1998). Molecular techniques also offer the potential to study effects of landscape structure on contemporary rates of dispersal and may be particularly useful for taxa in which dispersal rates and distances are difficult to measure directly.

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CHAPTER 4

Establishing native woodlands in former upland conifer plantations in Ireland

George F. Smith, Daniel L. Kelly and Fraser J. G. Mitchell

Summary

Because of land-use change and the advent of sustainable forest policies, former upland conifer plantations may provide opportunities for native forest restoration at the landscape scale. To investigate the feasibility of restoring Irish oakwoods to such sites, we established, in 1999, 21 pairs of fenced and unfenced permanent plots in clearfelled conifer plantations in the Wicklow Mountains and Killarney National Parks. Browsing damage from deer, sheep and other animals caused significant mortality to planted sessile oak and downy birch seedlings. Mortality at first sampling was 11.3% higher for oaks and 22.4% higher for birch planted in unfenced plots than in fenced plots. Damage from small herbivores, mostly hares, also caused significant mortality in the first year. While mortality of undamaged birch seedlings was 34.7%, mortality of seedlings damaged by small herbivores was 58.8%. A reduced cover of felling brash was associated with higher birch mortality and higher frequency of small herbivore damage. Brash may thus play an important role in sheltering trees from browsing from certain animals. Natural regeneration of tree species was highly variable across the sites and appeared to be limited mainly by dispersal. The most abundant species were the invasive exotics Sitka spruce and *Rhododendron ponticum*. The majority of Sitka spruce reseeding seems to be limited to a ‘window’ of a few years before developing vegetation reduces recruitment opportunities.

Introduction

Restoring or creating native forests in Ireland and Britain at the landscape scale obviously depends on the availability of land. In the upland regions of Ireland, the National Parks and Nature Reserves networks are expanding and have acquired large areas of exotic conifer plantations. Because native woodlands occupy less than 1% of the land area of Ireland, Dúchas, the Irish conservation agency, is interested in the possibility of restoring and/or creating native woodlands in clearfelled conifer plantations. Recent interest in sustainable forest management may also encourage native woodland restoration on conifer sites in private and semi-state hands. Even if species and provenances suited to the site are selected, however, many technical problems can interfere with successful restoration, and former conifer stands represent a largely new situation for managers.

Difficulties likely to be encountered in restoring native woodlands on clearfelled conifer plantation sites include those common to any restoration effort and some which are unique to upland or former plantation situations. The former conifer crop and attendant silvicultural and harvesting practices alter vegetation development in ways that may affect woodland restoration (Rodwell and Patterson, 1994; Olsson and Staaf, 1995; Wallace and Good, 1995; Bergquist *et al.*, 1999). Browsing by deer, sheep, hares and, more locally, feral goats is a well-known obstacle to tree regeneration in deciduous woodlands (Mitchell and Kirby, 1990; Gill, 1992), and is likely to be a problem in upland clearfells as well. The brash remaining from the felling operation will affect establishment and growth of woody and herbaceous species and may influence the behaviour of grazing mammals (Grisez, 1960). Many upland conifer plantations are located long distances from existing broadleaved woodlands or other significant native tree populations. In these situations, planting will probably be necessary to achieve tree cover. Clearfelled sites are likely to provide opportunities for the invasion of exotic species, particularly conifers such as Sitka spruce *Picea sitchensis* (McNeill and Thompson, 1982; Clarke, 1992; von Ow *et al.*, 1996; Dagg, 1998), and also *Rhododendron ponticum*.

The objectives of our three-year research project are to determine: (1) in what situations natural regeneration will be sufficient to establish native woodlands, and (2) under what conditions will more intensive management options, such as tree planting, be successful. In this chapter we present first-year results on natural succession in clearfelled sites and survival of planted tree seedlings.

Methods

In 1999, 15 pairs of 20 x 20 m experimental plots were established in the Wicklow Mountains in the east of Ireland and 6 pairs in the upland areas of Killarney National Park in the southwest (Figure 4.1 and Table 4.1).

Figure 4.1

Locations of the Killarney National Park and Wicklow Mountains National Park study sites.

One of each pair of plots was fenced to exclude large herbivores, including deer, and was situated adjacent to an unfenced control plot. The climate of Killarney is extremely oceanic, with over 200 rain days per year and mild temperatures year-round (Mitchell and Ryan, 1997). The central Wicklow Mountains are slightly more continental, receiving 150–200 rain days per year and generally not exceeding 2000 degree days per year (Mitchell and Ryan, 1997). The plots were located in former plantations of Sitka spruce, lodgepole pine *Pinus contorta* and Japanese larch *Larix leptolepis*, felled between 1987 and 1998 (Table 4.1). The distance from the plots to surrounding native and non-native tree seed sources varied greatly. Table 4.1 shows the estimated mean distance from the plots in each study site to the nearest significant source of native tree and Sitka spruce seed. Here, a 'significant' source of seed is a group of more than 20 mature trees, such as woodland, a riparian strip of woodland vegetation, or a standing remnant of the conifer crop. Soils included deep peats, rankers, podzols and brown earths; these are being investigated in more detail at a later stage in the study.

The most appropriate woodland type for restoration in the Irish uplands on acid soil is oak woodland dominated by sessile oak *Quercus petraea* and downy birch *Betula pubescens* (cf. Kelly and Moore, 1975). Rowan *Sorbus aucuparia* and holly *Ilex aquifolia* are also important components, but Scots pine *Pinus sylvestris* has not been a native species in Ireland since the first millennium AD when it is generally believed to have become extinct (Mitchell and Ryan, 1997). In the spring of 1999, 48 oak seedlings and 24 birch seedlings were planted in each 400 m^2 plot. The Killarney plots were planted in March while the Wicklow plots were planted in May. The oak seedlings were two- or three-year-

Table 4.1 *Study site details and estimated mean distances to the nearest significant source of native species and Sitka spruce seed.*

Site	No. of plots	Geology	Elevation range (m)	Former crop	Felling years	Seed source distance (m)	
						Native species	Sitka spruce
Killarney							
Torc-Mangerton	3	Devonian Old Red Sandstone	240–255	Sitka spruce	1987–1988	100	200
Looscaunagh	3	Devonian Old Red Sandstone	215–240	Sitka spruce	1994	250	70
Wicklow							
Glendalough	4	Ordovician schist and phyllite	220–285	Japanese larch, Sitka spruce	1991–1995	125	100
Glenmalure-Baravore	3	Granite	160–225	Sitka spruce	1998	180	100
Glenmalure-Fraughan Rock Glen	6	Granite	320–350	Sitka spruce, lodgepole pine	1997–1998	600	150
Glenmalure-Benleagh	2	Granite	350–380	Sitka spruce	1993, 1998	500	80

old bare-rooted transplants ranging in height from 30 to 100 cm. Birch seedlings were 60–90 cm tall two-year-old bare-rooted transplants. Data on planted and naturally regenerating tree seedlings were collected in the summer and autumn of 1999. Data on herbaceous and shrub vegetation were collected from eleven 1 m^2 quadrats in each plot. Site data collected include brash height and density, slope, aspect, elevation and exposure. Also presented in this chapter are results from tree seedling data collected from nine of the Wicklow plots in the summer of 2000. Univariate statistical analysis was conducted using Data Desk (Data-Description, 1997) after consulting Sokal and Rohlf (1995). Vegetation data were investigated using beta-flexible clustering, an agglomerative clustering technique, and non-metric multidimensional scaling (NMDS), an ordination method appropriate for identifying patterns in complex data (Legendre and Legendre, 1998). These multivariate analyses were conducted using PC-ORD (McCune and Mefford, 1997).

Results and discussion

Vegetation development

The herbaceous plants and low shrubs present on a site can influence the development of woodland and the success of tree planting efforts. Vegetation, or its absence, can change the water and nutrient regimes of a site, may influence the behaviour of grazing animals, and can alter the availability of safe sites for germination. For example, birch regenerates poorly in grass or under a closed *Calluna* canopy (Kinnaird, 1974; Atkinson, 1992). *Pteridium*, *Calluna* and grasses such as *Deschampsia flexuosa* and *Holcus mollis* have been found to reduce the growth of oak, but *Ulex europaeus* and *Rubus fruticosus* agg. can provide protection from grazing and *Vaccinium myrtillus* has been observed to improve the germination and growth of oak seedlings under experimental conditions (Jones, 1959; Ovington and MacRae, 1960; Shaw, 1974; Evans, 1988; Humphrey and Swaine, 1997). In our study, each 100 m^2 plot quarter was assigned to a vegetation type based on the 1 m^2 vegetation quadrat data using beta-flexible clustering (with $ß = -0.5$). The last five clusters formed appeared to be the most ecologically meaningful (Table 4.2).

Table 4.2

Vegetation types occurring on former conifer plantations felled between 1987 and 1998 determined by beta-flexible clustering (ß = –0.5).

Vegetation type	No. of 100 m² plot quarters[a]	Dominant species	Years since felling (± standard error)
Grass	27	*Agrostis capillaris* *Holcus mollis* *Pteridium aquilinum* *Galium saxatile*	7.2 ± 0.3
Mixed grass–heather	14	*Agrostis capillaris* *Calluna vulgaris*	6.9 ± 1.3
Rush	32	*Juncus effusus* *Juncus bulbosus* *Digitalis purpurea*	3.6 ± 0.3
Heather	19	*Calluna vulgaris* *Erica cinerea*	10.1 ± 0.8
Early successional	72	*Calluna vulgaris* *Carex* spp.	1.8 ± 0.2

[a]Three plot quarters dominated by *Luzula sylvatica* and one dominated by Molinia caerulea were omitted because of their very different vegetation compositions.

Vegetation types based on additional clusters seemed to represent variants distinguished from the last five clusters by species composition, but not by notable differences in environmental variables such as soil type or time since felling. A non-metric multidimensional scaling (NMDS) ordination of the vegetation data (not included here) showed reasonable cluster separation and indicated that years since felling and slope had the greatest influence on vegetation types.

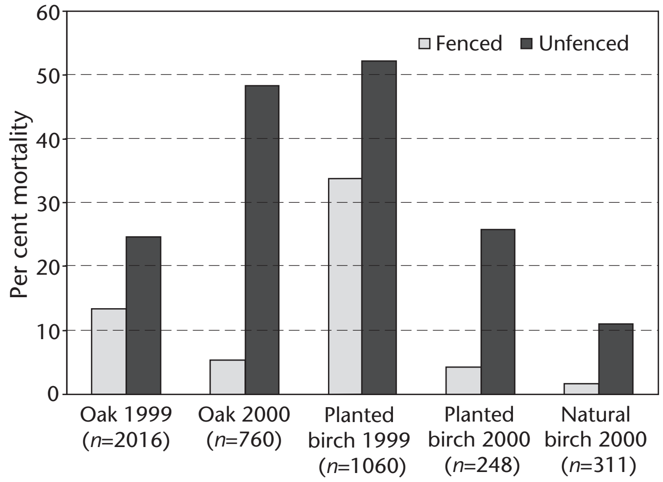
The early successional vegetation type generally occurred on former Sitka spruce sites that had been felled within the past year or two; this type was characterised by very little vegetation cover and high cover of needle litter. The rush type was found on flatter, more poorly drained ground. An intermediate stage in succession was represented by the mixed grass–heather type. The heather vegetation type appeared on old clearfells that seemed to be progressing well towards heathland, and was dominated by large *Calluna vulgaris* and *Erica cinerea*. The grass type was typical of older clearfells formerly occupied by larch. Judging by existing larch stands nearby, bracken and the dominant grasses almost certainly colonised the larch stands prior to felling. This vegetation type also existed in some places where Sitka spruce had been removed for several years.

In the early successional vegetation type, mortality of birch from planting to 1999 sampling, in fenced and unfenced plots, was significantly higher than in the rush and heather types ($P < 0.05$ according to an ANOVA on arcsine transformed data). Similarly, oak mortality was significantly greater in the early successional vegetation types than in the rush, heather and grass types. These results may be caused in part by higher severity of small herbivore damage and/or greater levels of seasonal water stress in early successional sites, which have little vegetation to shade the dark brown needle litter.

Herbivore damage

Unsurprisingly, browsing by large herbivores including deer, sheep and, in some locations, feral goats caused much greater mortality of trees planted outside than within fenced exclosures (Figure 4.2). Mortality of planted seedlings inside the exclosures was higher in 1999 than in 2000, doubtless a result of transplantation shock which was exacerbated by unavoidably late planting in the Wicklow plots. Late planting was almost certainly a large contributor to high birch mortality in 1999. Mortality of naturally regenerating birch seedlings between sampling in 1999 and 2000 was also significantly greater outside the exclosures (Figure 4.2).

Numbers of other naturally regenerating tree species were insufficient for meaningful analyses. Much of the natural regeneration in the older clearfells had become established prior to fencing, but was suppressed by high browsing pressure.

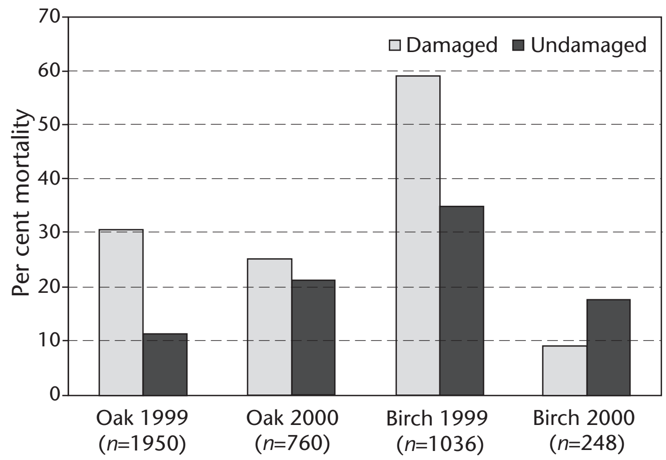


[a] Sample size (*n*) for the interval from planting to 1999 sampling is the number of trees planted; for the 1999–2000 interval, sample size (*n*) is the number of live trees in 1999 in the nine Wicklow plots resampled.

Figure 4.2

*Percent mortality inside and outside exclosures of planted oak and birch for the intervals from planting to sampling in 1999 and from sampling in 1999 to sampling in 2000. Also per cent mortality of naturally regenerating birch between 1999 and 2000[a]. Differences in mortality between fenced and unfenced seedlings in all five categories are significant (*P *<0.01) according to Fisher's exact test.*

Small herbivores were not excluded by the fences, and so severe damage from these animals occurred in several plots. Judging by browsing symptoms and abundant droppings, hares *Lepus timidus hibernicus* were responsible for the majority of the damage. Bank voles were introduced to Ireland around 1950 and are present in Killarney, but do not appear to have spread to Wicklow (Carruthers, 1998). Small herbivore damage was of two types: (1) bark stripping which often girdled and killed the trees, especially birch, and (2) browsing which often severed the leader, but usually did not cause the death of the tree. It was observed that bark stripping was the dominant form of damage in early successional vegetation, while in grass–bracken plots, bark stripping was rare. Mortality of oak and birch from planting to 1999 sampling was significantly higher for seedlings damaged by small herbivores (Figure 4.3).



[a] Sample size (*n*) for the interval from planting to 1999 sampling is the number of trees planted for which small mammal damage could be determined; for the 1999–2000 interval, sample size (*n*) is the number of live trees in 1999 in the nine Wicklow plots resampled.

Figure 4.3

*Percent mortality (in both fenced and unfenced plots) of planted oak and birch damaged and undamaged by small herbivores for the intervals from planting to sampling in 1999 and from sampling in 1999 to sampling in 2000[a]. Differences in mortality of oak and birch sampled in 1999 were significant (*P *≤0.0001) according to Fisher's exact test.*

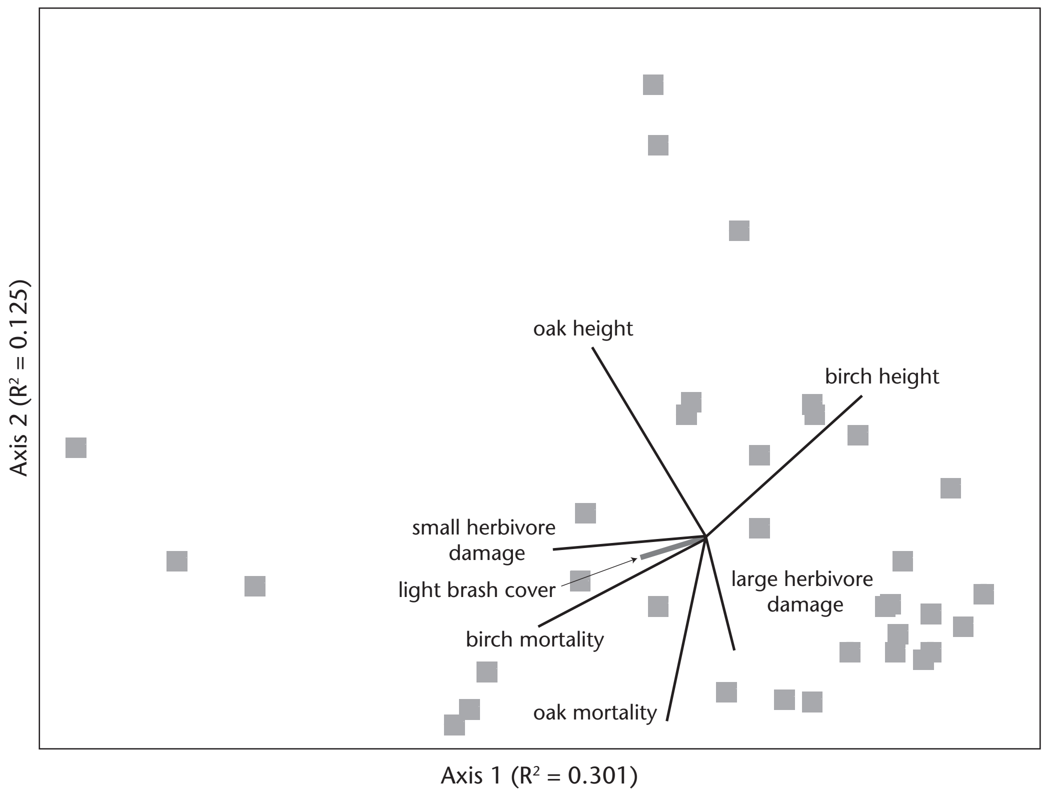
Mortality between 1999 and 2000, however, was not affected in the same manner. The differences between years may indicate an interaction with transplantation shock, may reflect increases in abundance of alternative food sources in the early successional vegetation type, or may simply be an artefact of the smaller 2000 sample size.

Brash influences

Brash left behind from felling operations is regarded with mixed feelings by those engaged in reforestation or attempting to encourage natural regeneration. Retention of brash on-site is suggested by some to maintain soil nutrient status (Fahey *et al.*, 1991; Titus and Malcolm, 1999). On the other hand, a heavy brash cover may inhibit natural regeneration and make tree planting more difficult (Rodwell and Patterson, 1994). Yet again, dense brash may be important in providing refugia for woodland plant species (Olsson and Staaf, 1995; Bergquist *et al.*, 1999; Humphrey and Nixon, 1999) and protecting tree seedlings from browsing (Grisez, 1960).

To investigate the influence of brash on young tree browsing, brash height and density (in three classes: light, moderate, heavy) were sampled at 28 points in each unfenced plot. These were compared with several tree seedling variables: planted oak and birch height and mortality, frequency of browsing by large and small herbivores, and natural regeneration of birch, rowan, holly and Sitka spruce. Because of correlations within the data, NMDS ordination was used to identify the axes of greatest variation in the tree data (Figure 4.4).

Figure 4.4 *NMDS ordination biplot of 1999 tree seedling data; plots are shown by squares. R^2 values indicate the proportion of variation in the tree data 'explained' by the ordination axis. Length and direction of the tree variable lines reflect their correlations with the ordination axes. Brash data were overlaid on the ordination. Variables poorly correlated with either axis (including abundances of natural tree regeneration and frequency of heavy brash cover) are not shown.*



The brash data (mean brash height and frequency of each of the three density classes) were overlaid on the ordination to investigate the relationships between the two sets of data. Frequency of hare damage and birch mortality had high correlations with axis 1 ($r = -0.622$ and $r = -0.660$, respectively). Although frequency of heavy brash cover was not well correlated with either ordination axis, light brash cover was correlated with axis 1 ($r = -0.413$). These results suggest that areas of lighter brash cover favour small mammals, chiefly hares. The density of brash occurring in this study, however, may not be sufficient to deter large herbivores. Most of the sites in this study were felled motor-manually and the trees removed to the roadside by cable skidding, thus leaving only moderate amounts of brash on-site.

Natural tree regeneration

Abundance of natural regeneration of trees and *R. ponticum* in fenced and unfenced plots was highly variable (Figure 4.5). In addition to the species in Figure 4.5, sessile oak, lodgepole pine, Scots pine,

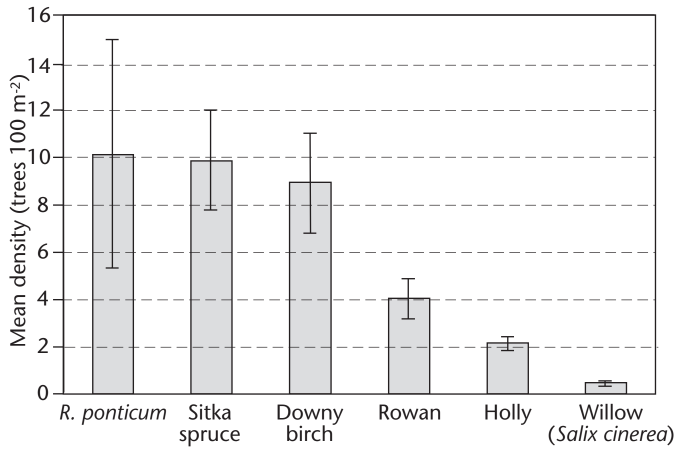


Figure 4.5

Mean density (trees 100 m^{-2}) and standard error of naturally regenerating tree seedlings and R. ponticum *appearing in greatest numbers in 1999.*

silver fir, Japanese larch and beech appeared in lower abundances. Apart from responses of unfenced birch regeneration to herbivory (and the influence of felling year and former crop on Sitka spruce discussed below), no environmental variables measured appeared to have a significant influence on natural regeneration abundances. Based on observations in the field, it seems that the primary factor at work is seed source availability. Birch seedlings were frequent in most plots within 250 m of native woodland or riparian strips of trees, but were virtually absent from the Fraughan Rock Glen and Benleagh plots, where a single tree represented the only birch seed source within 500 m or more (Table 4.1). Similarly, oak and beech seedlings were not seen in plots more than about 100 m distant from significant native seed sources. Rowan and holly seedlings, however, were found in nearly every plot and their abundances did not seem dependent on the location of woodland or other groups of trees. Mature rowan and holly trees occurred as scattered individuals clinging to the hillsides around all of the study sites. This distribution together with their bird-dispersed seed probably facilitates wide dispersal about the landscape. Most birch seed, in contrast, lands within 50 m of the parent tree (Atkinson, 1992).

The most abundant woody species in the early natural succession of clearfells were exotics (Figure 4.5). Regeneration of these species was also influenced by seed source availability. Invasion by *R. ponticum* is a much greater problem in Killarney than in Wicklow (Cross, 1982), and mature individuals were more abundant in the Killarney study sites than in Wicklow. In 1999 *R. ponticum* regeneration was only found in the Killarney plots, where it reached densities of up to 557 individuals 100 m^{-2}. Although *R. ponticum* is not as yet a serious threat in most of Wicklow, the appearance of *R. ponticum* seedlings there in 2000 suggests that it might be able to take advantage of the disturbance created by felling and become more invasive than before. Of the former conifer crops, Sitka spruce was the only species to reseed in substantial numbers. In some more recently felled spruce stands, new Sitka spruce regeneration occurred in high densities, while in earlier felled stands, the size distribution skewed towards larger individuals suggests that very little new regeneration is taking place (Figure 4.6a).

This supports earlier studies which found that Sitka spruce has a narrow ‘window of opportunity’ for colonisation before a site becomes dominated by competing vegetation (McNeill and Thompson, 1982; Clarke, 1992; von Ow *et al.*, 1996; Dagg, 1998). In former lodgepole pine and larch plantations, Sitka spruce regeneration was much lower (Figure 4.6b). One reason for the abundance of Sitka spruce seedlings in former spruce stands is probably the release of seed by the crop in the months prior to felling. When felling comes shortly after a good seed year, very high densities are likely (von Ow *et al.*, 1996; Dagg, 1998). Another contributing factor to higher abundances of Sitka spruce seedlings in former Sitka spruce stands may be the location of existing seed sources. In this study, plots in felled spruce stands tended to be closer to existing Sitka spruce plantations than plots in felled pine or larch stands. The latter plots probably receive an insufficient seed rain to yield problematic densities of Sitka spruce regeneration. Even light-seeded species like Sitka spruce have poor dispersal abilities over long distances (Mair, 1973).

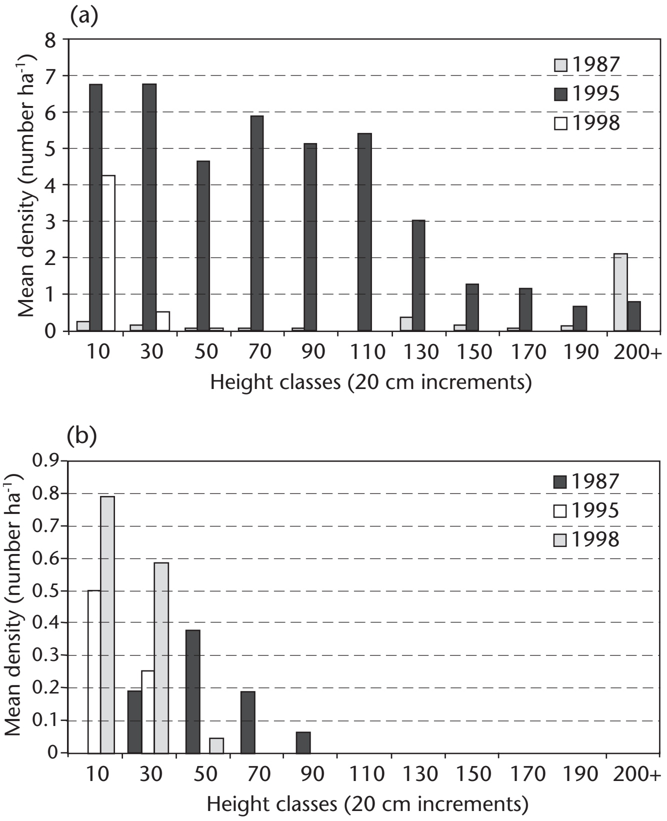


Figure 4.6

Mean density (number per ha) of Sitka spruce regeneration in 20 cm increment height classes. (a) former Sitka spruce stands felled in 1987, 1995 and 1998. (b) former pine or larch stands felled in 1991, 1993 and 1997 (note change in y axis scale).

Conclusions

What conclusions can be drawn from this study that are of use for those attempting to establish native woodlands on former conifer sites? Where protected from large herbivores, early survival of planted sessile oak and downy birch appears to be sufficient to form a woodland canopy. The potential effects of developing vegetation on establishment and survival of planted and naturally regenerating trees, however, should be kept in mind. Recently felled stands may be hostile environments for young trees, but on the other hand, well-established vegetation may reduce opportunities for natural regeneration. Where Sitka spruce reseeding is a potential threat, vegetation cover could be encouraged to reduce recruitment opportunities. Brash management, which this study has only begun to analyse, may have important implications for woodland restoration. Where only light brash cover exists, small mammal damage to trees can be high. In many former plantation areas, natural regeneration probably cannot be relied upon to establish woodland cover because of the scarcity of native seed sources. Bird-dispersed species such as rowan and holly will have little difficulty in reaching a site, but usually not in sufficient densities to form a woodland canopy. The presence of pockets of mature birch within a few hundred metres of the site may be necessary for adequate densities of birch regeneration. Where native seed sources are few, allowing succession to progress to heathland may be a more economically feasible conservation goal. The abundance of Sitka spruce and *R. ponticum* invasion, however, implies that succession to heath may not be an easy or assured option.

Our ongoing research should clarify many of the above issues. By measuring distances to seed sources and estimating abundances, we will provide a quantitative analysis of the role of dispersal in establishing native woodland cover and in Sitka spruce invasion. We are also currently investigating the influence that soils and the soil seedbank have on natural succession and tree survival and growth. A woodland soil translocation trial is under way to test whether or not this method can be effective in introducing native woodland ground flora to clearfells. Former conifer stands felled and abandoned for longer periods of time will be sampled to provide a longer-term view of succession.

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CHAPTER 5

Modelling the potential distribution of woodland at the landscape scale in Scotland

Alison Hester, Willie Towers and Ann Malcolm

Summary

The Native Woodland Model has been developed to link woodland and scrub habitat requirements with digital biophysical data, to predict the potential occurrence and distribution of a range of semi-natural woodland communities across Scotland. The model has been developed with the explicit aim of predicting patterns of woodland site capability over large areas without the need for ground survey. Thus the model provides a strategic planning tool, designed to be used at scales from 1:50 000 up to national level. Through the use of specific case studies, we outline the process of dataset integration, interpretation and model development, giving examples of the output generated. We discuss the important issues of uncertainty in woodland modelling, together with the options for validation when so little native woodland now remains in Scotland. Results of critical examination of our choice of datasets illustrate how soils and land cover data can act as robust surrogates for a range of climatic factors not directly used in the model. Validity testing of the model output to date has shown very good agreement with remaining native woodland and expert assessment on site suitability where woodland is no longer present. The merits and potential constraints of the approach used are discussed in the context of the applications for which the model is designed, and potential future applications are outlined.

Introduction

Over the past four millennia, the native woodland cover of Scotland has declined dramatically, and today only about 4% of the land area of Scotland is covered by semi-natural woodland and scrub (Mackenzie, 1999). The main reasons for this decline are agricultural development, timber harvesting, heavy grazing and climate change, but their relative importance remains controversial (McVean and Ratcliffe, 1962; Birks, 1973; Mackenzie, 1987; Smout, 1993; Fenton, 1997). In view of this decline, restoration and expansion of native woodland is now a conservation priority in many areas and a number of initiatives are in place to protect and expand this resource. These include the LIFE Caledonian Partnership, Habitat Action Plans and the Millennium Forest for Scotland Trust.

Despite the keen interest in woodland expansion in Scotland, little is known about the potential distribution and extent of different woodland types under current conditions; yet this kind of information is crucial to guide native woodland expansion at regional and local scales. Most surviving native woodlands provide only partial information since they tend to be highly fragmented and their composition has often been radically altered by heavy grazing, timber extraction and underplanting, and invasion by introduced species (Rodwell, 1991). Comparisons with historical reconstructions from palaeobotanical studies (McVean and Ratcliffe, 1962; Birks, 1973; Bennett, 1996) are of limited value as current site conditions have generally been modified by factors such as climate change and environmental pollution, the removal of the original forest cover, and agricultural cultivation of soils, as well as the fact that some taxa are more poorly preserved in pollen records than others.

We propose that a more useful and practical approach is to predict woodland distribution for current environmental conditions using site suitability models. A Native Woodland Model (NWM) has been developed which links published data and expert knowledge on woodland and scrub habitat requirements with digital biophysical data to predict the occurrence and distribution of a range of woodland and scrub communities at the landscape scale. Model development started in 1996 when the need for such a tool was identified, as part of a wider project in the Cairngorms area (MacMillan *et al.*, 1997). The main aim of the model was to predict general patterns of woodland site capability over large areas without the need for ground survey. As the model developed further, the aim has continued to be that of creating a strategic planning tool, designed to be used at scales from 1:50 000 up to the national level.

To make a strategic level model such as this, whole-Scotland datasets were needed, preferably in digital form to allow integration within a Geographic Information System (GIS). There was also a need for a common language to allow wider use of the model, so that all users could understand the predictions made. In this chapter we describe the development of the NWM, the datasets used, the validation of the model approach and output, and the purposes for which the model has been, and should continue to be, applied.

Model development

Two digital data sources are used in the model: the 1:250 000 scale national soils map (Macaulay Institute for Soil Research (MISR), 1984) and the 1:25 000 scale Land Cover of Scotland 1988 (LCS88) dataset (Macaulay Land Use Research Institute (MLURI), 1993). Both datasets cover the whole of Scotland and are in digital format. Together they contain a range of information relevant to the prediction of woodland and scrub communities, thus they are ideal for a strategic overview model such as this.

The 1:250 000 scale national soil map comprises 580 soil map units, differentiated on geological (soil association), pedological (component soils) and physiographic criteria (landforms). The soil map is also underpinned by an extensive database from which information on soil (and vegetation) properties important for the growth of different woodland types, such as base status, nutrient status and moisture regime, can be readily inferred. The system of soil classification and the terms used in soil description can be found in the handbook which accompanies the map series (MISR, 1984). Information of particular relevance to the definition of woodland suitability includes: (a) parent material and base status of different soils, (b) soil nutrient status, moisture regime and depth, and (c) landform features such as slope, rockiness or morainic deposits, which influence the proportions of different soil types within complex and heterogeneous landscapes.

The 1:25 000 LCS88 map provides information on Scotland's land cover as it was in 1988 and was captured from the visual interpretation of aerial photographs. The hierarchical classification allows for 126 single land cover features including all the major semi-natural vegetation communities. There are also over 1 000 mosaic categories used largely to describe the heterogeneous semi-natural vegetation resource. The LCS data provides valuable additional information, in particular by:

- providing higher resolution data than the soils data, so adding detail to the soil units and allowing some soil complexes to be disaggregated into their component parts in the landscape;
- locating cultivated land which allows delineation of soils where nutrients or moisture may have been altered by ploughing and fertilising (thus affecting woodland site suitability);
- delineating montane vegetation types and allowing the separation of the montane scrub zone from land considered unsuitable for trees/scrub, which is not possible using the soil data alone.

The 'common language' selected as most appropriate for the woodland and scrub descriptions was the NVC (Rodwell, 1991; Rodwell and Paterson, 1994), but we enhanced this with extra categories where we considered the NVC to be deficient, for example montane scrub types and more open woodland mosaics: for these we drew upon other available literature, in particular McVean and Ratcliffe (1962), McVean (1963, 1966), Wormell (1968), Birse (1982), Hester (1995) and Gilbert *et al.* (1997). Furthermore, because most NVC types also encompass much variation, we further refined the NVC predictions with more detail in the text accompanying all NWM maps, to guide the actual use of the model predictions specific to any area. Much of the original forest cover of the UK has been replaced over many hundreds (to thousands) of years by a number of semi-natural heath, grassland and bog vegetation communities. Each open vegetation community has different relationships with climate, soils, terrain, and many are variously described as the optimal precursor vegetation for specific NVC woodland types by Rodwell and Patterson (1994). It must be stressed that, although there is considerable knowledge about the relationships between many NVC woodland types and site conditions, this knowledge requires careful translation when applied to the integrated dataset derived from the soils and land cover data used in this study. By the very nature of what these data describe, they are imprecise and consequently some 'expert' judgement, interpretation and an understanding of the opportunities and constraints of the data are required.

The soils and land cover datasets were overlaid within a GIS forming a new integrated dataset to produce over 30 000 soil/land cover combinations. These combinations, which are essentially a description of the present site conditions, form the basis of the NWM predictions. Each combination is allocated to an NVC woodland type or to a mosaic of NVC types, based on the relationships between biophysical properties and woodland requirements, using available literature and expert opinion. It is important to note that the NWM thus predicts the potential for woodland and scrub types under current soil and vegetation conditions, i.e. with no or minimal intervention. The rationale underpinning this process is described in more detail below. Site requirements for some woodland and scrub types are better understood than for others, and thus the degree of certainty attached to specific predictions is also detailed in the text accompanying all NWM maps.

The philosophy, assumptions, uncertainties and methodology underpinning the model are described in the Appendices which accompany each report describing the model output, and will be published by the Macanlay Institute (Towers *et al.,* in press).

Model output

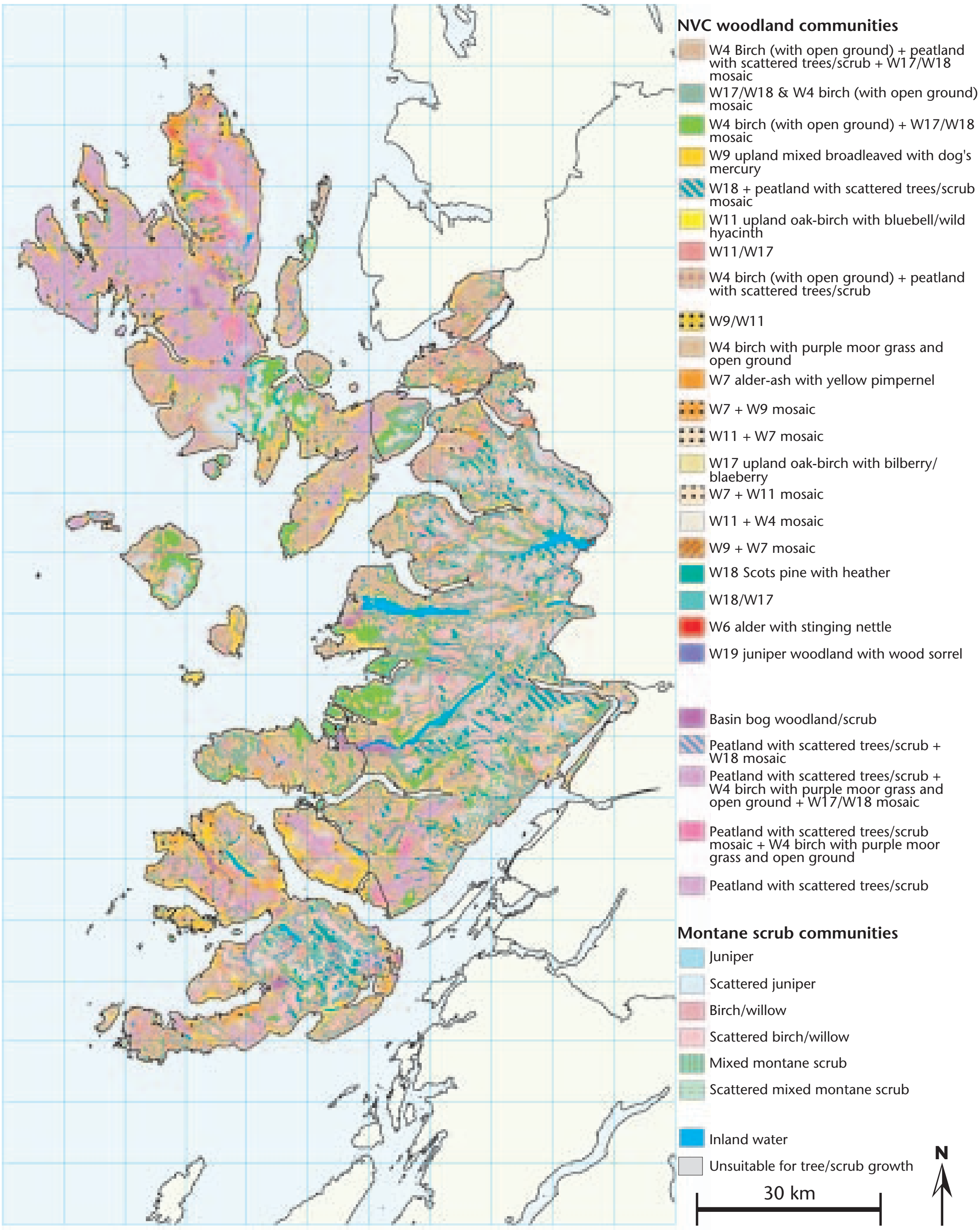
Figure 5.1 illustrates the NWM predictions for part of north-west Scotland. We can group the model output into three main types:

- **Single woodland types** These have been predicted where site conditions are clearly suitable for one woodland type alone at the scale of resolution of the model.

- **Mosaics** Mosaics of woodland types have also been extensively mapped under specific instances, in the following ways. Over large areas of upland Scotland, the soil and land cover pattern can be very variable even over short distances. The soil map data also contains information on the component soils below the scale of resolution of the map itself, and by integrating that information with LCS88 the complex soils and land cover can be disaggregated into a range of different 'soil landscapes' (Bibby *et al.*, 1982; see Figure 5.1 and point 3 below), which give information on the proportions of each component soil in different locations (Macaulay Institute for Soil Research, 1984: Macaulay Land Use Research Institute, 1993). Thus, each soil landscape contains a range of different site conditions with different potential for woodland or scrub, but below the scale of mapping used. By matching woodland/ scrub types to the different components of the soil landscapes, mosaics of NVC woodland types have been defined within the NWM as the most appropriate approach at this scale. The site-level requirements for each soil landscape are detailed in the text accompanying the NWM maps. For example, on moundy moraines, W18 is considered most suited to the peaty podzols on the mounds, whereas peatland with scattered trees/scrub is predicted on the peat in the intervening hollows and channels. This mosaic is indicated on the map legend as 'W18 + peat-land with scattered trees/scrub', with W18 being the dominant component as it is listed first.

- **Interchangeable types** Classifications, by definition, have to 'pigeon-hole' elements of the physical environment, whereas in reality many of these elements, such as vegetation and soils, are part of a continuum in the landscape. This problem manifests itself in the NVC, both in the similarity between descriptions of some woodland types and the overlap of species within

Figure 5.1 *Native Woodland Model output for Skye and surrounding area (SNH Natural Heritage Zones 6 and 8). The map shows the NWM-predicted site suitability for a range of woodland and scrub types.*



them (Rodwel, 1991; Rodwell and Patterson, 1994), and in the similarity or overlap of site conditions described as suitable for the different NVC types (Rodwell and Patterson, 1994). Indeed, Rodwell (1991) describes and illustrate many zonations of one woodland to another where soil conditions grade into others. The ‘interchangeable category’ has been used in situations where the soils are considered equally suitable for two woodland types and where the geology and/or overlying drift on which the soils are developed is intermediate between the optimal conditions for either woodland type. In our model predictions, we feel that it is important to distinguish these from mosaics, as described above, where different components of heterogeneous landscapes support quite distinct woodland types (but at scales more detailed than the mapping scales used here).

In addition to illustrating the woodland and scrub types defined by the model, the information displayed in Figure 5.1 illustrates the following five main points about the model output:

1. The model output can be very complex; in this part of NW Scotland it reflects the huge variation in geology and site conditions, even within very small areas.
2. Different landscapes tend to give very different patterns of woodland and scrub potential; for example, contrast the NWM output for the steeply dissected mountains of the mainland with the more open, rolling moorland areas of central Skye.
3. The Ardnamurchan area contains good examples of 'soil landscapes' where mosaics of different woodland types have been predicted, for example the W4 + peatland with scattered trees + W17/W18 mosaic on rock-dominated landscapes where peat is a common component. The rock outcrops have potential to support W17 or W18, whereas the flat channels and basins between the rock outcrops would support only scattered trees, with potential for W4 on intermediate ground.
4. The landscapes in NW Scotland are predominantly rocky and heterogeneous and thus woodland mosaics predominate (Figure 5.1). As we move east across Scotland, the landscapes tend to become more homogeneous and this is reflected in the NWM output by increasing proportions of single and interchangeable woodland and scrub categories (e.g. Cairngorms Partnership, 1999).
5. Finally, the altitudinal zonation in different areas is notable, for example from W9/W11 on steep coastal valley sides on mineral soils, through a range of open woodland mosaics on less steeply sloping ground and organic soils, up to montane scrub and open, exposed mountain tops.

The model output thus provides strategic overviews of woodland and scrub potential both within and between different areas of Scotland.

Uncertainties in woodland modelling

It is important to consider that there are a number of potential sources of error and uncertainty involved in all modelling work. In the NWM, these include:

- **Errors in source data.** By their very nature, soils and land cover maps are simplifications of reality; classes can merge into each other, there are uncertainties of definition and of boundary location. These difficulties are present even when describing such features on site. Some features are more accurately identified and located than others, for example, alluvial soils or arable land, as compared to soils with subtle drainage distinctions or bracken.

- **Uncertainties within the literature regarding optimum site conditions for different woodlands.** It is not yet possible to define with precision what is required for establishment of all woodland/scrub types, because the precise limits for growth and survival of most native tree/scrub species and woodland types are not fully understood. This is partly due to a lack of relevant research and partly because we have so little native woodland remaining in Scotland that it is hard to define and test such limits.

- **Limitations of woodland classifications.** The NVC is, of course, desirable to use as the standard UK classification system, but it is only based on relatively small samples and therefore has inadequacies in its application to some Scottish woodlands. Other Scottish woodland classifications, most notably that of McVean and Ratcliffe (1962), give valuable additional information to alleviate some, but not all, of the problems associated with the NVC.

- **Impacts of 'external' pressures.** Grazing, for example, can complicate woodland predictions which are based only on the biophysical attributes of a site.

Most of these potential sources of error are very difficult to quantify, and within our reports describing model outputs we make explicit recognition of them in order that the reader/user is aware of their implications for the model output. It is also important to note that, in view of the extreme scarcity and highly modified state of most of our native woodland remnants, it is currently not possible to fully validate any model predicting woodland potential in the UK. There are two main ways to partially validate the approach and the output:

1. To examine how well the datasets used predict site conditions and how the inclusion of other data might improve those predictions.
2. To test the predictions against the remaining native woodland areas.

It is *fundamental* to note that the latter is only valuable where the management history is well understood, in order to examine the *reason* for a match or mismatch of woodland type. Most of our remaining native woodlands are highly modified, thus if oak, for example, has consistently been felled from a woodland then its absence at a site does not indicate site unsuitability, simply management history. Equally, if many years of grazing have resulted in loss of a 'key species', such as ash (Gray and Stone, Chapter 7), then the current woodland type remaining is also not a true representation of what that site can support if not limited by heavy grazing. Thus, validation of these models can only be partial and, as new woodlands develop, new information should be used to regularly improve our limited understanding and update the models as appropriate. Where published information does not exist, this type of feedback loop has already led to some important revisions of detail within the NWM to date.

Validation of the NWM

Our ideal aim for the NWM is to maximise model simplicity without unduly compromising quality. The model is currently relatively simple, using only two main datasets (soil and land cover). Thus in the process of development we wished to test whether our approach is sufficiently robust at the scales for which the model is designed. We outline below the process of examination of other (climatic) datasets which could be included in the model, and discuss our conclusions from this exercise in relation to the current model structure.

Little or no research has been done on the definition of climatic limits for any but a few of our native tree species. The only available estimates of climatic thresholds for different NVC woodland types have been published by the Forestry Commission as part of the Ecological Site Classification (Pyatt,1995; Pyatt and Suárez, 1997; Hale *et al.*, 1998). Pioneering work on climate classification in the early 1970s (Birse and Dry, 1970) considered that annual accumulated temperature (above 5.6 °C) and potential moisture deficit were the two most fundamental parameters required to classify regional climate. Thus the accumulated temperature limits (and moisture – see next section) for different tree species as defined by Pyatt and Suárez (1997) were compared with the NWM predicted woodland categories in SNH's Natural Heritage Zones (NHZs) (SNH, 1999) 10–15 (at the 10 km National Grid Intersects) and on the Island of Rum (at 1 km intersects) (Hester *et al.*, 1999) as shown in Table 5.1.

Table 5.1 *Comparison of NWM predicitions (excluding unsuitable land) with accumulated temperature thresholds for tree growth. NHZ numbers refer to SNH Natural Heritage Zones (SNH, 1999). ESC = Ecological Site Classification. An indication of the location is also given in parentheses.*

Study area	ESC accumulated temperature classification			
	Optimal	Suitable	Unsuitable	Total
NHZ 14 (Argyll)	23	21	-	**44**
NHZ 13/15 (Lochaber, Trossachs and Breadalbane)	26	18	-	**44**
NHZ 10/11/12 (Cairngorms area)	47	38	2	**87**
Island of Rum	55	28	-	**83**
Total	**151**	**105**	**2**	**258**

From the sample results, the NWM predictions compare extremely well with the limits of accumulated temperature for tree growth, so we concluded that inclusion of accumulated temperature data would give no significant improvement to the NWM output. This is not unduly surprising, as different soils require specific temperature and moisture regimes to develop, thus the soils present in an area directly reflect the biophysical environment in which they are found. Therefore, when considering the requirements of more demanding species such as oak, they are predicted by the NWM on soils which themselves are more demanding in terms of the temperature regime required for them to develop.

We conducted a similar comparison between the NWM and the estimated soil moisture deficit limits (Pyatt and Suárez, 1997), and found that moisture deficit is not actually a limiting factor for woodland and scrub growth within the Scottish uplands, and is very unlikely to be limiting even in the Scottish lowlands. Very little of Scotland has a moisture deficit greater than 100 mm and with 140 mm being the upper threshold for most native woodland types (Pyatt and Suárez, 1997), inclusion of soil moisture deficit data would give little improvement to the model within Scotland.

The third component of climate which strongly influences woodland growth is exposure. Here we compared the NWM predictions with the Detailed Aspect Method of Scoring (DAMS), a scoring system which considers the wind zone, elevation and topographic shelter (Topex) of a site, as well as the effect of aspect and funnelling of the wind in valleys (Quine and White, 1993). DAMS exposure ratings were calculated using the Forestry Commission Forest*GALES* program (Dunham *et al.*, 2000), for a total of 561 grid intersects (both 10 km and 1 km), including those previously assessed for temperature. Table 5.2 illustrates well the gradual increase in woodland potential as the DAMS score reduces.

Table 5.2 *Comparison of NWM predictions with DAMS exposure scores. A score of 22–24 (and above) is considered to represent the limit of tree growth (Pyatt and Suárez., 1997; Hale* et al.*, 1998).*

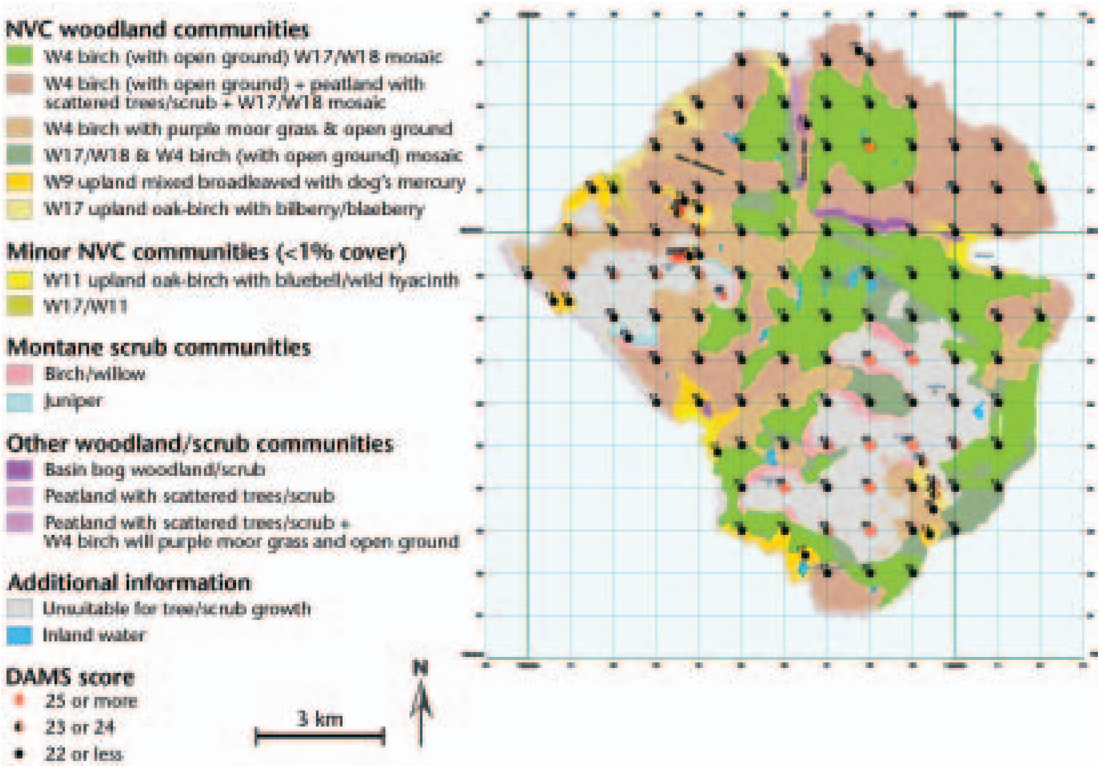
Increasing potential for woodland cover →

Increasing exposure →

DAMS score	Unsuitable for trees/ scrub	Montane scrub	Peatland with scattered trees and scrub	Birch with open ground	NVC closed woodland types	Total (number of grid intersects)
30	1					1
29						
28	1	1				2
27	2					2
26	2	1		1		4
25	4	3	3	2		12
24	2	2	4	3	1	12
23	5	2		5	1	13
22	3	3	2	9	2	19
21	8	5	6	8	2	29
20	3	4	4	18	1	30
19		6	9	37	7	59
18			4	41	17	62
17	2	2	7	24	17	52
16	1	2	2	34	22	61
15			2	25	23	49
14		1	1	16	36	54
13			1	11	26	38
12	1		3	4	26	34
11				1	9	10
10				1	9	10
9					5	5
8					3	3
Total	**35**	**32**	**48**	**240**	**206**	**561**

Importantly, the NWM did not predict closed woodland suitability in any areas where the DAMS score was greater than 24, which is considered by Hale *et al.* (1998) to be the maximum limit of tree growth. We originally considered that exposure would be the most likely measure to highlight any deficiencies in the NWM, as it can have strong impacts on the limits to tree growth. Therefore on the island of Rum we also tested clusters of points up and down exposed, steep mountains where altitude and exposure would change rapidly, to see how the NWM performed under these extreme situations. Figure 5.2 shows DAMS scores and NWM predictions for grid square intersections and the extra selected points. In this further test we found remarkably good agreement between model predictions and DAMS scores.

Figure 5.2 *Native Woodland Model output and DAMS scores for the island of Rum, Inner Hebrides. The map shows the NWM-predicted site suitability for a range of woodland and scrub types, as listed in the legend, together with the DAMS scores for all 1 km grid intersections and selected exposed localities where closer comparison with the model was considered to be particularly important. Red circles denote where DAMS (>24) indicates the site is too exposed for tree growth; red/black circles indicate DAMS score (22–24) close to limits of tree growth (Pyatt and Suárez, 1997).*



From the above tests we concluded that the combined soils and land cover datasets and associated rule base provide a robust surrogate for the main climatic variables at the resolution of the model output. However we must stress again that the NWM does not replace the need for detailed site surveys, which are essential to fine-tune the model predictions on the ground.

Validation of the model output against existing native woodland types has been carried out by Macmillan *et al.* (1997) for part of the Cairngorms, by external contractors to SNH in parts of western Scotland (see unpublished reports, D. Stone) and is now being more widely examined across Scotland by SNH (see Gray and Stone, Chapter 7) with very good results to date.

Model uses

The NWM has been employed in a range of uses to date, from the individual estate level to the whole of upland Scotland. At the estate level, we have run the model as part of whole estate audits, to give information and guidance on woodland and scrub potential (at the 1:50 000 scale) which has then been considered along with all other land-use options. This has proved particularly useful and has resulted in applications for a range of woodland establishment/regeneration grants from different estates. We must stress here that at the stage of site-specific woodland planning, the strategic use of the NWM is then supplemented by the usual processes of FC liaison and on-site survey. In other specific areas, such as Sunart, we have run the model using 1:50 000 soils data, which is available for a few parts of the country (Towers *et al.*, 1999). This has provided more detailed information on woodland and scrub potential than that produced using the 1:250 000 soil data. At smaller (less detailed) scales, one major use of the model was as part of the Cairngorms Forest and Woodland Framework project, where we extended its application to directly address key management options in the following ways, as illustrated in the Cairngorms Forest Framework publication (Cairngorms Partnership, 1999):

1. **Potential for natural regeneration** By combining the NWM site suitability predictions with information on current woodland extent and main tree species, we created 500 m wide 'regeneration potential' zones around potential tree seed sources (only where site conditions within that 500 m zone were considered suitable for the seed-source tree species). This allowed areas with greatest potential for natural regeneration to be specifically targeted.

2. **Potential through new planting** We then filled in the remaining areas (>500 m from current seed sources) with the normal NWM output, i.e. giving information on the potential for different woodland and scrub types, but this time any new woodland would be established by planting rather than achieving expansion through natural regeneration.

3. **The composite map of (1) and (2)** This gives detailed strategic level guidance on the potential for natural regeneration and planting options in different parts of the Cairngorm Partnership area. The text which then accompanies the maps in the Framework publication considers the woodland potential in relation to other land-use priorities, such as grouse shooting, commercial forestry, agriculture, and thus builds up suggestions for priority areas for the active encouragement or discouragement of new woodland. Priority locations for different woodland types have then been identified and used along with guidance within the Framework on how best to integrate with other landscape and land-use interests. Thus, the enhanced model output provides a strategic frame of reference for woodland grant applications within the Cairngorms Partnership Area.

Finally we have recently completed a contract for SNH where we have used the NWM to map native woodland and scrub potential within all their upland Scotland Natural Heritage Zones. This work and some of the uses to which it is being put are described in more detail in Chapter 7.

Conclusions

To conclude, the Native Woodland Model has proved itself to be an important strategic planning tool for a range of end-users (e.g. SNH, Tayside Native Woods, Highland Birchwoods, individual estates, Cairngorms Partnership, Forestry Commission). From tests against climatic variables described above, we conclude that there is currently no significant advantage in adding them to the NWM. Moreover, we advance the suggestion that interpretation of the soil and land cover datasets currently used in the NWM actually act as robust surrogates for those climatic variables of greatest relevance to the prediction of woodland site suitability. Climate is indeed one of the principal soil forming factors and one of the major factors driving land-use and land cover distribution in the uplands (with the additional influences of factors such as grazing, as discussed). Moreover, the currently available climate datasets are extrapolated over large areas from relatively few samples, and there is still uncertainty in setting climatic limits for all the different woodland and scrub types. From all tests to date, we conclude that the NWM appears to give robust predictions with at least 70% accuracy of main woodland types at the target scales of resolution (Chapter 7), bearing in mind the data/knowledge currently available. Thus we believe that we are currently achieving our aim of maximising simplicity without unduly compromising quality of output.

It is important also to reiterate here that we view the NWM as dynamic, i.e. the model is specifically designed to be easily updated whenever better data become available and/or new research improves our knowledge of woodland and scrub requirements. This has already been the pattern in the model development so far, as can be seen, for example, by comparison of the first Cairngorm map (MacMillan *et al.*, 1997) and one made in 1999 for the Cairngorms Partnership (Cairngorms Partnership, 1999).

The dataset testing exercise highlights the important issue about areas of uncertainty in woodland predictions where there is still insufficient knowledge about the exact conditions limiting growth of different woodland/scrub types and their associated flora, as well as the limitations of the NVC classification itself. We consider it very important when producing a predictive model such as this to state all uncertainties up-front in all outputs, making clear which predictions are more or less robust. But we must also stress that from the model ground-truthing tests so far, the need and the advantages of producing a model like this still far outweigh the uncertainties involved.

Acknowledgements

The NWM has been developed by ourselves at the Macanlay Institute with contributory support and funding largely from SNH and the Macaulay Development Trust plus, to a lesser extent, many other organisations through the range of contracts which we have carried out.

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CHAPTER 6

Applying an Ecological Site Classification to woodland design at varying spatial scales

Duncan Ray, Jonathan Clare and Karen Purdy

Summary

Ecological Site Classification (ESC) has been developed as a forest planning tool at the stand scale and the forest landscape scale. The ecological variables linking the National Vegetation Classification (NVC) and ESC have been coded to indicate the degree of suitability of alternative native woodland communities on any site type. The ESC methodology and relevant decision support systems do not solely rely on precursor vegetation, and so will extend the application of NVC woodland to restoration and expansion on plantation and non woodland sites. In this chapter we discuss the benefits and limitations of ESC predictions for suitable NVC woodlands types at three different scales: landscape scale regional planning, forest design planning and operational level site planning.

Introduction

British forest policy (Anon., 1998b) aims to increase the area of woodland as a sustainable resource for timber production, wildlife conservation, recreation and public amenity. Since the mid 1990s, grant targeting initiatives have attempted to provide more balanced wooded landscapes in Britain by emphasising broadleaved and new native woodlands. The area of broadleaved planting has increased by approximately 10 000 ha per year since 1995, with many new native woodland schemes (Rollinson, Chapter 1). In addition the expansion of new native pinewoods in Scotland has increased to 5 000 ha per year since the introduction of the scheme 10 years ago (Rollinson, Chapter 1). A large proportion (approximately 66%) of Britain's woodland is privately owned (Anon., 1998a) including many thousands of small woodlands (Anon., 1998b). Farmers in particular are planting more woodland with a large percentage on land previously used as rough grazing. The Scottish Executive June 2000 Agricultural Census (www.scotland.gov.uk) showed a 300 000 ha reduction in the area of rough grazing between 1991 and 2000. In the same period farm woodlands increased by 100 000 ha, improved pasture increased by about 40 000 ha, while the area of land in agriculture decreased by about 129 000 ha; some of this will have changed to forest land.

The National Vegetation Classification (NVC) (Rodwell, 1991) recognises 25 woodland and scrub communities, classified as native woodland types. The development, introduction and acceptance of the NVC has probably contributed to an awareness of the potential for expansion of native tree species on agricultural land. At the same time, the conservation of native woodland remnants and expansion of the planting of native tree species is encouraged by the Forestry Commission as a component of commercial forest planning (Anon., 1998b).

With sustainable forestry practice in mind, it is clear that species choice (native and non-native, conifers and broadleaved trees) is more important now than previously, when it was common to modify site conditions to suit any particular tree species. Coupled with the increase in farm forestry and particularly the interest in broadleaved grants for planting new native woodlands, there is a need for a methodology to help select species and woodland communities best suited ecologically to site conditions.

The introduction of the UK Woodland Assurance Standard (Anon., 2000) has highlighted the need for the auditing of species and woodland choice, and recommended Ecological Site Classification (ESC) as a tool suitable for this purpose. ESC can be used to assess the suitability of existing woodland sites and open managed vegetation communities (including pasture, rough grazing and moorland) for conversion to a native woodland community.

This chapter describes ESC suitability modelling for native woodland communities at different scales within an area of north-eastern Scotland. ESC is a decision support tool, designed to help managers make decisions objectively, in situations where large amounts of biophysical data are combined to assess species or woodland suitability. Professional judgement is an important factor in ESC as in all decision support tools (Twery *et al.*, 2000), and users are encouraged to analyse results and check the validity of spatial data, assessing the impact of variation in input variables on suitability. The use of ESC at various scales eases the ground-truthing procedure because the ESC factors and model terminology are consistent throughout the range of analysis scales.

ESC methodology

The ESC system was developed (Pyatt and Suárez, 1997; Pyatt *et al.*, 2001) using a similar methodology to the Biogeoclimatic Ecosystem Classification (BEC) of British Columbia, Canada (Pojar *et al.*, 1987). BEC was developed as a classification system for natural forests of British Columbia (Krajina, 1969), whereas in Britain a major objective has been to develop the methodology to include plantations as well as native woodland.

ESC addresses the problem of matching the ecological requirements of tree species and native woodland communities to site conditions (Pyatt *et al.*, 2001). This was first demonstrated in Grampian Region, Scotland (Pyatt and Suárez, 1997) as a methodology that brings together climatic data with soil quality data. There are six ESC factors which describe site quality, namely: four climate factors – Accumulated Temperature above 5°C (AT) (1961–1990 average), maximum summer Moisture Deficit (MD) (1961–1990 average), Windiness (DAMS), Continentality; and two soil factors – Soil Moisture Regime (SMR) and Soil Nutrient Regime (SNR).

Climate factors

The AT and MD factors have been calculated and interpolated from 10 km square digital data of monthly mean temperature, monthly mean rainfall and monthly mean evaporation throughout Britain (Barrow *et al.*, 1993), at a resolution of 1 ha. The ESC digital climatic analysis of Britain is similar in concept to the bio-climatic maps based on older climate data published in Scotland by the Macaulay Land Use Research Institute (Birse and Dry, 1970; Birse and Robertson, 1970; Birse, 1971) and in England and Wales by Rothamsted Experimental Station (Bendelow and Hartnup, 1980). Windiness scores using the detailed aspect method of scoring (DAMS) wind have been calculated for the whole of Britain at a resolution of 1 ha using the method described by Quine and White (1993). Continentality, describing the intensity of warmth and coldness, and the length of the growing season has been calculated at 1 ha resolution across Britain using a method based on the work of Conrad (1946).

Soil quality factors

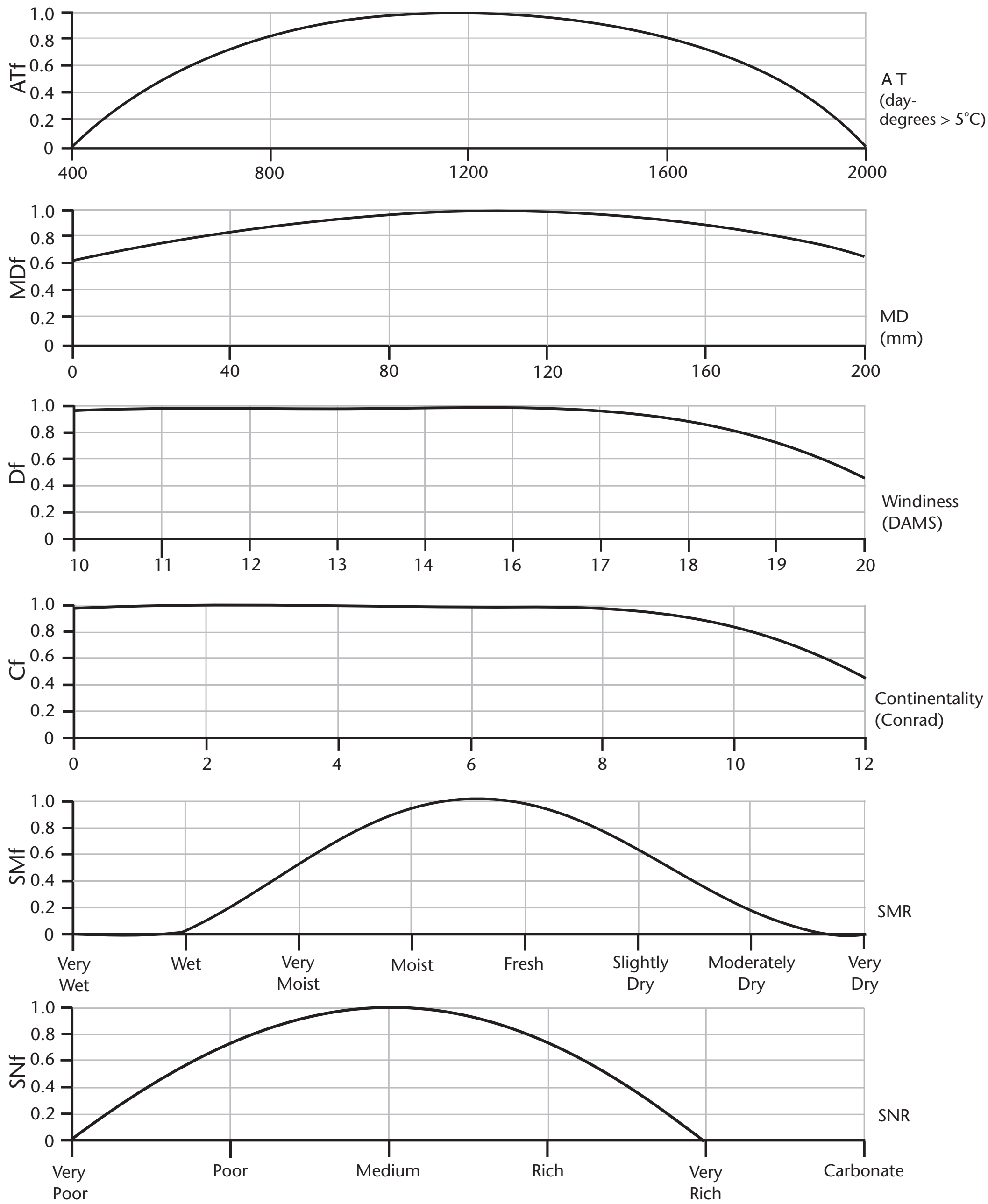
The factors SMR and SNR describe soil quality. SMR gives the relative wetness or dryness of the soil, and is calculated using one of two methods depending on whether the soil is a ‘wet’ type or ‘dry’ type. Wet soils are defined as having a winter mean water table depth of less than 80 cm, and the SMR is linked to the wetness classes described by Robson and Thomasson (1977). Dry soils have a mean winter water table deeper than 80 cm. Dry soil SMRs are calculated using a combination of available water capacity (AWC) (Hall *et al.*, 1977) and MD, described by Pyatt *et al.* (2001).

SNR describes the fertility of a site in terms of the available nitrogen in relation to pH (Pyatt *et al.*, 2001) and can be assessed by taking a default value from the soil type and phase or, more accurately, from a survey of the humus form or plant indicator species occurring on a site (Wilson *et al.*, 2001). This data provides the most accurate assessment of the site quality, and is used for stand scale ESC analyses using the ESC decision support system (ESC-DSS) (Ray, 2001). For forest landscape analyses using the spatial ESC system on a GIS (Clare and Ray, 2001), the Forestry Commission 1:10 000 scale soil maps can be used. About 50% of the FC estate managed by Forest Enterprise, as well as some privately managed forests, use the FC soil mapping techniques. The maps record soil type, phase and lithology using the Forestry Commission Soil Classification (Pyatt, 1982; Pyatt *et al.*, 2001) and have been assigned default SMR and SNR classes to allow the assembly of soil quality themes on a Geographical Information System (GIS) and subsequent input layers for SMR and SNR in the spatial ESC.

ESC native woodland models

ESC suitability models for 20 of the 25 NVC woodland communities (W1–W20) have been developed (Pyatt *et al.*, 2001). For each NVC woodland community, a fuzzy membership function approach (Zadeh, 1992) describing the suitability of the community to each ESC site factor has been constructed (Ray *et al.*, 1998). This knowledge-based representation stems from an assessment of the climatic variation, using each ESC climate factor, within the distribution of the community. A knowledge-based representation of soil quality suitability for each NVC community has been made from an ordination analysis of frequency-weighted 'Hill-Ellenberg' F against R+N values (Ellenberg, 1988; Hill *et al.*, 1999), based on Rodwell's list of vascular plants for NVC sub-communities (Rodwell, 1991). The set of six continuous suitability functions for *Quercus petraea – Betula pubescens – Oxalis acetosella* woodland (W11) are shown in Figure 6.1.

Figure 6.1 *Continuous suitability functions (0–1.0) for Quercus petraea – Betula pubescens – Oxalis acetosella woodland (W11 oak, birch and wood sorrel). ATf = Accummulated Temperature function; MDf = Moisture Deficit function; Df = DAMS function; Of = Continentality function; SMf = Soil Moisture function; SNf = Soil Nutrient function.*



ESC spatial modelling: results from analyses at different scales

An ESC analysis can be made at a range of scales depending upon the intended requirements of the user. At the stand scale, a soil description with rooting depth, stoniness and soil texture gives the most accurate estimate of SMR; a list of plant species and percentage cover, and/or humus form gives the most accurate estimate of SNR. A major consideration for users in applying ESC is the reduction in the resolution of data as the area under consideration increases, and a corresponding reduction in the accuracy of NVC woodland suitability estimates. In using digital soil maps to estimate soil quality (e.g. the 1:250 000 scale national soil map of Scotland), this usually means that soil types have been combined as soil complexes and, as a consequence, site variation tends to be masked.

The following three analyses demonstrate how ESC can be used at different spatial scales. We start at the regional scale, covering an area of about 250 km^2, using the Land Cover Scotland 1988 (LCS88) digital dataset of vegetation classes (Anon., 1993) to derive the soil quality data required by ESC. This is followed by an analysis at the forest landscape scale using 1:10 000 FC soil maps. The section concludes with a stand scale analysis that can be used to ground-truth the spatial analyses.

A regional scale analysis

LCS88 was used to estimate SMR and SNR over the region of Strathdon in Grampian, Scotland, an area in excess of 28 000 ha (Table 6.1). SMR and SNR were extrapolated from vegetation and soil information collected in a similar study in the Ochils (Ray *et al.,* 1999). Nine out of the twelve modified LCS88 classes occurring in Strathdon also occurred in the Ochils. The SNR and SMR of the remaining three classes: montane vegetation, peatland vegetation and low scrub were estimated from the vegetation communities of known sites and a ground-truthing field visit. The soil quality for the areas of woodland mapped within LCS88 were unknown and the conifer plantations were assigned a general soil quality of SMR Fresh and SNR Poor. The distribution of SNR classes is given in Figure 6.2, showing the range from Very Poor to Rich.

The climate data were calculated on the ArcView Spatial Analyst Map Calculator (Environmental Systems Research Institute – ESRI) using a 250 m x 250 m resolution grid of elevation, latitude and longitude as input variables for the multiple linear regression equations developed in ESC to interpolate climate data (Pyatt *et al.*, 2001).

Map Calculator was used to assess the suitability of each native woodland community for each site from rules developed from the function graphs (Figure 6.1). A woodland is classed as suitable if the smallest suitability score derived from each of the six continuous functions is greater than 0.7. If more than one NVC woodland community was found to be suited in a grid cell, then the community with the greatest suitability score was chosen.

The suitable NVC woodland communities at the regional scale are shown in Figure 6.3. Three 'altitudinal' zones featuring different forest types are prominent:

1. High elevation sub-alpine zone of krummholz Scots pine, juniper and peatland vegetation (W18 Krumholz). The high elevation ridges classified as montane vegetation show a borderline suitability to W18, designated as krummholz Scots pine woodland in the Figure 6.3 legend. This vegetation class is normally associated with skeletal soils, both peaty and podzolised mineral rankers of Moist SMR and Very Poor SNR. A patchy tree-line of scrubby Scots pine and juniper (*Juniperus communis*) would be found in these areas. Some peatland vegetation also falls within the high elevation krummholz zone and, with an SMR of Very Wet or Wet and SNR Very Poor, is unsuited to any native woodland community.

2. Mid-elevation zone Scots pine woodland (W18) with some birch and oak (W17) on the undifferentiated and dry heather moor associated with podzols, ironpan soils and podzolised brown earths of Moist or Fresh SMR and Very Poor SNR.

3. Low elevation oak wood (W11) on the lower slopes and valley bottoms with occasional mixed broadleaved woodland with ash (W9) and wet woodlands of willow (W3), downy birch (W4)

Table 6.1 *Summary of site data derived from surveys of LCS88 classes in the Ochils.*

Location Number	Grid reference	Elevation (m)	LCS88 vegetation class	Mean site nutrient indicator value	Std. error	SNR	Mean Ellenberg F value and SMR
1	NO 004 028	210	Smooth grass	3.00	0.05	Poor	5.10 Fresh
2	NS 955 993	320	Smooth grass	2.86	0.02	Poor	6.53 Moist
3	NS 956 992	300	Smooth grass	2.66	0.11	V. Poor	5.16 Fresh
4	NO 007 027	220	Smooth grass	2.95	0.02	Poor	5.00 Fresh
5	NN 964 044	260	Smooth grass	3.23	0.10	Poor	5.26 Fresh
6	NN 821 001	225	Smooth grass	3.19	0.17	Poor	5.42 Fresh
7	NS 821 999	230	Smooth grass	3.73	0.07	Poor	5.00 Fresh
8	NS 878 992	280	Heather moor	2.10	0.14	V. Poor	5.58 Fresh
9	NO 018 111	280	Heather moor	2.02	0.04	V. Poor	5.11 Fresh
10	NN 941 045	300	Heather moor	2.28	0.10	V. Poor	5.00 Fresh
11	NO 007 027	210	Heather moor	2.41	0.08	V. Poor	5.00 Fresh
12	NN 941 061	300	Heather moor	1.94	0.04	V. Poor	5.00 Fresh
13	NO 018 063	350	Heather moor	2.07	0.08	V. Poor	5.00 Fresh
14	NS 818 990	225	Heather moor	2.17	0.18	V. Poor	7.81 V. Moist
15	NN 853 051	270	Heather moor	1.81	0.05	V. Poor	5.00 Fresh
16	NN 941 047	350	*Nardus molinia*	2.91	0.10	Poor	5.11 Fresh
17	NS 953 994	320	*Nardus molinia*	2.71	0.02	V. Poor	7.00 V. Moist
18	NN 945 065	380	*Nardus molinia*	3.14	0.11	Poor	5.86 Fresh
19	NO 018 061	340	*Nardus molinia*	3.00	0.12	Poor	6.23 Moist
20	NS 883 984	270	Smooth grass and low scrub	3.71	0.04	Poor	5.38 Fresh
21	NO 017 111	280	Smooth grass and low scrub	3.43	0.13	Poor	5.11 Fresh
22	NN 944 065	300	Smooth grass and low scrub	3.54	0.07	Poor	5.00 Fresh
23	NN 963 044	320	Smooth grass and low scrub	3.47	0.04	Poor	5.29 Fresh
24	NS 823 998	250	Smooth grass and rushes	4.70	0.18	Medium	6.86 Moist
25	NO 015 061	260	Bracken	3.49	0.16	Poor	5.10 Fresh

and alder (W7). On slopes the presence of smooth grass of *Agrostis* and *Festuca* species indicates an SMR of Fresh and an SNR of Poor to Medium. Improved pasture within this zone, often on surface and ground water gleys of the valley bottom, would have an increased nitrogen availability (SNR of Rich) and is suitable for the W9 upland or northern mixed broadleaved woodland and W7 alder woodland, where the SMR is Moist or Very Moist. The wetter peaty soils of the valley bottom and higher elevation plateaux (Very Moist and Wet) provide suitable sites for W3 willow woodland where eutrophic flushing occurs and W4 downy birch woodland on poorer sites.

Because LCS88 identifies coniferous woodland within a single class, the large blocks of existing woodland have been assigned an SMR of Fresh and SNR Poor resulting in an ESC analysis showing mainly W11 upland oak wood (with small areas of W17) best suited over much of the forest area. However, if these blocks are examined in more detail, additional variability appears (see below).

A forest landscape scale analysis

Within the south-eastern area of Strathdon lies Tornashean Forest managed by Forest Enterprise (lower right of Figure 6.2), an area of 2 000 ha. This is an area of rather complex lithology within the Eastern Dalradian basic igneous site region. Climatic data layers were calculated as previously, using a 100 m x 100 m grid, and the higher resolution of the data defines the minimum area (1.0 ha) of a site at the forest landscape scale.

Figure 6.2 *Strathdon: Soil Nutrient Regime draped over terrain, from the reclassified LCS88 map.*

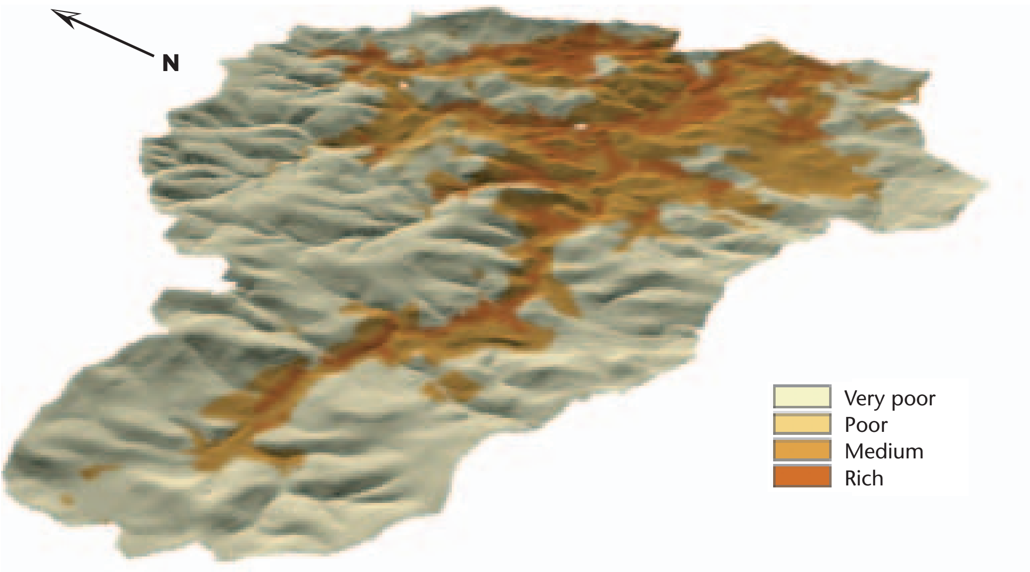
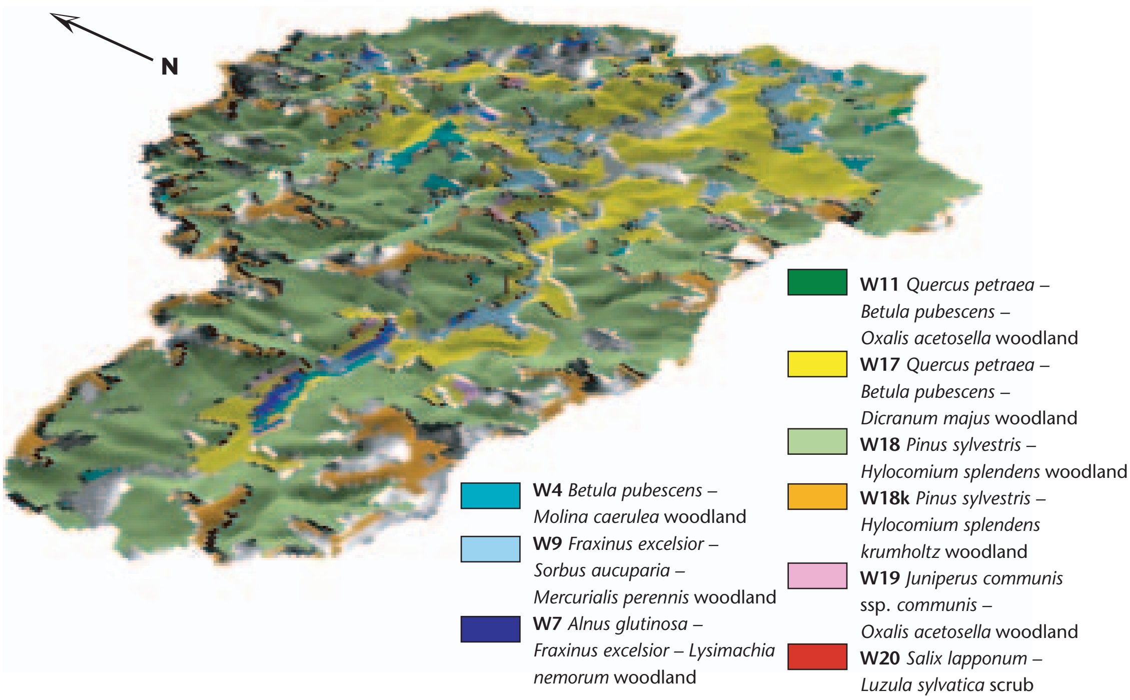


Figure 6.3 *Strathdon: suitable native woodland communities derived from the ESC analysis draped over terrain. Grey areas are unsuitable for woodland.*

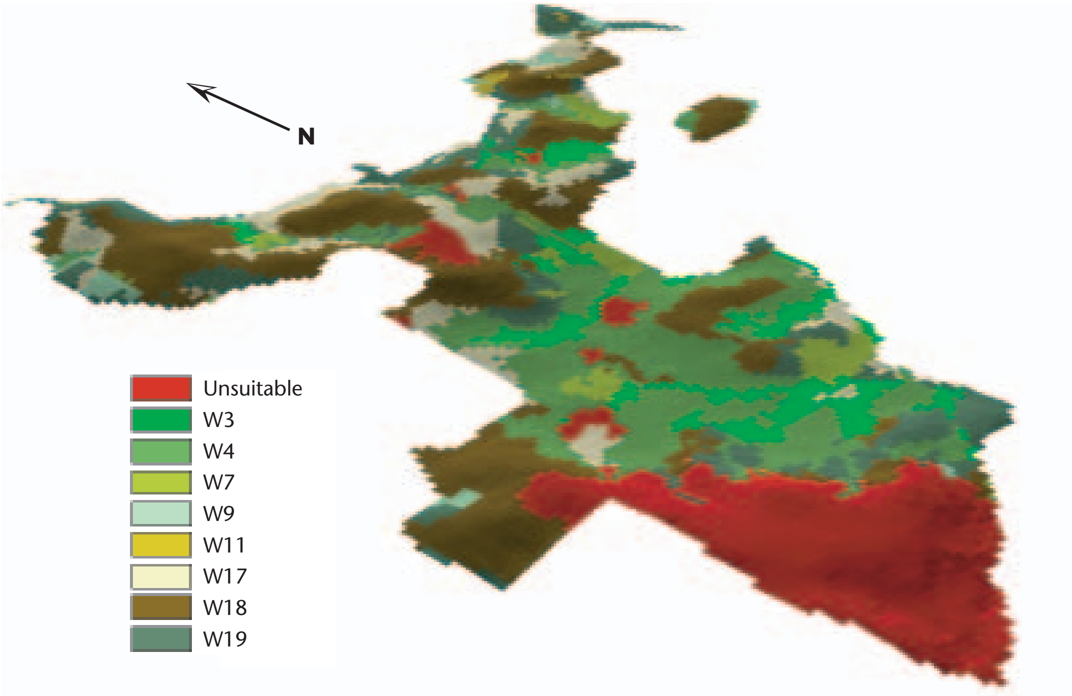


Maps of soil types and phases based on field survey were used to set defaults of SMR and SNR. The six ESC factors were calculated as GIS layers using the same rule-based models. Soil quality was much more variable than predicted by the LCS88 classification resulting in a more complex NVC woodland mosaic than predicted by the regional analysis (Figure 6.4). The upper slopes and ridges of the forest covered by hill peat and peaty rankers were classed as Very Moist and Wet and generally Very Poor, and consequently are unsuitable for native woodland communities. At lower elevations woodland communities ranged from W18 and W17 on iron pans and podzols to W9 and W11 on richer soils at lower elevations. Wet woodland (W3 and W4) was predicted to occur on the wetter soil types.

Stand scale analysis

Within Strathdon a 28-year-old stand of hybrid larch was selected (courtesy of Gordon Woodlands) for a stand scale analysis using ESC-DSS (ESC Decision Support System Version 1.7). The site is located at National Grid Reference NJ373137 – latitude 57° 12′N, longitude 3˚ 2′W, at an elevation of

Figure 6.4 *Distribution of NVC woodland communities in Tornashean Forest predicted by ESC at the forest landscape scale using 1:10 000 scale FC soil maps for soil quality. Key to woodland types as in Figure 6.3; W3 –* Salix pentandra – Carex rostrata *woodland.*



390 m above sea level, on an easterly and moderately steep slope of 20˚. From the site location, ESC estimates: AT=838 day-degrees above 5°C; MD=48 mm; Windiness=12 DAMS; Continentality=6.

The site factors describe a cool moist but relatively sheltered upland climate. The soil is a well-developed deep, freely draining upland brown earth, with a rooting depth of more than 100 cm, a moderate stone content (estimated at 15% v/v) and a sandy loam texture, giving an SMR of Fresh. The presence of a few slowly weathering fragments of gabbro throughout the profile give a dark brown colour to the soil matrix. The basic igneous parent material suggests a relatively richer soil. A moder humus form (Pyatt *et al.*, 2001) is indicative of the ESC SNR Poor and the mean site indicator value of 3.29 (Wilson *et al.*, 2001) shows that the plant indicator species on the site agree with an SNR of Poor (Table 6.2). Nutritionally, the soil is slightly more fertile than the podzol or podzolic upland brown earth that might usually occur on a granite lithology (SNR is Poor rather than Very Poor).

The soil quality of SMR Fresh and SNR Poor suggests the site is suitable for four native woodland communities. The woodland with the highest suitability score generated by ESC-DSS is W19 juniper scrub, followed by W17 upland birch oak woodland, W11 oak and birch woodland with wood sorrel and lastly Scots pine woodland (W18) is only marginally suitable. ESC-DSS flags all the suitable woodland types to allow the opportunity (whenever possible) to manage sites more flexibly. For example, here, juniper might be managed in glades and less shaded parts of a W17 woodland, if this is considered to be consistent with the overall management objectives.

Discussion

The relationship between scale and data resolution is one of the most important considerations when undertaking a spatial modelling exercise. Usually, the data quality increases as the area being considered decreases, e.g. towards the stand scale for an ESC analysis. A unique feature of ESC is that the scale issue does not complicate the analysis, because consistent methods and terminology are used in ESC at all scales. This attribute positively encourages ‘ground-truth’ surveying to check spatial datasets. In Strathdon, at the regional scale, the area of Tornashean Forest appears as a single polygon unit in the Land Cover Scotland 1988 digital dataset, in which soil quality was assessed as

Table 6.2 *Floristic table of plants under hybrid larch in the Gordon Woodlands, Strathdon.*

ESC indicator species		Wilson indicator value	Mean per cent cover
Deschampsia flexuosa	wavy hair-grass	2.86	44
Holcus mollis	creeping soft-glass	4.00	19
Galium saxatile	heath bedstraw	3.06	18
Oxalis acetosella	wood sorrel	3.74	9
Anthoxanthum oderatum	sweet vernal-grass	4.39	8
Agrostis capillaris	common bent	3.15	6
Anemone nemorosa	wood anemone	4.78	3
Viola riviniana	common violet	3.74	3
Festuca ovina	sheep's fescue		2
Holcus lanatus	yorkshire fog	3.94	2
Potentilla erecta	tormentil	2.58	1
Deschampsia cespitosa	tufted hair-grass	5.04	1
Agrostis canina montana	brown bent		<1
Campanula rotundifolia	harebell		<1
Calluna vulgaris	heather	1.70	<1
Cerastium fontanum	common mouse-ear		<1
Veronica chamaedrys	germander speedwell	5.25	<1
Rumex acetosa	common sorrel		<1
Site mean indicator value		**3.29**	

SMR Fresh and SNR Poor. This resulted in W11 upland oak wood (with small areas of W17) being identified as best suited over much of the forest area (Figure 6.4). This type and scale of analysis is suitable for regional planning and strategic consideration of suitable areas of main forest types.

Because ESC soil quality can be defined by the indicator plants within a vegetation community, it is possible to link LCS88 mapping units and ESC soil quality units by a simple ground survey. Surveys to sample the vegetation will give the mean SMR and SNR and the variance or standard error of the ESC soil quality within LCS88 vegetation classes. If sites show variation between soil quality classes, the soil quality scenarios can be assessed in the ESC analysis to investigate the effects on woodland suitability. This method is best suited to woodland expansion onto open ground.

The digital soil map of Tornashean Forest converted to soil quality showed a complicated pattern of SMR and SNR at the forest landscape scale. The effects of lithology and topography, adding to the effect imposed by the climate, gives a complex mosaic of suited woodland communities. The approach using LCS88 data was unable to distinguish this detail, and the national soil maps at scales of 1:50 000–1:250 000 are also unlikely to capture the fine mosaic. Both classifications group information into the dominant class within the scale of the mapping unit, thus masking the variation in soil quality found within the forest landscape.

An ESC analysis at the stand scale uses exactly the same methodology as in a spatial analysis. The difference is in the type and amount of information that can be considered, since at the stand scale a range of data can be used to estimate soil quality. At the hybrid larch site presented earlier, we were able to use the soil type, site lithology, humus form and the ESC plant indicator species present to assess SNR. The range of SNR was between Poor and Medium, with the best estimators of SNR (plants and humus) both suggesting a Poor SNR midway in the class. This gives confidence to the ESC analysis. The lithology alone had indicated that the site was slightly richer, and the implication of this can be explored by running a scenario for SNR Medium through the program. A medium SNR scenario increases the suitability of W11 upland oak with wood sorrel woodland above W17 oak birch woodland, but W19 juniper continues to have the highest suitability score. Thus the scenario analysis also suggests that the site is tending towards a W11 woodland.

Vegetation suitability models, including ESC, are decision support tools, and must be used with care and professional common sense. It should be obvious that the existence of a high suitability score for a particular community certainly does not guarantee the community will be found on the site. Neither does it guarantee that the community will regenerate on the site. There are many other factors that would need to be investigated to predict regeneration likelihood.

Conclusions

ESC links regional and stand scale forest planning by providing objective ecologically-based interpretations of climate and soil quality that are transferable between scales. In this chapter we have only presented and discussed NVC woodland community links with ESC. ESC evaluates the site suitability for all NVC woodland communities, and so can be used by managers to assess the potential for more than one community, depending on particular management objectives. However the methodology also allows an assessment of the site suitability for plantation-grown timber species. ESC suitability models and site-yield (index) models are available for 26 species of tree (native and exotic). It is at the stand scale that ESC-DSS is particularly useful for audit assessment for meeting the UK Forestry Standard (Anon., 1998b) and the UKWAS accreditation standards (Anon., 2000) for species or woodland site suitability. At present ESC-DSS is available for analysis at the stand scale, and with further development, ESC-GIS will extend the method to the forest scale, although this is dependent upon the availability of digital soil maps.

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CHAPTER 7

Applications of spatial data in strategic woodland decisions: an example from the Isle of Mull

Helen Gray and Duncan Stone

Summary

Driven by a mix of international conservation legislation, community interest and a developing hardwood timber trade, current woodland strategies require increasingly detailed information about existing woodland and the potential for woodland expansion. Historically, large-scale woodland information has been categorised in broad terms such as conifer/broadleaf or plantation/semi-natural, with little attention paid to species composition. Combinations of newly available tools and data (namely the Native Woodland Model, the Scottish Semi-Natural Woodland Inventory and the Ancient Woodland Inventory) permit an estimate of the woodland type of the current semi-natural woodland resource, along with the extent and patterns of potential expansion. A worked example (Upland Ashwood identification, restoration and expansion, in line with Habitat Action Plan targets) is presented to illustrate the range of support tools available and some of their potential combinations.

Introduction

At present there is an increased emphasis on strategic forestry issues, led by the new Scottish Forestry Strategy (Scottish Executive, 2000) and the revisions across Scotland of local authority Indicative Forestry Strategies. Scottish Natural Heritage (SNH) is a key consultee and source of information for such strategic plans, and is making extensive use of new data tools and concepts such as Forest Habitat Networks (Cairngorms Partnership, 1999; Ratcliffe *et al.*, 1998; Peterken *et al.*, 1995; Peterken, Chapter 9). Woodland Habitat Action Plans (HAPs) and their associated Local Biodiversity Action Plans (LBAPs) contain targets for the protection of the condition and current extent of different semi-natural woodland types alongside targets for restoration and expansion.

The power of geographic information systems (GIS) to combine and analyse datasets is becoming apparent in the strategic planning of woodland development. However, there are relatively few digital woodland datasets with national (Scotland) coverage, although many more local and regional sources of data exist. The key national datasets are the Scottish Semi-Natural Woodland Inventory (Caledonian Partnership, 2001), the predictions of the Native Woodland Model (Towers *et al.*, 2000), the Ancient Woodland Inventory (Kupiec, 1997) and the Land Cover for Scotland 1988 (Macaulay Land Use Research Institute, 1993). Contained within these datasets is key information on existing woodland cover and continuity, potential woodland type and pattern, and current land-use outside woodland. In this chapter we present an example of how the Native Woodland Model (NWM) can be applied to the setting of strategic planning priorities for woodland restoration at a regional scale. Developed by The Macaulay Land Use Research Institute (MLURI) with SNH, the NWM uses geological, edaphic and land cover data to model a woodland suitability map for most of Scotland (see Hester *et al.,* Chapter 5; Towers *et al.*, 2000). It is intended for use at broad forest level, at scales of 1:50 000 and above.

Applications of spatial data to the restoration of Upland Ashwoods in north Mull

Large ashwoods and their western hazel-dominated variants are now extremely rare and fragmented in Scotland, and the remaining examples are internationally important especially for lower plants. The characteristic biodiversity supported by larger extensive ash/hazel woods is likely to be richer than in small ravine ashwoods by virtue of the size, stability and interior conditions that larger woods generate. Thus the conservation and restoration of these larger woods carries a high priority. The following example explores the opportunities for Upland Ashwood HAP development in north Mull.

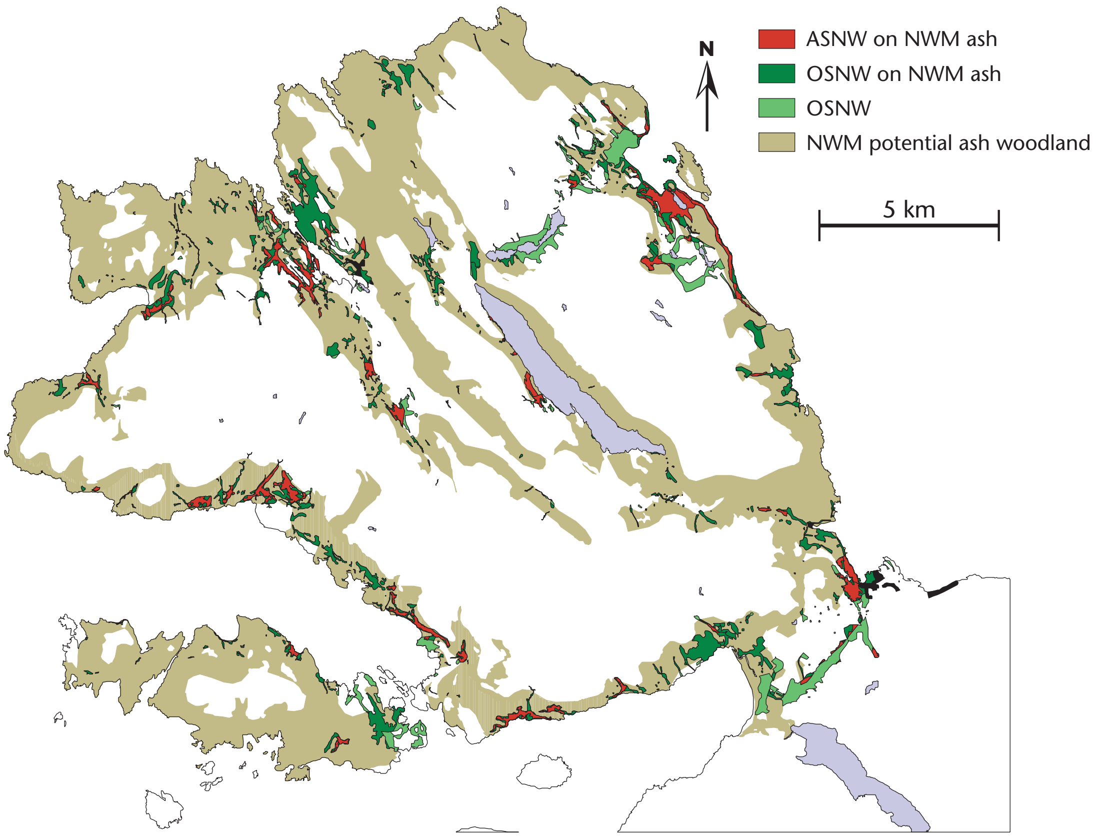
Current semi-natural resource

Figure 7.1 represents the existing semi-natural woodland in the area, and explores its likely woodland type. The map shows areas classed as suitable for the establishment and development of native ash and hazel woodland by the NWM. *Existing* semi-natural woods are mapped on top of this, identified using the Scottish Semi-Natural Woodland Inventory (SSNWI). Unlike most national surveys whose smallest unit size is 2 ha, SSNWI captures woodland down to 0.1 ha. This permits the identification of highly fragmented remnant semi-natural woodland often missed from other inventories. These existing semi-natural woods are shown divided into those occurring on NWM potential ashwood sites, and others. The information from the Ancient Woodland Inventory (AWI) is then incorporated to identify *ancient* semi-natural woods lying on NWM potential ashwood. The ancient and other semi-natural woods lying on NWM potential ash sites are those most likely to contribute to the current Upland Ashwood resource. It would therefore be logical to first consider these – what might be called 'high value' woods – when seeking to identify existing Upland Ashwoods in order to maintain and enhance them as part of HAP targets.

Existing semi-natural woodland	ha
Ancient semi-natural woodland (ASNW) on NWM ash	430
Other semi-natural woodland (OSNW) on NWM ash	940
OSNW	403
Total	**1 773**

Figure 7.1

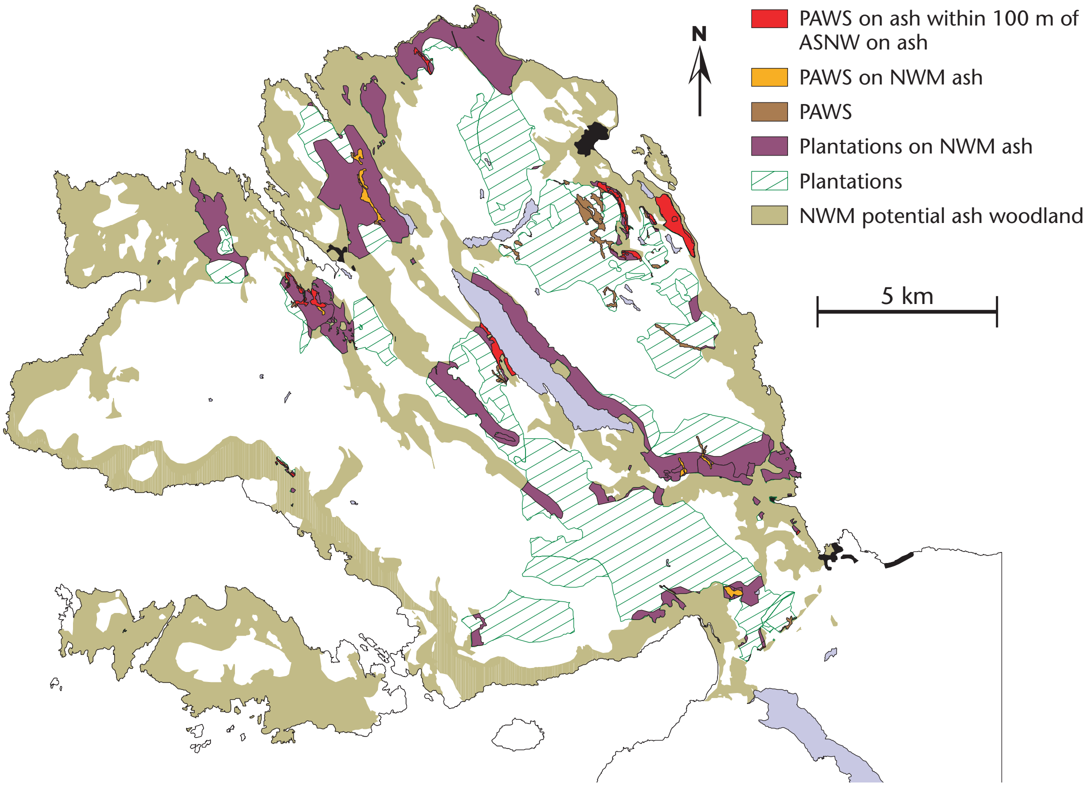
Distribution of current semi-natural woodland on North Mull in relation to existing and potential ashwood sites.



Existing semi-natural woodland	ha
PAWS on NWM ash within 100 m of ASNW on NWM ash	143
Plantations on Ancient Woodland Sites (PAWS) on NWM ash	45
PAWS (not on NWM ash)	110
Other plantations on NWM ash	2 195
Other plantations (not on NWM ash)	6 257
Total	**8 750**

Figure 7.2

Distribution of plantations on North Mull in relation to existing and potential native ashwood sites.



Restoration targets

The UK is committed to restoring a minimum of 2400 ha of plantations on ancient woodland sites (PAWS) back to Upland Ash woodland by 2015 (UK Biodiversity Group, 1998). The NWM provides a means of identifying the most suitable candidates for restoration. Figure 7.2 shows the distribution of all plantation woodlands on north Mull. Using the same methodology as for the mapping of existing semi-natural woodlands, SSNWI and AWI are interrogated to pinpoint ordinary and PAWS plantations which lie on NWM potential ashwood.

Successful restoration of a PAWS site back to native woodland will greatly depend on its proximity to an appropriate colonising seed source. Therefore, the selection of PAWS on NWM ashwood has been further reduced to select only those woods within 100 m of high value woods – ancient semi-natural woodland also lying on NWM ashwood. The resulting 143 ha of PAWS on NWM ashwood in close proximity to high value woods represent 'high opportunity' woods, and as such should be among the first to be considered for restoration.

Expansion targets

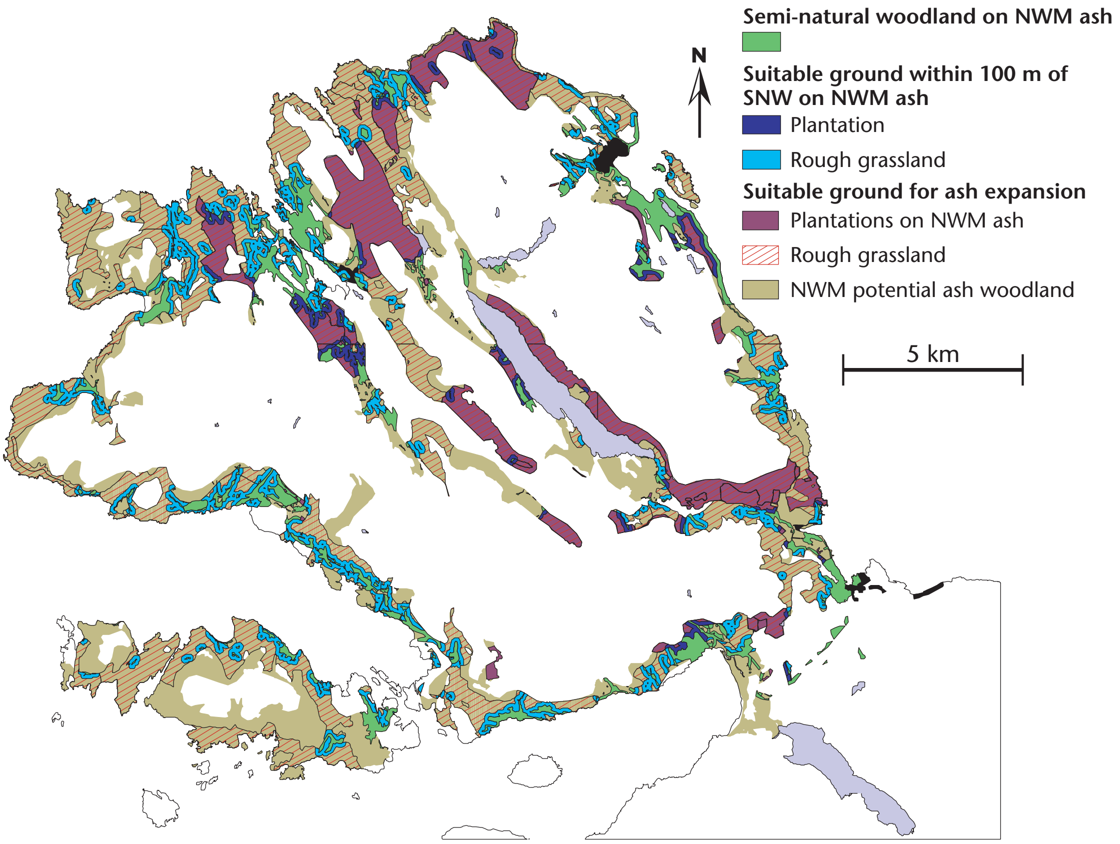
In addition to the restoration targets for PAWS, by 2015, a further 6 000 ha of Upland Ashwoods will be created either through establishment on currently open ground, or through conversion of existing plantations that are not on ancient woodland sites (UK Biodiversity Group, 1998). Figure 7.3 summarises the opportunities for targeted ashwood expansion. Land cover Scotland 1988 (LCS88) data was combined with NWM potential ashwoods to identify those LCS88 categories of suitable open ground (in this instance rough grassland and bracken categories were selected as ecologically appropriate, following Rodwell and Patterson, 1994) corresponding with NWM potential ashwood sites. Whether establishment is achieved through natural regeneration (the preferred option) or planting, successful establishment of the woodland community (as opposed to just the trees) will, as for restoration targets, also depend upon proximity to a suitable seed source.

Existing semi-natural woodland (both AWI and other) on NWM potential ashwood is displayed in Figure 7.3. A buffer of 100 m has been created around each wood, and only that part of the buffer that falls on grassland or plantation (which are both on NWM potential ashwood) has been selected. Ground suitable for expansion within 100 m of semi-natural woodland on NWM ash that is currently open ground covers 1 722 ha; ground that is currently under plantation covers 350 ha. If brought into the Upland Ashwood expansion targets, this 2 072 ha of land may give the greatest conservation benefit in the shortest time, while at the same time linking remnant woods to address the problems of fragmentation. In addition, expansion around semi-natural woods will increase woodland cover in areas where woodland is an established part of the natural landscape, rather than colonising land from which woodland may have been absent for many years.

Figure 7.3

Expansion opportunities for Upland Ash woodland on North Mull.

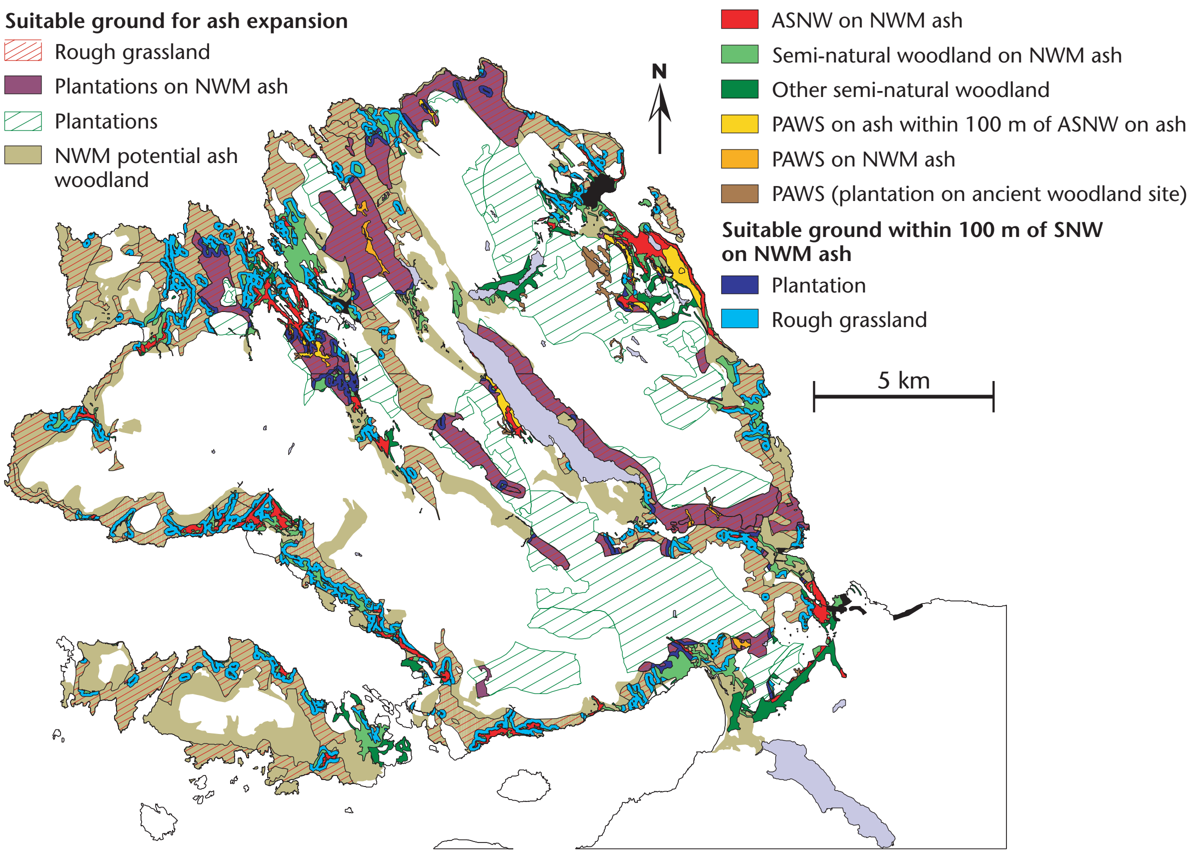
Expansion opportunities	ha	within 100 m of SNW on NWM ash
Plantation on NWM ash	2 195	350
Rough grassland on NWM ash	5 665	1 722
Total	**7 860**	**2 072**



Combined HAP targets for strategic woodland development

Woodland restoration and expansion activities should not be carried out in isolation from each other or from the current semi-natural resource. Figure 7.4 attempts to group all the information from Figures 7.1, 7.2 and 7.3 to give an overview of the opportunities for Upland Ashwood immediately available in north Mull. The graphic combination of information onto one map facilitates prioritisation. For example, fairly undifferentiated areas of plantation on NWM potential ashwood

Figure 7.4 *Distribution of current semi-natural woodland on North Mull in relation to existing and potential ashwood sites.*



sites displayed in Figure 7.2 are shown in Figure 7.4 to possess different potentials for contribution to biodiversity targets by virtue of their relationship to other nearby priority sites.

This kind of modelling brings useful information to the development of strategic woodland planning, be it a Forest Habitat Network (FHN), an Indicative Forestry Strategy or a Local Forestry Framework. At the simplest level, the most colourful areas of the map in Figure 7.4 show the parts that the woodland conservation effort should consider first – for the greatest gain in the short to medium term. In the longer term, FHN considerations must dominate.

Forest Habitat Networks

The requirements of the Habitat Action Plans emphasise the area targets for maintenance, restoration and expansion, but there is considerable work (e.g. Kirby *et al.*, 1999) suggesting that an uncoordinated approach to delivering these targets may miss real opportunities. The fragmentation of formerly extensive habitats has been shown across the world to lead to a decline in characteristic woodland biodiversity; the corollary of this is that a reversal – a defragmentation – is likely to sustain such biodiversity, and in some cases to lead to recolonisation and range expansions. Thus the targets derived from pressures such as HAPs will be most valuable when combined to form networks which link similar habitats; these have been articulated more thoroughly as FHNs (Peterken *et al.*, 1995).

The raw materials for promoting the development of a FHN are derived from the discussion of HAP targets: identifying the existing native woodlands, especially those of high value; identifying areas suitable for restoration from plantation; and identifying areas suitable for expansion, especially by natural regeneration. How the various actors – landowners, public agencies, communities, NGOs, local authorities – can use these raw materials to bring about this intelligent woodland expansion is likely to be a function of strategies (IFS, local Forest Frameworks), incentives (WGS challenge funds and targeted grant schemes), data and information availability, market pressures (certification, local value-added initiatives), other land-uses and pressures (designated sites and landscapes, agricultural decline or expansion and associated incentives). However, the GIS-based approach to strategic planning illustrated in this chapter demonstrates how the integration of spatially explicit woodland datasets and models can provide a powerful tool for setting priorities for woodland restoration and expansion at the regional level.

The analysis of these datasets in order to illuminate HAP priorities for the whole of Scotland is now under way, with the expectation that this information will be available for all the upland HAP types in map and digital form in the near future.

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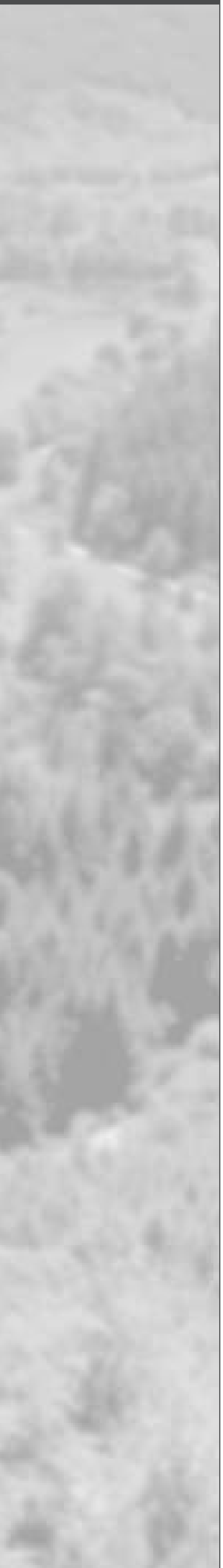
SECTION THREE

National and regional planning

CHAPTER 8



The contribution of the Millennium Forest for Scotland initiative to forest restoration

John Hunt

Summary

The role and scale of the Millennium Forest for Scotland initiative and its contribution towards native woodland restoration and woodland habitat networks is described. One major woodland inventory project is discussed, the Millennium Guide to Scotland's Forest Resource, with an example of the data which this can provide. Two core forest areas (Affric/Cannich and Loch Lomond/Trossachs) have a concentration of Millennium Forest for Scotland projects and these are used to illustrate the type and scale of restoration work being carried out. The lessons learnt to date are outlined and the issues likely to influence the success of future woodland restoration projects are considered.

Introduction

The Millennium Forest for Scotland Trust (MFST) was set up in late 1995 in support of a portfolio of projects throughout Scotland to further the restoration of native woodlands and enhance their value for people. Funding has been provided mainly by lottery money distributed by the Millennium Commission (MC), but Millennium Forest for Scotland (MFS) projects have also been supported by a wide range of other organisations. The considerable interest aroused by the MFS initiative resulted in over 300 applications being received for woodland restoration projects. In the event the grant provided by the MC (£11.34 million) only allowed about a quarter of these projects (72) to be supported. These will result in the creation and improved management of over 22 000 ha of predominantly native woodland. The stimulus provided by MFS also led to many other projects being developed in detail and then proceeding in some form without MFS grant aid.

The publication of *A forest habitat network for Scotland* (Peterken *et al.,* 1995) was very timely and provided a valuable rationale when developing criteria for promoting the concept of MFS at the beginning, and then later when assessing applications. It helped to confirm that the emphasis should be on improving and expanding existing remnants of native woodland rather than creating new isolated native woods. A number of Core Forest Areas were identified early on as locations where there would be particular value in supporting woodland restoration schemes. These included:

- Cairngorms
- Beauly catchment
- Torridon area
- Loch Lomond and the Trossachs
- West coast oakwoods.

The development of projects in these areas was encouraged by approaching key landowners and organising regional meetings. The desirability of providing linkages between existing woodlands and along riparian and other natural corridors was also recognised and promoted. However MFST was not able to design projects itself but had to consider the applications submitted to it. These were then subject to a process of assessment which often led to revision of the proposals to ensure they met MFS criteria. Projects were all expected to consult with and obtain the support of their local communities, though a number of project proposals originated from local community or woodland initiatives.

Figure 8.1

Map showing locations of MFST Core Forest Areas in Loch Lomond and the Trossachs, and the Beauly (Affric/Cannich) catchment.

In the event, the delivery of the Core Forest Areas did not go as far as MFST would have wished, but in two particular areas a valuable grouping of projects did emerge. These were in part of the Beauly catchment (Affric/Cannich), and the East Loch Lomond and Trossachs area (Figure 8.1). These are used as examples to illustrate what has happened and the issues that arose.

Millennium Guide to Scotland's Forest Resource

The Millennium Guide to Scotland's Forest Resource is an inventory of all woodlands in Scotland over 0.1 ha in size. The project is funded by MFST, Forestry Commission and Scottish Natural Heritage, and is carried out by the Caledonian Partnership under the auspices of Highland Birchwoods. Information from aerial photographs and other data sources has been brought together and digitised, including a wide range of woodland attributes as well as woodland boundaries. The data can be readily mapped and analysed in a way that has not been possible before. This inventory will provide a framework to which can be added additional information about individual woods, such as field surveys. Selected data will be available to the public on the Internet now available at www.scotlandswoods.org.uk.

The inventory is not yet complete, but it may be of interest to illustrate its use with some provisional data for the Affric/Cannich area (Table 8.1). Within the approximately 44 000 ha which encompass the catchments of Glens Affric and Cannich, the inventory identifies 12 557 ha of woodland (28.5% of the total area).

Woodland type	Area (ha)	% of woodland
Broadleaf	1 685	13.4
Mixed broadleaf and conifer	1 808	14.4
Mainly conifer	3 074	24.5
Conifer	5 925	47.2
Scrub	65	0.5
Total	**12 557**	**100**

Table 8.1

Provisional data from the Millennium Guide to the Forest Resource for the Affric/Cannich area, showing woodland types and naturalness.

Woodland naturalness	Area (ha)	% of woodland
Semi-natural	3 844	30.6
Mixed semi-natural and planted	941	7.5
Mainly planted	3 105	24.7
Planted	4 667	37.2
Total	**12 557**	**100**

Example 1: Affric/Cannich catchment

This is an area long famous for its substantial remnants of native pinewood set in a magnificent Highland landscape. However, until the 1960s, these woods suffered considerably from felling and replacement with non-native conifers, while uncontrolled grazing by deer and livestock contributed to a long-term decline. However in the 1960s the Forestry Commission fenced 1 400 ha of the old pinewoods to encourage natural regeneration and since then their management has been increasingly aimed at conservation, with further major restoration work taking place in the 1990s (Wield, 2001). Two other nearby areas have been purchased in recent years by conservation bodies – West Affric Estate by National Trust for Scotland and Corrimony by RSPB. The conservation charity Trees for Life has been championing forest restoration in this area for some years, using volunteers and raising funds to carry out a number of projects. Some of these have taken place on private land with the support of sympathetic landowners concerned to protect their native woodlands. MFS funding in this area has helped to extend or bring forward restoration work on a number of properties and has complemented initiatives already funded by European Union LIFE money, Woodland Grant Scheme and private funding. Table 8.2 gives details of the work that has been carried out from the mid 1990s to 2000 for selected landholdings (covering 57% of the catchment and 85% of the existing woodland in the catchment).

Forest Enterprise (FE) is now well advanced in a massive conversion of plantations to native woodland by removal of non-native tree species and 'naturalisation' (varied thinning and restructuring) of planted Scots pine. Some plantations near to Cannich will remain as primarily production woodland, but most of the FE's huge holding will become native woodland with future re-stocking to be achieved by natural regeneration.

The extent of native woodland is being increased substantially on several properties by a combination of planting and natural regeneration, protected in part or completely by deer fencing. This new woodland will be mainly Scots pine and birch, but further west in Upper Glen Affric the emphasis shifts to native broadleaves with comparatively little pine. This reflects recent palaeoenvironmental work (Tipping, personal communication) which suggested that pine has probably never been an important component of woodland in West Affric because of past climate and soil conditions.

Ground to the north and south of Glens Affric and Cannich is deer forest with high densities of red deer. Further expansion of woodland is severely restricted by deer browsing and, apart from the lower part of Glen Affric, is only possible within deer fences. It is hoped that collaborative management with the sporting estates will lead to future reductions in deer numbers and allow at least some natural regeneration to take place outside exclosures. This will not be easily achieved.

The series of exclosures on National Trust for Scotland ground continue westwards through to Kintail and provides the beginnings of a corridor of woodland from the west coast that links with the more continuous woodland in Affric and Strathglass, and so through to the east coast. The planted exclosures are also intended to create seed sources for future natural regeneration over wider areas, although to succeed, deer numbers will need to be reduced.

Table 8.2 *Native woodland restoration on selected landholdings in Glens Affric/Cannich.*

Ownership	Land area (ha)	Existing wood-land (ha)	New planting (ha)	Natural regene-ration (ha)	Non-native removal (ha)	Naturali-sation (ha)	Other works and comments
Forest Enterprise, Affric and Cannich	*c.*14 000	*c.*10 000	260	2 500+	1 550	184	Strategic deer fencing and deer control to achieve natural regeneration throughout forest. 80%+ of Forest Enterprise forest will become 'native' in due course.
National Trust for Scotland, West Affric	3 700	0	154	67	-	-	Ten enclosures (221 ha). Reduction in deer numbers intended. More enclosures on Kintail to the west.
Royal Society for the Protection of Birds, Corrimony	1 530	490	100	240	80	Some intended	Deer numbers reduced; deer fences removed; low level stock grazing on part of site.
Mullardoch Estate	2 000	120	10	90	-	-	Four enclosures (105 ha) mainly for natural regeneration. Deer reduction planned to achieve more.
North Glen Affric Estate	3 700	30	0	30	-	-	Three deer exclosures (90 ha). Estate primarily a deer forest.
Total	***c.*24 930**	***c.*10 640**	**524**	**2 927+**	**1 630**	**184+**	

1. Figures apply to woodland restoration completed from the mid 1990s to 2000.
2. The table does not include all the private estates which lie within the 44 000 ha Affric/Cannich catchment.

Example 2: East Lomond and the Trossachs

The semi-natural broadleaved woodlands in the Loch Lomond and Trossachs area have long been recognised for their landscape and conservation value and their past management has been well documented (Tittensor, 1970). However for most of the last 100 years they have been neglected with, apart from the Loch Lomond islands, most woods open to grazing by sheep, deer and goats so that natural regeneration has rarely occurred. Where semi-natural woods were fenced against grazing they were usually hemmed in by conifer plantations and unable to expand.

However, over the last 15 years much has changed. Substantial areas have been acquired by conservation bodies (National Trust for Scotland, Jensen Foundation, RSPB, Royal Scottish Forestry Society Forest Trust and Woodland Trust) who are all involved with native woodland restoration, while Forest Enterprise and West of Scotland Water have recently started conservation management of their woodlands. On a smaller scale some private landowners have also taken steps to expand native woodland. The Ben Lomond Memorial Park and forthcoming National Park status for the wider Loch Lomond and Trossachs area should provide further impetus to continue this process.

Table 8.3 gives a breakdown of the main landholdings on the east side of Loch Lomond and in the west Trossachs, and details the woodland restoration work which has been carried out since the mid 1990s or which is under way. Only part of this work is supported by MFST.

Table 8.3 *Native woodland restoration on selected landholdings on east Loch Lomond and west Trossachs.*

Ownership	Land area (ha)	Existing wood-land (ha)	New planting (ha)	Natural regene-ration (ha)	Non-native removal (ha)	Rhodo-dendron removal (ha)	Other works and comments
Forest Enterprise, East Loch Lomond	1 800	1 400	25	Some	260	55	Deer exclosures (40 ha); 5 ha broadleaves thinned. Aim to convert remaining 800 ha conifers to native broadleaves over 40 years; 250 ha to be completed by 2003.
National Trust for Scotland, Ben Lomond	2 173	23	-	36	-	-	Several exclosures (49 ha). Reduction in overall sheep numbers has taken place.
Jensen Foundation, Comer	2 428	110	150	98	13	16	Series of deer fenced exclosures; sheep operation plus enclosed deer population; remaining conifers (22 ha) to be removed.
Royal Society for the Protection of Birds, Inversnaid	400	100	12	Intended	-	-	Deer exclosure (40 ha). Deer and goat control to achieve natural regeneration.
Royal Scottish Forestry Society, Cashel	1 240	23	360	Intended	-	-	Woodland all within a ring deer fence. Full altitude range of woodland planned.
West of Scotland Water, Loch Katrine	9 576	900	201	176	67	-	Restoration within deer or sheep exclosures; bracken control; sheep farming still major land-use.
Woodland Trust, Glen Finglas	4 039	166	538	460	-	-	Grazing some ground for wood pasture. Further sheep reduction planned. Potential for 2 000 ha more natural regeneration.
TOTAL	**21 656**	**2 722**	**1 326**	**770+**	**340**	**71**	

The figures apply to woodland restoration work carried out since the mid 1990s or under way in late 2000.

Grazing management has also been a key issue in this area. Sheep farming is still an important land-use, while feral goats are also present in some locations and raise particular sensitivities when control is necessary to protect woodlands. Red deer are at the southern end of their Highland range and densities are low, so it has been disappointing that deer fencing has been felt necessary in most cases to ensure the success of planting or natural regeneration. Even though the organisations involved have a common interest in achieving low deer numbers there has been a reluctance to take a risk or give offence to sporting interests by relying on deer control alone; instead the safer option of deer fences has generally been adopted.

Restoration work that is now under way will substantially increase and improve the strip of oakwoods up the east side of Loch Lomond and around Loch Katrine. There is potential to extend this further north to Glen Falloch where other native woodland schemes are taking place. In the long-term natural woodland should be established over its full altitudinal range on Ben Lomond and at Cashel, and there is every expectation that in due course woodland links will be established across Comer and Cashel to the Trossachs and Loch Ard Forest to the east. At Glen Finglas the long-term intentions of the Woodland Trust are to create very large areas of native woodland, including the restoration of former wood pasture. The latter aim will be assisted by the Trust retaining a limited farming operation.

Why are these particular forest areas being restored?

The Affric/Cannich and Lomond/Trossachs areas outlined above are subject to a large and sustained effort aiming to restore native woodland. The following are suggested as the main reasons why they are receiving such favourable treatment:

- Significant remnants of semi-natural woodland have survived and these have provided a focus on which to build.
- Statutory Designations such as SSSIs have provided protection and helped to stimulate restoration.
- Large parts of the area are owned by sympathetic landowners – including conservation bodies who have been attracted to acquire land because of the conservation interest and potential.
- Public opinion has influenced government policies and grants, thus providing a favourable climate of change as well as increased funding.
- Landowners have the confidence and commitment to undertake restoration based on increased knowledge and resources.

The MFS experience

The MFS initiative will result in a substantial increase in the extent and quality of native woodland, which in turn will help to strengthen woodland habitat networks. Much should be learnt from future developments and there is considerable potential to extend the woodland restoration into adjoining areas. The main lessons learnt to date and some thoughts for the future are given below:

Lessons learnt

- The timetable imposed by the Millennium Commission required that all major works were completed by the end of 2000 and this has meant that in some cases management has been carried out more quickly than was ideal. With some projects it would have been preferable to take longer to plan or carry out restoration work.

- Grazing by deer and/or sheep has been a major consideration with most projects and has required substantial expenditure on fencing. MFST encouraged projects to avoid the use of deer fencing but accepted that it was necessary in many cases. Where deer fencing has been used it has usually been a condition of grant that future deer management should seek to achieve woodland regeneration in the long-term without the use of deer fencing.

- In carrying out woodland restoration, great care is needed to ensure that valuable non-woodland habitats such as wetlands, heathlands and botanically rich grasslands are not damaged. The protection of archaeological interests is also an important consideration. It is believed that these interests have been safeguarded and in some cases enhanced by MFS projects.

- The level of communication and collaboration between landowners involved in woodland restoration projects within the same general area has been less than hoped. Despite encouragement from MFST, the organisations and individuals concerned have usually preferred to operate independently.

Issues for the future

It is hoped that extension and improvement of native woodland and their associated habitat networks will continue in the future. If so, some of the key issues will be:

- The need to involve local communities at an early stage and to maintain their support and involvement if opposition is to be avoided and local benefits are to be realised.

- The impact of grazing by sheep and deer which prevent woodland restoration in many upland areas. Deer present particular problems because of their high populations and mobility. Significant reductions in deer numbers are needed but the necessary control measures often require a collaborative approach between adjoining landowners and this is hard to achieve. Deer fencing has many disadvantages and its continued widespread use may not be acceptable in future.

- The desirability of encouraging natural processes (for environmental and practical reasons) wherever possible in woodland restoration schemes. This should include the use of natural regeneration which has ecological benefits and may be more acceptable on landscape grounds than new planting. Changes are required to the Woodland Grant Scheme if natural regeneration is to be more widely adopted for woodland establishment.

- The need to strike a balance between native and non-native species in order to maximise conservation and other benefits. There is still much to learn about the management of mixed native and non-native woodland, and changes would be required to the Woodland Grant Scheme to encourage this approach.

- The need to ensure continuity of management with clear objectives including the use of long-term management plans.

- Adequate resources will be needed for ongoing woodland management to ensure that the initial restoration work is not wasted. It will be desirable to generate income from timber and other sources at an early stage in order to assist woodland management and generate economic benefits.

- The need to learn from existing restoration schemes which should be well monitored and used for demonstration purposes.

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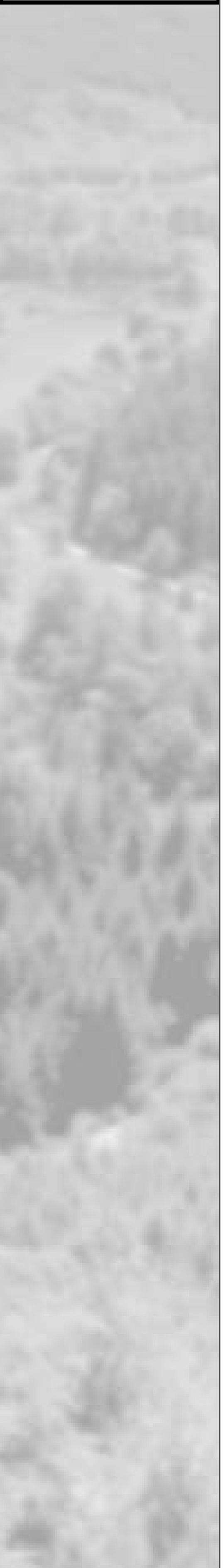
CHAPTER 9

Developing forest habitat networks in Scotland

George Peterken



Summary

The concept of a forest habitat network provides a useful basis for reversing forest habitat loss and fragmentation, thereby providing opportunities for populations of wildlife species to become more resilient. This chapter summarises the concept and describes how it might be applied in Scotland as a whole and in particular regions (Cairngorms, western Highlands) and districts (Middle Clyde Valley). An effective network would also benefit timber production, amenity and water quality, and afford foresters greater freedom in management.

Introduction

Woodland was once the matrix within which other habitats in Scotland occurred as patches or corridors. Millennia of forest clearance reversed this: woodland was reduced to patches within a matrix of unwooded land. As a result, populations of most woodland species have been reduced and fragmented to various degrees into separate, isolated sub-populations, which are vulnerable to local extinction and genetic change.

The restoration of woodland habitat networks aims to reverse the effects of habitat fragmentation. By adding woodland to the landscape, we aim to reduce ecological isolation, thereby enabling populations to expand and become more resilient, i.e. capable of occupying all available habitats, adjusting rapidly to habitat and other environmental changes (such as global warming), and maximising genetic diversity. There may, however, also be costs; for example, networks might facilitate the spread of introduced species.

The concept of habitat networks has developed out of landscape ecology. Landscape ecology itself developed in the 1950s from the work of Carl Troll, as a way of looking at large-scale patterns and processes and a means of considering the context – human and natural – of individual sites (Forman and Godron, 1986; Forman, 1995). Habitat networks were seen as a basis for habitat restoration from the 1960s onwards. Ecological isolation was appreciated as a conservation threat at about the same time (e.g. Moore, 1962). A European Ecological Network was proposed in Bennett (1991).

This chapter considers the basis and practicalities of restoring woodland habitat networks in Britain, particularly in Scotland. Early suggestions that the network approach would be useful (Wightman, 1992) were considered in some detail on a whole-Scotland scale by Peterken *et al.* (1995). Since then, further studies in the Cairngorms (Ratcliffe *et al.*, 1997), western Highlands (Peterken, 1999) and the Clyde catchment (Peterken, 2000a) have provided opportunities to develop the concept for particular districts and regions. A more English perspective is given in a complimentary paper (Peterken, 2000b). Throughout, the term 'forest habitat network' (FHN) is preferred to 'woodland habitat network', because it more accurately conveys the large-scale of the concept, includes plantation forests as well as native woodland, and covers both wooded ground and other habitats among the trees.

Structure of a network

Landscapes can be regarded as the sum of patches, corridors and matrix. A habitat network comprises patches and corridors of one habitat in a matrix of distinctly different habitats, or it is a network of natural and semi-natural habitats in a matrix of intensive agriculture, residential and industrial development. The quality of the matrix varies in respect of forest species from hostile habitats that support few woodland species, such as arable fields, short-term leys and urban concrete, to benign environments containing many semi-woodland habitats supporting numerous woodland species, such as hedges, incised stream banks and meadows.

The habitat network can be viewed at a range of scales, from a continent to a parish or individual farm or forest. At a national scale, large forest concentrations, or 'Core Forest Areas' (CFAs), are linked by densely- or well-wooded corridors, i.e. 'Large Landscape Links' (LLLs). Within CFAs and LLLs, individual woods form smaller-scale nodes, linked by related habitats, such as hedges, scrub-covered streamsides and shelterbelts. Outside CFAs and LLLs, individual woods may be isolated from the network, or tenuously linked through a benign matrix.

Reference points for woodland in the landscape

It is important to recognise thresholds in the relationship between woodland cover and the whole landscape. Checkerboard simulations of land transformations (Franklin and Forman, 1987) have shown that thresholds occur at about 30% and 60% wooded cover (Peterken, 2000b). In a landscape in which woodland is increasing randomly, the former is the point above which almost all additional woodland will be close to existing woodland, i.e. at which the ecological isolation of individual woods becomes minimal. It is also the threshold at which mammals tend to use the landscape as a single wood (Andrén, 1994), and at which the pattern of woodland herbs is not strongly determined by the pattern of ancient and secondary woodland. The 60% threshold is the point at which almost all woodland coalesces into a single wood with holes.

Below 30% cover, woods are generally small and isolated. In the poorly wooded 0–15% range, most woods will be small, edge habitats will be minimal, there will be little or no interior habitat, landscape-scale movement of woodland species must largely take place outside woodland, and ecological isolation will be maximal. In the well-wooded range from 15–30%, large woods will be present and edge habitats will be substantial. In the densely wooded range between 30% and 60% cover, woodland comprises a mix of large and small woods. Edge habitats are maximal, but there is little interior habitat. Woodland species can move through the landscape without crossing other habitats, particularly above 40% cover. Above 60% cover, woodland forms the matrix within which other habitats are isolated. The amount of edge habitat decreases as 100% cover is reached, and the amount of interior forest habitat increases rapidly above 80–90% cover, depending on the depth of edge effects.

The basic requirement for CFAs and LLLs is that ecological isolation within them must be minimal. This implies that forest must cover at least 30% of the ground, though there are advantages for forest species in having a larger proportion. Put another way, CFAs can include up to 70% cover of open ground.

Thresholds in the size of individual woods

At most realistic levels of forest cover even CFAs are likely to include individual discrete woods. Even within a CFA, a balance must be struck between the extremes of many small woods or few large woods, for both patterns have merits and disadvantages for biodiversity. In most landscapes, however, the critical need is to develop some large woods (Forman, 1995). There are also indications that size thresholds operate in the interactions between diversity and management. For example, a study in the English lowlands suggested that: 3 ha is the minimum size at which managed woods are more likely than not to include open spaces, such as access tracks and rides; 20–30 ha is the minimum size at which at least some part of the rides and glades is likely to be unshaded, due to forestry operations, etc.; and these thresholds influence the number and relative importance of species dependent on open spaces (Peterken and Francis, 1999). The size at which lowland woods are almost certain to contain breeding marsh tits is 25 ha (Hinsley *et al.*, 1994).

Development of forest habitats in Scotland

Scotland has long been 'poorly wooded', save for a few districts which have remained well wooded, e.g. central Speyside (Anderson, 1967; Smout and Watson, 1997). Millennia of forest reduction were arrested in the 18th–19th centuries by widespread planting and coppice management of many remaining native woods, but, even after three decades of state-sponsored afforestation, the total area of woods and plantations in 1947 was still only 6.7% of the land area (Forestry Commission, 1953). Massive afforestation after 1947 increased the total area to 19%, and changed some districts from treeless moorland and pasture into substantial plantation forests (e.g. central Galloway, Nairn and Moray), but 'natural-origin' woods remained at 2% of the land area (MacKenzie, 1999). Today, plantations dominate the forest cover and native woodlands occupy only a tiny fraction of their natural range. However, during the last decade a substantial effort has been made to increase native woodland cover (see Rollinson, Chapter 1, Figure 1.3).

Native woodland is now concentrated in and around the Highlands, with a scatter of small woods elsewhere. Boundaries of Highland woods are often poorly defined and mobile over centuries; nevertheless most woods have retained a core of permanently wooded land. In the lowlands, woodland is more evenly scattered as small woods in a matrix of farmland. Boundaries are now sharply defined and static, but this is a condition imposed by enclosure in the last 200 years.

The composition and structure of native woodland has been modified by usage. Substantial tracts were converted by wood-pasturage to open parkland, and this is still visible in the granny pines, surviving grazed woodland at Glen Finglas, Creag Megaidh and western Highlands generally. Remote Highland pinewoods were logged in the 17th–19th centuries (Steven and Carlisle, 1959). Native broadleaved woods were converted to coppices in the 18th century and many were planted as oak at that time (Lindsay, 1975). Coppicing survived until the 1950s in Perthshire, but in most areas it ceased in the 19th century and the woods were exposed to grazing by deer, sheep and cattle. Today, most native woods comprise well-defined age-classes, which can be related to pulses of regeneration following felling, enclosure, or fluctuation in grazing pressure (McVean, 1964). In terms of composition, many have been simplified into single-species dominance by planting, selective felling, selective effects of pasturage, and the different longevities of native tree species. Birchwoods are the most common (Kirby, 1984), but their ecological status varies from (1) ancient birchwoods which probably never had other dominant trees, and (2) simplified mixed woods from which oak and pine have been removed, to (3) secondary birch scrub, which would probably develop a mixed composition if allowed.

During the 18th and 19th centuries numerous 'policy' woods were planted. These generally included some native trees, but they also introduced beech, sycamore and other non-native trees, some of which have become naturalised. Beech and sycamore in particular are now also widespread and well established in ancient and long-established native woods, while conversely the older plantations have been colonised by native trees and shrubs. Norway spruce, Sitka spruce and western hemlock became naturalised during the 20th century.

Applying network concepts in Scotland: background considerations

The theoretical structure of a network must be reconciled with the particular characteristics of the region under consideration. In Scotland, certain key features require modifications to the theoretical structure:

1. Core Forest Areas (the large-scale nodes) have a strongly linear form in and around the Highlands, principally because they are associated with the major valleys and the mountain margins, and in the west they are shaped by the strongly indented coastline and elongated lochs. Examples include central Speyside, upper Deeside, Sunart and the Great Glen. Relatively compact CFAs do occur, e.g. Galloway. Nevertheless, CFAs generally function as both nodes and large-scale links.

2. A high proportion of existing forest comprises conifer plantations originating in the 20th century. As forest habitats for native wildlife these are both immature and poor in quality, yet

they cannot be ignored in a FHN. Key components of network development must be improvements in habitat quality and the completion of links to native woodland outside plantation boundaries.

3. Native woodland boundaries tend to be diffuse, particularly in the Highlands. This is partly due to the prevalence of shade-intolerant trees, which generate 'mobile woods', degenerating in closed stands, but regenerating in open areas. It is also due to the historical prevalence of woodland pasturage, which generated a low-definition landscape of intimately intermixed trees, shrubs, grassland and mire (Smout and Watson, 1997).

4. The unwooded matrix contains numerous semi-woodland habitats, notably streamsides, rocky gullies, ledges, screes and cliffs, which provide shelter and some refuge from grazing. Even in the intensively cultivated lowlands, woods tend to be associated with other semi-natural habitats on valley sides (e.g. mid-Clyde), and farmland still has orchards, boundary trees, hedges and remnants of 'unimproved' grassland, all of which provide habitats for some forest species. Except in intensively arable districts, the matrix is not totally hostile to forest species. In some Highland districts and along major valleys it makes an important contribution.

Ecological and conservation limitations

It is easy to assume that all land within a forest network will provide habitats for forest species and that forest networks are invariably beneficial for forest species. However, there are limitations in the use of networks by forest species and disadvantages in achieving networks, both of which must be taken into account in network design.

Limitations on colonisation arise from the need for the habitat to mature before certain species can colonise, e.g. dead wood species; poor powers of colonisation of many woodland species, some of which colonise only those new woods formed close to existing woods; and internal barriers to movement within a forest network, such as wet tracts that form a barrier within a forest corridor to species that cannot tolerate wet ground. Experience on the Isle of Rum (Wormell, 1977) has shown an encouraging capacity for invertebrates to colonise, which is particularly significant on an almost totally deforested, remote island. On the other hand, the spread of many woodland plants may be limited, e.g. even in a region dominated by birch woodland, some plant species were found associated with ancient woodland (Miles and Miles, 1997). These limitations imply a need to locate new woodland close to existing woodland, and for either patience or measures to accelerate habitat maturity.

Many species are specialised within forest habitats, e.g. species associated with particular vegetation types or site conditions, phytophagous species associated with particular plant species, and saproxylic species associated with large trees and dead wood. The consequences can be inferred from the maps generated by the Native Woodland Model (Hester *et al.*, Chapter 5), which show the potential forest pattern in terms of the National Vegetation Classification. Some woodland types occur as small patches within a matrix of other types; if any species depends on such a type, it will be subject to isolation effects, whether or not the whole landscape is forested. Even after woodland types are converted to hypothetical contour maps for individual tree species (Ratcliffe *et al.*, 1997), it is clear that, even in a heavily forested landscape, species associated with particular tree species will encounter internal barriers, isolation from small patches of their specialist habitat, and limitations on their potential range. Montane habitats inevitably occur as islands in a matrix of lowland habitats. Network design must take these limitations into account. In particular, riparian locations provide the best opportunities for LLLs, since they are the most heterogeneous sites within the landscape, they include the spatially limited base-rich sites, and they are the natural focus of movement in a landscape. Collectively, however, they form dendritic networks, which have their own inherent forms of isolation and hierarchies of connections (Forman, 1995).

Forest species include specialists dependent on either pre-thicket (young-growth) stands or old-growth stands. Such species will only experience a FHN as a network if the various growth stages are well distributed through the forest. This requirement for networks-within-the-network implies particular specifications for forest management:

- a mosaic of all age classes, i.e. continuity of management;
- well-linked long-rotation stands and retentions in parts of the forest, and/or;
- more extensive deployment of coppice-with-standards, shelterwood regeneration and two-storey high forest, all of which provide both maturity and young growth in one stand.

Network development in Scotland may not invariably be beneficial. There are two particular points of concern:

1. Introduced species might also spread, notably rhododendron, cherry laurel, grey squirrel, beech and sycamore. This danger may, however, be less than it seems. Thus, grey squirrel and rhododendron have, and will continue to spread even in the absence of a FHN. Spread of beech and sycamore will be facilitated by a FHN, but there is a case for accepting them as part of the mixture (Peterken, 1996). Nevertheless, even though networks have a net benefit, there may be a case for retaining the isolation of some woods. For example, increasing the connectivity of broadleaved areas can pose a threat to red squirrel populations by facilitating the dispersal of grey squirrels, and to avoid this broadleaves are being removed from Clocaenog Forest (North Wales).

2. Fragmentation of non-forest habitats. If woodland is linked over long distances, other habitats will be reduced and could be fragmented. Leaving aside the point that increased forest cover would be a return to a natural state of affairs, there are three mitigating points: (a) the forest network would take on the same dendritic network form of valleys, which implies a complementary network of non-forest habitats; (b) forest networks would still contain up to 70% open ground within them, so should be porous to non-forest species; (c) the mesh size of any forest network can be very large where non-forest habitats are of critical importance.

A forest habitat network at a whole-Scotland scale

The rationale and basis for a whole-Scotland network were set out by Peterken *et al.* (1995) and Hampson and Peterken (1998). Particular priorities were: to reinforce links around and within the Highlands to complete an already well-developed network based on major valleys; to link this to existing concentrations of woodland in the western Highlands (e.g. Sunart); and to construct a link across central Scotland to the concentrations of forest in and around the Southern Uplands. Within the lowlands, the need was for improved small-scale connections between the scatter of small woods. Apart from identifying a need to concentrate new woodland along the Highland Boundary Fault to complete the 'ring', the focus of development at both large and smaller scales was riparian, in the sense that the forest network would take on the pattern of the drainage network. This would not only reinforce existing forest patterns and help to link native woods that are strongly associated with valley and stream sides, but also generate links within the most heterogeneous parts of the landscape.

Regional scale forest networks

Ratcliffe *et al.* (1997) developed network ideas for the Cairngorms. This is a massif fringed by two elongated CFAs concentrated on the Spey and Dee. Boreal pine and birch forests form a major component of potential and actual native woodland, but there has been extensive conifer afforestation, especially on the north side. The report identified the need to complete the ring of forest around north and north-east sides, and reinforce links with areas outside (e.g. via Loch Laggan to Spean Bridge). Guidelines based on the colonising powers of wildlife species were developed for planning insertions of new woodland into the established pattern, and large-scale requirements for managing all semi-natural woodland and plantations.

The Western Highlands raise many issues for network development (Peterken, 1999). Forest patterns will inevitably be linear and fragmented. The native woods have a long history of wood-pasturage before coppicing became important in the 18th and 19th centuries. The widespread and characteristic oceanic oakwoods are probably the least natural kind of native woodland in the region, having been formed during the era of coppicing. Within the potential matrix of birch and oak woodland, wet woodland and ash–hazel–elm woodland on base-rich soils will take the form of narrow, linear patches and isolated concentrations. Against this background, there seemed a strong case for planning networks on a catchment basis, and for emphasising expansion in the form of wood-pastures, i.e. inserting more scattered trees into lowland farmland. This would not only respect tradition and reduce competition for land between forestry and agriculture, but would also provide optimum conditions for epiphytic lichens and other key wildlife species.

The middle Clyde Valley possesses a concentration of native woodland in a poorly wooded region, and offers the best opportunity to develop links between the Highlands and southern uplands (Peterken, 2000a). It is dominated by the dendritic network of the Clyde and its tributaries, and this pattern is faithfully reflected by the ancient woods, which are mostly base-rich upland mixed ashwood stands on steep slopes and ravines. This concentration of native woods is, however, separated from the new plantations on the uplands by treeless headwaters and plateau farmland. Downstream, existing woods merge with urban greenspace on the fringes of the Glasgow conurbation. The measures proposed for developing a forest network start with reinforcing the concentration of ancient woods (Box 9.1, component 1), then continue with five components designed to forge links beyond this base.

Box 9.1 *Components in the development of a forest habitat network in the middle Clyde valley.*

1. **Consolidate the main river network; mainly ancient woods.**
2. **Link minor tributaries to the main river network.**
3. **Extend the main river network into headwaters; contact with upland plantations.**
4. **Extend main river network into plateau farmland.**
5. **Redesign upland plantation forests.**
6. **Develop wooded habitats in urban open spaces.**

Conclusions

Forest expansion over the last 200 years could be construed as a step towards a Forest Habitat Network for Scotland. It has reinforced some woodland concentrations, generated new CFAs, and achieved substantial long-distance links in and around the Highlands. However, it has mostly been progressed without native tree species; many native species remain isolated in small woods; and the management of plantations and native woods is far from ‘natural’, either collectively or individually. We still need more woodland, but it should have a higher native component, be directed to filling gaps and raising individual woods above threshold areas, and management should be more sustained and balanced.

The key need is to secure the base, i.e. ensure that ancient woods are retained, form all or part of a large enough block of contiguous woodland and are well managed. Then there is a need to construct links between small native woods, and to link native woodland within upland plantations to native woodland lower down, both kinds of link having a strong component of native trees and shrubs. There is a case for planning on a whole-catchment basis, and concentrating new woodland links on riparian corridors. CFAs need to be reinforced and managed in a balanced fashion.

If all these objectives are achieved, then not only will woodland species be more resilient, but there will also be many other benefits. Most, if not all additional woodland should be managed to yield timber and other material products. Landscape values are subjective, but for many people additional woodland with a strong native component and a pattern based on natural features would be attractive. Within agricultural catchments, riparian woodland would improve water quality. Moreover, since the populations of wildlife species would be more resilient, foresters would be able to manage with greater freedom, secure in the knowledge that species would respond to changes in structure and pattern.

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CHAPTER 10

A management framework to optimise woodland biodiversity in Wales

James Latham

Summary

Four basic structural states can be recognised for woodland in Wales – Natural Woodland, High Forest, Coppice, and Wood Pasture. These are the endpoints of different forms of management, and have varying, and sometimes mutually exclusive, benefits for biodiversity. A balance of woodland management, and hence structural states, is desirable to improve overall conditions for biodiversity and to reduce problems associated with management of woodland in isolation. To achieve this, the Countryside Council for Wales is developing a management framework. This will be based on units of ecologically linked woodlands, defined by criteria such as proximity and connectivity, woodland type, geology, soil type and river catchments. An example is presented which identifies 23 units in Wales. For each unit, the representation of woodland structural states will be evaluated, and management recommendations made to refine their overall balance and arrangement within the landscape. The framework will be applied to protected sites (SSSIs) initially, but should have the potential to accommodate woodland in the wider countryside as well. This chapter describes the background to the framework, its development to date, and discusses wider applications.

Introduction

Semi-natural woodland is a complex habitat, and can exist in quite different structural states depending on the type of management it receives. In British woodland, four basic structural states have been recognised (e.g. Hill *et al.*, 1998; Reid, 1998), and these are described below. Although transitions and intermediate forms occur between these states, they form a useful classification against which to consider management.

1. **Natural Woodland** In this state, structure, composition and regeneration result from, and are controlled by, natural processes. Such woodland may provide analogues of the original British 'wildwood' and has similarities to remnant virgin forest in continental Europe. Peterken (1996) discusses these concepts in detail. Generally, late-successional organisms are favoured, for example litter organisms, saproxylic beetles, bryophytes, fungi, bats, woodpeckers and owls. Additionally, the natural processes themselves have intrinsic natural conservation value. Natural Woodland is developed principally by some form of minimum intervention. This is not to be seen as abandonment of the woodland, but as management to control anthropogenic and external factors. For further discussion see Latham (2000) and Peterken (2000).

2. **High Forest** This structure consists predominantly of mature trees which are managed silviculturally. This may be for timber production and/or the conservation of organisms that require a predictable proportion of open space or particular canopy form. Tree and shrub species can be manipulated to promote rare species and discourage undesirable features (e.g. non-native species). The structure may be less good for late-successional groups than that of Natural Woodland because generally most of the trees are harvested before biological maturity and there is less potential for the development of deadwood habitat.

3. **Coppice** Early successional habitats and a high proportion of open space are maintained indefinitely in coppiced woodland. These benefit a range of organisms, most notably flowering

plants, butterflies and some other invertebrate groups. The dense, multi-stemmed regrowth provides a habitat for dormice and nesting sites for some birds, e.g. willow warbler *Phylloscopus trochilus* and blackcap *Sylvia atricapilla*. The 'losers' in coppice systems are deadwood specialists and any organisms reliant upon old growth, late successional stages, or continuously damp and humid conditions. Coppicing provides opportunities for some economic return. For a broad overview of coppice management and its ecology see Buckley (1992).

4. **Wood Pasture** This state combines woodland habitat with productive forage and shelter for grazing animals, and in this chapter refers to any woodland in which grazing pressure is high enough to significantly restrict regeneration and to be the dominant factor affecting structure (typically > 0.6 Live Stock Units ha^{-1} $year^{-1}$: Latham, 1999). Mature examples typically have many large, old and often open grown trees, and at their most open, wood pastures blend into parkland. Grazing maintains short field and sparse shrub layers. These structural components benefit light-demanding lichens, some fungi, deadwood specialists, litter-sensitive bryophytes (Rose, 1994; Kirby *et al.*, 1995) and birds such as pied flycatcher *Ficedula hypoleuca* and redstart *Phoenicurus phoenicurus* (Stowe, 1987). There is a growing opinion that this structural type has similarities to many of the original Holocene forests of northern Europe (Vera, 2000).

Each structural state has benefits for biodiversity, but the organisms benefiting are different for each. Optimum conditions for biodiversity and opportunities for production are likely to occur if all structural states relevant to local conditions are present within a representative area, or series of woods. This overall balance of representation requires co-ordination of management at a landscape or regional scale.

Co-ordination of management rarely happens at present, and management of a wood in isolation can cause various problems. For example, there may be no clear reason to choose one management type over others, which can lead to uncertainties and ambiguities in management planning. There may be real losses of structural and biological diversity if particular management treatments and their resulting structures are or become absent from a series of woods. At present there is no mechanism for knowing how well these structural types are represented and the scale of this problem. Managers may seek the best of all worlds by applying several management treatments within a single wood and, although this increases diversity for that individual site, for a group of sites the effect is to increase structural homogeneity, with each woodland containing fragmented examples of each structural state. In addition, none of the areas of a particular treatment may be individually large enough to provide the biodiversity benefits sought. There are economic implications of management in isolation too: opportunities for productive management may be missed, because groups of small sites are not being linked to give viable production overall. Conversely, timber harvesting, or 'management for management's sake' may take place in Sites of Special Scientific Interest (SSSIs) without real evaluation of its suitability.

The Countryside Council for Wales (CCW) is a developing a management framework to address these problems, and to achieve an overall balance of structural states. The idea is not new, and many of these issues have been discussed for some time (Nature Conservancy Council, 1983; Oliver, 1993; Fuller and Warren, 1995). They are mentioned in the UK Habitat Action Plans for woodland (UK Biodiversity Group, 1998), and have been acknowledged as necessary to achieve Favourable Conservation Status (EC Directive, 1992). However, until now, no such frameworks have been developed. This chapter describes the ecological basis for the Welsh management framework and outlines proposals for its implementation.

The framework principle

The basis of the framework is that woods are considered in groups, which are in some way linked ecologically. Within these groups, the representation and location of existing structural types and associated management are assessed. This information is then synthesised to make recommendations for management to improve the proportion and location of structural types to benefit both biodiversity and sustainable production. The emphasis is on recommendation; the framework is intended as a guide and not as a rigid prescriptive system. For example, landowners' interests and the specific requirements of rare species always have to be considered.

Identifying Ecological Woodland Units

Ecological Units are groups of woods, or defined areas of land containing woodland, that are ecologically interactive or in some way depend on one another. There are already geographical divisions of Wales, for example the Areas of Search used for SSSI selection (Blackstock *et al.*, 1996) and areas used for Local Biodiversity Action Plans (Natural Science Group, 2000). While it would be convenient to use pre-existing structures like these, they may not have the ecological linkage necessary for the framework to succeed. Therefore, it is proposed that new geographic divisions are identified using primarily ecological criteria, and these are provisionally called Ecological Woodland Units (EWUs). Criteria which may be used to identify EWUs include:

1. **Physical proximity** Woods are likely to be ecologically linked if they are close together. This is because organisms may disperse between them, or the woods may hold meta-populations of species. Dolman and Fuller (Chapter 3) review the theoretical background to these processes, and consider the dispersal ability of different woodland taxa. Additionally, natural processes and disturbances may take place at scales larger than individual woods, but can be fully expressed within a series of local woods. For example, wind is an important agent of disturbance in upland British forests (Quine *et al.*, 1999), and patterns of windthrow and the successional structuring resulting from it may occur on a landscape scale (Spies and Turner, 1999, and *inter alia* Perry, 1994; Forman and Collinge, 1996; and Peterken, 1996). Information to allow analyses of overall woodland distribution is available from CCW's Phase I survey (Blackstock *et al.*, in prep), Ancient Woodland Inventory (digitised version of the inventory described in Spencer and Kirby, 1992), and the Forestry Commission's National Inventory of Woodland and Trees (Forestry Commission, 2002).

2. **Corridors and stepping-stones** Woods may be ecologically linked (for the reasons above) if wooded corridors, such as hedges, scrub, or stepping-stones of individual trees, connect them. Phase I survey and aerial photographs provide information for this criterion.

3. **Woodland type** Woods with similar communities of plants and animals will inevitably share species and may be subject to similar ecological pressures. In some situations they may be considered part of an ecological continuum, and therefore ecologically linked. Plant communities are generally better recorded than animal communities and provide a practical basis for analysis. Soils, geology and climate are intimately linked with biological communities and these may be used to infer ecological affinities of woodland. CCW keeps extensive Phase II woodland survey information (using the National Vegetation Classification methodology – Rodwell, 1991), which is held in a GIS database (Latham, 2001) and can be used to identify groupings of woodland types. This can be done from the broad distributions of Habitat Action Plan (HAP) woodland types (as defined by Hall and Kirby, 1998), but more subtle patterns can be demonstrated by multivariate statistical techniques such as Detrended Correspondence Analysis and Cluster Analysis. Most vegetational groupings are likely to be spatially defined, and to relate to geographic clusters of woods. However, there may be instances of rare woodland types or communities that occur as dispersed fragments. In such cases there may be a justification for regarding all such sites as the unit, and allocating management types across all of them, independently of geography.

4. **Topographic position** Linkage may come about through common location in a valley or other hydrological catchment – plant and animal dispersal often occurs within these systems, particularly along watercourses. Woods at similar altitudes may have specific adaptations, which need to be considered.

5. **Physical separation** Barriers such as mountain ranges and large, unwooded areas are likely to prevent ecological linkage, and so make effective boundaries between EWUs. A preliminary analysis shows that a sharp cut-off occurs at about 300 m, with very few semi-natural woodlands found above this altitude in Wales.

6. **Management history** Woodland in some areas has had a common tradition of management to which the composition of flora and fauna may be adapted, and might form the basis for EWUs.

There is a further question of scale. EWUs must be small enough to ensure that woodlands within them are truly ecologically linked, but large enough to be practical on an administrative scale. Possibly, a nested system will arise, with a coarse structure for broad recommendations, within which more detailed units are identified as appropriate.

Figure 10.1 shows an example of a GIS analysis of these datasets, which divides Wales into 23 units (descriptions of each are given in Latham, 2000). Various analyses have been carried out with different weighting given to particular parameters. Certain EWUs regularly appear in different analyses and seem to be relatively insensitive to variations in the emphasis placed on the parameters. These are likely to have an ecological reality and to be a firm basis for considering management.

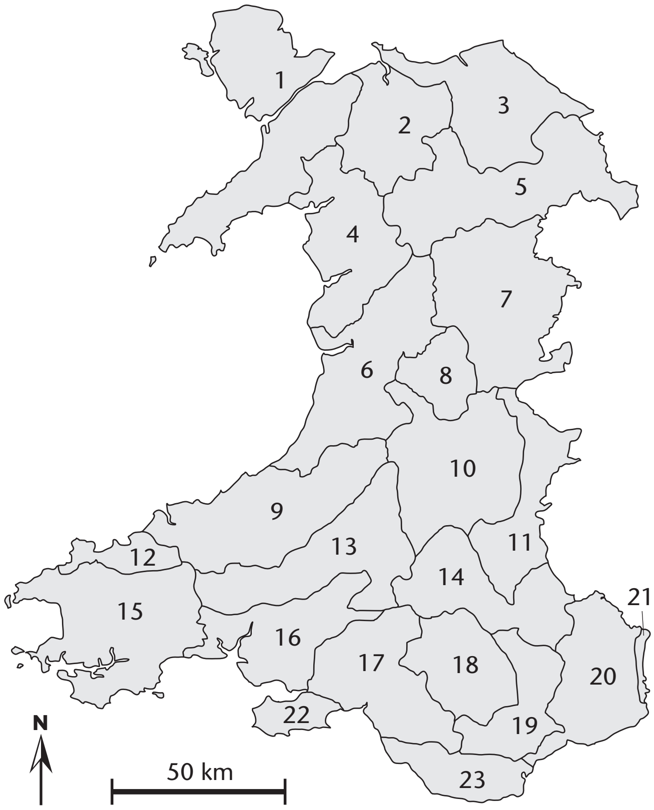

Figure 10.1

An example of an Ecological Woodland Unit (EWU) classification of Wales. The 23 units are based primarily on river catchments and separation by ground over 300 m. Secondarily, divisions have been added which correspond to obvious floristic and geological discontinuities.

Relative proportion of structural types

At the simplest level, the framework should indicate which structural states are absent, or severely over- or under-represented in each EWU. However, with further work it may be possible to develop some broad guidelines of suitable proportions. These can never be definitive, as the relative suitability of structural states will vary geographically, with management tradition, and with the types of woodland involved. Additionally, the balance of emphasis between production and nature conservation interests will vary between SSSIs and non-statutory sites. Table 10.1 suggests the relative emphasis that should be given to each structural state in different woodland types (HAP types, after Hall and Kirby, 1998).

Location of structural states

The location of structural states within an EWU may be important. The value of structural states often increases with size, and the framework gives the chance to promote linkage of management across woods to achieve an effective larger size. This is especially important for Natural Woodland, as natural processes may take place at a landscape scale. It is also important for High Forest, as the production from a series of linked woodlands may become economically viable. This issue is addressed by Edwards and Kirby (1998). Natural Woodland should also be present in a range of representative locations. It is often selected passively when inaccessibility precludes other management. However, Natural Woodland in more accessible locations (perhaps with deeper soils and more sheltered conditions) may develop structures and associated biodiversity, which are currently rare. The framework allows the current locations of structural types and their representivity to be assessed. For

Table 10.1 *A suggestion of the relative emphasis to be given to different structural states across a range of woodland types (the more spots, the greater the emphasis) for protected sites (SSSIs). The strong emphasis on Natural Woodland would decrease in non-statutory sites.*

Structural state	Woodland type (UK Habitat Action Plan category)				
	Upland Oakwood	Upland Mixed Ashwood	Wet Woodland	Lowland Beech and Yew Woodland	Lowland Mixed Deciduous Woodland
Natural Woodland	••••	••••	••••	••••	••••
High Forest	•••	•••	•	•••	•••
Coppice	•	•	••	•	•••
Wood Pasture	••••	•••	•••	•••	•••

coppiced woodland, for example, which through its dynamic nature may contain ephemeral populations of plants and animals, clusters or linked areas are likely to be more valuable than dispersed and isolated areas.

Discussion

Future development

The framework project has three broad developmental stages. The first is the development of the framework concept, consultation and analysis to identify the EWUs. At the time of writing, this stage is well advanced, and progress is described in Latham (2000).

The second stage will be a practical application of the framework to the example EWUs. A similar approach has already been employed in the Lower Wye Valley (EWU Number 21 in Figure 10.1: Hellawell, 1999), and has been suggested for southern Snowdonia (EWU Number 4) by Oliver (1993). These EWUs are obvious starting points and an iterative process of data collection, interpretation and recommendation is envisaged. Without examples like these, it is not possible to assess how well structural types really are represented, and hence how valuable a fully developed managed framework might be. Ideas of suitable proportions for each structural type are likely to develop as information becomes available on the current proportions and locations of management. The framework will be first applied to SSSIs. CCW has reasonable influence on the management of these sites and a relatively large amount is known about their ecology and special interest. If the process is shown to work for the SSSI series, it provides a solid foundation for extending the framework to include woodland in the wider countryside. An operational structure will be needed to administer the framework, and it is likely that using real examples will help this to evolve.

The third stage is the application of the framework to woodland throughout Wales. Again, the SSSI series provides a useful starting point, but woodland in the wider countryside needs to be accommodated. This could be achieved through any management programme that involves woodland. Tir Gofal – the Welsh agri-environment scheme – could influence the management of a very large woodland area, and it is important that the framework integrates with it. Similarly, linkage to management carried out with Forestry Commission grants should be explored.

Wider applications

Although the framework is being developed primarily to guide management, it may provide a sound ecological structure with wider applications, for example for directing woodland expansion and restoration, and it has relevance to woodland network projects (e.g. Hampson and Peterken, 1998; Good *et al.*, 2000). Areas within EWUs could be suggested for new woodland, so that existing woodlands have increased areas or are ecologically linked; restoration effort may be targeted in the same way. Suitable future management of new woodland could be identified, and this may influence their location, design, decisions on planting versus natural regeneration, and choice of any planted species. The EWUs may provide a basic structure for other woodland conservation programmes, for example, as selection areas for SSSIs. This function is currently carried out by areas of search (Blackstock *et al.*, 1996), which, while providing a functional breakdown of Wales are based on administrative and not ecological principles. Site selection and evaluation on an EWU basis may give a more thorough representation of woodland types in protected sites. A similar function is carried out by English Nature's Natural Areas and Scottish Natural Heritage's Natural Heritage Zones.

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CHAPTER 11

Woodland restoration and forest planning in Forest Enterprise

Wilma Harper

Summary

This chapter describes how the restoration of native woodlands can contribute to the objectives of Forest Enterprise. It explains how the planning structures which operate at the landscape and regional levels form a framework for deciding how best to deliver a restoration programme. Questions relating to the monitoring of restoration are also discussed.

How does woodland restoration contribute to delivering Forest Enterprise's objectives?

Forest Enterprise (FE) is an executive agency of the Forestry Commission, charged with managing the public forest estate in Great Britain. Like all agencies of Government departments, it has a Framework Document (Anon.,1996), which defines its remit, objectives and performance measures. The current Framework Document was produced in 1996 and the 5-yearly review of the Agency is currently under way. Box 11.1 lists the FE objectives*.

Box 11.1 *Forest Enterprise: Objectives 1996.*

Financial

- To maximise financial returns on the assets of the estate through wood production and the exploitation of commercial opportunities using private capital wherever appropriate.

Providing environmental, social and other outputs

- To develop the recreational and educational potential of the estate.
- To take the action needed to facilitate access by the public on foot, extending it as widely as is consistent with the safety of users and with the Commissioners' legal obligations.
- To enhance the environmental conservation and amenity value of the estate including biodiversity and landscape and to seek and realise opportunities to further the Government's environmental policies.
- To conserve and manage sympathetically areas of special natural and heritage interest.

An efficient service

- To be efficient, cost effective and businesslike in all its operations.
- To set and achieve the standards of service set out in its Citizen's Charter Standards Statement.

Two of these objectives are particularly relevant to restoration of woodland landscapes. FE is charged with 'enhancing the environmental conservation and amenity value of the estate'. In addition, it has to actively 'seek and realise opportunities to further the Government's environmental policies', for

*The Quinquennial Review of Forest Enterprise, which set revised aims and objectives, was published in July 2001. See www.forestry.gov.uk/forestry/hcou-4wee7x example, delivering the UK Biodiversity Action Plan. In addition, FE must 'conserve and manage sympathetically areas of special natural and heritage interest'.

The 1996 Framework Document sets FE within the context of sustainable forest management (SFM). In the past 5 years, the commitment to sustainable development has, if anything, grown in importance for Government, its departments and agencies. Sustainable forest management combines economic, social and environmental objectives which can be depicted as three interlocking circles (Rollinson, Chapter 1). Figure 11.1 shows this model applied to woodland restoration.

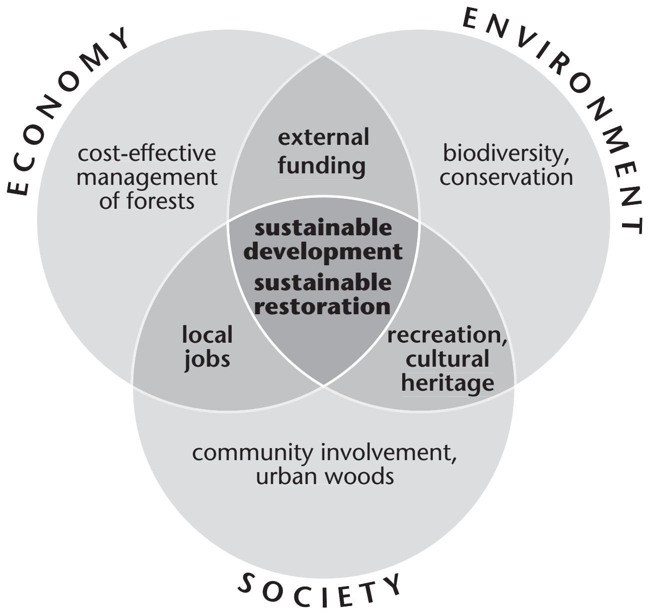


Figure 11.1

How woodland restoration relates to the three facets of sustainable forest management.

Woodland restoration makes a major contribution to increasing the environmental value of the forest, by conserving and enhancing existing habitats and adding to the biodiversity of the forest as a whole. But native woodlands in Britain, and the landscape itself, are as much about the cultural heritage and lives of the people of these areas. Restoration needs to be sensitive to this heritage. Some landscapes may be much more recent and it may not be appropriate, for example, to try to recreate the 'wildwood' in an 18th century designed landscape. Restoration can put archaeological sites back in context, for example making the link between charcoal, iron bloomeries and oakwoods. Restoring the Sunart oakwoods, near Ardgour on the north-west coast of Scotland, has enhanced the conservation value of one of the most prized habitats in Europe. It has also been the catalyst to rediscover the history of the area, from archaeological sites, Gaelic place names and recording the more recent history from recollections of local people. Restored woodlands have a role in providing opportunities for recreation and educational use, so long as the site is robust enough or access can be channelled away from the most sensitive areas.

Woodland restoration may often be seen as an uneconomic activity. Conversion of plantations or removal of underplanted conifers is expensive and many schemes only go ahead with the assistance of external funding. For example, the Millennium Forest for Scotland initially funded the work at Sunart but since 1996/97 funding has come from EU LIFE-Nature. (For more details of this programme see europa.eu.int/comm/life/nature/.) This has allowed neighbouring landowners to become involved in restoration and assist with a wider range of work. Such funds provide an incentive for managers to undertake significant capital expenditure to kick-start the conversion regime in their woods. In addition, in the west Highlands of Scotland, the pinewood and oakwood restoration programmes have encouraged the development of partnerships with other bodies and have underpinned training packages to encourage local people to acquire the necessary skills.

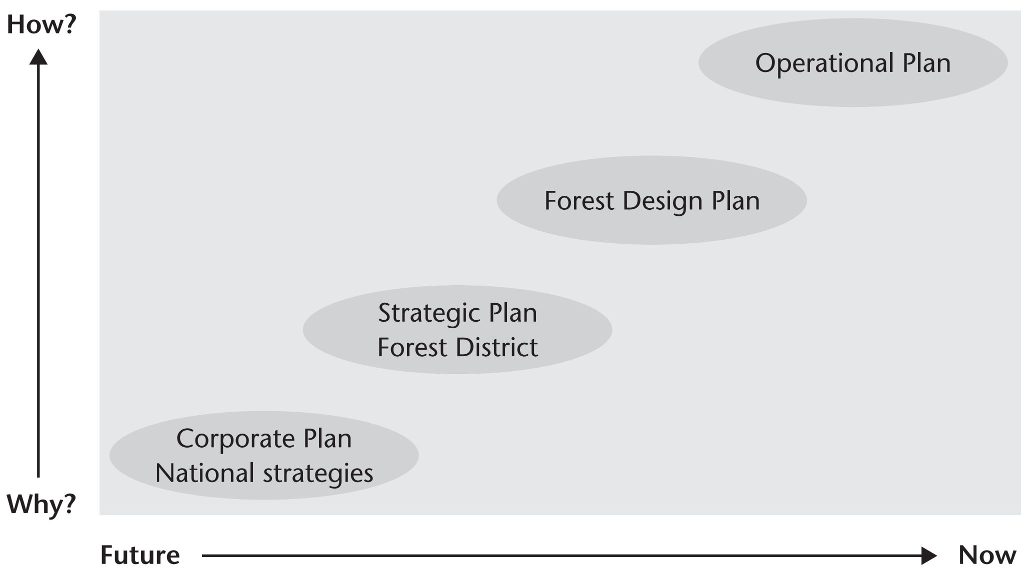
In recent years, most new native woodlands on open ground have been created with the help of grant-aid through the Woodland Grant Scheme and Forest Enterprise has done very little new planting. FE has concentrated its new planting activity in the urban fringe, helping to address the regeneration of these areas. Over the next 2 years, this will expand rapidly with funding from the Government's Capital Modernisation Fund. The priority is rapid transformation of the landscape to improve the quality of life for the local people. The choice of species may depend on the severity of the site, and often these are landfill sites and old coal spoil heaps, but most planting designs include an element of native woodlands.

How do we use our planning structures to deliver these objectives?

Forest Enterprise manages more than 1 million hectares of land throughout Great Britain. This is a very diverse estate, with sand dunes and peat bogs, limestone pavement and forests on old coal workings. It also includes 387 Sites of Special Scientific Interest (SSSI) covering 62 000 ha. There is around 200 000 ha of ancient woodland of which about a third is classed as both ancient and semi-natural. Not only are the sites diverse, we also have a wide and potentially conflicting set of objectives (Box 11.1). To manage this, a structured approach to planning has evolved to create a framework in which managers can address questions at the appropriate level. At the strategic level, we need to consider how to deliver sustainable forest management, how to balance the objectives, and how to 'balance the books'. Operationally, there are the specific questions of what do we do, and how, where and when do we do it?

Essentially there are four levels of plan as outlined in Figure 11.2. The highest level plans are mainly about *why* we do things, with more detail about *how* things will be done in the operational plans. In broad terms, operational plans are about *now,* and strategic and corporate plans about the *future*. This hierarchical planning structure does not necessarily correspond with any organisational hierarchy as staff throughout the organisation will have an interest and input into parts of all plans.

Figure 11.2 *Planning framework for Forest Enterprise.*



Corporate Plan

Forest Enterprise has a Corporate Plan and produces a report of progress annually. Since the publication of the Corporate Plan in 1996, responsibility for forestry has devolved to England, Scotland and Wales. The three countries have national strategies for forestry or are in the process of producing them. (For the latest versions see: www.forestry.gov.uk.) There is a very important role for FE, as the manager of the people's forests, in delivering the aspirations set out in these strategies.

Strategic Plans

Each of the 32 Forest Districts has a Strategic Plan. These were introduced in 1999 to provide a means of relating the high level objectives to local conditions and priorities. The Strategic Plan sets out the key policies for the district and gives an overview for staff and the wider community. It brings together plans on particular subjects such as conservation or thinning, either by including the relevant material, or by establishing the link to a more detailed plan, e.g. for managing an SSSI. The Strategic Plan also allows policies which apply widely to be defined and thus reduces repetition in other plans.

The Strategic Plans help Forest Districts to define priorities for woodland restoration. They identify areas of existing conservation value, and set out the criteria for deciding where there will be the greatest benefit from restoration or other work. The Strategic Plans also address the key management issues and the Forest District's approach to these. Strategic Plans are usually the most appropriate level at which to consider how to deal with balancing multiple objectives. Are the objectives complementary or conflicting? One way of approaching these issues is to consider a simple, four box diagram as shown in Figure 11.3.

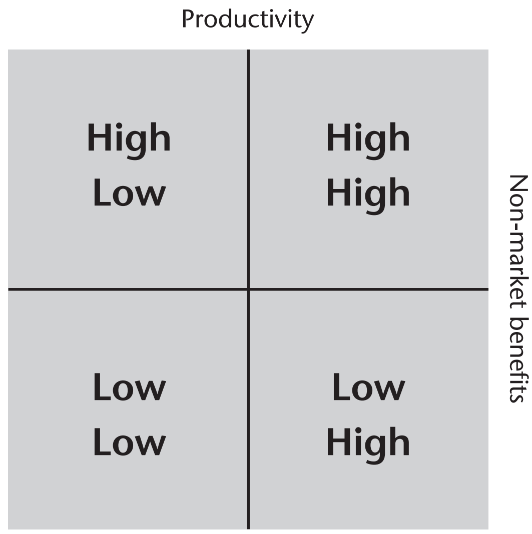


Figure 11.3

A simplified way of looking at the relationship between productivity and non-market benefits.

In this example we have some measure of productivity as the *y*-axis, such as yield class or discounted revenue per hectare. The *x*-axis represents the non-market benefits – the social and environmental products of the forest, which have no direct monetary value, but are no less real. It is then possible to place the diversity of woodland types within a Forest District, which may have been grouped together as zones in the Strategic Plan, in their relative positions on this diagram. When looking at priorities for woodland restoration for example, this might show native pinewoods as being of high conservation value but low productive potential and so in the lower right quadrant. The preferred areas for pinewood expansion might be those with a low/low rating such as poor lodgepole pine or Sitka spruce in check on heather.

Some areas of plantations on ancient woodland sites (PAWS) may be highly productive but also have the potential to be of high conservation value if restored. Here a balance has to be struck between foregoing revenue from timber and not realising the non-market value of the site. There is no 'right answer' in these cases, nor a magic figure of what is an acceptable loss. In the end, it is the public accountability of FE which helps guide where the balance will lie. In this we are no different to other countries. The US Forest Service tried a range of mathematical solutions but these lacked transparency and have now been largely replaced by a political process (Sedjo, 1999).

The Strategic Plans give the opportunity to set out the issues and explain potential difficulties. The areas of genuine conflict may turn out to be quite small. It may be possible to concentrate restoration work in the buffer zone round an existing native woodland – planning in space – or plan for restoration to take place when the current trees reach maturity and are felled and can be replaced by native species – planning in time.

Forest Design Plans

Forest Design Plans are crucial to the management of the forest. They set out how the existing forest is to be managed, especially the shape and timing of felling coupes. By describing the proposed replanting, they give a vision of the future forest, as it will evolve over the next 50 to 100 years. For Forest Enterprise, they are the main way in which our activities are given formal approval by the Forestry Commission. Over 70% of the estate has an approved plan. Plans will be drawn up for the remaining areas as they reach the age when significant felling is proposed.

The starting point for a Forest Design Plan is the brief. This comes from the Strategic Plan and how it relates to this particular area. The importance of the area to the local community and other stakeholders will also be considered when drawing up the brief. Each Forest Design Plan unit has a 'sensitivity score' marking, on a scale of 1 to 5, how important it is for conservation, landscape and people. The higher the score, the greater the importance given to that factor in the design brief. In addition, the volume and revenue from the trees can be modelled using *Forester,* Forest Enterprise's custom GIS (Coppock, 2000; Ditchburn, 2000).

The planning process starts with 'analysis', which identifies the key features of the area, both the strengths and weaknesses. This is matched with the brief to draw up a 'design concept', which gives the broad outline of how we see the forest developing in the future. Often this design concept is used in consulting with stakeholders to ensure that there is a consensus of agreement before more detailed work is done. The felling and restocking maps translate the concept into a more detailed framework for the proposed work in the area, and, in due course, become the basis for operational plans.

Forest Design Plans help in the management of woodland restoration by setting out the permanent structure of the forest such as existing native woodlands, areas along watercourses to be planted with native species, or areas where natural regeneration will be encouraged. The felling plan defines the rate of change in the forest. For some restoration work, a rapid change may be desirable, for example to prevent further shading out of woodland groundflora; in other circumstances, where key species are less robust, the adoption of a continuous cover management system, where small numbers of trees are felled, may favour shade tolerant species. The felling plan may also be key in defining the management of other species, for example making sure that invasive species are cut back from key sites, or felled before reaching an age when prolific seeding might occur. The restocking plan sets the vision for the next generation. It can include plans to plant appropriate native species. These may be set as indicative plans, possibly related to NVC type (Rodwell and Patterson,1994), to be given more detail when the current trees are felled and the underlying site conditions are easier to identify.

Monitoring of woodland restoration

Monitoring and evaluation are an essential part of the planning cycle. The planning process is a way of addressing some simple questions:

- What have we got?
- Where are we going?
- How do we get there?
- How do we know when we have arrived?
- How do we respond to change?

As for many simple questions, the answers can be complex. With woodland restoration work, some of the difficulties arise because of the uncertainty about defining the basic resource. We need clear definitions, which can be interpreted in the field or from other sources such as aerial photographs. We may have to accept a workable but imperfect definition today while waiting for a more exact definition at some time in the future. An important principle, which the development of GIS has highlighted, is to separate *description*, which can be verified by anyone who can identify the key features, from *prescription* which needs an awareness of the intentions of the manager. Thus an area may be described as Sitka spruce in mixture with naturally regenerating birch. The prescription will depend on the objectives of management. A failing conifer crop will be handled rather differently to a developing native woodland which is suitable for restoration. Reassessing the description of the area in 10 years time will help measure how successful we have been in meeting the objectives.

The science of woodland restoration may be poorly developed but managers on the ground are having to make decisions on what to do with their woods. Even doing nothing is a decision, and what happens to the woodland will depend on grazing levels and the nature of adjacent stands. There is a need to continue co-operation between research and practice to get the best current advice in to the hands of the managers.

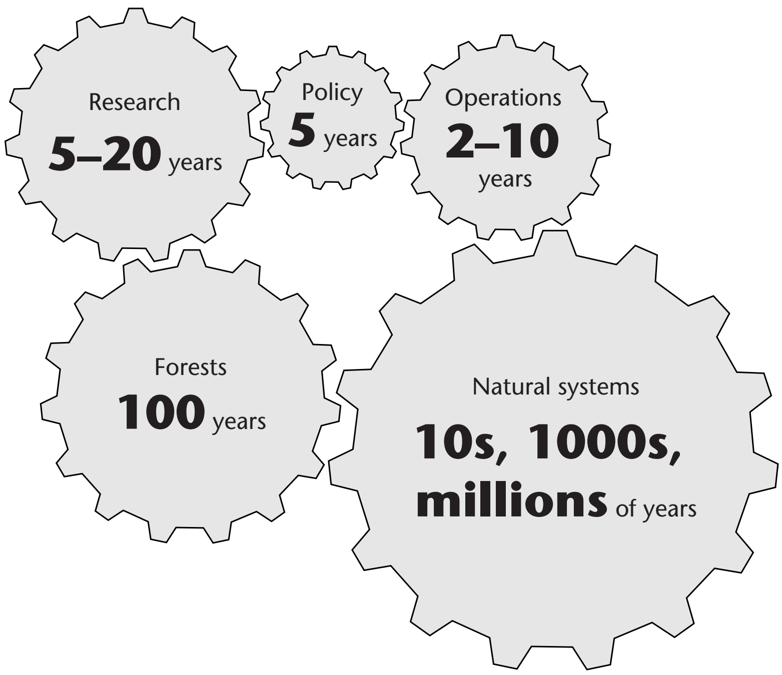
The Living Dance

Recently I spotted a book in the University of British Columbia bookshop, Vancouver, was a book with the title '*Policy and practices for biodiversity in managed forests*' on the spine. Interesting, but what was really appealing was the subtitle on the front cover – '*The living dance*'. The book was the proceedings of a conference in 1994 (Bunnell, 1998) and the introduction by Fred Bunnell began as follows:

> 'We tend to view most events occurring in and around a forest as moving slowly, even quietly. The truth is different. Forest are parts of a living dance of constantly changing steps, many rapid. Dance is an appropriate metaphor. Like medicine, forestry is as much an art as a science. Among our art forms, painting is two-dimensional, sculpture three-dimensional, and dance four dimensional, through its changes in rhythm and frequencies.'

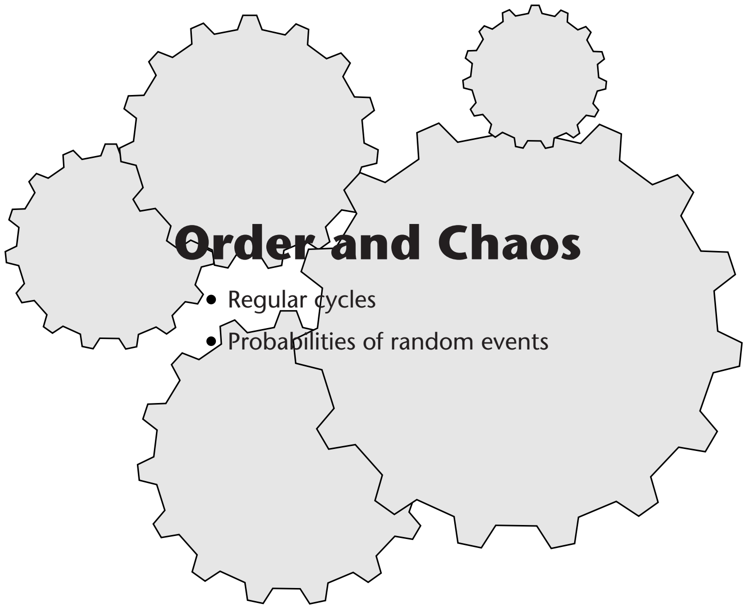
The rhythms of the dance operate at different rates (Figure 11.4). The time needed to get meaningful results from research may not sit comfortably with the changes in policy or the capacity to bring these changes about in the forests themselves. But the simple cyclic model ignores the shocks and catastrophes that are inherent in natural systems (Figure 11.5).

Figure 11.4 *The Living Dance – cycles of activity operating at different timescales (after Maini, 1998).*



To take an example from Lady Park Wood in the Wye Valley – perhaps the nearest to an undisturbed native wood we have in Britain (Peterken and Mountford, 1996) – a severe drought in the mid 1970s killed many large beech trees and stopped the growth of others over a prolonged period. This in turn led to gaps in the canopy and a change in the species composition of the stands. Single events can cause significant changes – a late spring snowfall in 1984 killed young ash and flattened the young leafy shoots of the understorey lime which are now layering and regrowing. Peterken and Mountford

Figure 11.5 *The Living Dance (2) – other events occur at different times and disrupt the system.*



note that: 'Since the disturbances are essentially irregular and unpredictable, the natural state of the wood can be stated only as a set of probabilities.' Because Lady Park Wood has been studied over a long time period, we can relate what we see now to events many years ago. It also serves to warn us of the hazards of generalising from limited observations. Even where we have good information, the best we can hope to do is work out a probability of an event. The ForestGALES model (Gardiner and Quine, 2000) allows us to predict the probability of a stand being subject to windblow; it does not allow us to say which stand will blow down and when. We can predict a likely NVC type for a regenerating native woodland (Gray and Stone, Chapter 7), but what we will get, and when, is less certain. To return to the 'living dance' analogy, only someone with a deep understanding and familiarity with the ballet would be able to look at a single bar from the score of Swan Lake and describe what is happening on stage at that point.

In managing the forest to restore native woodlands, forest planning provides a framework for the management of change. The changes may be in the nature of the resource itself, the climate, the policy environment, or the best practice arising from research. It is the job of the forester to try to keep this ramshackle machine on some sort of defined path.

To quote Bunnell (1998) again:

> 'The dance will continue with or without our participation, but policy-makers, practitioners, and researchers are usually partners in it. We participate to enjoy, conserve, and shepherd the value we desire. To do this well, we must somehow try to match our rhythms with those of the ongoing dance.'

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CHAPTER 12

The National Trust for Scotland approach to woodland restoration

James Fenton and Chris York

Summary

The National Trust for Scotland has been heavily involved in woodland restoration throughout upland Scotland in recent years. This chapter summarises the major projects and also the Trust's policy framework within which such work is undertaken. A review of the planning aspects of these restoration projects has highlighted that:

1. Restoration schemes at the landscape scale, and the cumulative effect of many small schemes, result in major landscape changes that will be with us for a long time: caution is needed, and if we have 'grand visions' we need to be sure that they are grounded in ecological reality.

2. Woodlands are just one component of large properties and strategic planning is essential. A clear rationale should be developed as part of the project plan, and agreed by all parties prior to proceeding.

3. Research or survey work should be done before defining objectives or submitting grant applications. It should be recognised that our knowledge of the long-term vegetation dynamics of much of the Scottish uplands is poor.

4. Projects can develop their own momentum and can change direction by degrees, but this can result in the eventual project being different in character from that planned at the outset. Reviews of progress are needed to keep people informed of changes, and if it emerges that the eventual project is likely to be significantly different from the initial plan, then the option of abandonment should be considered.

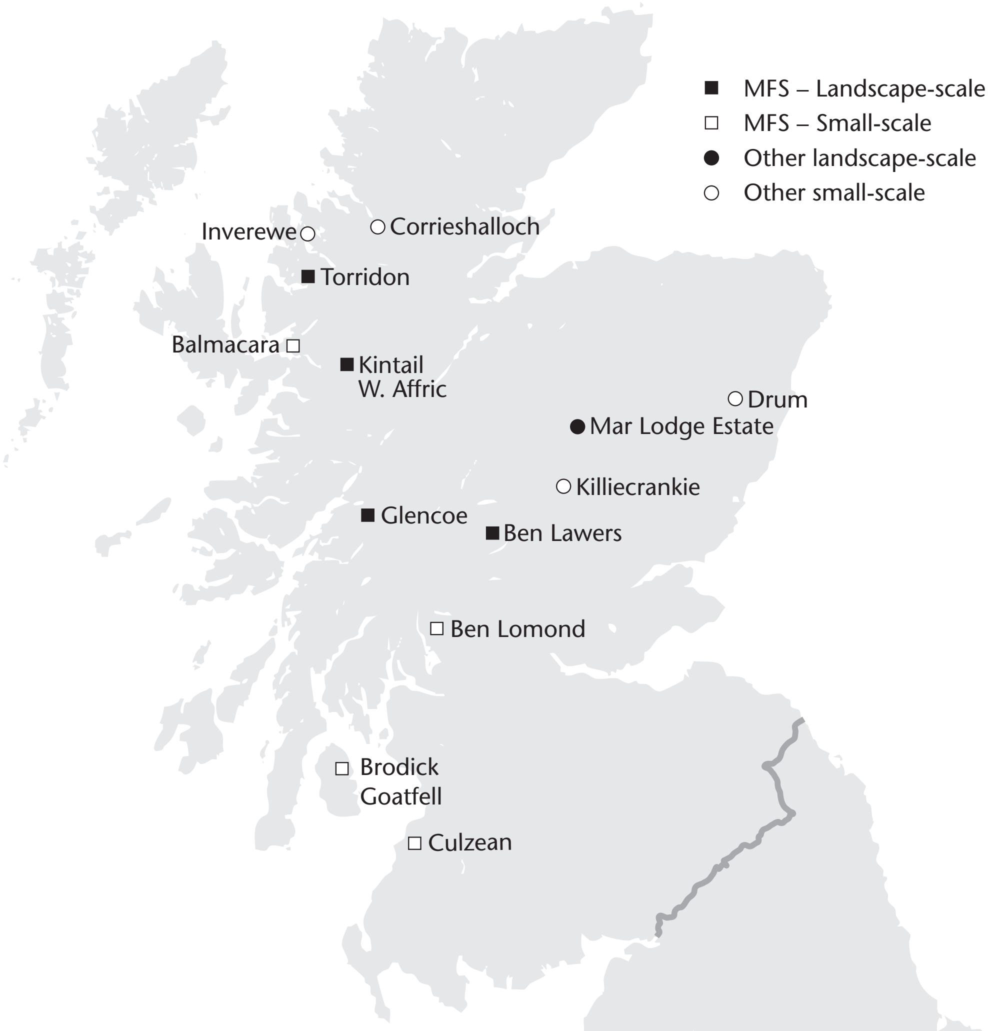
Recent initiatives and opportunities on NTS Land

In recent years, The National Trust for Scotland (NTS) has been heavily involved in woodland restoration projects, and many have been funded by the Millennium Forest for Scotland (MFS). Although most of the recent schemes have involved natural regeneration, there have been planting schemes at Glencoe, Kintail, West Affric and Torridon. At Ben Lawers a major project is under way to restore montane willow scrub, planting being necessary as there is very limited seed-source.

Projects have been wide-ranging and have included:

- Regeneration of existing native woodland fragments at Goatfell, Ben Lomond, Glencoe, Kintail, West Affric, Balmacara and Torridon.
- Major native pinewood restoration at Mar Lodge, through natural regeneration (deer control).
- Creation of new native woods by planting at Glencoe, Kintail/West Affric and Torridon.
- Montane willow scrub restoration at Ben Lawers (large-scale) and Glencoe (small-scale).
- *Rhododendron ponticum* control at Brodick and Torridon.
- Restructuring of existing conifer plantations to native-type at Mar Lodge, Ben Lawers, Glencoe, and Torridon.
- Crofter-forestry schemes at Kintail, Balmacara and Torridon.
- Research into the post-glacial woodland history of West Affric.

Figure 12.1 *Sites of recent NTS woodland restoration work.*



The NTS policy context

The National Trust for Scotland was set up by an Act of Parliament, and an extract from its 1935 Confirmation Order is given below [italics added for emphasis]:

> 'The NTS shall be established for the purposes of promoting the permanent preservation for the benefit of the nation, of lands and buildings in Scotland of historic or national interest or natural beauty…and as regards lands for *the preservation* (so far as practicable) *of their natural aspects and features* and animals and plant life'

Management of Trust properties is guided by management plans, which are based on identifying the *key features* of properties. Key features are classified under the following headings:

1. Natural Heritage
2. Cultural Heritage
3. The Landscape
4. The Visitor
5. Social and Economic Context.

Additionally, each plan now builds on a *Statement of Significance*, which encapsulates the importance of a given property. A clear rationale is developed to ensure that all management actions accord with protecting and enhancing the identified key features and take account of the significance of the property.

There are various policies, agreed by the Trust's governing council, that guide its property management, and extracts from certain relevant ones are given below.

From *Grazing Working Party Final Report (1993):*
In future, management plans should state the desired objectives of conservation management, and suggest the appropriate grazing level at the property. Plans should also identify scope for habitat restoration.

From *Woodlands: Policies for Management (1994):*
Woodland management objectives reflect the primary Trust aims of the conservation of nature and landscape:

- Natural regeneration should be the restocking method where it is practicable in woods in the wider countryside.
- Where new woods are being created … then the choice and proportion of trees should be based on the appropriate NVC woodland type, augmented where possible by detailed research into the site's vegetation history.
- In extensive upland properties, reduction in grazing pressure should always be the first option to be considered in woodland regeneration schemes; fencing should only be used where there is no alternative.

From *Deer Management Policy (1997):*
The Trust will undertake a full assessment of each relevant Trust property to determine the status of the habitat, the required grazing regime and the culling levels required to reach the aims and objectives for flora and fauna, and historic/archaeological remains, which will be specified in the Property Management Plans.

From *Crofting Working Group Report (1998):*
The Trust should support crofter forestry schemes which are broadly in line with the Trust's woodland policy.

Experience gained from recent projects

A review of recent projects undertaken by the Trust has identified the following issues:

1. Landscape planning in nationally important sites is a complex process, and these sites need a strategic approach. Opportunistic schemes, while enabling much needed work to go ahead, can have tight deadlines, and may not be taken into account in the existing management plan. Hence, they can bypass the full management planning process, and the next management plan has to provide a *post hoc* rationalisation.

2. The rationale behind woodland restoration projects is sometimes fragile, and is rarely clearly articulated; the schemes are often not based on holistic evaluations of the key features of the sites, e.g. the impact of planted woods on the wild-land quality of open, semi-natural landscapes has often not been considered.

3. The timescale of restoration projects can relate to human aspirations rather than ecological processes. Projects have to be seen to achieve outcomes – funding deadlines make it difficult to proceed with caution. This can lead to planting where natural regeneration may arguably be as effective in the long-term.

4. Additionally, if grant-aid is sought, then the eventual conditions attached can result in a different scheme to that originally planned: thus, other parties have 'taken over' the Trust's agenda!

5. Income can be derived from action, so there is a tendency towards doing as much as possible; rewards are often a percentage of total expenditure, not quality- or outcome-based. This can conflict with a policy of 'minimal intervention'.

6. Large schemes have facilitated research, survey and monitoring. However, the timing has not always allowed informed decisions to be made, e.g. in West Affric a palaeo-environmental survey was funded to guide woodland restoration; however, to fit in with deadlines, tree planting has had to take place prior to an analysis of the ecological history produced by the survey.

7. Evaluation of the different semi-natural vegetation types can be difficult. Who is to say that woodland is more important than moorland? Can we be certain in many upland habitats that woodlands, under the current climate, would be the 'climax' vegetation in need of 'restoration'? Or could open moorland be more 'natural' than most people think? This applies particularly to wet heath, where the Trust has encountered conflicting advice as to whether this should be wooded or open.

8. 'Grand visions' of woodland restoration should be treated with caution (see note below), as they may end up conflicting with ecological reality on the ground, or our visions may have to change as our knowledge of woodland history in Scotland increases.

Avoiding future problems – the benefits of hindsight

Hindsight has indicated that future woodland restoration projects would benefit if the following learning points from projects at NTS properties were adhered to:

1. Woodlands are just one aspect of large properties and strategic planning is essential (see ICOMOS, 2000). Projects should be based on the *Statement of Significance* for a property and take account of potential impacts on other key features.

2. Projects should relate to the objectives for that property, rather than be determined in character or size by the amount of funding available.

3. A clear rationale should be developed as part of the protect plan, and agreed by all parties prior to proceeding.

4. Research or survey work should be done before defining objectives, submitting grant applications or signing contracts. It should be recognised that our knowledge of the long-term vegetation dynamics of much of the Scottish uplands is rudimentary. A problem is that, while the availability of funding provides a mechanism for research that may otherwise be difficult to undertake, funders as well as applicants need to be aware that the research should inform projects, which may subsequently need to be altered in light of the findings.

5. Projects can develop their own momentum and, as a result of suggested modifications, can change direction by degrees. However, the cumulative effect of these many small changes can result in the eventual project being different in character from that planned at the outset. Reviews of progress are needed to keep people informed of changes, otherwise those involved at the outset may be concerned that the eventual project is not the one they originally signed up to. Projects that are found to be ill-conceived, or which end up differing significantly from the initial terms of reference, should be abandoned; reasons for abandonment should be documented for future reference.

6. Small-scale natural regeneration schemes around existing woodland fragments are generally less contentious than the planting of trees on unwooded open moorland, although if natural regeneration is to be achieved by a major reduction in grazing animals, then contention can arise. Restoration schemes at the landscape scale, and the cumulative effect of many small

schemes, result in major landscape changes that will be with us for a long time: caution is needed, and if we have 'grand visions' we need to be sure that they are grounded in ecological reality.

Note on 'grand visions'

'Visions' do have their place in nature conservation to guide management direction, but perhaps they should be flexible and not be too prescriptive – allowing for the vicissitudes of nature! However, in his book *The Open Society and its enemies*, Karl Popper argues that centralised planning of society, including utopias and grand visions (such as advocated by Plato or Karl Marx), is inherently dangerous and anti-democratic: reality is not allowed to intervene in the grand plan. The alternative approach, a piecemeal one of tackling problems as they arise, albeit in pursuit of given aims, 'alone makes it possible to apply the method of trial and error to our political actions… it alone allows us to find out, by experience and analysis, what we actually were doing when we intervened with a certain aim in mind' (Popper, 1945). It is suggested here that the above arguments apply at least in part to some of our 'grand visions' of woodland restoration in the Scottish Highlands. The woodlands have had a long and complex history, much of which has still to be discovered: it is not a straightforward theory – i.e. 'humans destroyed the forest, we have a duty to put it back.'

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CHAPTER 14

Geltsdale, Cumbria: restoring wood pasture at the landscape scale

Iris Glimmerveen

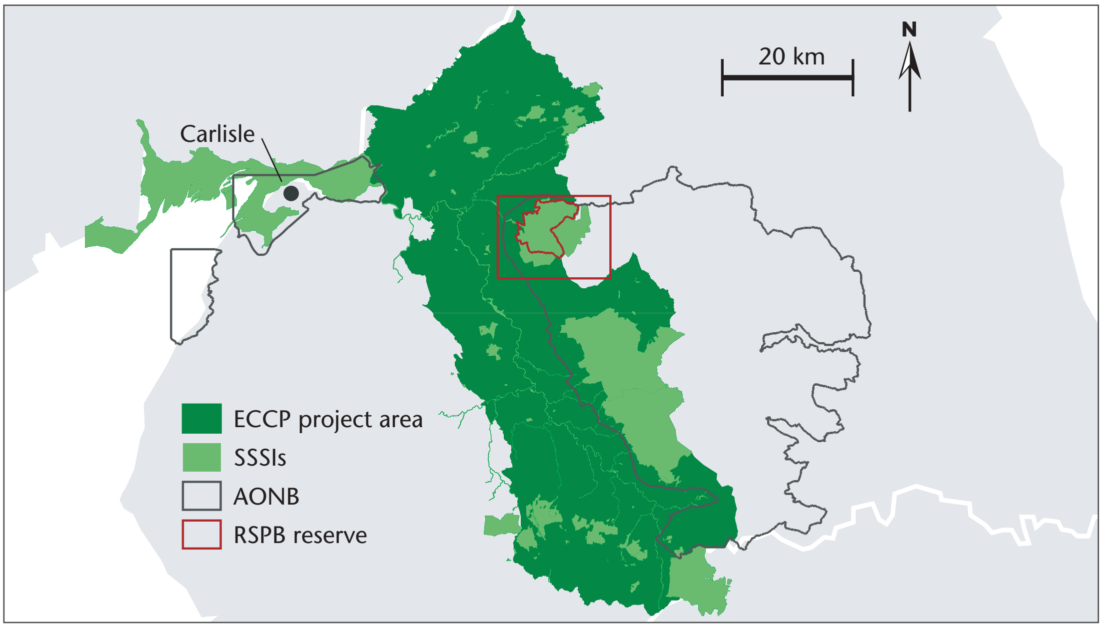
Summary

This chapter looks at the opportunity to restore wood pasture over a significant part of the Geltsdale Fells in the northern Pennines. It discusses the wide ranging landscape and its diversity of habitats which support a large number of bird species. Surveys are required to understand the historic and ecological features of the site before any management prescriptions can be drawn up. Habitat network principles will need to be applied to take other important semi-natural habitats into account, and the incompatibility of high levels of grazing with tree establishment will also have to be overcome. Funding upland wood pasture establishment and management is very difficult because neither the Forestry Commission nor the Department of the Environment, Food and Rural Affairs grants cater directly for this woodland type. East Cumbria Countryside Project has therefore put together a partnership bid to the Heritage Lottery Fund.

Introduction

Geltsdale Fells (5 516 ha) lie at the north end of the Pennines about 20 km due east of Carlisle (Figure14.1). The Geltsdale and Glendue Fells SSSI is within the East Cumbria Countryside Project (ECCP) area and forms part of the North Pennines Area of Outstanding Natural Beauty (AONB). It contains a sizeable Royal Society for the Protection of Birds (RSPB) reserve of 4 723 ha, supporting a large number of bird species, many of which breed, including waders, raptors and game birds. Since 1994 a black grouse population has re-established itself in the area. Preliminary investigation has shown Geltsdale to be a good example of northern wood pasture in an upland site and especially interesting because of its history as a medieval hunting forest. The woods, maintained by livestock

Figure 14.1 *Geltsdale in East Cumbria.*



grazing, have a remarkably natural species composition, containing hundreds of veteran trees, which contribute greatly to the historic character of the landscape. There are some important historic landscape features, as well as coppiced stands, pollards and old stools indicating long-term use. Like most valleys in northern England, Geltsdale was carved by water as the ice receded. The valley bottoms and slopes became colonised by woodland, while the tops of the hills became mostly moorland with a few trees. Man probably entered the valley 9 000 years ago (Anon., 1991). Over time the area was mined for lead, quarried for lime and farmed with cattle and sheep. On the slopes the woodland ebbed and flowed for hundreds of years in response to pressure from farming. This has resulted in heather cover on top of the hills, a mosaic of woodland and hay meadows in the valley bottoms and extensive upland wood pasture on the hill slope.

The current ground vegetation suggests that the wood pasture was once more extensive than it is today. Woodland indicator plants, such as wood sorrel *Oxalis acetosella* occur in several places on the hill slopes underneath bracken *Pteridium aquilinum*. Current maps and historical records refer to the area as the King's Forest of Geltsdale. This hunting forest is thus clearly a historic wooded landscape, much of which has been continuously grazed. 'Gelt' from the Irish 'geilt' means wild or mad (Anon., 1995), but whether that refers to the river, to the landscape as a whole, or to both is not clear. Forest reduction and expansion could be linked to climate, but also to man. For example, nomadic mesolithic hunters initiated the reduction in woodland cover, which was intensified after 3 000 BC by neolithic farmer-hunters (Anon., 1991).

Examples of historical forest use are still visible today, such as the remnants of lead mining buildings and a lime kiln. Forest expansion, however, could have occurred soon after 1750 (Anon., 1750), when the then owner, the Earl of Carlisle, stipulated in the terms of the lease to James Skaife that he: 'will not, nor at any time or times thereafter during the granted term, lopp, topp, fell, cut down, destroy or make away any of the woods, underwoods, timber or other trees now growing, or which hereafter shall grow, without the licence and consent of said Earl of Carlisle'. No doubt, an archaeological survey could shed more light onto this, but even so the woodlands themselves tell a story.

Upland wood pastures

Simmerson Gully is the most westerly part of the wood pasture sites. The valley bears the usual signs of overgrazing: a few remaining trees left in the gully with some straggly hawthorns clinging onto the hillside. East of Simmerson Gully are Knotts Wood and Binnie Banks, which are different because they have clearly been worked in the past. Both woodlands have some areas where land has been cleared cleared of stones with adjacent cairns, improved tree-lined boundaries and some specific boundary marker trees.

Knotts Wood appears to be older than Binnie Banks, principally because the veteran trees are taller with larger girths. There is the odd oak and ash, but the woodland is mostly dominated by alder and hazel. From visual inspection, they are estimated to be about 300 years old, but resources are unavailable to age the trees. Knotts Wood also contains hazel coppice which has been neglected for a long time. Some of the hazel now has a diameter at breast height (dbh) of 20 to 30 cm and although this in itself would not necessarily preclude it from being reworked, the astounding amount and variety of lichens on them would be lost if the hazel were to be recoppiced at this stage.

In contrast, Binnie Banks is dominated by birch veterans, estimated to be 200 years old. The trees are growing out of crevices in between the rocks and boulders, and it is suspected that because of the Earl of Carlisle's stipulation, the farmers and foresters stopped grazing or working the area. The ensuing conditions would probably have allowed the birch to regenerate and establish itself.

There are still signs of pollarding within the woodland however, and many of the hawthorn, ash and rowan appear to have been used for the odd timber pole and/or fodder – perhaps by the farmer just cutting a branch of timber as and when he needed it. Just outside the enclosure, to the east of Binnie Banks, is another unenclosed area with mostly alder trees. It does not have a specific name, but the name 'King's Wood' is proposed, because all the trees here seem very old indeed, making a truly majestic impression. Records from 1712 (Anon., 1712) refer to the presence even then of the large number of old trees on this site.

Veteran trees

Evidence that all the veteran trees in Geltsdale are very old abounds. There are:

- buttresses, basal swellings and burrs which are formed through many decades of grazing of epicormic buds and suckers at the base of the trees;
- old rowans growing out of the centre crook of the alders (known by ancient tree affectionados as 'rowan air trees');
- veterans with large diameters, the largest alder having a dbh of 2.65 m (circumference of 5.80 m)
- species of lichens on the trees that are indicators of antiquity and continuity of tree cover;
- alder coppice stools which are a metre or more in diameter.

Because Geltsdale's woods and wood pastures have not been managed intensively for a very long time, wildlife has flourished and hence there is a tremendous biodiversity. The main species include black grouse, otter, red squirrel, badger, adder, peregrine, lapwing and curlew.

Site management: now and in the future

Without any management input in the future the veteran trees will eventually die and without a respite from grazing, tree seedlings will not survive and the veterans will not be replaced. There are therefore two questions: how should this vast and exciting site be managed, and from where will the resources come to do this?

There is no doubt that the answer to the first question is in partnership. The site is too big and complicated for just one organisation to tackle. The answer to the second question is also likely to depend on partnership. Building a partnership was indeed the first step that led to the eventually successful implementation of the first phase (phase 1) management in Geltsdale. ECCP accomplished this by bringing together existing management ideas and resources for the site. The management objectives decided upon were basically to secure the continual existence of the wood pasture core area (phase 1), from which later the wood pastures could be expanded (phase 2). These ideas and resources were put forward by:

- the RSPB, who own the shooting rights and manage Geltsdale as a bird reserve, particularly for black grouse;
- English Nature: because of Geltsdale's SSSI status English Nature and a tenant have entered into a management agreement;
- the Forestry Commission, through two new planting grants and one woodland improvement grant through the Woodland Grant Scheme (WGS);
- the ECCP who have liaised with all these partners to maximise and realise the conservation potential for the site, while minimising the impact on the agricultural business of the farmers.

Phase 1

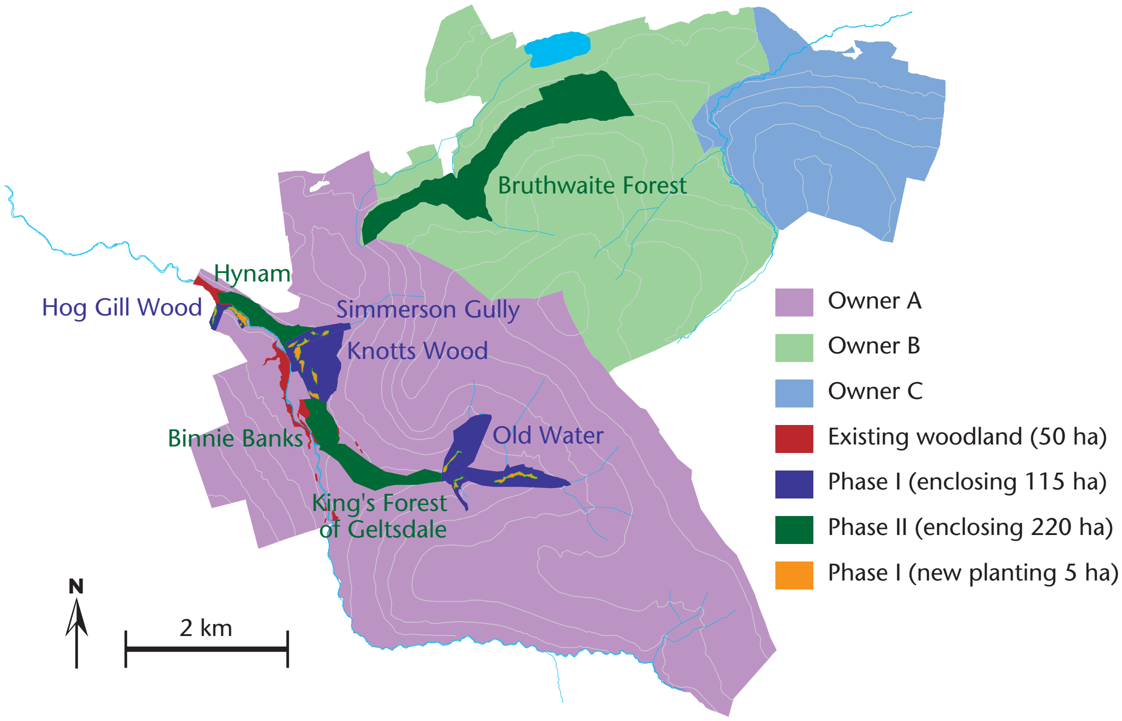
The first phase began in 1998, when 115 ha was excluded from grazing and some 5% of this was planted with locally native trees (Figure 14.2).

Trees were planted to enhance the link between the existing wood pasture, i.e. between Simmerson Gully and Knotts Wood and between Knotts Wood and Binnie Banks, using species suitable to the NVC communities found on site. A small area along Old Water was planted to provide a future seed source for natural regeneration.

Phase 2

ECCP is now ready to tackle the second phase, which in addition to boosting the existing woodlands will focus on the expansion of the upland wood pasture. It is intended to do this on a large-scale: there is the

Figure 14.2 *Geltsdale in East Cumbria.*



potential temporarily to exclude grazing from all of Binnie Banks and perhaps even the King's Forest, and to re-create another wood pasture of 220 ha. This is much more difficult because both woodland and pasture habitats will have to be dealt with at the same time *and* on the same area of land. There is still a lot more work to be undertaken to establish the baseline information on which management decisions can be based (see Fenton and York, Chapter 12). Surveys are required for archaeological and historic woodland features, i.e. past woodland boundaries. The number, species, size, condition and location of the numerous veteran trees need to be determined and some tree ageing needs to be carried out. Surveys for lichens, ground vegetation, tree regeneration and birds need to be undertaken, and both grazing levels and timing for cattle and sheep, as well as acceptable damage levels from rabbits, hares and deer, need to be determined.

The issues faced by ECCP and its partners include:

- What is the optimum tree planting density to create wood pasture? Since the idea is to create a wood pasture right from the start, the most cost-effective answer is likely to be the planting of fewer trees than the 1100 trees ha^{-1} required for the Forestry Commission's Woodland Grant Scheme, but who could fund this?
- Individual or small groups of tree protection at the landscape scale is likely to be too costly to be practicable everywhere, so how long should stock be excluded by fencing, before it could have a detrimental effect on the vegetation?
- What is the optimum size for such enclosures?
- What species mix should be used? Should it be representative of the present mix, i.e. nearly 100% alders, or of the species and percentages of an NVC alder mix, or should it be something in between?
- Is there any way that grazing can be emulated artificially while the trees become established?

Potential contributors: funds and in kind

Since there are as yet no specific funds available through the WGS or Countryside Stewardship Scheme for either the management or the establishment of wood pastures, this work may have to be established as a demonstration project, which could show the way for future Forestry Commission or

Department for the Environment, Food and Rural Affairs (DEFRA) incentives. It may therefore be possible to attract partnership funding from organisations such as Heritage Lottery, DEFRA, the Countryside Agency, the European Union, in addition to the current partners, i.e. EN, RSPB, FC, ECCP, together with smaller sums from individuals who have an interest in Geltsdale, such as landowners, tenants and the public. If it is possible to fund phase 2, then together with phase 1, total of 335 ha of upland wood pasture could be protected and enhanced.

Geltsdale is highly valued by individuals and conservation organisations, but even so it will be difficult to fund this enhancement project because the specific aim is to maintain the *wood pasture* character of the area. In doing so, the work required is not going to fit any of the standard rules currently set out by either agricultural or woodland funding packages. Although it is encouraging to know that upland wood pastures are now included in the Lowland Wood Pastures Habitat Action Plan, there is to date no funding mechanism to implement the Action Plan objectives. Some innovative thinking is therefore required to make this project happen, because if a wood pasture project as proposed above cannot be undertaken on a site like Geltsdale with so many designations, then the chances of restoration of other wood pasture sites seem slim indeed.

Acknowledgements

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CHAPTER 15

Corrimony: an example of the RSPB approach to woodland restoration in Scotland

Neil R. Cowie and Andy Amphlett

Summary

The Royal Society for Protection of Birds (RSPB) approach to woodland restoration is described with reference to the Corrimony nature reserve in the northern Scottish Highlands. Details of the management planning process are outlined together with a description of survey and monitoring methods for trees, vegetation and other faunal groups to ensure successful restoration. A future vision for Corrimony is espoused which includes enhancing black grouse populations and habitat diversity, while maintaining close liaison with statutory bodies and local communities.

Introduction

The Royal Society for the Protection of Birds (RSPB) is involved in management of many native woodland sites throughout the UK, although it is only in Scotland that we are carrying out woodland restoration at the landscape scale. Here the RSPB has a number of ongoing woodland restoration projects on its reserves including Abernethy (Taylor, 2001), Wood of Cree (in South-west Scotland), Inversnaid (on the eastern side of Loch Lomond) and Corrimony (Figure 15.1).

There are also other reserves (Figure 15.1) where restoration work is taking place at a smaller scale. This chapter gives an overview of woodland restoration over the past three years (up until 2001) at Corrimony, a 1 520 ha site in the Beauly catchment, south-west of Inverness, and the work planned for the future.

Figure 15.1

Woodland restoration on RSPB reserves in Scotland.

Background

RSPB took ownership of Corrimony in 1997. Our primary reason for acquisition of this site was to increase native woodland in a strategically important area for capercaillie and black grouse. Both these species are in decline nationally (Batten *et al.*, 1990; Catt *et al.*, 1994). The importance of landscape scale areas of woodland and habitat networks for capercaillie and black grouse is one of the drivers in our efforts to reverse the decline of these species in the UK (Baines, 1991; Moss and Picozzi, 1994). The surrounding Beauly catchment pinewoods form one of the core areas of remaining native pinewood in Scotland (Steven and Carlisle, 1959; Forestry Authority, 1994). There is therefore the potential at Corrimony to restore a large enough patch or network of patches of woodland, within the context of the wider forested landscape, that would make a difference to both native woodland habitat and species conservation.

Assessing the existing resource

The long-term management objectives for the Corrimony nature reserve have been guided by detailed environmental assessments. Key to the production of these objectives was the formulation of a Management Plan for the site (O'Hara, 1998). The first thing we did when we acquired the site was to make a careful assessment of what was there and where everything was. This involved detailed surveys and mapping of species and habitats as well as an assessment of the current levels of tree regeneration across the site. An audit of all archaeological sites was also made. RSPB carries out site assessments on all sites it manages regardless of whether they have any conservation designations. It is important to take this precautionary approach where major changes in management are intended.

Habitat survey

A 1:10 000 scale NVC survey of the site, including the plantation areas, was undertaken in 1998 and the data were captured onto a Geographical Information System (Hutcheon *et al.*, 1998). This core dataset has formed the basis of much of our planning. Sixty-three different plant community types or mosaics were identified. The site is dominated by wet heath interspersed with areas of dry heath and bog. Small patches of woodland, grassland and base-rich flushes are also present together with the blocks of plantation woodland. Some of the older Scots pine *Pinus sylvestris* plantations were mapped as W18 pinewood as they exhibited many of the floristic attributes of this community (see Table 15.1).

Table 15.1

NVC communities and other habitats at Corrimony RSPB reserve.

NVC community/habitat	Area (ha)
M15	634
H12	275
Planted non-native conifers	185
W18	159.5
M17	70
W11	40
Other planted Scots pine	36
W17	25
M25	25
M18	23
U20	14.5
M19	11
H10	8
U4	4
M6	3
G10	2.6
S9	2.5
M1	1
M10	1
H16	0.5
W4	0.5
W7	0.5
S4	0.1

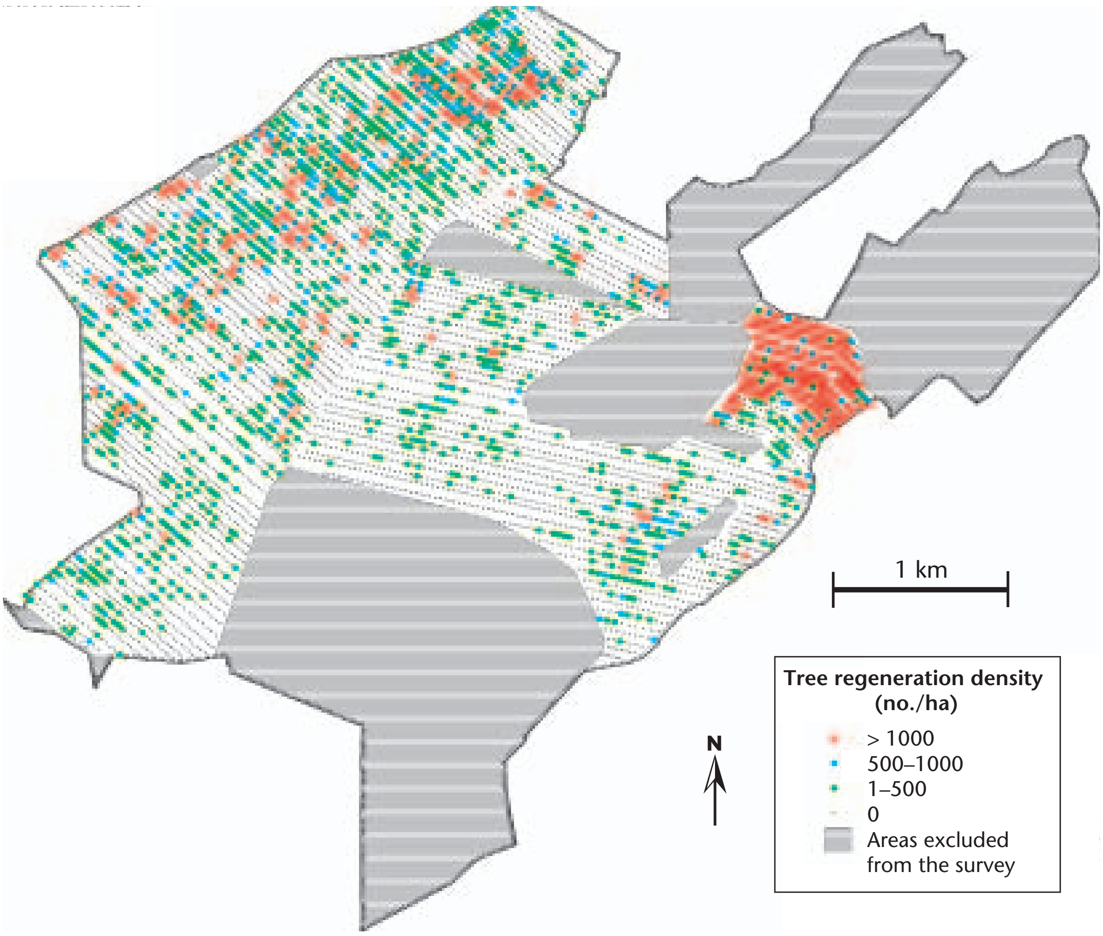
Monitoring vegetation change

Using the NVC survey data, the reserve has been stratified into three broad habitat types: dry heath, wet heath and bog. Further stratification was also done by grazing type (cattle, sheep and deer) and in terms of areas to be planted, areas burnt and areas where exotic conifers had been felled. Sixty-five permanently marked 20 x 20 m plots have been established. Within each plot, four permanently marked 2 x 2 m quadrats have been established to allow monitoring of ground vegetation and seedling microsite availability. All seedlings and saplings (878) within the plots have been measured, and a proportion (204) tagged to allow individual plants to be followed. Resurveying of these plots in future years will provide important information about the effects of different grazing regimes, with interest focusing on creation of regeneration gaps in the vegetation, growth rates of different native species under different treatments and in different habitat types, and the success rates of planted trees.

Regeneration survey

In 1998 a systematic sample survey of the distribution and density of tree and shrub regeneration was undertaken over 900 ha of the site, excluding areas of existing plantations. Data were collected along 50 m spaced, parallel strip transects, subdivided into *c.* 30 m sections (see Box 15.1 for methodology) and added to GIS. Figure15.2 shows the density of tree regeneration in 1998 for all species combined for each of the transect sections.

Figure 15.2 *Tree regeneration survey across the Corrimony reserve. All species have been added together.*



Regeneration is extensive but patchy. Over the whole survey area the mean density is 404 ha^{-1}, but the median density is zero. Areas of high density regeneration are particularly noticeable in the eastern corner, and on the north-west side of the reserve. Many of the areas, with little or no

Box 15.1 *Regeneration Survey methodology.*

Parallel transects were walked at 50 m spacing over 900 ha of the site (all areas outwith existing plantations), and all tree and shrub seedlings and saplings within 2.5 m of the transect line were recorded (thus sampling 10% of the survey area). Transects were walked at a slow steady pace and no detailed searches for seedlings were made, therefore density estimates are taken to be minima. The transect lines were subdivided by pacing into *c.*30 m sections. Transect start and end points were recorded on 1:10 000 scale maps.

Eight figure grid references for the mid point of the first section of all transects and the lengths of all transects were calculated from these maps. Given the grid reference of the first transect section, the compass bearing walked and the mean transect section length, basic geometry allows calculation of the grid references of all subsequent transect sections along that transect.

Calculation of the large number of individual grid references generated by this survey (6015) required use of a spreadsheet. To allow this, the map bearing (i.e. not magnetic) for each transect was converted from degrees to radians. A formula for this conversion was:

$$\text{radians} = \left(\frac{360 - (\text{bearing} - 90)}{360}\right) \times 2\,\pi$$

To calculate **X** co–ordinates for successive section grid references the formula used was:
co–ordinate of last section + (mean section length X cosine transect bearing in radians)

To calculate **Y** co–ordinates for section grid references the formula used was:
co–ordinate of last section + (mean section length X sine transect bearing in radians)

Note that the grid reference and mean section length values must be in the same units, e.g. section lengths in metres requires use of 10 figure (to 1m) grid references.

Given the resources a number of improvements could be made to this survey method. Detection of small seedlings obviously decreases with distance from the observer / transect line. Distance sampling (Buckland *et al.,* 1993) would allow a more accurate estimate of density to be calculated. This would also be a useful refinement if the survey were to be repeated by different observers, or after a period of reduction in grazing, when vegetation height and structure might have altered, hence altering the detection rate of seedlings / saplings. This method of locating sample locations along a transect line, compatible with GIS or other computer mapping software, has a wide variety of other possible applications. It is necessarily restricted to terrain in which it is possible to follow a compass bearing, and requires accurate grid references of start and end points. Since May 2000, hand held Global Positioning Systems give satisfactorily accurate grid references to be used to locate transect end points, especially in open country.

regeneration, coincide with areas of bog and the wetter types of M15 wet heath. Of seedlings / saplings recorded, 79% were birch (*Betula spp.*). Willow (probably all *Salix aurita*) was the next most abundant species (19% of regeneration). Although much less abundant than birch, willow is equally as widespread across the reserve (both species being recorded in 21–22% of transect sections). Rowan (*Sorbus aucuparia*) was the only other species recorded in any appreciable numbers (2% of regeneration and in 4% of transect sections). Juniper (*Juniperus communis*) and Scots pine accounted for <0.1% of regeneration.

Biodiversity audit

Ninety-one species of bird (59 breeding) have been recorded. Breeding species of particular note include black grouse, merlin and hen harrier. Recent botanical and entomological surveys by specialist contractors, staff and volunteers have significantly improved our knowledge of species on the site. A number of notable records have come out of this ongoing work.

Four Red Data Book (RDB) or nationally notable dragonfly species have been recorded within the bog areas, *Aeshna caerulea, Leucorrhinia dubia, Somatochlora arctica* and *S. metallica*. Pearl-bordered fritillary has been recorded from open birch woodland. There is also a considerable lower plant interest, which is concentrated along the gorge section of the River Enrick, on the areas of blanket bog and in relatively base rich flushes. A number of oceanic bryophytes are found here close to their eastern limit in Scotland.

Archaeological audit

Where we have proposed small areas of tree planting and are encouraging natural regeneration we have worked closely with Historic Scotland to ensure that we take account of all sites of archaeological interest. The areas that have been surveyed so far have produced a number of new archaeological sites including some Bronze Age settlements (Kenworthy, 1999). One important archaeological site is being kept open and natural regeneration discouraged. This Bronze Age hut circle is on a patch of ground that is also an important black grouse lek. With the help of a local grazier and some electric fencing this site is grazed in the autumn.

Restoration work

Our long-term vision for the site is to have at least two-thirds woodland cover. Our objectives are based on a strong preference for encouraging natural processes to restore native woodland cover on the site, primarily through natural regeneration. However we have decided to use limited planting of Scots pine as part of our efforts to restore native woodland and considerable management of existing plantations is also being carried out (Figure 15.3). Although our management plan has not been driven by grant-aidable schemes for woodland restoration, various proposals have been drawn up to fund aspects of the work we are doing. Extensive consultation has taken place with local communities and adjacent landowners, taking on board their views about our proposals. There are three ways in which we are planning to restore native woodland to this site.

Existing woodland and plantations

An accidental fire swept through 175 ha of young plantation at the south end of the site in 1997. The effects of this fire have probably been neutral/beneficial to our plans for the site as a whole. It has provided us with a good seedbed for natural regeneration and already there are birch, rowan and willow seedlings appearing. It also killed plenty of non-native conifers! Other recently planted non-native conifers have either been removed already or will be through phased removal.

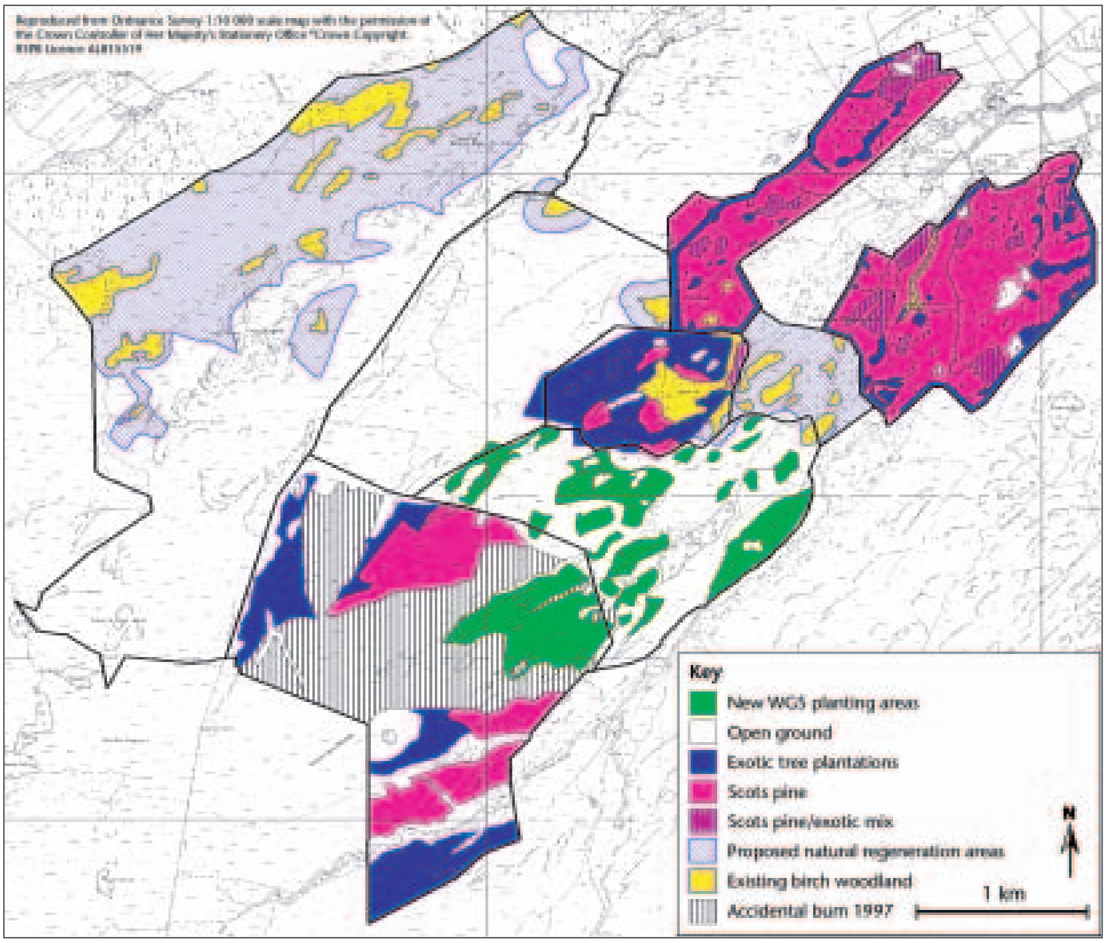
We also plan to manage the older plantation areas in the NE corner of the site. As well as removal of exotic tree species a certain level of restructuring and thinning is planned to encourage more ground cover, particularly blaeberry and broadleaves. The gorge woodland is currently dominated by natural Caledonian pinewood with some mixed broadleaves including stands of aspen. Parts of the Scots pine plantation bordering the gorge woodland already show some characteristic features of a more natural pinewood and will remain unmanaged. There are no plans to manage these plantations for long-term income. It is our preference to allow natural processes to operate (e.g. self-thinning, windthrow), in time allowing a woodland with more natural attributes to develop.

A screen of larch surrounds most of our Scots pine plantations. Some concerns were raised initially over our wish to remove some of the larch screening mainly because of the potential for windthrow in the plantations. After close consultation with the Forestry Commission some blocks of larch will be removed to soften the plantation edge and improve the landscape character of the plantations.

Natural regeneration

Our expectation is that most of the woodland restoration will be through natural regeneration. Areas with sufficiently high regeneration densities have been or will be submitted as WGSs for natural regeneration. Some of these are within the vicinity of existing woodland fragments whereas others are planned for much larger areas of open hill ground where regeneration is present but patchy due to the mosaic nature of habitats present.

Figure 15.3 *Woodland restoration work planned for the Corrimony RSPB, part-funded by the FC Woodland Grant scheme (WGS). Note that open ground areas outwith WGS areas for natural regeneration area also developing significant amounts of tree regeneration, though this is patchy.*



We are proposing to continue summer grazing the north-west side of the reserve with sheep at low density (200 sheep and 12 cattle over 525 ha from June to October). Here our objectives are to encourage tree regeneration while maintaining the mosaic of open habitats important for the black grouse and other, non-avian, species of conservation interest. Despite many years of sheep grazing, regeneration is widespread within this compartment, and at fairly high mean density (*c.* 350 ha^{-1}). Our detailed browsing survey work has indicated that this regeneration is actually growing well. Despite 28% of tree seedlings being browsed to some degree, there has been a significant overall increase in the height of existing regeneration. Survey work in 2000 showed that in monitoring plots set up in 1999 there was a mean annual height increment of 7 cm for birch tree regeneration and 17 cm for willow, representing a 20% increase since the 1999 survey (Cowie *et al.*, 2001). Integrating grazing with woodland restoration across this compartment, eliminates the need for additional fencing, and hence reduces the risk of mortality to the black grouse population through fence collisions. The Forestry Commission has been very supportive of our objectives for this area and we are working very closely with local staff to bring this application together.

Planting

Our NVC survey data were used to predict potential woodland cover for the site based on a knowledge of the open ground plant communities present. We compared this technique to the Ecological Site Classification being developed by Forest Research (Pyatt and Suárez, 1997) and found that it gave similar results for the major woodland types. However, we adopted the NVC approach because it was more sensitive for our needs in predicting the smaller scale areas of woodland (Bates and Perkins, 1997).

Much of the area could potentially carry a mixture of pine and broadleaf woodland types, however 99% of current regeneration is broadleaf. A decision was therefore made to plant comparatively small copses of Scots pine in areas which were relatively remote from seed source and in a non-uniform way. These trees will eventually reach cone-bearing age and act as a source of seed for further expansion of the pinewood habitat. Areas being planted are quite small and planting is patchy even within these areas. The main reason for planting in the selected areas was to help link the recent Scots pine plantations and existing semi-natural woodland and older plantations (see Figure 15.3). Planting is restricted to H12 (dry heath), M15c (wet heath) and better drained areas of M15b communities.

There is very little fencing across the site as a whole; 8 km of deer fence have been removed and 3.5 km reduced to stock height. However, where it is needed (e.g. march fence on south side of the reserve) we have been testing 3.5 km of a new grouse-friendly fence design to discourage deer from entering the site (stock fence with offset electric wire). In conjunction with this there has been a cull of red and roe deer within the site to keep numbers at an acceptable level to allow tree growth and regeneration.

The future

This is obviously a long-term project and RSPB has managed the site for only 3 years. Most of this time has been spent carefully assessing the site and planning the restoration work. This has been particularly important as there is a diverse range of habitats and features that should be conserved. Although Corrimony does not have any statutory conservation designations we have involved Scottish Natural Heritage as well as FC from the start. Close liaison with local communities has also been important. This restoration project has not been driven by the grant-aidable schemes, but we are using them to achieve our conservation objectives. Some difficulties have arisen in using the various grant schemes but these have been overcome by working closely with local Forestry Commission staff. Corrimony is already one of the best black grouse sites north of the Great Glen. We have seen numbers increase from 16 to 23 lekking males between 1997 and 2000. Restoration of suitable woodland habitats will hopefully further enhance this population as well as potentially providing in the long-term for species such as capercaillie. Corrimony has also attracted a lot of interest from other research bodies in relation to the woodland restoration work being carried out, for example the Macaulay Land Use Research Institute, the Scottish Agricultural College, Forest Research and the University of Edinburgh.

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CHAPTER 16

Woodland improvement on the Woolhope Dome, Herefordshire

Humphrey Smith, Mark O'Brien and Elizabeth Vice

Summary

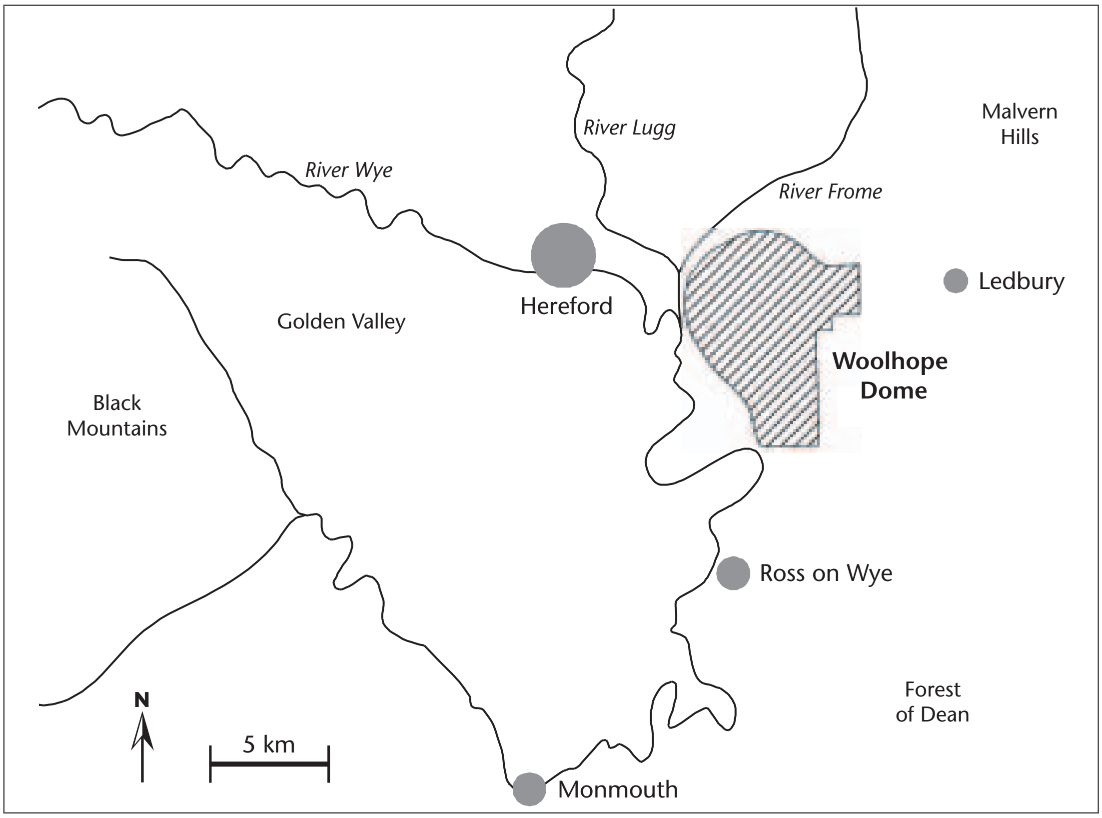
The Woolhope Dome is a distinctive feature of south Herefordshire with high potential for integrated development of sustainable forestry, wildlife conservation, eco-tourism and education. The Dome consists of a mixture of land-uses over a 4 000 ha area. It contains abundant woodland, with some 50 properties of varying area covering about 1 000 ha in total. All the woodland is either semi-natural or planted ancient woodland sites. The plantations give good yields of high quality softwoods and sessile oak. The western fringe woods lie within the Wye Valley Area of Outstanding Natural Beauty, an area of exceptional wildlife conservation value in Britain. The whole Dome area has been designated Herefordshire's Prime Biodiversity Area. There have been biodiversity losses over the past century due to agricultural intensification, afforestation and neglect, resulting in habitat fragmentation and isolation. A comprehensive statement of the potential for restoration and improvement is due soon with the publication of the Biodiversity Vision Project by Herefordshire Nature Trust. Meanwhile, some improvements have been initiated on both planted and semi-natural sites by Forest Enterprise and private owners, aimed at enhancing the habitats of Biodiversity Action Plan priority species. The area has not hitherto attracted intensive tourism and there is scope for sensitive development of eco-tourism and educational activities.

Introduction

The southern Herefordshire part of the Welsh Marches is a complex area of downland, ridges and valleys, located between the Herefordshire lowlands (north-west), the Forest of Dean (south), the Malvern Hills (north-east) and the Black Mountains of Gwent (west). Within this area, the Woolhope Dome (SO 600 360) is an upfolded, roughly circular area of *c.* 4 000 ha (Figure 16.1), in which alternate beds of Silurian shales and limestone surround an acid sandstone core (Earp and Hains, 1971). Its steep edges form a scarp landscape which falls to the valleys of the Rivers Frome and Lugg to the north-west, while the south-western edge lies within the Wye Valley Area of Outstanding Natural Beauty (AONB) (Countryside Agency, 2001a) and contains a 6 km section of the Wye Valley Walk (Burton, 1998). The Woolhope Dome is described as 'a very distinctive feature of the Herefordshire landscape' in the Countryside Character Initiative, Area 104b (Countryside Agency, 2001b). Woodland is abundant within the Dome area, covering about 1 000 ha in total. Much of this woodland is located on the ridges and uplifted plateaux, as well as the valley slopes, and so dominates the skyline from most vantages. Other key characteristics of the Dome area are the mosaic of hamlets, farmsteads, cultivated arable and pasture fields, historic parks, cider orchards and both brick and timber framed buildings. Twentieth century changes to the area include:

- agricultural intensification with loss of hedgerows and species-rich grasslands;
- fragmentation of semi-natural habitats;
- afforestation of some ancient woodland sites and neglect of some traditional orchards and woodlands.

Figure 16.1 *Location of the Woolhope Dome in southern Herefordshire.*



Woodlands and forestry

The Woolhope Dome bears some 50 woodland properties of between 5 ha and 400 ha in area variously managed by Forest Enterprise, the National Trust, Herefordshire Nature Trust and private owners. All of the woodland properties are either ancient semi-natural woodlands (ASNW) or planted ancient woodland sites (PAWS), and most of them have Site of Special Scientific Interest (SSSI) notification by English Nature. Extensive areas were afforested by Forest Enterprise and private owners, mostly in the 1960s, with Norway spruce (*Picea abies*), Douglas fir (*Pseudotsuga menziesii*, Japanese larch (*Larix kaempferi*), Western red cedar (*Thuja plicata)* and Scots pine (*Pinus sylvestris*), with smaller areas of sessile oak (*Quercus petraea*), beech (*Fagus sylvatca)* and sweet chestnut (*Castanea sativa)* . The largest Forest Enterprise planting (*c.* 400 ha) is Haugh Wood SSSI (SO 590 365), which stands on the old red sandstone of the central plateau of the Dome, while smaller private policies are on the more peripheral Silurian shale and limestone beds. The resulting fertile loam and clay soils, together with a mild western climate and low windthrow hazard, support high timber yields.

Table 16.1 *Yield data gathered for stands in the Woolhope Dome.*

	NS	DF	JL	SP	WRC	Oak
Local yield class	14–16	14–20	12	12	18–22	4–6
Production class	a	a/b	a	a	a/b	a
Rotation age (years)	68–74	52–60	44	72	60–66	82–88

Mensuration of sample plots has given the estimated yield classes and rotation times shown in Table 16.1. Fellings in 2000 and 2001 of 38-year-old Douglas fir in Siege Wood (SO 599 345) and 85-year-old sessile oak in Mabley Grove (SO 599 345) have yielded high quality timber, realising £34 per tonne and £3.75 per hoppus foot respectively at roadside, reflecting the best prices available in the currently depressed markets. While the long-term future of softwood forestry in Britain is open to debate, the stands currently at pole-stage should certainly yield excellent maincrops from about 2020. Sessile oak should have an assured future indefinitely, given the quality achieved. The recent acquisition of Forest Stewardship Council certification for Siege Wood and Mabley Grove should help to ensure continued consumer confidence.

Wildlife conservation value

The Woolhope Dome has been designated a Prime Biodiversity Area by the Herefordshire Nature Trust and English Nature in recognition of its species-rich semi-natural ancient woodlands, old English meadows and traditional cider orchards – all priority habitats in both the national and local Biodiversity Action Plans (BAPs). In particular, the woodlands of the lower Wye Valley form one of the most important areas for woodland conservation in Britain, having been ranked with the New Forest, Caledonian pinewoods and oceanic western oakwoods (Peterken, 1977). One of the richest woods of the Wye Valley AONB is Lea and Paget's Wood with Mabley Grove (SO 600 345). It has impressive species lists including many ancient woodland indicators:

- Flowering plants *c.* 300 spp., excluding grasses and sedges.
- Broadleaved trees *c.* 20 spp., including small-leaved lime and wild service.
- Birds *c.* 140 spp., including wood warbler, tree pipit, pied flycatcher and goshawk.
- Butterflies *c.* 30 spp., including grizzled skipper, pearl bordered fritillary and wood white.
- Bats 10 spp., including lesser horseshoe and noctule.
- Other mammals including polecat, dormouse and yellow-neck mouse.

While biodiversity has been exceptionally sustained on the Woolhope Dome, there has been fragmentation and isolation of the richest habitats and the potential for biodiversity gains through woodland improvement is therefore considerable.

Biodiversity Vision Project

This project was launched in Summer 2000 by Hereford Nature Trust in partnership with English Nature, Herefordshire County Council and the Wye Valley AONB. Its aims were to:

- Produce a GIS map of the Woolhope Dome identifying the location and extent of known Biodiversity Action Plan priority species and habitats.
- Make generic prescriptions for optimal management of existing habitats and species.
- Identify the location and extent of areas that have potential for restoration in order to reduce isolation/fragmentation by habitat recreation and buffering.
- Calculate the costs of delivering the management and creation targets of the vision and indicate the benefits to the rural economy.

Meanwhile, a number of initiatives are currently in progress. Since 1997 Forest Enterprise has been operating a programme of felling compartment margins in Haugh Wood in advance of economic rotation age. The aim here is to create wide gladed zones throughout the wood in order to restore the ancient woodland flora and associated invertebrate fauna (especially for butterflies on south-facing slopes), and to initiate 15-year rotation coppicing of hazel and oak.

New management of Mabley Grove SSSI started in 1999 with a view to its restoration to a medieval hazel coppice with oak standards structure: stand type 6Cc (Peterken, 1993). Two coppice cycles have been initiated: a 7-year cycle of larger coupes (0.75–1.25 ha), primarily for hazel production and associated biota, and a 23-cycle of smaller coupes (0.1–0.5 ha) to favour dormice (a BAP priority species). Currently overstocked with 85-year-old oaks, the reduction to 5–7 trees per ha will progressively release the hazel understorey, while generating income to fund management for biodiversity including ride restoration and glade creation. About 100 fallow deer utilise the woods and surrounding area; browsing on coppice regrowth is being restricted by building dead-hedge exclosures and by selective culling, which generates a modest venison yield.

Opportunities for future woodland development

Given the long-term nature of woodland processes, new developments need to be fully consistent with the principles and standards for sustainable forestry set out in the *UK forestry standard* (Forestry Commission., 1998) and by English Nature (Kirby and Reid, 1996). To be successful they must also take account of the critical state of the agricultural and softwood timber markets and the pressures these exert for diversification of rural land-use. Encouragement for appropriate developments is evident in the grant aid incentives operated by the Forestry Commission, English Nature and DEFRA Countryside Stewardship. In this context, the Woolhope Dome has considerable potential for woodland conservation, restoration and improvement. Most of the currently afforested PAWS will reach main crop rotation age between 2020 and 2030. If second rotations of planted softwoods are then still seen as economic, then consideration could be given to converting the even-aged monoculture stands of Haugh Wood to a continuous-cover silvicultural system. One such system with considerable potential, using mixed species in uneven-aged stands, is that used in the Tavistock Woodlands, known as Bradford Plan forestry (Wigston, 1980). In this system compartments are replaced by 18 m x 18 m units divided into 6 m x 6 m plots. Within each unit, plots are planted at 6-year intervals, initially with nine nursery transplants, which are then progressively thinned to a single main crop tree per plot over a 54-year rotation. Alternatively, wholesale conversion of all plantations on ancient woodland sites (PAWS) to ancient semi-natural woodland (ASNW), with a long-term programme of restoration to coppice-with-standards by managed natural regeneration, should prove feasible, following the pattern initiated in Siege Wood (SO 607 343) during 2000.

In the event of land which is currently devoted to improved agriculture being taken out of production, there will then be scope for linking the Dome's core areas with its fringes, in particular with the riparian Capler Wood (SO 589 325), with hedgerows, spinneys and areas of wood-pasture. While the capacity of habitat corridors to act as conduits for species dispersal remains questionable (Dawson, 1994; Dolman and Fuller, Chapter 3), an increase in woodland edge habitats will be beneficial in itself and help to reverse some of the habitat isolation currently experienced. Such developments would be highly compatible with conservation of ancient oak pollards which are characteristic of the regional landscape; they could also be integrated with conservation of some 30 stands of traditional cider orchard on the lowlands of the Dome edge at Dormington (SO 584 400), Putley (SO 640 373) and Much Marcle (SO 645 332).

The valleys of the Rivers Wye, Lugg and Frome experienced unprecedented episodes of flooding in Autumn 2000. With the potential for such incidents to occur more frequently in future with climate change, it would seem prudent to establish riparian strips of willow/alder carr in order to obtain benefits for water quality and soil conservation as well as for biodiversity.

Potential for ecotourism

The Woolhope Dome and adjacent areas of the Wye Valley AONB have not attracted tourist visitors on the same scale as the nearby Malvern Hills and Forest of Dean; indeed its sense of quiet remoteness is one of its attractive features. However there is scope for sensitive development of education and ecotourism in the woodlands. A programme of guided walks and talks on ancient woods, butterflies, mammals, birds and wildflowers is currently offered (www.wyevalleywildlife.co.uk). It is hoped to expand these in the future with the eventual aim of establishing a field centre offering residential courses.

All these proposed developments are consistent with national and local policies on sustainable forestry and Biodiversity Action Plans, but will necessarily involve the sympathetic co-operation of landowners and agents, many of whom experience fiscal pressures to maximise returns from intensive agriculture and potentially damaging woodland uses. If conflicts of interest can gradually be resolved, there is every prospect of woodland improvement on the Woolhope Dome making a valuable contribution to sustainable forest production, biodiversity, education and rural employment.

Acknowledgements

For information about the Woolhope Dome and for their sympathetic interest and advice on the management of Mabley Grove and Siege Wood, we would like to express grateful thanks to Sarah Ayling and Sue Holland (Herefordshire Nature Trust), Dick Gosling (Forestry Commission), Charlotte Padgenham (English Nature), David Sykes (Forest Enterprise) and officers of the Forestry Stewardship Council.

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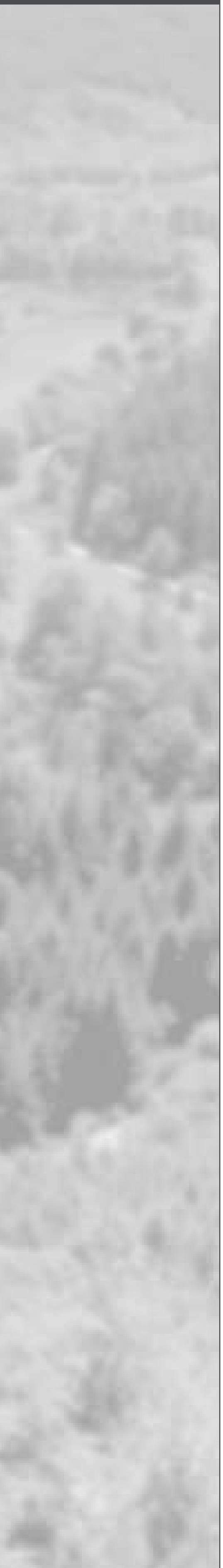
CHAPTER 17

Ettrick: a habitat network in the Scottish Borders

Andrew McBride



Summary

This chapter describes an ongoing agri-environmental project in the Scottish Borders that aims to create and restore floodplain woodland and other semi-natural habitat at the landscape scale. The lessons learned during the implementation of the project are discussed, and recommendations made for the improvement of project management practices.

Introduction

Floodplain forest habitats are highly diverse ecological systems. These habitats have disappeared from much of Britain with a few surviving in Scotland. During 1995, WWF Scotland commissioned a review (Smith and McGhee, 1995) of the status of Scottish floodplain forests. This identified the Upper Ettrick (near Selkirk in the Scottish Borders) as one of the finest examples of existing floodplain habitat that also offered great potential for restoration and expansion. Borders Forest Trust launched the Ettrick Habitat Restoration Project with funding provided by The Millennium Forest Trust for Scotland, The Forestry Commission, Scottish Natural Heritage and The World Wildlife Fund (Scotland).

Project development and progression

The project centres upon a 5 km stretch of floodplain with a fragmented mosaic of habitats that follows the Upper Ettrick and Tima Water. This area possesses wide biodiversity of high conservation value. The River Ettrick is a shifting and dynamic watercourse with gravel bar formation, ox bow siltation and bank undercutting.

Ecological principles form the basis of site management decisions within the project which aims to achieve both environmental and economic benefits. The co-operation of nine farms and Forest Enterprise made it possible to put the project concept into action. The participation of enlightened and willing landowners and land managers enabled the implementation of a range of land management techniques, and the linkage and extension of a range of habitats, to benefit the overall natural environment of the Ettrick forest floodplain.

The involvement of the local community of Ettrick (approximately 100 inhabitants) has proved central to the progress and subsequent success of the project. Professional/technical and community steering groups enabled extensive but structured consultation with statutory bodies and members of the local community. Such consultation has been invaluable when faced with management dilemmas and controversial decisions; for example, when developing the formal access network or closing the main road route in and out of the valley to allow tree felling. This integrated approach also provided an opportunity for raising wider awareness of the ecological nature of the project.

Establishment of native woodland through planting and regeneration

During the early stages of the project, it became apparent that conservation value could be enhanced by linking the valley bottom with small areas of woodland within steep valleys (cleuchs) running into the floodplain. Through negotiation with local farmers, approximately 30 ha of new native woodlands were planted in total. The planting season 1999–2000 saw the establishment of 50 000 native tree species, including ash, oak, alder, birch, willow, aspen, hawthorn and blackthorn. To protect the trees from stock, local contractors erected 7.5 km of fencing. Spiral guards and bamboo canes provided the young trees with minimal individual tree protection. Variable planting densities of between 2 000 and 2 200 per ha allowed for subsequent losses from herbivore damage. After two winters, minimal damage to seedlings was recorded and survival rate was 98%. On the recently cleared floodplain, where the lush vegetation attracts deer, Forest Enterprise implemented an intensive programme of deer control, resulting in low levels of browsing damage.

Fifty-three hectares of mainly Sitka and Norway spruce were felled between 1997 and 1999 on a selected area of the floodplain to encourage natural regeneration of native broadleaves. Following the felling operations, regeneration of native species, namely willow, birch, bird cherry and rowan has occurred across the site. Less welcome regeneration of sycamore and Sitka spruce has also occurred, particularly close to conifer tree stumps and on the mounds left by drainage ditches.

The management of floodplain meadows

The funders accepted the proposal to include meadow management as part of what was essentially a woodland project. The creation of a woodland network, which included other associated habitat types, was relevant to that particular landscape. To date 30 ha of floodplain meadows are under conservation management. After consultation with farmers, basic prescriptions were drawn up for each site. The prescriptions are not rigid, enabling the site manager to judge when to introduce grazing stock according to ground conditions and ecological interest. The site manager is therefore unrestricted by specific calendar dates as imposed by other agri-environmental schemes.

The meadow management programme encompasses:

- late hay/silage cutting;
- supplementary seeding of locally collected wild flowers;
- no application of artificial fertilisers;
- stock grazing of fresh grass one month after hay/silage removal.

Access

From the beginning, the community were keen that the project maintained a low public profile. They did not wish to attract large numbers of people, particularly day visitors, to a place of low population density accessible only by a single minor road. However, they were keen to improve facilities for visitors who regularly come to stay in the area and particularly walkers who enter the valley via the long distance footpath known as the Southern Upland Way. Access improvements included the construction of 5 km of new footpaths that link with an existing 4 km of forest tracks. The high rainfall of the upper valley combined with heavy soils made necessary the installation of nearly 0.5 km of boardwalk.

Hydrology

For much of the year the River Ettrick has the appearance of a gentle burn, but looks are often deceptive. The annual rainfall for the Upper Ettrick is in the region of 2 000 mm. Consequently, after periods of heavy rainfall the movement of high volumes of water dramatically alters the character of the river. The rapid rise and fall of the river level quickly turns the gentle burn into a raging torrent, flooding the haughland approximately 10–12 times each year. Following several hydrological surveys, a policy of non-intervention was decided, allowing the river to follow an unhindered and variable course through the wetlands, creating a wide variety of shore and standing water features.

Ettrick Project: achievements

A summary of the project achievements, as put into action through landowner and community negotiation includes:

- Removal of 53 ha of exotic conifers from the floodplain – some conifers were retained outwith the floodplain.
- Planting of 28 ha of riparian woodland.
- Reinstatement of traditional meadow management on 27 ha.
- A policy of non-intervention of floods over project land.
- Through livestock exclusion and the selective removal of exotic tree species, 10 ha of existing woodland brought into management.

Components of an effective project

The consultation process

Consultation with stakeholders is a vital component of any restoration project. Some consider the Ettrick Project in particular as a potential blueprint for consultation, drawing as it did upon both the local community and professional bodies for input. This process enabled the project managers to overcome a series of obstacles, including one very determined objector.

Community consultation can work either for or against a project. On the plus side, communities usually possess a latent skill base that a project can utilise to its benefit. In addition, inclusion of the community at the outset creates a sense of ownership, and empowerment of the scheme. On the other hand, excessive empowerment and consultation may limit the project's potential and lead to less than ideal decisions. These may run contrary to management guidelines and landowner needs, for example felling trees, on the basis of a decision made by 'committee' is not ideal.

The timing of any consultation process requires careful consideration. Is it more appropriate to consult during the early or later stages? Early consultation that has received a positive response needs to be followed up quickly or goodwill may be lost, especially if several months pass between the consultation and practical action. This is often the case for restoration projects, as securing funding can be a laborious and time-consuming process. This type of scenario can mean that a supportive community loses heart and interest wanes. Maintaining contact throughout these 'quiet' periods is crucial to overcome this type of negative malaise. On the other hand, consultation during the later planning stages may mean that the opposite happens. Things will often need to happen quickly on the ground in response to funding deadlines and this can preclude the community from an active role in the decision-making process.

Community consultation is a financial burden on project management time. It is estimated (author's own experience) that the inclusion of community consultation requires an additional 30% increase in project management input. This in itself can cause problems with funding agencies who are more used to between 10 and 15% of total project costs being ascribed to project management for non-consultative projects. It is also important to stress that project management continues even when results are not apparent on the ground.

Dealing with opposition

Managers of large-scale projects often discover, at some point, a detractor or group of detractors ready to test their mettle. The abilities of the opposition should not be underestimated, and managers should not allow themselves to become contemptuous or complacent in situations of conflict. In this age of transparency, providing a free flow of information can create problems and a consultation process with associated openness will fuel the exploits of detractors. These detractors are usually well informed, know their rights, and are often prepared to utilise aspects of the law (particularly planning) and news media to their advantage. They also know all too well that delays cost and may even lose the project money. On the plus side, vociferous opposition means that each aspect of the scheme is put under close scrutiny and the activities of an objector can even galvanise the local population to back the project more solidly. That collective wealth of experience and knowledge held by a community can both help and hinder.

Field staff

It is not practical to manage the landscape at arm's length. There is a pressing need for more people working in the field, with the skills and abilities to become part of the community and provide some element of continuity for environmental projects. All too often, local knowledge, goodwill and the contacts amassed during the months or even years it takes to implement a large-scale project are lost when the project officer moves on. Being on the ground really can make a difference by providing opportunities to observe, liaise and respond to the situation as and when required. A project officer on the ground can inform and educate thus leaving farmers able to make their own conservation management decisions, for example, helping with site-specific aspects of management such as the grazing of meadows after the removal of hay/silage. This approach has real potential long-term benefits for both farming and wildlife, but we have to learn to educate and to trust.

Conclusions: improving the way environmental landscape projects are implemented and managed

An important concept of the wooded landscape is one of multi-usage, a working landscape that encompasses agriculture, forestry, fisheries, tourism, game and access management. Working within the current grant structure, the Ettrick project has achieved its goals and more. However, if a truly diverse, sustainable and cost-effective landscape is our desire, this is not the way forward in the long-term.

In terms of woodlands, the answer appears quite logical. The main obstacle to establishing wooded landscapes is over-grazing by herbivores, in particular sheep. There is a place for fencing but is it always a necessity for achieving a wooded habitat network? We are forgetting, and are even reluctant to use, past management skills such as shepherding to benefit the natural heritage. Part of this problem is the disconnection of an increasingly urban society from the countryside and the loss of a collective understanding of practical land management methodology. The tendency is to opt for blanket management techniques, for example wholesale bracken spraying or fencing. All too often, we start reinventing the wheel, failing to transfer lessons learned from one area to another. By the very nature of the requirements of their jobs, land managers are often busy and dispersed. In an ever-changing world, they require accurate and regularly updated information. Currently in Scotland, the dissemination of conservation advice and information is sparse and often not in a form attractive or engaging to the target audience. In these circumstances, 'adaptive management' is often the most useful approach to management, particularly for grazing, where there is a continuous cycle of observation, consideration and response.

When looking at habitats on a landscape scale, we should seize the opportunity for adding diversity to land management. However, the current range of woodland and agri-environment incentives are not flexible enough to achieve this goal at the present time. There are a number of problems which need to be addressed. When accepted into agri-environmental schemes, farmers receive payments for following an environmentally responsible form of management. Yet there is no differentiation in payment for non-management (i.e. removal of grazing) as opposed to situations that require active time and consuming management (i.e. seasonal grazing). Therefore, non-intervention becomes the more economically attractive option, even though low intensity grazing may bring more ecological benefits.

Nature is an opportunist and, like nature, we must learn to respond to opportunities as and when they arrive. However, to do this will require the introduction of flexibility into grant schemes. In addition, we need to encourage patience and an understanding of regenerative processes, as these often take far longer than our short-term human perspective. Rather than insisting on fencing or individual tree protection, why not allow the natural processes of predation and browsing to have a limited effect and invest in local skills to care for the trees as they grow and develop into woodland. In this way, not only can we create a diverse wooded landscape, but also sustainable rural employment.

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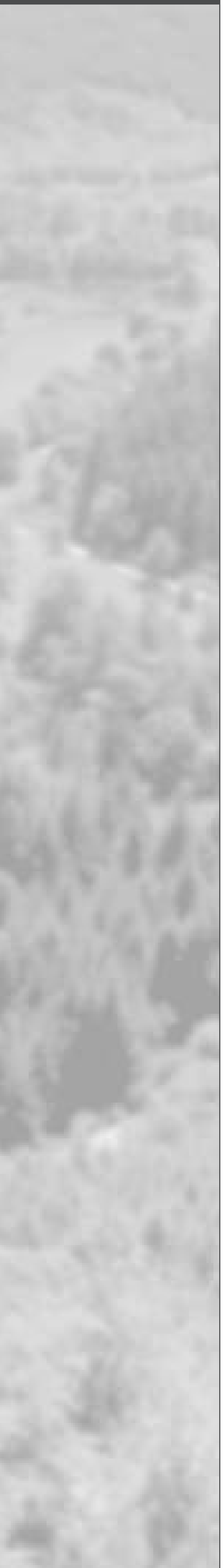
SECTION FIVE

Conclusions

CHAPTER 18

The restoration of wooded landscapes: future priorities

Jonathan Humphrey



Introduction

This chapter draws together the main problems and issues relating to woodland restoration at the landscape scale as highlighted in these proceedings and during discussions at the conference. It is not intended to be a comprehensive review of the science and policy of restoration ecology or a synthesis of the practical initiatives taking place across the UK as a whole. The views expressed reflect impressions gained from the conference and are therefore a subjective appraisal of issues and priorities.

The conference was successful in its main aim of bringing together researchers, policy-makers and practitioners and in fostering fruitful discussions and exchange of views. There was a consensus that woodland restoration at the landscape scale was important to fulfilling ecological, economic and social aspects of sustainable forestry objectives (Rollinson, Chapter 1), but the focus at the conference was primarily on ecological and strategic policy issues, the delivery of biodiversity benefits and the implementation of UK Biodiversity Action Plan targets for woodlands (Anon., 1995). Economic and social issues only really came to the fore in the case studies of individual restoration schemes (Chapters 14–17). Therefore, in this concluding chapter the emphasis is on the ecological issues raised and how ecological knowledge might be translated into future strategic priorities for woodland restoration in Britain. Future research needed to support this process is also highlighted. The main components of a successful restoration project are identified by drawing on evidence from the case studies.

Ecological issues

Judging by the number of recent publications (e.g. Urbanska *et al.*, 2000; Perrow and Davey, 2002), it is clear that restoration ecology is a rapidly developing field. In Britain there has been a traditional focus on site-scale restoration, but the recent work of Peterken *et al.* (1995) has helped to promote a landscape scale approach to restoration through the development of the Forest Habitat Network (FHN) concept (Peterken, Chapter 9). Restoration in this wider sense includes woodland improvement, creation and expansion, as well as the restoration of planted ancient woodland sites back to semi-natural woodland.

The level of restoration activity in Britain is small compared to that taking place in other countries such as the US (Newton and Kapos, Chapter 2), but the area of restored woodland is set to increase considerably through initiatives driven by the forestry strategies for Scotland (Anon., 2000), England (Anon., 1999) and Wales (Anon., 2001). Biodiversity objectives within all three strategies incorporate habitat network concepts such as increasing the area of existing native woodlands, building linkages between woodlands and reducing fragmentation. For example, in England, habitat networks are being developed through the 'Jigsaw Challenge' (GLEAN, 2001a) a targeted grant scheme which aims to help reverse the historical fragmentation of ancient semi-natural woodland in a number of target areas and conserve Biodiversity Action Plan species. Restoration priorities for woodland are also being explored within English Nature's Natural Areas (Kirby and Reid, 1997; Ferris and Purdy, 1999). In Chapter 7, Gray and Stone provide an example of how future habitat networks might be planned at the regional scale in Scotland. Plans have already been developed for the Cairngorms and will be extended to the Loch Sunart and Clyde Valley areas (Peterken, Chapter 9). In Wales, Latham's (Chapter 10) proposed management units provide a context for restoration and woodland management in the future, based on a historical perspective of different woodland structural and management types.

Some of the ecological assumptions underlying the FHN concept still have to be tested further scientifically. In their review, Dolman and Fuller (Chapter 3) question the usefulness of making linkages between existing native woodland fragments to provide movement corridors for wildlife unless the characteristics of the individual species of interest are known. Specialist woodland species are poor colonists, and may make little use of corridors, if at all. A more effective strategy for enhancing woodland biodiversity might be to target the expansion around existing ancient semi-natural sites in order to decrease the proportion of habitat close to external edges. Good showed (in his conference presentation) that simple buffering of native woods in mid-Wales brings benefits in terms of habitat area increase, although connectivity also increases as a by-product. This assumption also underlines the Woodland Trust's approach to restoration as reported by Smithers at the conference (Woodland Trust, 2001). Considerably more resources have been invested in woodland creation rather than in restoration over the past two decades. Prior (Chapter 13) questions the ecological rationale for this approach and suggests that given limited financial resources, the best way to enhance woodland biodiversity is to focus on restoring planted ancient woodland rather than on woodland expansion or creation. While considerable areas of new native woodland have been created there has been little monitoring of ecosystem development in order to justify the emphasis on creation rather than restoration.

Another question to consider, is the role of plantations within FHNs. Research has shown that mature and over-mature stands of spruce and pine can have considerable value as a habitat for fungi, bryophytes, song birds, red squirrels, dormice and various raptors (Humphrey *et al.*, 2002; Petty, 1996; R. Trout, personal communication). Humphrey *et al.,* in their conference presentation, suggested that suitably modified planted stands could play a key role in the extension of habitat networks. Again, this hypothesis needs to be tested.

Future priorities

Focusing landscape restoration around core FHNs makes intuitive sense, and indeed this was the approach adopted by the Millennium Forest for Scotland Trust (MFST) in its initial years of operation (Hunt, Chapter 8). Creating new native woodlands outwith FHN core areas would seem be a lower priority for woodland biodiversity, but may be the easiest way of diversifying poorly wooded landscapes. In addition, other non-ecological factors may need to be considered when arriving at any future decisions, for example, the wishes of the local community, recreation and amenity. There are numerous examples of large, new, native woodland schemes established in landscapes which have had little woodland cover for centuries if not millennia, e.g. Carrifran in the Scottish Borders. These schemes have been criticised as being too artificial and based on the assumption that trees will always provide a better habitat for wildlife than the habitats they replace (an assumption that is questioned by Fenton and York, Chapter 12). Clearly, assumptions about the dispersal and colonising abilities of woodland species need to be tested, as different people make different assumptions! As part of this, more information on the habitat requirements of particular species and species groups is also needed.

Within core FHN areas it is important to consider the relative balance between the restoration of planted native woodland, the expansion of exiting native woods and the creation of new native woods. In addition, although not covered in any detail during the conference, improving the condition of existing woodlands is an important priority which should run alongside restoration activities. The current Habitat Action Plans (HAPs) for priority native woodland types give area targets for creation and expansion of roughly twice those for restoration (Table 18.1).

These targets are to be achieved over the next 10–15 years, depending on the woodland type. Inevitably the setting of these targets was influenced by a range of other factors as well as ecological considerations. However, based on the views expressed in these proceedings, the following set of four priorities (in decreasing order of importance) may well be more appropriate if the aim is to maximise future gains for woodland biodiversity:

1. Restore planted stands.
2. Expand existing woods to buffer core woodland areas.
3. Integrate adjacent 'naturalised' plantations with native woodlands.
4. Link existing woodlands by creation of wildlife corridors.

Table 18.1 *Published area targets for creation/expansion and restoration of priority native woodland types (Anon., 1995).*

Woodland HAP type	Creation/expansion (ha)	Restoration (ha)
Upland oakwoods	7 000	7 000
Native pinewoods	25 000	5 600
Upland mixed ashwoods	6 000	2 600
Wet woodlands	6 750	2 600
Beech and yew woods	3 000	1 500
Wood pasture	500	2 500
Lowland mixed deciduous woodland[a]	25 000	15 000
Total (UK)	**73 250**	**36 800**

[a]The figures for lowland mixed deciduous woodland are from the draft plan (K.J. Kirby, personal communication).
A plan is also in preparation for upland birchwoods, but targets have not yet been finalised for this woodland type (R.N. Thompson, personal communication).

If this set of priorities were to be accepted in principle, then greater emphasis should be placed on restoration rather than on expansion, creation and linkage, although the relative ranking of these priorities is likely to vary between the different woodland types. For example, native pinewood restoration in the Scottish Highlands has been extensive to date, and the priority here might therefore be to naturalise areas of Scots pine plantations (Mason and Humphrey, 1999). The appropriate time to re-consider restoration priorities will be when progress with the implementation of the woodland Habitat Action Plans is reviewed as a whole in 2005.

Implementation

Developing models

The research and modelling papers (Chapters 3–7) demonstrate the power of GIS as a modelling tool, allowing a variety of different ecological analyses to be carried out for specific areas. GIS systems can also be linked to visualisation software, enabling stakeholders to 'see' the results of different ecological analyses. The factor limiting the development of these models is not the technology but the availability of data. In particular, there is a lack of information on soils and vegetation at the landscape scale in a form which can be used for the modelling of future woodland cover. Inventories of soils, vegetation and other wildlife are needed to: determine the scope for future woodland; judge whether some habitats are best left unwooded; and provide the basis for design and management plans and the submission of grant applications.

In Scotland, digital vegetation and soil information exists in the form of the Land Cover of Scotland 1988 dataset and the national soil dataset (see Gray and Stone, Chapter 7; Hester *et al.*, Chapter 5) but similar national coverage of vegetation is not yet available for Wales and England in a usable form. There is a clear need therefore to address this problem.

The need for a flexible 'vision'

The adoption of a shared vision and the development of strategies for realising the vision seem to be important components of successful projects. However, the vision has to be flexible because nature is unpredictable, and chance events can have effects on the direction and nature of forest development. Harper emphasised (Chapter 11) that we must be prepared to modify plans in response.

Partnerships

Restoring woodland at the landscape scale is not cheap and rarely within the budget of individual organisations or landowners. Partnerships are usually essential mechanisms for securing realistic levels of funding; the range of LIFE-Nature projects part-funded by the European Union are good examples of successful partnership projects. There is also the potential for drawing down more funding from the Heritage Lottery Fund.

Securing agreement of stakeholders and ensuring community involvement

This can be expensive, but it is a necessary part of the planning process, and there is a need to strike a balance between too much and too little consultation. Too much consultation can generate excessive costs and inactivity; projects may grind to a halt because there are too many people involved in decision-making. In contrast, too little consultation can engender a feeling of lack of 'ownership' of the project among key contributors, leading to lack of support and possible opposition (Gray *et al.*, 2001).

Controlling grazing

Nearly all the case study schemes had to address the problem of over-grazing either by domestic stock and/or deer. While fencing and individual tree protection remain options in some areas, more innovative approaches are needed where fencing is deemed inappropriate on landscape or ecological grounds (i.e. where woodland grouse populations will be adversely affected) or where it is simply impractical. The restoration of wood pasture provides a particular challenge (Glimmerveen, Chapter 14) as there is a need to encourage natural regeneration while also maintaining the typical open structure of scattered mature trees. The foot-and-mouth disease outbreak which occurred since the conference has brought the whole question of levels of stock in the uplands into sharper focus.

More flexible and 'joined-up' incentives

The structure of the current grant schemes for woodland establishment has been delineated according to traditional land-use divisions, with agriculture and forestry seen as the responsibilities of separate government departments. Restoration at the landscape scale invariably means the inclusion of a range of different land-use types over a wide area, and habitat types such as wood pasture simply do not fit into the current grant structure. With the advent of devolution and the merging of government departments, the prospects for more 'joined-up' incentives have improved considerably, and indeed a number of grant schemes are currently under review (GLEAN, 2001b).

Dissemination of best practice

Practitioners rarely have the time to trawl the ecological literature for guidance on restoration at the landscape scale, and there is clearly a pressing needed to produce simple user-friendly advice. This might include:

1. An explanation of the Forest Habitat Network concept in an accessible form for a wide readership.
2. Guidance on how to plan Forest Habitat Networks at various scales while also recognising the limitations and uncertainties of the approach.
3. Advice on how best to access targeted incentive schemes.
4. Help with the use of, and interpretation of the output from, GIS modelling packages such as ESC (Ray *et al.*, Chapter 6) and the Native Woodland Model (Hester *et al.*, Chapter 5).
5. Suggestions as to how to integrate woodland restoration with the restoration of other habitats at the landscape scale.

Monitoring ecosystem development

Monitoring ecosystem development is an essential element in assessing the success of a restoration project. Often this is interpreted solely as a need to measure tree growth and survival, but other floral and faunal aspects of the ecosystem should be monitored as well, possibly using surrogate measures such as indicator species or vegetation structure (Ferris and Humphrey, 1999). Future monitoring should be planned at the outset of any scheme and linked with any initial survey work carried out. The MFST has produced guidance on monitoring for the schemes which it supports (Millenium Forest for Scotland Trust, 1999), but this approach needs to be extended more generally.

Final remarks

The conference attracted over 120 delegates, reflecting the growing interest in landscape scale restoration both in the UK and abroad. Although the number of restoration schemes in the UK may still be small in a world context, the prospects for achieving large-scale native woodland restoration seem to be better now than at any point in the last 100–150 years. With changes in agriculture and rural land-use gathering pace, significant opportunities will arise. Judging by the knowledge and practical experience demonstrated during the conference and in these proceedings, we are in an good position to make the best of these opportunities, and make a major contribution to UK biodiversity conservation.

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