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Managing Woodlands and their Mammals

Forest Research

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Proceedings of a symposium organised jointly by
The Mammal Society and the Forestry Commission

Edited by Chris Quine,
Richard Shore and Roger Trout

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THE Mammal SOCIETY

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Preface

This publication represents the proceedings of a symposium on 'Managing woodlands and their mammals' organised jointly by The Mammal Society and the Forestry Commission. The aim of the meeting was to bring mammalogists and foresters together to review knowledge, foster understanding and enhance co-operation between two enthusiastic groups interested in the welfare of the British countryside. The meeting was over-subscribed (approximately 250 people attended) and one of the aims in publishing these proceedings is to bring the knowledge to a wider audience.

The publication seeks to summarise the meeting and act as a source of information for those interested in the topic; all but one of the talks at the conference are presented here. Contributors were encouraged to provide an accessible overview, emphasise management recommendations, and provide further reading to supplement the references necessary to justify their statements. In addition to these suggestions, the web sites of the two sponsoring organisations will provide much useful background – for The Mammal Society www.mammal.org.uk, and for the Forestry Commission www.forestry.gov.uk (and its research arm Forest Reserach www.forestresearch.gov.uk).

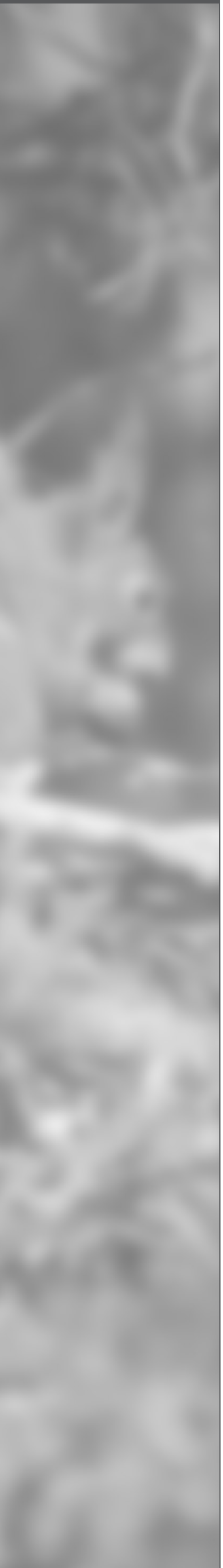
The papers reflect the intriguing variety of interactions between mammals and woodlands – spanning a variety of timescales (as far back as the last glaciation), locations, and woodland types. A number of common strands emerge – how much we know and yet how little we know; the complexity of management choices; the substantial shift in foresters attitudes to woodland mammals; and the ever-changing dynamics brought about by the shifting population and assemblage of British mammals interacting with the changing nature and extent of British woodlands. There was considerable cause for optimism with new approaches to woodland management and ambitious conservation plans for the future, including re-introductions and landscape-scale restoration and management. Nevertheless, there are some substantial and apparently intractable problems such as grey squirrels, and some dynamic situations, such as levels of grazing by domestic and wild animals, and disease incidence – emphasising the need for vigilance and further investigation.

The 19 papers are organised into 5 sections – providing background to the current interaction between woodlands and mammals; management to benefit mammals; management of conflicts between mammalian impacts and woodland objectives; complex interactions and finally a view of the future. While the papers provide many examples of practical application, such recommendations cannot be taken as representing the policy of the sponsoring organisations.

SECTION ONE

Background to managing woodlands and woodland mammals

CHAPTER 1



British woodland: a historical perspective

Keith Kirby

Summary

Trees and woods, ranging from pine-dominated forests in Scotland, to lime and hornbeam woods in the southeast, once covered most of Britain. Their structure and dynamics also varied by region with small-scale gap dynamics leading to relatively closed forest in some, while in others the activities of large herbivores may have produced a more open landscape of wood-pasture.

Significant clearance and management of forests took place from the Neolithic period onward. Management systems had been formalised by the medieval period, with coppice and wood-pasture management maintaining many elements of the former wildwood, but in separate less dynamic landscapes. Continued forest clearance increased the isolation and fragmentation of our woods up until the early 20th century.

History helps us to understand where our woodland wildlife has come from, but may not always be the best guide to the future treatment of our woods.

Introduction

Trees and woodland would cover much of Britain in the absence of humans. However, even with the expansion of forest cover that took place in the 20th century, woodland is currently a minor component of our landscape (*c.* 12%). In this paper I explore the nature of the former forest cover, and its transformation from the Neolithic period to the present day, and describe some implications for wildlife. The next ten years are likely to see further changes in woodland and how it is treated: the opportunities for biodiversity need to be realised.

The wildwood

During the last glaciation conditions over the whole of the Britain were unsuitable for tree growth. Therefore virtually all the species (both plant and animal) that we associate with our native woodland have had to spread back as climate conditions improved. Despite appearing immobile under current circumstances, these organisms must have (or have had) some capacity for significant long-distance spread given sufficient time periods.

Species colonised singly, not as pre-existing assemblages and for some species there may have been more than one point of entry. Broadly, Scots pine (*Pinus sylvestris*) and birch (*Betula* spp.) preceded oak (*Quercus* spp.), while lime (*Tilia* spp.) and hornbeam (*Carpinus betulus*) were relatively late arrivals, but at any one place this sequence would have been far less apparent.

By 7000 years ago most of our native trees and shrubs appear to have been present. Variations developed in the woodland composition across the country reflecting climate, soils and topographic patterns. While much of Britain is classed as within the temperate broadleaved vegetation zone, the central and northern Highlands are an outlier of European boreal forests. Eastern England could be classed as 'semi-arid' with less than 50 cm annual rainfall, while parts of western Britain fall into 'temperate rain forest zones' (250 cm rainfall). Within any one climatic zone the woodland that would have developed on the chalk would have differed from that on heavy clay soils, and topography would have influenced woodland development – for example, the woods of swamps and flood plains would differ from those at the tree line. The composition of our woods was therefore very varied (Peterken, 1996; Rackham, 1980).

Box 1.1 *Some sources of information about the former natural woodland cover. Note that all can only give partial information and that survival of remains may not be uniform across the landscape.*

Pollen analysis
Pollen records give a broad picture of the botanical composition of the former vegetation and its change over time. Early work using lake sediments gave only coarse, regional-scale patterns. More recent small hollow work allows stand-scale changes to be followed. Low growing and insect pollinated plants may however be under-represented.

Modern analogues
'Near-natural' forests in continental Europe and north America may provide models of how British natural woodland functioned. These forests have however been altered directly or indirectly by humans and have developed under slightly different climatic conditions, so cannot be taken as direct equivalents to the 'wildwood'.

Buried trees and forests
Remains of actual trees from the former forests have been recovered from bogs and estuaries.

Invertebrate remains
Remains of invertebrates in bog deposits and similar can be identified and conclusions drawn from them about the likely habitat conditions in the surrounding countryside.

Variations in woodland composition across the country have long been accepted, but the past structure of the woodland has become a particular subject for recent debate. What were the disturbance processes that shaped its dynamics? How much of it might have been open with only scattered trees and scrub rather than closed-canopy forest?

Different disturbance regimes probably operated in different types of woodland and at different times (Peterken, 1996). The 1987 great storm (Kirby and Buckley, 1994) illustrated how a single event could leave the trees undisturbed in one wood, create a scatter of small canopy gaps in another, while a third wood was largely flattened. Fire may have been a major factor in pinewoods (whether in Scotland or on bogs further south) but would have been unlikely to be significant in broadleaved woods on boulder clays.

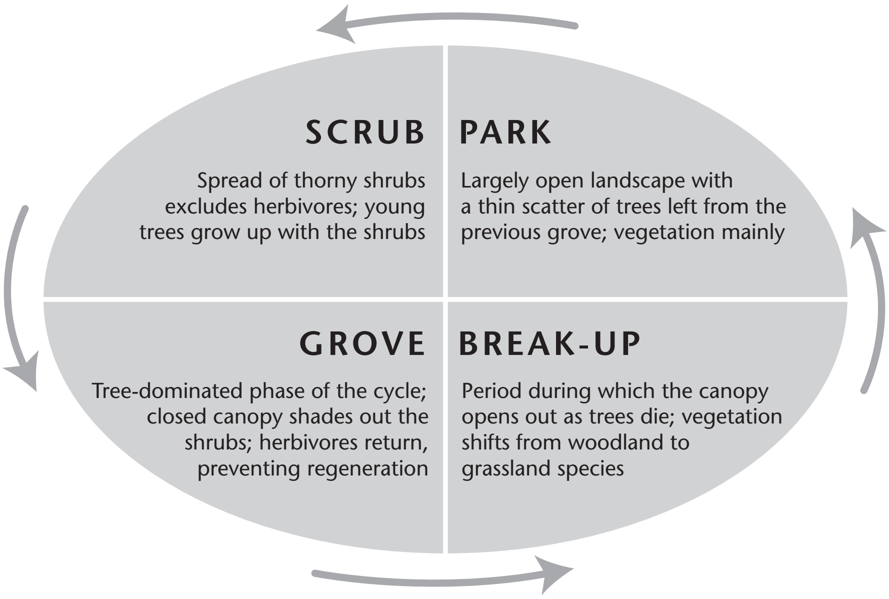
Frans Vera, (2000, 2002) has proposed that the role of large herbivores in driving the woodland regeneration cycle has been underestimated. This cycle (Figure 1.1) may have operated in parts of Britain although it does not automatically follow that these necessarily had a 'savannah-like' appearance. Rather there may have been a shifting pattern of glades and groves, some of the latter tens or hundreds of hectares in extent, with trees and woodland perhaps covering more than half the landscape (Kirby, 2003). Such a mosaic would allow continuity across time and space both for species of the open phase (glades) and of the wooded groves.

Forest clearance

Mesolithic peoples almost certainly had an impact on the forest cover in which they lived. Certainly by the Neolithic period there are indications that woodland may have been managed as coppice (Rackham, 1980; 1986) and quite large areas may have been cleared of trees, for example in the downlands. Many heathlands seem to have originated in the Bronze Age; traces of large-scale Iron Age ranching-systems have been found in Yorkshire. The pollen records show increasing amounts of grasses and non-woodland herbs.

At the time of the Roman invasion, much of Britain was already given over to agriculture. However, forest clearance was not necessarily a once and for all event, spreading inexorably across the country.

Figure 1.1 *Vera's cyclical turnover of vegetation in response to large-herbivore activity (after Vera, 2000, modified Kirby, 2003).*



Patches of woodland might escape clearance and management because they were relatively inaccessible or because they fell within the borderlands between different groups of peoples. Local recolonisation and forest regrowth would occur if areas were abandoned because of war, famine or disease. There is evidence from archaeological timbers that large diameter trees with narrow ring widths, typical of old growth high forest, could still be found close to London in the Dark Ages. The rapidity with which both deforestation and subsequent regrowth can happen has been illustrated over the past 150 years in the Eastern States of America (Whitney, 1994).

Formalisation of 'traditional woodland management'

From the 11th century the nature of our woodland becomes increasingly well documented. Rackham (1980) used the entries in the Domesday Book to estimate a total woodland cover of only about 15% for England (excluding some northern counties). Wales and Scotland were also relatively well populated at this time, so it is unlikely that woodland cover in these countries would have been very much greater (Linnard, 1982; Smout, 2002).

The Domesday entries also make it clear that woodland was generally owned and managed in well-defined ways, typically either as coppice or wood-pasture. These management systems remained widespread throughout the medieval period. There are various accounts of the methods by which coppice was cut, the regrowth protected, and who had the rights to graze or lop the pollards in particular wood-pastures. In some instances, e.g. at Bradfield Woods (Suffolk) and Hatfield Forest (Essex) these practices do seem to have been maintained fairly consistently. However, at other sites (Bernwood Forest, Oxon) the management of the woods was much more irregular, with periods of neglect, and considerable variation in the coppice cycle length (Kirby and Watkins, 1998).

Centuries of change 1700–1900

Although traditional management helped to maintain continuity of structure in some woods from the medieval period through to the 19th century, there was an ongoing reduction in woodland cover: in England from about 15% in 1100 AD, to about 4% in 1900 AD. Furthermore, particularly in the second half of the 19th century, the structure of many woods was transformed to high forest, reflecting the shifts in demands for wood products, an increasing use of softwood, and reliance on coal and coke for fuel rather than charcoal.

A wide range of animals and plants were introduced to our woods, of which the most notorious are perhaps grey squirrel (*Sciurus carolinensis*) and rhododendron (*Rhododendron ponticum*). Another change that was particularly significant for the distribution of many mammals and birds was the increased interest in game management.

20th century – a turning point

The first half of the 20th century saw the final collapse of wood-pasture and coppice as significant elements of the rural economy, except in a few places. Stimulated by the emergencies of the First World War the Forestry Commission was established; new forests, primarily of introduced conifers, were planted on a massive scale to meet timber production needs. The nature conservation movement expanded and, after the Second World War, agencies within government were formed to promote wildlife conservation in our woods (Sheail, 1998).

A key concept established in the 1970s was the distinction between ancient woods (those where there is a continuity of some kind of woodland cover back to *c.* 1600 AD) and recent sites where tree cover has been established on open ground in the past 400 years. The former were found to be much more likely to contain rare species and features (Peterken, 1981).

Later papers, particularly Chapter 3, in this publication cover subsequent development of forestry policy and practice.

Some implications of our woodland history for modern conservation:

- Woodland cover was formerly extensive and there would have been a high degree of continuity of habitat across the landscape in post-glacial times. The doubling of woodland cover over the last century (to 8.4% for England, 12% for Great Britain; Forestry Commission, 2001), has reduced overall woodland fragmentation and isolation, but this positive change is partially offset by loss of hedgerows, meadows, heaths and other semi-natural habitats in the countryside.

- Continuity of woodland cover by area over time has been lost except for the minority of woods that are ancient (2.7% in England; Spencer and Kirby, 1992). Veteran trees outside of woodland provide additional links to the past, but these are frequently in highly modified landscapes of improved grassland and arable, isolated from other elements of the woodland cycle.

- Ancient woods are widely scattered and occur in most ten-kilometre squares. They span the range of soils and climate in Britain and have probably retained much of the variation in the broad tree and shrub composition found in the wildwood. However, woodland types of lowland floodplains and tree lines have almost completely disappeared. To the remnants of the former natural woodland is now added a range of new woodland types created by afforestation, particularly upland spruce forests, that have no historical precedent.

- The wood-pasture and coppice systems may between them contain most of the structural elements, albeit in different abundances, that were present in the wildwood. However, these systems tend to have been perpetuated separately on different sites. These structural elements have become rather fossilised in their occurrence and distribution. There is far less dynamism in many of our woods than would have been the case under natural conditions.

- Most ancient woods are small and isolated: species within them are therefore more vulnerable to extinction through chance events; and recolonisation is less certain than when the woods were more connected.

Even within ancient semi-natural woods, changes are occurring through the impact of introduced species, e.g. sycamore (*Acer pseudoplatanus*), rhododendron, muntjac (*Muntiacus reevesii*) and grey squirrel that have no historical analogue.

Conclusions

There are reasons to be optimistic about the prospects for woodland conservation, despite the losses to semi-natural woodland habitats and species that have occurred in the past 50 years. The forestry and nature conservation sectors increasingly share objectives. More people now care about our woods, and what goes on in them, than was the case even 30 years ago. The management of all types of woodland is more likely to include conservation as one of its objectives. Plantations on ancient woodland sites are being restored to native trees and shrubs. Species are being re-introduced to areas from which they had been lost.

Conservation policy and practice is moving beyond a narrow focus on protected sites to consideration of whole landscapes. Working at a larger scale provides opportunities to explore a wider range of management options (including re-uniting the habitats currently separated in pasture-woods and coppices). Novel and large-scale solutions may be needed to cope with the implications of climate change for our woodland wildlife. In moving on we must not forget where our woods have come from. Recent work on the role of large herbivores in natural forest systems is exciting and highlights the potential significance of wood pasture. However, many features in our woods are the products of cultural rather than natural processes. We may or may not choose to perpetuate them everywhere but that decision should be based on an understanding of their significance. Moreover, not everything has historical precedents. The story of how our woods were managed in the past can help us to understand their present condition, but it is not always the best guide to their future management.

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CHAPTER 2

The use of woodlands by mammals – past and present

Derek Yalden

Summary

The present interactions between mammals and forestry need to be considered in the light of historical information on how they came about; this is particularly true given the current emphasis on creating something more like a 'natural' broadleaved woodland cover in lowland Britain. Just what was that 'natural' cover, what mammals inhabited it, and how have six millennia of human interference changed it?

The changing mammal fauna

The maximum of the Devensian (last) Glaciation about 20 000 years ago saw a high arctic fauna in Britain of lemmings, reindeer and musk ox. Neither trees nor any of the forest mammals could have lived there. The record of the pollen rain shows first birch scrub, then hazel and pine, followed by oak, elm and alder recolonising over about 2 000 years from 10 000 to 8 000 years ago, by which time the English Channel had probably been flooded, forming the British Isles. From about 8000 to 5500 years ago (when Neolithic farmers and their livestock appeared, and started to interfere seriously with the landscape), high forest of species such as oak (especially in the west and north), lime and elm (in lowland England), ash (on chalk and limestone), and alder and willow (in the river valleys) blanketed most of Britain. In Highland Scotland, and as altitudinal bands around higher mountains further south, Scots pine would have persisted, with birch scrub even higher uphill and further north (Bennett, 1988). Only on very high northern mountains would open ground, perhaps analogous to moorland, have been evident.

If most of Britain was blanketed by high forest, what exactly would that high forest have looked like? We know, from the archaeological record of such sites as Star Carr in Yorkshire and Thatcham in Berkshire, that Mesolithic hunters were pursuing a large mammal fauna that included roe and red deer, elk, aurochsen, wild boar and beaver. Though archaeological remains of elk are few, those of aurochsen are quite numerous, as are red deer (Yalden, 1999). High tree foliage could not have supported these; there must have been river valley grasslands, glades where soil conditions or accidents of windthrow or lightning provided grazing, and considerable amounts of low-level scrub and young growth to support browsers. Indeed, the mammals themselves must have played a role in creating (beavers) or maintaining (large ungulates) such conditions, and the pollen analysts point out that the pollen rain, which creates the illusion of almost complete forest cover, is itself deceptive. Because most trees are wind-pollinated, they produce copious amounts of pollen which therefore dominates the pollen rain. A fairer assessment of the actual vegetation associated with any particular pollen flora has to be derived by adjusting the figures: dividing birch, alder, pine and hazel by 4, multiplying lime by 4, grasses by 3.33, heather by 5 and sedges by 2 (Maroo and Yalden, 2000). On this basis, the high forest seems to have included rather more grassland, and could have supported the mammal fauna. Deciduous woodland is likely to have covered 43.2% of the landscape (cf. 3.7% now), but there might have been about 19.3% grassland.

Many of the larger mammals here then are now extinct, perhaps hunted out, but perhaps because their habitat has gone. Red deer and roe deer survive, and in sufficient numbers to cause problems for forest management, though they too were nearly exterminated in the 17th century. That red deer prefer a forest habitat is indicated by their greater densities and better reproductive rate in woodlands (see Figure 7.2 in Clutton-Brock and Albon, 1989). We may regard roe deer as marauders of farmland, using woodland to provide shelter, but they take territories that neatly divide the woodland between themselves, so they certainly regard it as their prime habitat. Other surviving forest mammals include the red squirrel, now reduced to a remnant of its former range and population size, but then much more widespread, and responding not only to the density of pine cones in its favoured conifer habitat, but also to the hazel crop (but not the acorn crop) in deciduous woodlands (Kenward *et al.*, 1998). Hazel dormice too are now reduced to a remnant of their former range and abundance, perhaps out-competed for hazel nuts by introduced grey squirrels, but dependent on a range of flowers, fruits and berries, and making much greater use of tree-holes for nesting than we formerly suspected (Bright and Morris, 1992; 1993). Tree holes are also critical to a number of woodland bats, among them some of our rarest species – Bechstein's bat, barbastelle and Leisler's bat – while foraging conditions inside woodland are also the preferred hunting grounds for several of them, including lesser horseshoe and Bechstein's bats. Others, including Leisler's bat, prefer to forage outside or around, rather than within, woodland.

Population changes

If we take some guidance from the nearest analogue to our Mesolithic countryside, the Bialowieza Forest in eastern Poland, together with what we know from the archaeological record, we can hypothesise about the Mesolithic mammal fauna and the extent of the changes since then. We do not know enough about the densities of bats, then or now, to estimate their population changes, though we know that Bechstein's bat used to be more widespread, and relatively much more numerous, in earlier times (Yalden, 1999). For the other mammals, we estimate that numerically, small rodents and insectivores dominated the fauna then, just as now, though bank voles and common shrews would have been the most numerous, rather than field voles. However, the greatest ecological impact would have come from the large biomass of large ungulates already mentioned (Maroo and Yalden, 2000). There would perhaps have been 1.2 million red deer, rather than 0.36 million, 0.8 rather than 0.5 million roe, and in addition 0.9 million wild boar, 80 thousand aurochs and 60 thousand elk. There might have been 11.8 million red squirrels, rather than 0.16 million, and 25.8 million dormice rather than only 0.5 million. Numerically, the mammals might have then totalled 535 million, rather than 281 million (53%), but their biomass has reduced rather more, by loss of the larger species, from 304 kt to 129 kt (42%); the modern figures include an introduced mammal biomass (e.g. rabbits, fallow deer, brown hares) about equal to that of the surviving natives. However, this latter figure still overlooks the enormous biomass of ourselves and our domestic species, and the total biomass of mammals in the British countryside is now some 6743 kt, 2218% of what it then was. It is not surprising that our woodland mammals are squeezed into a rather small segment of the countryside, nor that, when they become too numerous within it, or extend outside it, we notice them and categorise them as pest species.

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CHAPTER 3

An overview of contemporary woodland management

Bob McIntosh

Summary

This paper gives a broad overview of contemporary woodland management to provide context for others that look in more detail at woodland management and its effects on wildlife. It describes some of the policy drivers and the impact these are having on the nature of our woodland. Four topics of particular concern, representing current challenges for woodland management, are highlighted.

Introduction

The nature and character of any woodland is a function of many influences but principally it is the result of a combination of:

- the effects of the forestry policy drivers, including incentives and legal constraints;
- the site limitations imposed by bioclimatic factors;
- the objectives of the owner and hence the management techniques and intensity adopted.

These factors have combined in different ways over the centuries and will no doubt continue to do so; the lifespan of trees dictates that changes in practice often take some considerable time to be apparent. The concentration of the remainder of this paper is on policy and management – further information on site limitations is well summarised in the Ecological Site Classification (Pyatt *et al.*, 2001), while the Forestry Commission Handbook 6, *Forest Practice* (Hibberd, 1991) gives a general introduction to management techniques.

Current forestry policy

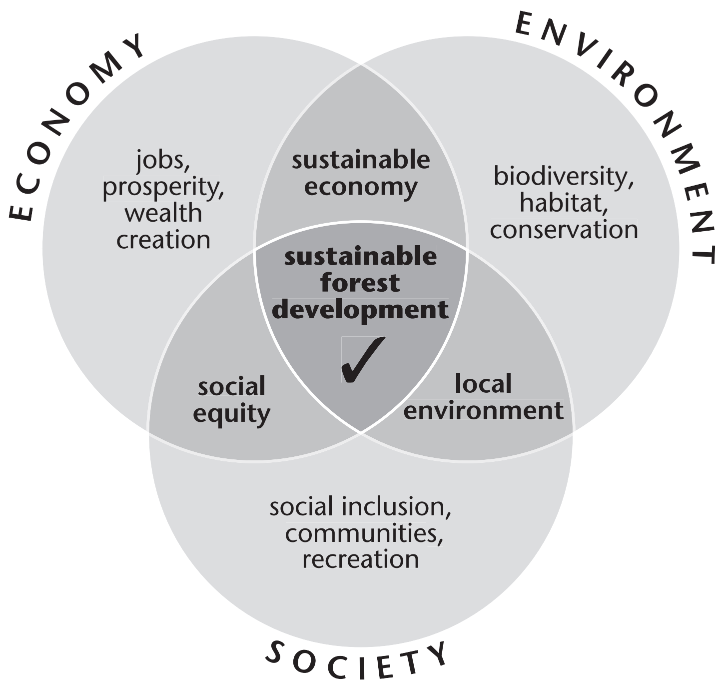
Forestry in GB is a devolved subject. The three countries have recently agreed to continue the system where the Forestry Commission delivers on behalf of the three ministers but each country has control over forestry policy. However, each country (plus the Northern Ireland administration) does subscribe to the international protocols and commitments to sustainable forest management adopted by the UK.

The UK Forestry Standard is the top-level policy document (Forestry Commission, 2004). The Standard sets out the UK Government approach to sustainable forest management within the context of European and Global protocols for sustainable development. Beneath this sit the individual country forestry policies as outlined in the country forestry strategies (Forestry Commission, 1998, 2000, 2001a). These reflect views on how *The UK Forestry Standard* should be interpreted and applied in each country. At present, they are relatively similar but greater diversity can be expected with time (after devolution), and the emergence of different priorities and emphases. The first signs of this can be seen in the new forestry grant schemes being developed in Scotland, England and Wales.

Sustainable forest management

Although the emphasis may differ from country to country, all 3 strategies are firmly based around the principle of sustainable forest management and the production of social, economic and environmental outputs (Figure 3.1).

Figure 3.1 *A schematic of sustainable forest management.*



The outputs can be summarised as:

- Economic: in the form of timber production and other forest products.
- Social: in the form of access and recreation opportunities, the creation of attractive surroundings in which people live, a greater degree of stakeholder involvement in the decision making process, and the production of greater benefits for local communities.
- Environmental: in the form of protection and enhancement of special sites, the enhancement of key species and habitats and the creation of more biologically diverse forests.

The priorities for contemporary forest management therefore appear to be:

- to ensure that it is soundly based on the principles of sustainable forest management (SFM)
- to make the decision-making processes more inclusive and transparent
- to think about outcomes rather than outputs.

The latter means that the emphasis is not so much on outputs, such as timber production or hectares planted or number of picnic sites, but on the ultimate outcomes – rural development, improvement in the health and well-being of the population, increased biodiversity.

The most difficult job for forest managers is to find the appropriate balance of objectives in each situation, particularly when there is no satisfactory way of putting a monetary value on social and environmental outputs.

Recent changes and the current character of GB forests

The woodland area of Britain has increased over the past century and is now approx 2.7 million hectares, of which 0.8 million hectares is in state ownership and 1.9 million hectares is in private (Forestry Commission, 2001b). Woodland expansion is continuing at about 17 000 ha per year, but there have been significant changes in the character of the new woodlands. In the latter part of the 20th century much of the expansion was of coniferous plantations in the uplands, planted largely for timber production. Recently, there have been a much higher proportion of non timber-related schemes, often involving broadleaved tree planting on high quality former agricultural land.

Two recent types of planting deserve special mention:

- There has been a significant expansion in the creation of woodlands around towns, particularly by the public sector and also by targeted grant incentives.

- There have been a significant number of large new planting schemes aimed at creating new 'native woodlands' – with particular success in establishing substantial areas of new native pinewood in Scotland.

The community woodlands present particular challenges and opportunities. Significant areas of new woodland are being created on the edge of urban communities, on both former industrial sites and on former high-grade agricultural land. These woodlands are being designed with a high proportion of broadleaves and open space, with a high degree of public involvement and with a view to a high degree of public usage. They will become excellent habitat for a wealth of wildlife and present an exciting opportunity to re-connect urban communities with woodlands and the wildlife supported by them. They may also, however, bring some interesting challenges associated with the need for wildlife management in such situations, for example of deer and grey squirrels.

Management of existing woodlands

Management of existing woodlands is being influenced by a more inclusive decision-making process. Much of this is based on the production of long-term forest design plans which chart the future nature of the woodland. The production of these plans seeks to reach a consensus on the appropriate balance of objectives for the woodlands in question.

The planning process is leading to an increase in both species and structural diversity within woodlands. This has been particularly marked in the uplands where many single-species, even-aged plantations were established in former times.

Restructuring is ensuring the gradual conversion of large uniform plantations to more diverse and interesting forests through careful design and management – within the constraints imposed in the uplands by infertile soils and exposed site conditions. (McIntosh, 1995).

Current challenges for forest management

The discussion above indicates a number of positive changes to woodland management, many of which should generally be viewed as being beneficial as regards the creation of diverse habitats for the future. However, there are many challenges and I will highlight a number of current issues which reflect some of the remaining difficulties and uncertainties.

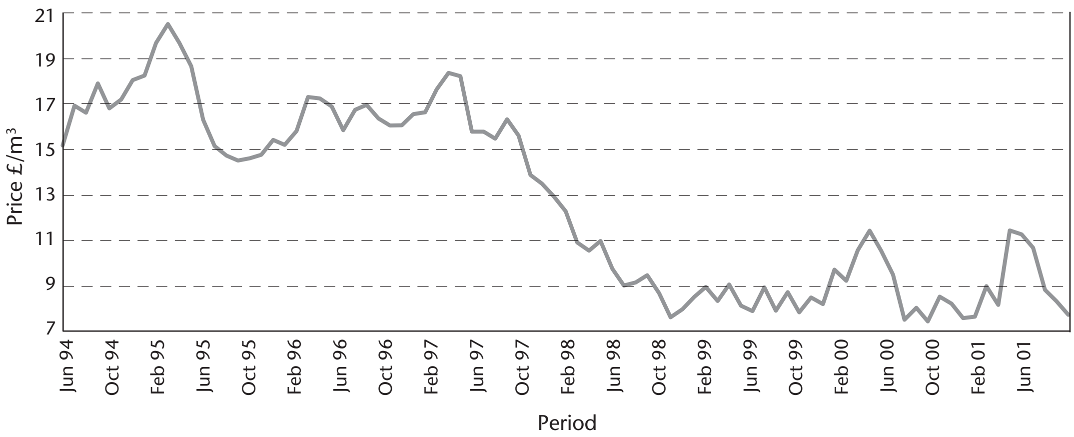
Timber prices

Timber prices in the UK are at their lowest ever level and look like remaining so for the foreseeable future (Figure 3.2). There are many reasons, including the global nature of the timber trade, timber supplies from former Eastern Bloc countries, and the impact of paper recycling. The main effect has been to weaken the economic base of forestry. This has in a way helped to focus more interest on the social and environmental aspects of forestry but it has highlighted the fact that the traditional forestry operations cannot currently pay for the delivery of the wider non-market benefits. It seems inevitable that public funding will have to be increased if woodland owners are to be encouraged to produce more public benefits.

Naturalness

There is much current interest in 'naturalness', in terms of both species and woodland structure. This is reflected in a push for more use of native species in new planting and the replacement of exotics with natives at restocking. There is increasing interest in the use of local seed origins too and these pressures are in direct conflict with the commercial pressures which reflect the dominant requirement

Figure 3.2 *Index of monthly standing sales (timber sold to merchants whilst still standing in the forest not at roadside) illustrating decline in value of timber over the period 1994–2001.*



for softwoods by the UK forest industry and the superior form and growth rate of foreign provenances of native species. This will continue to be an interesting challenge.

With respect to forest structure, low-impact silvicultural systems are perceived as being more natural and less intrusive. There has been a significant increase in the area of woodland where some form of continuous cover management system is planned. Such systems are less disruptive visually and give rise to more stable habitat than clearfelling systems. However, it is important to remember that the processes operating in Britain's natural forests, long ago when such forests existed, may have been driven by periodic perturbations in the form of fire or windblow. In many cases, a patch clearcutting system is a better mimic of such natural processes. Many of our native woodland species would have evolved in such woodlands and may indeed be disadvantaged by continuous cover forestry systems which maintain land under high forest cover. This has been recognised, for example, in Thetford Forest, by English Nature who requested a continuance of clearfelling because of the importance of restock areas to woodlarks and nightjars.

Impact of deer

The third issue I have chosen to highlight is deer, which other papers will cover in more detail. However, the key issue for the forest manager is that deer species are increasing both in range and in numbers and are having a major influence on woodland development. Deer populations are responding to the increase in woodland area and to the increasing diversity of woodland habitats. There has long been concern over their effect on planted tree species but there is growing concern about their effect on other plant communities within the forest and the general biodiversity of woodlands. Maintaining deer density at an acceptable level is a difficult and expensive business, particularly if the same management objectives are not shared by adjoining owners.

Management or neglect

The final challenge is the extent of woodland in this country which is effectively unmanaged. This is perhaps a particular issue in the south of England where there are extensive areas of mainly broadleaved woodland where no management takes place at all. Conscious decisions to leave woodland unmanaged can bring benefits, but in other circumstances the relative benefits and disbenefits are unclear. Some such woodlands will be a valuable reserve of important species but the lack of management is probably adversely affecting some species groups, e.g. butterflies, which rely on more mixed age woodland with periodic opening of the canopy. Further information is probably required to assess the magnitude of this issue.

Conclusions

In conclusion, there is no doubt that the general direction of woodland management is positive. Changes in policy and practice will result in forests and woodlands which become increasingly diverse and interesting with time, and in which biodiversity and wildlife management are viewed as equally important as the economic and social aspects.

Transforming our woodlands to more diverse composition and structure is a long process and foresters embark on that journey without a really clear idea of the endpoint. The mixed conifer-dominated forests in the uplands represent a completely new habitat which has not previously existed in this country. There is still much to learn about how best to develop them and how our resident wildlife will adapt and react to this new habitat. We do need more research and more understanding of the effect of management practices on woodland wildlife, and some of the priorities are identified in subsequent papers in this volume.

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SECTION TWO

Management of woodlands to benefit mammals

CHAPTER 4

Management of broadleaved and coniferous woodland to benefit hazel dormice

Fiona Sanderson, Paul Bright and Roger Trout

Summary

Research into the effects of woodland management on the hazel dormouse (*Muscardinus avellanarius*), a priority species in the UK Biodiversity Action Plan, has been carried out over a number of years at a large number of mainly broadleaved woodlands. Results from this work suggest that traditional coppice-with-standards is likely to provide more benefit to dormouse populations in oak (e.g. National Vegetation Classification W10) than in ash (e.g. W8) woods. Dormice are also known to inhabit coniferous woods and recent research shows that they make use of the conifer-dominated stands as well as the broadleaved wood edge habitat. Planted ancient woodland restoration may therefore impact upon dormice and research on the effects of such restoration on dormouse populations is ongoing. A list of woodland management guidelines is presented.

Why manage woodlands for dormice?

The hazel dormouse (*Muscardinus avellanarius*) is a priority species in the UK Biodiversity Action Plan and is protected by both UK and European law. It is considered near-threatened globally by the World Conservation Union. Since the late 1800s it has disappeared from six counties in Britain, about half of its former distributional range. This has been due to the loss, fragmentation and deterioration in quality of its woodland and hedgerow habitat and probably climate change (for a review see Bright and Morris, 1996). In most landscapes in Britain dormice probably occur mainly in ancient woodlands, i.e. those existing since at least 1600 AD (Spencer and Kirby, 1992). But, where habitat (especially hedgerow) connectivity permits, this strongly arboreal species is also resident in recent woodland, hedgerows and scrub (Bright, Mitchell and Morris, 1994; Bright, 1996). The majority of ancient woodlands in lowland Britain (i.e. where dormice occur) have in the past been subject to coppice woodland management (Rackham, 1986). This creates a sere of wooded habitats that can be highly suitable for dormice.

The dormouse is a sequential specialist feeder on ephemeral food sources, largely tree and shrub flowers, fruits and phytophagus insects (Bright and Morris, 1993; for a list of valuable plant species in dormouse habitat see Bright, Morris and Mitchell-Jones, 1996). Dormice therefore require woodlands and hedgerows with high tree and shrub diversity to provide them with a continuous succession of food sources. Dormouse abundance is often highest in mid-aged coppice (6–10 years of re-growth) (Bright and Morris, 1990), apparently because food biomass and diversity, and hence continuity of availability, is highest in this age of coppice. However, some woodlands not coppiced for at least 20 years also provided excellent habitat for dormice, provided the understorey was unshaded (Bright and Morris, 1990).

The management of lowland ancient woodlands in Britain underwent two great changes during the 20th century. Firstly coppicing was discontinued nearly everywhere, save for some sweet chestnut coppice in Kent and Sussex (Peterken, 1992). This resulted in the woodland understorey, where dormice find much of their food, becoming heavily shaded by canopy trees and much less productive of flowers, fruits and insects. Growth of brambles in the field layer was also strongly suppressed. Food availability for dormice would thus have been greatly reduced (Bright and Morris, 1990). Shading also resulted in shrubs assuming a dominantly vertical growth habit, which probably hinders the efficient movement of dormice; the sprawling growth habitat assumed by unshaded hazel shrubs is likely to be more beneficial (Bright and Morris, 1990). Secondly, much lowland ancient woodland was felled and replanted with non site-native coniferous trees. Felling *per se* probably led to the local extinction of dormouse populations, which are small, temporally highly variable and therefore very vulnerable to loss (Bright and Morris, 1996). Conversion to conifers would almost certainly have removed much of the dormouse's food supply; radio tracking, where it has been possible to identify tree species that dormice utilise, showed that conifers were seldom visited (P. W. Bright and T. Smithson, unpublished; note however that dormice readily construct nests in coniferous trees – see below). Furthermore, commercial coniferous woodland management is usually at a much larger spatial scale than traditional coppicing. This has almost certainly been inimical to the needs of dormice, whose home ranges are small, largely irrespective of habitat quality (Bright and Morris, 1991; 1992, P. W. Bright, unpublished).

Data from 83 sites throughout the dormouse range in Britain (part of the National Dormouse Monitoring Programme, NDMP; Bright and Morris, 1996) show a 19% decline in dormouse abundance between 1991 and 2000 (Sanderson and Bright, in preparation). This decline may in part be due to climate change, as populations in northern England have declined much more (40% between 1993 and 2000; cf. Bright and Morris, 1996). It also probably results from (meta) populations reaching equilibrium at the landscape scale following the loss of ancient woodlands and hedgerows in the 20th century. However the change in management of ancient woodlands, cessation of coppicing and replanting with alien conifers, has almost certainly had the greatest impact on dormouse populations nationally (Bright and Morris, 1990; Bright and Morris, 1996). Thus woodland management has been, and will continue to be, of pivotal importance to dormouse conservation.

Dormice and woodland management

Coppicing and dormouse populations

Current recommendations for managers suggest that coppicing is beneficial to dormice provided it is done on a long rotation in small, widely distributed compartments (Bright and Morris, 1990; Bright, Morris and Mitchell-Jones, 1996). However it is clear that dormouse populations do not respond to coppicing in the same way at all sites (Bright and Morris, 1990). Furthermore, current management recommendations are based on a short-term study, confined to sites in the southwest of England. Here we summarise research recently undertaken at Royal Holloway, University of London on a much larger, longer-term dormouse population data set with national coverage. We examined the influence of coppicing on dormouse populations at 29 broadleaved woodland sites from the NDMP over 5 years (1996–2000). The study included both derelict and active coppice sites and compartments. A very large proportion of UK dormouse sites are comprised of such habitat.

Study sites were classified as either 'oak', which is largely analogous to the National Vegetation Classification (NVC) W10 category (*Quercus robur–Pteridium aquilinum–Rubus fruticosus* woodland), or 'hazel', which corresponds to the W8 category (*Fraxinus excelsior–Acer campestre–Mercurialis perennis* woodland; for details of NVC in woodland see Rodwell *et al*., 1991). W8 and W10 do not cover the whole spectrum of dormouse habitats in broadleaved woodland, but they represent a useful shorthand for managers who already have NVC information.

Clear differences were found in dormouse response to coppicing between oak and hazel sites. Both hazel and oak woodlands that had a high proportion of young coppice (less than six years old) supported relatively few adult dormice at the beginning of summer, prior to breeding. Hazel woods with a greater proportion of mid-aged (6–25 years) coppice were also associated with lower dormouse numbers. However, in oak woods, it was sites with a high proportion of mid-aged coppice that had higher dormouse densities. This difference between oak and hazel woods is likely to be related to food supply. Where coppicing is greater than 25 years old, hazel and other shrubs produce less dormouse food in oak woods than in hazel woods, probably because of heavier shading. We found hazel to be significantly less productive of hazel nuts at oak sites, supporting this conclusion. Coppicing may therefore not provide great benefit to dormice at hazel sites, as an already abundant food source (hazelnuts) is made unavailable to them until understorey shrubs are sufficiently mature to fruit again. Large-scale coppicing is unlikely to benefit dormice in this type of woodland, though a small amount of coppicing to encourage brambles to fruit is likely to be useful. Coppicing is clearly highly beneficial at oak sites.

Analysis of dormouse abundance within coppice compartments showed that they make some use of very young coppice, so this stage of coppice regrowth is not entirely lost to them. However, they were more likely to be found in older coppice with 9–25 years of regrowth. Maintaining a mosaic of coppice age classes apparently enhanced juvenile abundance at oak sites. This implies that it is best not to coppice adjacent compartments in consecutive years, as a variety of age classes close together is needed. Dormice do not have large ranges (usually less than 1 ha in the course of a whole summer) and coppicing in a mosaic will retain a greater variety of food sources close together.

It should also be noted that more dormice were found in woods that had an extensive field layer, suggesting the need for some form of deer fencing around young coppice to protect the field layer from over-browsing.

Dormice in conifer plantations

It has been known for a number of years that dormice inhabit coniferous and mixed woodland as well as broadleaved woods. Forest Research is currently investigating how dormice use conifer plantations, particularly plantations on ancient woodland sites (PAWS) in order to inform management policies aimed at restoring broadleaved woodland. Ongoing studies in the west of England showed that dormice use stands of plantation conifer as well as making use of the broadleaved edge habitat. Radio tracking revealed that dormice spent up to 75% of their days in natural nests in conifer trees in summer and that they used overhead branches to cross forest tracks. In the future, this work will look at the effects of various management regimes and associated disturbance in order to assess the best approach to allow foresters to thin conifers and restore broadleaved woodland while retaining dormice.

Recommendations for managers and researchers

Broadleaved woodland

- All major management operations should take place during hibernation (November–March) to avoid disturbance of breeding animals.

- The coppicing of oak, W10, sites on a 15–25 year rotation is recommended. Coppicing should be carried out in a mosaic fashion – adjacent compartments should not be coppiced in consecutive years. It is advisable to retain at least 10 ha of mature hazel in a wood at any one time (Bright *et al.*, 1996).

- Standard trees that provide food should be retained at low density, e.g. oak, wild cherry.

- Some form of deer fencing (possibly brash or temporary deer fencing would be sufficient if budgets are tight) is strongly recommended to promote a well-developed field layer to provide food and above-ground pathways and allow rapid regeneration of coppiced shrubs.

- At hazel, W8, sites, non-intervention should be considered if hazel is still fruiting heavily. However, a few small (0.1 ha), widely dispersed coppiced compartments to promote growth of brambles will be beneficial to dormice.

- If a woodland site is small (less than 20 ha), it may form part of a network of dormouse-occupied woods where hedgerow connectivity is probably vital to allow the population to be maintained. Where woods and hedgerows are managed together in the landscape, we recommend that 30% of hedgerows are left uncut for at least seven years to allow food plants to fruit. For further details of hedgerow management, see Bright and MacPherson (2002).

Coniferous woodland

- Avoid clearfelling of PAWS that support dormouse populations.
- Maintain aerial connectivity where possible, by retaining branches crossing perimeter tracks and retaining branch contact with large broadleaves.
- Provide connectivity by pushing trees or brash across new racks (the rows of trees removed to allow machinery access) at approximately 100 m intervals following timber extraction.
- Scallop edges when widening rides.
- Dormice may use stacked timber as a hibernation site. Stack logs on the ride side away from the managed compartment and avoid moving timber in winter to prevent disturbance of hibernating animals. Cut ride edges of the compartment to be managed in autumn to discourage hibernation there.
- Our provisional advice is that management operations should take place in autumn or mid-winter to avoid disturbance of breeding animals.

These recommendations are necessarily brief but up-to-date, and the reader should consult the latest edition of the *Dormouse Conservation Handbook* for more detail.

Research recommendations

Continued monitoring of dormice in managed woodland sites using nestboxes as part of the NDMP is vital. Collection of accurate spatial management data concurrently would allow examination of the effects of various management regimes at more sites in the future. This is particularly important in the case of management of coniferous and mixed woodland, as little is known about the effects of such management on dormice and regimes are likely to change substantially in the near future.

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CHAPTER 5

Managing forests for red squirrels

Peter Lurz, John Gurnell and Steve Rushton

Summary

The red squirrel (*Sciurus vulgaris*) is considered endangered in large parts of Britain where it has been replaced by the alien North American grey squirrel (*Sciurus carolinensis*). In this paper, we first present an overview of red squirrel conservation strategies and red squirrel forest management guidelines. We then go on to emphasise the usefulness of minimum linked area (MLA) and spatially explicit population model (SEPM) approaches for informing forest managers in relation to managing red and grey squirrel populations.

Red squirrel decline

The loss of the native red squirrel, *Sciurus vulgaris*, throughout Britain as well as parts of Ireland and northern Italy is well documented (Gurnell and Pepper, 1993; Wauters *et al.,* 1997; Ó Teangana *et al*., 2000). Although habitat loss, habitat fragmentation, and parapox virus infection in Britain may have contributed to the loss (Rushton *et al.,* 2000, Sainsbury *et al.,* 2000), the main reason why the red squirrel has disappeared is the spread of the introduced North American grey squirrel (*Sciurus carolinensis*) (Gurnell and Mayle, 2003). A considerable amount of progress has been made in the past few years on understanding the processes involved in the replacement of red by grey squirrels (see Wauters and Gurnell, 1999; Wauters *et al*., 2000, 2001, 2002). Although this will not be reviewed here, it is clear that the ecological requirement for the persistence of red squirrels in an area is an absence of grey squirrels (Gurnell and Pepper, 1993), or, at the very least, that greys are maintained at sufficiently low numbers that they do not out-compete the native red squirrel (precisely what this level might be in different habitat types still requires further study).

Conservation strategy

Three conservation strategies have been identified (Gurnell and Pepper, 1993): the Area Exclusion Strategy, the Regional Defence Strategy, and the Forest Management Strategy. In brief, these are:

- The Area Exclusion Strategy. To protect relatively small areas against grey squirrels where only red squirrels exist, and where the geography of the areas provides some natural barriers to the immigration of grey squirrels.

- The Regional Defence Strategy. To defend regions where red squirrels still exist against grey squirrel presence. In this case, it may not be possible to eliminate grey squirrels completely but numbers would be kept low over a long period using appropriate control methods.

- The Forest Management Strategy. To design and manage forests in a way that are favourable to red squirrels, but that are unfavourable to grey squirrels. Most suitable areas are conifer forests without large-seeded broadleaves such as the Sitka spruce dominated forests in the north of England and Scotland.

Strategies 1 and 2 are reliant on controlling grey squirrels, and strategy 3 may involve some targeted grey squirrel control. Methods and the legal requirements for controlling grey squirrels have been published (Forestry Commission, 1998; Gurnell, 1999; Gurnell and Mayle, 2003; see also Chapter 9).

Importantly, only live trapping methods should be used for grey squirrel control where red squirrels are at risk, and, since red squirrels may inadvertently be captured, a licence to trap is required under the Wildlife and Countryside Act (1981). Management recommendations for strategy 3 have been outlined (Gurnell and Pepper, 1993; Lurz *et al.*, 1998; Pepper and Patterson, 1998).

Managing forests for red squirrels

Controlling grey squirrels for red squirrel conservation requires a large input of money and resources, particularly manpower, that need to be sustained over long periods of time (Gurnell and Steele, 2002). Thus, managing forests in such a way as to favour red squirrels and discourage grey squirrels is generally believed to be the best way forward, bearing in mind that forest managers are faced with achieving the often conflicting requirements of timber production, wildlife conservation and recreation within what are increasingly becoming multi-functional forests. Broadleaved and mixed broadleaved forests hold little prospect for red squirrels in the face of invading grey squirrels (e.g. Gurnell, 1987; Skelcher, 1997) and so we believe that the design and management of large areas of conifer forest of suitable species and age structure hold the key to the long-term survival of red squirrels in Britain.

Current guidelines for managing forests for red squirrel conservation are outlined in Table 5.1. Tree seed abundance drives the ecology of squirrels (Gurnell, 1983; Lurz *et al.*, 2000) and management decisions need to be defined in terms of harvesting, thinning and restocking operations against the backdrop of natural fluctuations in seed crops and the potential for grey squirrel competition. This is a complex procedure, especially as forest design plans need to look 20 or more years into the future. In addition, forests have to be viewed in the landscape context, particularly with respect to the infiltration of grey squirrels. This makes it difficult to assess the effects of changes in forest design over the rotation period on red squirrels by looking at forest harvesting and restocking maps and applying general guidelines to the local situation. To help understand the current status and future trends in red and grey squirrel distribution and abundance within a landscape, and therefore to inform management decision regarding forest design and harvesting, we have developed population models linked to geographic information systems (GIS).

In particular, we have developed two types of model: the first is called the minimum linked area (MLA) model and the second the spatially explicit population model (SEPM). The models are based on 10 years of fieldwork on red and grey squirrel ecology in spruce-dominated plantations in the north of England and experiences collected at Thetford Forest in east England and Clocaenog Forest in north Wales. Some of the studies which demonstrate the utility of these models are outlined in Table 5.2. Both approaches use information provided by digitised forest stock maps and can be used to investigate the impact of different forest design plans on red squirrel population viability. In order to implement the models, expert knowledge of the ecology of red and grey squirrels in different types of forest habitat is required.

A minimum linked area (MLA) encompasses suitable sub-compartments of trees necessary to maintain one or more resident squirrels. It can be used to determine the total available area of suitable habitat that will hold a viable population of red squirrels for current and future design plans. A detailed description of the methodology and application has been prepared (Gurnell *et al.*, 2002a). Spatially explicit population models (SEPM) are process-based models and simulate the dynamics of populations through deaths, births and dispersal in the landscape and the distribution and population viability of red squirrels arises as an emergent property. We successfully applied the approach in a pilot project at Kidland Forest in the north of England (Lurz *et al.*, 2003). Currently, we are working on a Mammal Trust UK supported project in collaboration with Forestry Commission (Kielder Forest District) in the north of England to identify the areas that are most suitable for red squirrels and those areas that are most vulnerable to grey squirrel incursion. This will enable us to advise on the current forest design and management plans in terms of conserving red squirrels, and to develop, in partnership with forest managers, a long-term management strategy. Kielder Forest District contains the largest and most important remaining strongholds for red squirrels in the north of England.

Table 5.1 *General guidelines for the conservation management of red squirrels in British conifer forests (modified from Gurnell & Pepper, 1988, 1993; also see Pepper & Patterson, 1998). These guidelines should be adapted to the requirements of individual forests. For example, although it is recommended that large areas of forest >2000 ha are desirable for red squirrel habitat management, smaller forests than this may be acceptable. However, the smaller the forest, the more likely it is that grey squirrel control will be necessary around and within the designated area, unless there are natural barriers to incursion.*

1. Conservation plan

The conservation of red squirrels should be incorporated into the other management objectives of the forest. The strategy should be to manage forest habitat to encourage red squirrels and discourage greys. Areas of forest may be designated as red squirrel reserves if not all the forest is going to be targeted for red squirrel conservation.

2. Habitat management

a) **Area** – set aside a large area of forest (e.g. >2000 ha) and, if necessary, designate as a Red Squirrel Reserve.

b) **Shape** – round, rather than long and thin, to minimise incursions of grey squirrels.

c) **Boundary areas** – at least 3 km of conifer forest or open land must be established as a buffer against the infiltration of grey squirrels.

d) **Broadleaved trees** – avoid planting or selecting areas containing large-seeded broadleaves (e.g. beech, oak, sweet chestnut) and remove existing mature trees if acceptable and compatible with policy and management objectives. Broadleaves should be confined to small-seeded species, e.g. willow, aspen, birch and rowan.

e) **Cover and tree seed availability** – in addition to the main commercial crop, other species (pine and spruce, larch, fir) should be planted in groups or ribbons along ride edges and in small patches within the forest. This will improve cover and seed availability. This is particularly important in Sitka spruce dominated plantations to provide squirrels with a continuous food supply throughout the whole year.

f) **Forest structure** – young and middle-aged plantations (e.g. 16 to 30 years) which provide food and cover are preferred by red squirrels. At least 50% to 60% of the forest should be of seed-bearing age. In a commercial timber forest, a forest structure should be developed such that <3rd of the forest is below the age that a good cone crop is produced, a 3rd is 'middle-aged' and a 3rd older than this (including long-term retentions), depending upon the length of the rotation (e.g. for Scots pine in Thetford Forest, East Anglia, respective ages may be <15 years, 16–30 years and >30 years).

g) **Harvesting** – leave some good single seed trees over a clearfell, and groups of trees for nesting sites. When possible, avoid harvesting during the breeding season, April to October, and avoid clearfelling large contiguous blocks.

h) **Connectivity** – seed-producing areas should be connected by continuous strips of trees to prevent isolation and facilitate movement between them.

i) **Thinning** – if possible, first thinnings to crops should be deferred to about 30 years of age. 1-in-4, 2-in-8 or 2-in-10 row thinning may benefit red squirrels by maintaining a closed canopy cover in the unthinned strips. Line thinning is preferred to selective thinning. Thinning should try and avoid gaps in the canopy of 4 m or more so that red squirrels can move through the canopy rather than coming to ground. Try and thin small blocks at a time and, when possible, avoid thinning during the breeding season, April and October.

3. Monitoring

Set up a regular monitoring programme to provide information on relative abundance and habitat use of red and grey squirrels (see Gurnell *et al.,* 2001).

Table 5.2 *Examples of the application of minimum linked area (MLA) and spatially explicit population model (SEPM) approaches to assess red squirrel population viability, predict grey squirrel spread and assist with red squirrel conservation.*

1. MLA approach	
a)	**Thetford, east England** – evaluated red squirrel population size in the red squirrel reserve in relation to changes in habitat suitability for different management prescriptions (Gurnell and Steele, 2002)
b)	**Clocaenog, Wales** – assessed current and future habitat suitability for red and grey squirrels based on potential felling and restocking scenarios (Cartmel, 2000; Gurnell *et al.,* 2002a).
2. SEPM approach	
a)	**East England** – described the squirrel SEPM and investigated the spread of greys in Norfolk and their impact on red squirrel decline (Rushton *et al.,* 1997).
b)	**Isle of Wight, England** – investigated the distribution of red squirrels on the island. Results suggested that dispersal was restricted on the island, and woodland size and the distance to the nearest next woodland were important factors explaining red squirrel presence (Rushton *et al.*, 1999a).
c)	**Britain** – identified potential areas to target red squirrel conservation (Rushton *et al.*, 1999).
d)	**East England** – modelled the spatial dynamics of parapox virus disease in red and grey squirrels (Rushton *et al.,* 2000).
e)	**Piedmont, Italy** – used the model to predict the likely future spread of grey squirrels in northern Italy and their possible range expansion into France and other neighbouring countries (Lurz *et al.*, 2001).
f)	**Island of Jersey, Channel Islands** – examined the conservation management of red squirrels on the island (Gurnell *et al.,* 2002b).
g)	**Britain** – modelled the impacts and costs of grey squirrel control regimes on the viability of red squirrel populations (Rushton *et al.,* 1999b, 2002).

General recommendations

- The selection of conservation areas needs to be carried out at the landscape level to target the most suitable forests and woods and to minimise potential conflicts with other interests (e.g. timber production, amenity and recreation, other conservation needs).
- There is a requirement for close co-operation between species experts and forest managers.
- General forest management guidelines are available. Management advice needs to be flexible and adapted to the local situation.
- Changes in economic or other circumstances may require reassessments from time to time of current management plans with regard to harvesting, thinning and restocking.
- Appropriate spatially-linked population models should be used to inform management decisions with respect to forest design and management.

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CHAPTER 6

Managing woodlands in the presence of otters

Rob Strachan, Geoff Liles and Tom Fairfield

Summary

Otters will make use of all watercourses and often find resting and den sites in wet woodlands, carr and adjacent woodland habitat if linked with strong linear features such as ditches or dense hedgerows. Woodland blocks may contain features for secure natal holts, particularly when disturbance is low. An otter holt is a protected feature and otters have been known to breed in every month of the year. Forestry operations may pose a real threat to the breeding success of otters. However, forestry operations need not be at odds with nature conservation and can be carried out to benefit otters. The key to success is planning, and there are a number of sources of advice. In addition, the Woodland Improvement Grant for biodiversity provides an opportunity to establish or re-establish sensitive management and specific features that benefit otters. New native woodland planting can be targeted to link up existing woodland and can also include wet woodland.

Introduction

Unlike some species of conservation concern, the otter (*Lutra lutra*) is regarded as a success story, currently making a comeback after a dramatic population crash in the 1960s and 70s. This was largely attributed to the widespread agricultural use of organo-chlorine insecticides and their subsequent pollution of watercourses. The otter is not generally thought of as a woodland species because it can survive in many riverine systems and historically, woodland management practices have rarely taken the species into consideration.

In this paper we first make the case for the otter as a species that readily uses woodland. We have reviewed the literature to assess the importance of woodlands within the home range of an otter; the features within woodlands that are used by otters, and the potential impact of woodland management. We recommend best practice procedure to benefit otters, and describe how the UK Forestry Standard and associated grant aid can be used to assist otters.

Woodland otters

Unlike many species of conservation concern, often restricted to tight habitat requirements and small home ranges, the otter is an extensive traveller and explorer and generally considered to be a multi-habitat user. In one radio-tracking study in Scotland a male travelled over the entire upper catchment of a river, totalling 84 km of stream length (Durbin, 1993). The average linear length of home range for otters is around 40 km for males and 20 km for females (Kruuk, 1995). The large scale at which otters operate is largely dictated by the opportunities to find food, and the most significant determinant of otter usage of freshwater habitats was abundance of prey (Durbin, 1993; Kruuk *et al.,* 1993).

Although otters are quite happy on rivers, reedbeds and coasts in the absence of trees, bankside trees and [especially wet] woodlands, if they are present, will be used by otters. Trees beside watercourses are important in determining river habitat quality as they provide shelter, shading, nutrient input for prey and bank stability. Of some 6000 River Habitat Survey (Environment Agency) reference sites, over half support semi-continuous or continuous tree cover along one or both riverbanks (Raven *et al.,* 1998). More than a third of the sites had extensive shading of the channel. Around half of the sites had exposed bankside tree roots. Underwater roots, fallen trees and natural stick piles were also noted at many sites, offering potential resting-places for otters. Extensive areas of carr woodland, however was recorded as a scarce habitat comprising only 3.5% of the total sites (Raven *et al.,* 1998). Isolated patches of woodland and scrub cover may be readily used by otters, particularly in areas of intensive agriculture, as they may represent the few places where they can best seek refuge.

Wet woodland and carr

Wet woodland and carr habitat is characterised by trees such as alders, birch and willows; a high water-table and a field layer of sedges, reeds, ferns and bramble. Although a relatively scarce habitat, radio-telemetry studies of otters in lowland England clearly demonstrate its importance: between 55–82% of day resting sites within an individuals home range were located in wet woodland and carr habitats (Mitchell-Jones *et al.,* 1984; Roberts, unpublished; Strachan and Bonesi, unpublished).

What is so important about wet woodland and carr? Otters find them easily as they are generally connected to watercourses, lie within the floodplain and have an abundance of natural cover, such as greater tussock sedges and common reed. They often contain fallen trees with raised rootplates, providing features suitable for dens and may support seasonally abundant prey such as frogs.

Radio-telemetry has also shown the lying-up places that otters use (Green *et al.,* 1984; Mitchell-Jones *et al.,* 1984; Durbin, 1993; Kruuk *et al.,* 1998; Roberts, unpublished; Strachan and Bonesi, unpublished). These have revealed that between 60–80% of day resting sites were actually above ground in couches among reeds, tussocks, shrubs or flood debris (stick piles). Only 20–40% of resting sites conform to the general idea of an otter holt as an underground feature – among tree roots, boulders, field drains or burrows.

Tree cover and diet

We summarised 12 diet studies of otters in the UK (excluding coastal otters) and found that, as expected, fish formed the major component of their diet, with frogs being seasonally hunted. Energetically, it has been calculated that an otter will consume 1–1.5 kg fish each day – approximately 15% body weight (Kruuk, 1995).

This begs the question – does riverbank tree cover influence foraging opportunities for otters? One consequence of woodland is that the presence of overhanging branches of trees significantly adds to the availability of invertebrate prey for fish populations, with up to a ten-fold increase in prey items when compared with riparian banks with no trees (Mason and Macdonald, 1982). Submerged tree roots, debris dams and leaf litter also provide refugia for fish and crayfish and provide foraging areas for otters. Thus, riverbank tree cover can positively influence where otters hunt for prey. Conversely, coniferous trees overhanging stream tend to contribute to acidification and a consequent paucity of fish prey – despite also having insects in the canopy.

Holts and natal dens

Woody branch debris, fallen trees and raised rootplates often occur as bankside features. On some rivers stick piles may become substantial features after flood events and offer convenient places of shelter for otters. The scouring effect of dynamic rivers may hollow out cavities below the roots of bankside trees and underwater entrances may make the hollows particularly secure for otters.

Old trees may also develop hollow trunks in which otters can conceal themselves. Holts such as these may be selected as natal dens. However, some observations suggest that natal dens may occur some distance away from major rivers or areas frequently used by other otters, in order to better protect the cubs.

Otters can be encouraged by building artificial holts. Evidence for their use has come from a study that examined hair samples from bedding found during log-pile holt rebuilding and refurbishment. Thirteen out of the 19 holts investigated (68.5%) on English and Welsh rivers had been used by otters (Cowell *et al.,* 2001). Note that once an artificial holt is being used by otters, it is a protected feature within the woodland. Under the Wildlife and Countryside Act, otters, and their resting-places are fully protected and intentional or reckless destruction or disturbance of holts could result in prosecution.

So when are the cubs born? We summarised the recorded birth dates into two-monthly periods at 168 riparian sites in Britain (Data from Shetlands and coastal areas have been excluded). Although there are more observations in the winter period, this was not statistically significant and births can occur throughout the year (Figure 6.1).

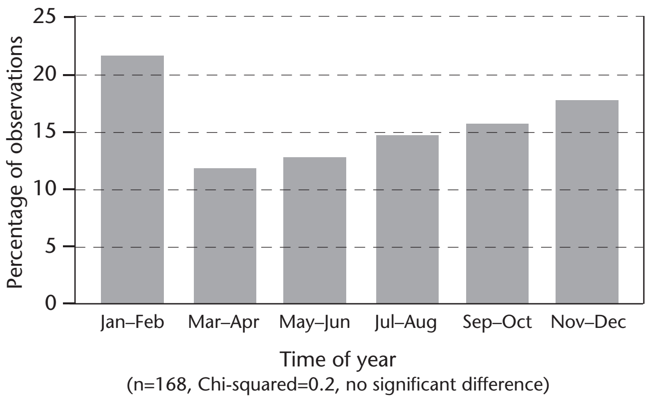

Figure 6.1

Recorded and extrapolated birth dates of otters in British river systems (Data from the Shetlands and coastal areas have been excluded).

As a consequence of their aseasonal breeding, natal holts in woodland need to be considered in terms of the potential impacts of forestry operations throughout the year. Female otters may be particularly vulnerable to disturbance around natal holts, leading to cubs being abandoned. Cubs may be especially vulnerable for the first six months after birth. Due care must also be used to ensure that no holts are destroyed.

Best practice woodland management

Potential impacts of forestry operations include direct loss of habitat (e.g. through clearfelling), damage to otter holts, and disturbance at or near holt sites. An indirect impact might be degradation of watercourses through soil run-off (diffuse pollution) affecting the prey base for otters.

Best practice guidelines already exist by way of the UK Forestry Standard (Forestry Commission, 2004) and its criteria for sustainable woodland management. Under the UK Forestry Standard the various species and habitats subject to EU Directives are recognised for conservation action or habitat enhancement. Designated sites and sensitive areas for threatened or rare species are clearly recorded and protected.

UK Forestry Standard – Criteria for sustainable woodland management includes:

- Ensuring protection of watercourses and aquatic ecosystems
- Ensuring general protection of wildlife and vulnerable features
- Making specific arrangements for protected species and habitats (including avoiding tree harvest during breeding seasons)
- Considering landscape in new plantings
- Preventing soil erosion during operations

The UK Forestry Standard recognises the otter as a flagship species of conservation concern and so the presence of this species has bearing on how forestry operations should be carried out. An important element in the whole process is a good survey and ecological appraisal designed to pick up the presence of protected species such as the otter. The key to success is planning, and the detailed survey work should underpin management objectives and proposals (including monitoring). The plan of operations will then accommodate any mitigation requirements that may be needed to minimise potential impacts on any otters present.

Woodland grant schemes provide opportunities to establish features for otters. A good example of this is *The Jigsaw Challenge* (a pilot project based on the Clun and Northwest Herefordshire Hills Natural Area). This is designed to encourage new native woodland planting adjacent to existing semi-natural woods. A feature of the project is a target for planting or regeneration of wet woodland adjacent or near to watercourses. Species that may be planted and will provide good cover for otters include alder, willows, ash, blackthorn, hawthorn, buckthorn and dogwood. Linking of woodlands along riparian corridors, and their sensitive management, will be of direct benefit to otters (and many other species).

Recommendations for managing woodland with otters

Thinning

- Thin out mature trees to ensure that sunlight reaches the understorey to encourage a dense field layer that otters may use as resting sites.
- Rhododendron, gorse and other scrub cover plants may be used by otters, so always check for evidence before thinning or eradicating rhododendron.
- Re-coppice mature stools and protect them initially from browsing to encourage a proliferation of young growth and additional cover for otters.
- Cut branches and brashings could be used to create stick-piles for otters.
- Encourage a continuity of thick scrub with species such as bramble.
- Ensure that young coppice and areas of scrub have minimal disturbance.

Felling

- Survey riparian woodland blocks before felling – check for signs of otters and look for active holts among tree roots.
- If otters are present, avoid direct disturbance and phase the felling operation to clear last any areas used by otters.
- Once felled, the timber should be moved away from watercourses for stacking as the stacks may be used as denning sites if left in-situ for any length of time.
- Carefully remove conifers from near watercourses.

Restocking

- Plantation woodlands may play an important role, especially when at the thicket stage, as young plantations may be favoured by otters for lying up above ground (couches may be encountered among tussock vegetation around the young trees).
- Re-plant or encourage the regeneration of shrubby and coppice species and encourage wet woodland areas where practicable.
- Species that may provide cover for otters include ash, willows, hazel, blackthorn, hawthorn, field maple, spindle, buckthorn and dogwood.
- Ensure that replanted areas and associated scrub have minimal disturbance.
- Make sure that conifer crops are not planted beside waterways.

Stock management

- Fence out livestock (there are grants available for new fencing, gates and hanging water-gates) as cattle and sheep can prevent an understorey of dense vegetation which provides otter resting sites.
- Deer management, possibly through physical barriers, will also encourage understorey growth.

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CHAPTER 7

Reintroducing pine martens – habitat constraints and enhancement opportunities

Johnny Birks, John Messenger, Steve Rushton and Peter Lurz

Summary

The pine marten (*Martes martes*) has a restricted distribution in Britain. While populations in Scotland are expanding, those in England and Wales are not and attention has recently focused on measures to promote the recovery of species' the there (Birks, 2002). The availability of suitable habitat is critical to recovery; in this paper we question the assumption that all woodlands provide the resources (food, cover and dens) required for successful recovery. We introduce a landscape suitability model based on the availability of foraging habitat and den sites for successful breeding. Recommendations for habitat improvements are proposed.

Pine marten habitat preference

The pine marten evolved as a habitat specialist. Its extraordinary agility enables it to exploit three-dimensional habitats in which climbing facilitates predator avoidance and allows access to elevated resting sites and sources of food. Throughout its Eurasian range the pine marten is typically associated with extensive woodland, often showing an affinity for 'old growth' forest habitats with an abundance of deadwood (Brainerd, 1990). The low and fragmented nature of woodland cover in many parts of the British Isles must be viewed as a major constraint to the recovery of the species. However, more open landscapes may be occupied where rocky features offer alternative three-dimensional environments to woodland and trees (Webster, 2001).

Pine martens select resting and denning sites in response to predation risks and energetic constraints (Brainerd *et al.*, 1995; Zalewski, 1997). Because the red fox (*Vulpes vulpes*) is a significant predator (Helldin, 1998), pine martens tend to choose sites well above ground level. Within woodland most pine marten dens are, predictably, associated with trees. Arboreal sites commonly used in Europe include natural cavities, nesting chambers excavated by woodpeckers and squirrel dreys or bird nests (Zalewski, 1997). A distinction should be drawn between den sites used simply for resting and those used for the birth and initial rearing of young (natal dens). One would predict that natal dens tend to be located in structures offering greater shelter and insulation than those used only by resting adults.

Availability of natal dens

Research in Europe has identified the availability of natal (breeding) dens as a critical factor for pine marten populations; throughout their European range breeding female pine martens prefer arboreal cavities as natal dens (Achterberg *et al.*, 2000; Balharry, 1993; Brainerd *et al.*, 1995; Zalewski, 1997). Suitable cavities tend to be associated with relatively large, old trees with a substantial deadwood component. Some studies have concluded that arboreal cavities are essential to the success of pine marten populations (e.g. Brainerd *et al.*, 1995). One study concluded that fox predation may limit pine marten populations where arboreal cavities are scarce (Zalewski, 1997).

Following heavy felling during the 20th century and the establishment of extensive new plantations, the great majority of trees in British woodlands are less than 100 years old. Consequently, compared with near-natural woodlands, deadwood volumes are very low and this is especially apparent in recent plantations (Hodge and Peterken, 1998; Humphrey *et al*., 2002). On this evidence we predict that arboreal cavities suitable as natal dens for pine martens are rare in British woodlands, and especially so in commercial forests. Similarly, in Scandinavia modern forestry practices are known to reduce the availability of marten denning and resting sites (Brainerd *et al*., 1995).

Drilling by species such as woodpeckers increases the abundance of cavities in trees. However, the accessibility of woodpecker nesting chambers to pine martens depends upon the species involved. The black woodpecker (*Dryocopus martius*), that is absent from Britain, excavates nest chambers that are commonly used as natal dens by pine martens. The widespread presence of black woodpeckers on the European continent leads to the availability of potential pine marten natal dens across a wide range of woodland types. The absence of the species from Britain begs the question as to which sites are used for denning and breeding by pine martens here.

Radio-tracking studies have revealed that pine martens in Britain commonly occupy dens in crags or other rocky features, as well as nests and dreys in trees (Balharry, 1993; Bright and Smithson, 1997; Velander, 1986). However, there is little recent information on natal den sites used by breeding females in Britain. Many 19th and early 20th century naturalists wrote of the pine marten's dependence upon rocky landscapes for survival, with rock crevices providing alternatives to tree cavities as breeding sites (Webster, 2001). Moreover, there is a strong historical association between pine marten distribution at its lowest point in the early 1900s and areas of extreme rockiness. Although woodland cover in Britain has increased to 11.6% from 4% in the early 1900s, that increase is composed mainly of softwood plantations in which tree cavities are predictably rarest. We suggest, therefore, that there has been no concomitant increase in natal den availability except where new woodlands have encompassed suitable crags. Anecdotal evidence suggests that pine martens in both Scotland and Ireland increasingly use the roofs of buildings as natal dens (R. Raynor and P. Sleeman, personal communication); this probably reflects the scarcity of more natural natal den sites.

A landscape suitability model

We constructed a pine marten landscape suitability model based on the availability of preferred foraging habitat (woodland) and potential natal den sites (crags). We ran this model in four study areas, two in southwest Scotland, one in northern England and one in northeast Wales. Measures of high habitat suitability as predicted from the distribution of woodland adjacent to crags were good indicators of the presence of pine martens and the success or failure of past releases of the species into southwest Scotland. This corroborates our belief that in the absence of suitable trees within woodlands, rocky features are important as breeding sites for pine martens. We intend to publish full details of this landscape suitability model elsewhere.

Conclusions

British woodlands provide very few of the arboreal cavities preferred by pine martens as natal dens. When considering reintroductions this must be regarded as a major deficiency in the availability of preferred habitat for the species. In view of this deficiency we suggest that rocky landscapes, offering alternative natal den sites, remain important as core breeding areas for pine martens in Britain. We hypothesise that while pine martens may disperse to occupy woodlands away from such landscapes, the scarcity of secure natal den sites results in low breeding success, especially where fox populations are high. This hypothesis is consistent with some differences in the survival and performance of pine marten populations within and beyond extensive rocky landscapes in Britain through the 20th century. We suggest that further recovery of the pine marten in Britain will depend partly upon habitat improvements designed to increase the availability of suitable natal den sites within woodland.

Recommendations for enhancing woodland for breeding pine martens

There are measures that woodland managers can take to increase the provision of natal den sites in the short, medium and long term:

- In the long term (50–100 years), retention of a suitable density (minimum 1 per hectare) of large (>55 cm dbh) woodland trees into senescence will increase the volume of standing deadwood in which cavities suitable for pine martens may develop through natural processes (see the strategy proposed by Humphrey *et al.*, 2002). However, given the current age structure of most woodlands, such deadwood strategies may take many decades to improve natal den provision.

- In the medium term (10–50 years), arboreal cavity development may be accelerated through deliberate damage to large woodland trees (special attention should be paid to health and safety considerations). There is some experience of this approach in the United States (e.g. Bull and Partridge, 1986).

- Also in the medium term, natal den site availability may be enhanced by promoting the establishment of scrub and non-commercial woodland adjacent to areas of existing senescent trees (e.g. parkland, hedgerow trees, arboreta).

- In the short term (0–10 years), provision of artificial, purpose-built natal den sites is the only viable option in most woodlands. Natal den boxes should be constructed and erected in ways that meet the breeding female pine marten's need for insulation and protection from predators. In collaboration with Forest Enterprise, The Vincent Wildlife Trust is testing a new design of such a box in Scottish forests.

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CHAPTER 8

The importance of woodlands for bats

Henry Schofield and Patrick Fitzsimmons

Summary

The contrasting roosting and foraging ecology of three bat species, Bechstein's bat (*Myotis bechsteinii*), the barbastelle (*Barbastella barbastellus*) and the lesser horseshoe bat (*Rhinolophus hipposideros*) are used to illustrate the importance of broadleaved woodland to British bats. Current information only allows the most general advice to be given to woodland managers.

Introduction

Recent reductions in the size of radio-transmitters have allowed an increasing number of studies to be conducted into the ecology of British bats. With a notable exception (the serotine *Eptesicus serotinus)* the majority of studies have highlighted the importance of broadleaved woodland as habitat for bats. The high insect abundance of broadleaved woodlands provides bats with important foraging opportunities. In addition, many species roost in holes or cracks in trees.

The importance of broadleaved woodland

Bechstein's bat

Bechstein's bat is a true woodland mammal, roosting within hollow dead branches, old woodpecker holes or rot holes in old deciduous trees. A study of female Bechstein's bats in summer showed them to have small home ranges (minimum convex polygons 6.9–50.5 ha) and to forage close to their roosts (maximum foraging distances 310–960 m).

Compositional analysis of broad habitat classes ranked broadleaved woodland and water significantly over pasture, tree-lines hedgerows and conifer plantations. Analysis of woodland micro-habitat use demonstrated that this species selected areas of broadleaved woodland with closed canopy and well-developed understorey as its key foraging habitat. This makes them vulnerable to changes in woodland management, particularly conversion to woodland pasture.

Barbastelle

The barbastelle is a specialist moth predator. Although it feeds in dense woodland during the winter, for most of the year it forages away from woodland, favouring riverine habitats in the spring and hedgerows and meadows during the summer. However, barbastelles are dependent on woodland throughout the year as a source of roosts. Although a few colonies are known to use buildings in the summer and a few individuals use caves as hibernacula, the majority of colonies are found in woodland.

In warm summer months they are found roosting in cracks in the trunks of storm damaged or old decaying trees. They will sometimes continue to roost in these trees even after they have fallen. In the spring and autumn they roost under loose tree bark and in the depths of winter they hibernate deep within hollow trees.

This species is highly dependent on woodland managers allowing old or damaged trees to remain in situ.

The lesser horseshoe bat

In contrast to the previous species, the lesser horseshoe bat roosts in old buildings during the summer and caves and mines in winter. In a radio-telemetry study of this species in south Wales bats were shown to have ranges 12–53 ha. They foraged 50% of their time within 600 m of their roost. Compositional analysis showed that broadleaved woodland was their principal foraging habitat. A further study in mid-Wales demonstrated the importance of stands of broadleaved trees within 2 km of the maternity roost.

Conclusion and recommendations

With so many British bat species utilising woodland as their key foraging areas it would be surprising to find them all competing in the same area. There is considerable evidence that microhabitat use may be predicted from wing morphology but few radio-telemetry studies have attempted to identify or confirm the types of woodland microhabitat used by these species. This currently makes it difficult to supply woodland managers with detailed advice on how to adapt or maintain woodland for species known to use a particular area. Summary recommendations are:

- Broadleaved woodlands are the key foraging areas for most British bat species. Future countryside policies should include the concept of strategic connections between woods, including ancient woodlands.

- More work is required on the microhabitat use of woodlands by bats in order to inform stand management.

- Wherever possible woodland managers should retain large old trees and any standing dead wood as these provide many species of bat with a range of roosting opportunities.

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SECTION THREE

Conflicts between mammals and woodland managers

CHAPTER 9

Grey squirrel management in woodlands

Brenda Mayle

Summary

Since its initial introduction in the late 1800s the grey squirrel (*Sciurus carolinensis*) has spread rapidly throughout Britain. This paper presents an overview of the impacts of grey squirrels on woodlands and woodland biodiversity, cost implications and ways of reducing these impacts. Priorities for research are suggested.

Introduction

Grey squirrels were introduced to Britain from North America between 1876 and the 1920s. Since then they have rapidly expanded in number and range, displacing our native red squirrel. Their introduction and expansion has been well-documented (Middleton, 1931; Lloyd, 1983; Gurnell, 1987). They now occupy almost all of Wales and England, and are continuing to expand northwards through northern England and west and southwards in Scotland (Pepper *et al.*, 2001).

This paper reviews the ecological impacts of grey squirrels in Britain, how these impacts may be reduced, the cost implications, and priorities for future research. Best practice guidance on control of damage in woodlands is provided in (Mayle *et al.,* 2003). Guidance on control of grey squirrels to benefit red squirrels is provided in Forestry Commission (1998).

Grey squirrel impacts

Tree damage

Damage to trees by grey squirrels was noted soon after their introduction. They strip the outer bark of trees from late spring; this is usually late April to the end of July, but damage may extend to early September in some years. Damage levels vary considerably between years and between sites in the same year. Although damage has been reported for almost 40 species of broadleaved and conifer trees (Rowe and Gill, 1985), thin-barked species, particularly sycamore, beech, oak, sweet chestnut, pine, larch and Norway spruce, are most vulnerable (Table 9.1).

Planted and naturally regenerated trees of between 10–40 years of age are most at risk, especially vigorously growing trees with a good sap flow and easily stripped bark. Younger trees may be stripped when large enough to support the weight of a squirrel. Bark on the main stem of older trees is usually too thick to strip, but squirrels may still strip bark from thinner branches in the crown (Mayle *et al.*, 2003). Trees established in Farm Woodland and similar schemes during the late 1980s are now becoming vulnerable to damage and a number of damage reports have been received recently.

Results from surveys of squirrel damage since the 1950s (Table 9.1.) suggest an increase in reports of damage to conifer species. This may be associated with an increase in the area of conifer woodland planted over this period as well as an expansion in the range of the grey squirrel into upland areas where more conifers are grown. Regional differences in frequency of attack to different tree species have also been suggested (Rowe and Gill, 1985).

Species	Survey year			
	1954	1955	1984	2000
Beech	46	51	87	66
Sycamore	70	43	85	100
Oak	21	14	41	40
Scots pine	6	2		16
Larch	7	4		8
Lodgepole pine	N/A	N/A		33
Sweet chestnut	9	4		33
Norway spruce	N/A	N/A		16
Ash	10	16	36	33
Birch	15	12	9	
Sitka spruce	N/A	N/A		0
Poplar	3	0	29	

Table 9.1

Relative frequency of damage by grey squirrels revealed in Forestry Commission surveys.

Effects of damage

Bark stripping damage may lead to:

- timber degrade, from callusing of wounds leading to uneven growth of wood and poor structural strength, or stain or rotting agents entering the wound site.
- deformed stems, if the main stem is damaged and branches take over apical dominance leading to structural weakness at the fork.
- growth rate reduction (due to tree response to the damage, or loss of 30% of the canopy).
- loss of the tree if ring-barked, or if infection weakens the stem such that it snaps in high winds.

Oak, poplar, Scots pine and Norway spruce are particularly vulnerable to weakened stems snapping in the live crown. Occasionally, stem snap occurs lower down, as recently illustrated in the Forest of Dean in western England (R. Guest, personal communication). Damage to 40-year-old Norway spruce has resulted in trees breaking off 4–6 m from the ground, resulting in a total loss of timber. Broome and Johnson (2000) estimated the likely cost of squirrel damage to the forest industry by considering the cost of damage to the area of trees currently in the vulnerable age class (10–40 years old). For the three most vulnerable species alone (beech, sycamore and oak) losses were estimated, in terms of value at the end of the rotation, of up to £10 million.

Biodiversity and other impacts

The costs in terms of woodland biodiversity, conservation, landscape and recreation are more difficult to assess. Lack of recruitment of trees into the canopy is of particular concern in semi-natural beech woodlands. This may be accompanied by a loss of associated fungi and invertebrates. Grey squirrels may compete for food with native species such as the red squirrel and dormouse, and are known to take birds eggs and nestlings (Gurnell and Mayle, 2003; Hewson *et al.*, 2004). They are also implicated in the transmission of parapox virus, an infection fatal to red squirrels (Tompkins *et al.*, 2002). Damaged trees may also pose a safety risk in areas of public access.

Triggers of damage

The reason that grey squirrels strip bark does not appear to be directly related to hunger or thirst (Kenward, 1983). Although they do eat some of the soft inner bark, damage has occurred where food and water are not limited, also when mineral licks have been available. Damage severity appears to increase with the density of squirrels, especially juveniles, living near or moving into vulnerable woodland during the damage period. Grey squirrels generally have two breeding periods: December–February, with a second later in the summer (Figure 9.1.) with between 1–7 young per litter (mean = 3). However, recent research has confirmed that individual females may have up to 3 pregnancies between February and October. Juveniles from the first breeding season become active during the late spring/early summer and move into the main population. It is thought that the greater the numbers of squirrels, the more interactions there are between them. This leads to an increase in 'displacement' behaviours that are 'redirected' into stripping bark. Trees of vulnerable species and age-class will be most at risk if adjacent to mature mixed woodland habitats supporting high populations of squirrels. Damage appears to be triggered at densities of 4–5 squirrels per ha, a relatively low density given recorded densities of 17 per ha in mature mixed broadleaved woodland.

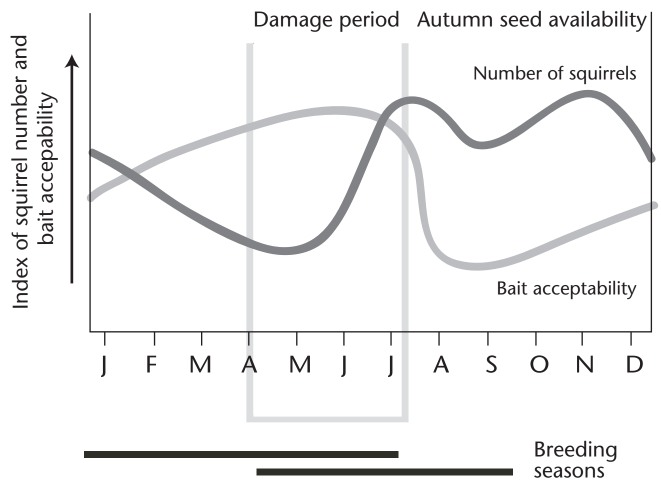

Figure 9.1

Relationship between grey squirrel density, breeding success, food availability and bait acceptability.

Reducing the impacts

Controlling grey squirrels

Unsuccessful attempts were made to eradicate the grey squirrel up to the mid 1950s. Now, the grey squirrel is frequently seen and enjoyed by the public and it is probable that eradication is not a desirable option even if it were practicable. Control to prevent damage to woodlands is therefore targeted at reducing populations around vulnerable areas just prior to and during the damage period. This is also the time when there is least natural food available and so bait acceptability is high (Figure 9.1). Starting control earlier than mid-March will have little influence on damage levels as squirrels are very mobile and rapidly recolonise cleared areas.

Control should be targeted in the mature woodland areas holding the squirrels rather than the damage vulnerable areas. The recommended methods are poisoning, using warfarin-treated wheat (unless red squirrels or pine martens are present), or live trapping (Mayle *et al.*, 2003). The warfarin bait is dispensed from hoppers designed specifically to allow access only to grey squirrels and may only be used legally between mid-March and mid-August. This reduces the risk to non-target species. The use of warfarin is generally more cost-effective because a larger area can be controlled by one operator. Once the hoppers have been sited, inspection is required between twice weekly and fortnightly, compared with the recommended twice daily inspection of live capture traps (minimum legal requirement once a day). Costs of control have recently been estimated as £7–11 per ha depending on the areas covered and method used.

In areas where game birds are reared, squirrels will be attracted to the bird feeding hoppers. This is likely to maintain higher overwinter densities of squirrels than would naturally be present. Squirrel control can be sited close to the hoppers, but also needs to be dispersed throughout the holding wood.

Future research

Control methods

More recently the use of immuno-contraceptives for grey squirrel control has been investigated (Pepper and Moore, 2001). This is an innovative technique, which has shown some initial promise. However, considerably more research is required into squirrel physiology and behaviour before a suitable vaccine can be developed that will have a sufficiently long-term effect on the squirrels, without risks to non-target species. Such research is costly and long term.

The continued availability of warfarin for grey squirrel control is currently under consideration by the European Union Plant Protection Directorate. Live trapping using multi-capture cage traps will become the most cost-effective method if warfarin is withdrawn. Current research is focusing on the development of simple predictive methods to improve targeting of control to years and sites when damage risk will be greatest. A better understanding of the relationship between seed crop availability, overwinter survival, spring breeding success, juvenile dispersal and damage the following summer is required (Gurnell, 1989). Studies are also seeking to improve bait attractiveness to squirrels where natural food availability may be less limited (e.g. in conifer areas).

The availability of data management systems such as geographical information systems (GIS) should allow managers to identify potential areas of risk (vulnerable species and age-class) in close proximity to areas likely to hold high densities of squirrels (mature mixed/broadleaved woodland). This should also allow better targeting of control. Monitoring of control efficacy through damage surveys in vulnerable areas following a control programme will also inform managers about the cost effectiveness of control.

Wider impacts

Grey squirrel impacts to woodland biodiversity both directly through predation and competition, and indirectly through changes to canopy composition and associated fauna, need to be evaluated. There have even been some suggestions that grey squirrel impacts to trees may be beneficial to woodland biodiversity by increasing the amount of deadwood present in the habitat. Initial studies are likely to focus on grey squirrel impacts on woodland birds (Hewson *et al.*, 2004).

Habitat management

Adjusting planting and management prescriptions to minimise the ease of bark removal and hence risk of damage to commercial trees have also been suggested (Kenward and Dutton, 1996), but little progress has been made to date. The high damage levels reported from Farm Woodland schemes suggest that these sites, with widely spaced, vigorously growing trees, may be particularly at risk. Studies are also required to evaluate the impact of 'continuous cover' woodland management systems on tree growth, bark character and grey squirrel damage levels.

Conclusions

The grey squirrel has become an established part of the British fauna. It continues to expand its range displacing the native red squirrel and causing damage to woodlands in terms of timber quality and, most probably, biodiversity. As a well-adapted and widely-distributed part of the British fauna, eradication is not a realistic option. Control to reduce woodland damage should be targeted in time and space both to increase cost-effectiveness, and to reduce the number of animals killed. Collaboration to ensure effective control should be encouraged between neighbouring landowners, game shooting and woodland management interests.

Lethal methods of control targeted at reducing impacts will continue to be the only option available to woodland managers unless methods such as fertility control can be developed to maintain low density populations which do not cause unacceptable impacts.

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CHAPTER 10

Controlling numbers of the edible dormouse in plantation forests

Pat Morris

Summary

The edible dormouse (*Glis glis*) has become a pest of plantation forestry in limited areas around the Chilterns, where it was introduced to this country 100 years ago. The animals will readily use nest boxes, but these are too costly for use as a means of removing the animals. A less expensive substitute has been developed (a 'nest tube'), offering a potential means of catching and controlling the numbers of *Glis*. However, woodland populations fluctuate widely, with reproductive failure in some years and extraordinary abundance in others. This may be linked to beech masting and could make control very difficult where beech trees are intermixed with other tree species. Possible widespread immigration to suitable forests in mast years is a further potential complication.

Introduction

Since its introduction to Tring in the Chilterns in 1902 (Morris, 2003), the edible dormouse (*Glis glis*) has not extended its distributional range more than about 30 km in any direction. However, illicit translocations, bypassing natural impediments to dispersal, make it likely that the species will become more widespread in the future (Morris, 1997). Where it does occur, the animal has become a significant forestry pest by gnawing plantation trees, particularly softwoods (Platt and Rowe, 1964; Jackson, 1994). Control has been difficult because trapping is sufficiently labour intensive that it is not cost-effective and no poisons have been type-approved for use on this species.

Studies on the native hazel dormouse (*Muscardinus avellanarius*) have shown that a significant proportion of the population may be attracted to nest boxes, especially in young woodland where secure nest sites such as natural tree cavities are scarce (Morris, Bright and Woods, 1990). Nest boxes have also been used successfully to assist studies of edible dormice in Bavaria (Mueller-Steiss, personal communication) and to remove them from nut and fruit plantations in Italy (Santini, 1978). In 1995, one hundred nest boxes were put up near Berkhamstead, in Hockeridge Wood (belonging to the Royal Forestry Society) to confirm that edible dormice would use them in Britain. They did (Morris, Temple and Jackson, 1997) and a population study has been carried out based on monthly monitoring of the boxes during each of the subsequent summers.

The nest boxes were of varying designs, but basically enclose a cavity about 10 cm x 15 cm and 30 cm high. Initially an entrance hole of 35 mm was incorporated, but within two years most of these had been enlarged by the animals to a diameter of 50–55 mm, suggesting that this is an attractive hole-size for *Glis*. In some years, nearly one third of the nest boxes can be found occupied in late summer, offering the prospect of large-scale removal of these animals if that was desired. However, wooden nest boxes are sufficiently expensive that their widespread deployment for pest control purposes would not be cost-effective, particularly in defence of relatively low-value softwood plantations.

An inexpensive substitute for wooden nest boxes is required if the principle of using artificial nest sites to catch this animal is to form a basis for its control. One such design, referred to here as a 'nest tube', has proved successful (Morris and Temple, 1998). This consists of a modified tubular tree guard that is both cheap and likely to be available to foresters in large numbers. These nest box substitutes cost less than £2.40 each, barely a quarter the price of their conventional wooden counterparts.

Design

Nest tubes are made from flat-sided tree guards, with an inner plywood tray slid inside the full length of the tube. The trays are about 60 cm long, made of exterior 3 ply, and fit tightly inside the width of the tree guard (approx 7.7 cm). Each tray has a vertical block of wood glued and nailed to one end. This forms a friction tight plug blocking one end of the tree guard. About 10–15 cm from this end plate is a transverse block of wood which forms a 'doorstep' partially closing off a nesting chamber between itself and the end block. Square tree guards are easier to adapt than the more commonly available round ones because of the relatively complex carpentry involved in closing off one end of the latter. This problem might be solved by cutting down discarded polycarbonate drink bottles of suitable dimensions to form the end plug. Each tree guard was shortened by 10 cm to allow the inner plywood tray to project that amount from one end to create a platform, facilitating entry, as plastic tree guards offer little purchase even for the gripping feet of dormice. Tubes were painted with black mastic roofing paint on the outside to darken the interior and to prolong the life of the plastic. Nest tubes were slung horizontally in two wire loops below a tree branch, about 2.5–3 m above the ground. They resemble a hollow branch, with the open end facing away from the tree trunk. An alternative arrangement is to mount the tube transversely across the tree, with about one third of the tube projecting each side of the trunk. The open end then appears like a broken-off hollow branch. Both arrangements allow easy inspection from a ladder.

Animals or nesting material are partially visible over the top of the 'doorstep' block inside, when viewed from the open end of the tube. A bag can be wrapped around the back end of an occupied tube. Pushing the platform part of the base tray into the tube ejects the animal and its nest out of the far end and into the bag. Blowing into the mouth of the tube also helps persuade the animal to leave the tube.

Testing nest tubes in the field

Fifty nest tubes were put up at Hockeridge Wood in May 1997, in mixed beech and conifer forest or very young spruce and young beech. Twenty tubes formed matched pairs with existing wooden boxes to permit comparison of the use of each type in the same area (the 'C Series'). The other 30 nest tubes were set out in three lines of 10 away from established wooden nest boxes (the 'F Series').

The conventional nest boxes and 50 nest tubes were checked monthly through the summers of 1997–2001 inclusive. The nest tubes were first occupied within 3 months of being put up, and among the first animals found was a female that had previously used a nearby wooden nest box and therefore must have made a deliberate choice in favour of the nest tube. The same female was found later with five nestlings, and another tube was occupied in August 1997 by a mother and her litter of seven. Both females appeared to have given birth in the nest tube, judging by the small size of the blind young (all less than 30 g). A few beech leaves had been installed as nest lining, but both nests were very wet after torrential rain. The plywood trays had been gnawed to release shavings to form nest-lining material. Another female ran from her wooden nest box to the nearby nest tube when disturbed. By September 1997, two other females were found using nest tubes, both of whom had previously occupied wooden nest boxes and therefore knew that these were available.

By October 1997, the first year of nest tube availability, five more nest tubes had been found occupied by single animals, including two juveniles newly dispersed from their family group. Seven of the F series of nest tubes were in use for the first time, three of them by newly independent juveniles and three by adult males that had not been caught before. One tube contained an adult female who had previously been caught about 100 m away in a wooden nest box in the summers of 1996 and 1997.

In summary, the dormice found and used the new nest tubes within three months and, where wooden nest boxes were each paired with a nest tube, there seemed to be no preference between them. Several animals had moved into tubes, despite the availability of wooden boxes nearby. In subsequent years, the tubes continued to be used, despite some of them becoming very wet inside, particularly after heavy rain, but also as an apparent result of condensation on the inside of the plastic surface. They were finally taken down in 2001 when some of the plywood trays had become very rotten and the plastic outer casings had started to degrade. During the four years 1998–2001, edible dormice used 14 of the 20 available wooden nest boxes in the ‘C series’ and 12 of the 20 nest tubes.

Nest tubes as tools for controlling edible dormice

Nest tubes are clearly acceptable to edible dormice as substitute tree holes in which to nest. They are inexpensive and could be installed in young plantations at risk from damage by gnawing dormice. They could be checked frequently during the summer until October (when most edible dormice will have entered hibernation), removing any animals present. If done repeatedly, population build-up could probably be minimised and after a few years vulnerable young trees might have grown old enough to reduce their susceptibility to attack, as edible dormice seem to do little damage in mature forests. *Glis* control might then no longer be needed.

However, catching the animals is not the end of the matter. *Glis glis* is a protected species (Wildlife and Countryside Act, 1981: Schedule 6) and removal of these animals from woodlands must be licensed. Release of live specimens into the wild is illegal (Wildlife and Countryside Act, 1981: Schedule 9), so they must be humanely destroyed although this appears illogical, if legal protection was desirable in the first place.

The problem of *Glis* as a forestry pest may actually be only a localised phenomenon, brought about by traditional planting practices. In the Chilterns, beech plantations have been established, using various conifer species as nurse crops. This is not a natural association of species in Britain, and perhaps if the animal had been introduced to another part of the country (dominated by birch or ash woods for example) it might never have become established here. Beech/conifer mixtures comprise ideal habitat for the edible dormouse in central Europe, and traditional dormouse hunters in Croatia and Slovenia claim to catch many thousands of the animals in such habitats in certain years. The implication for forestry management in Britain is that damage to softwoods is particularly likely where beech grows nearby, but beech is the more valuable crop and unlikely to be removed as a protective measure.

A further complication is that the nest box studies in Hockeridge Wood have shown that *Glis glis* fails to breed in some years, as it does also on the Continent (Bieber, 1998). Yet in other years enormous numbers of young may be produced. For example, there were over 350 juveniles present in the nest boxes in September 2002, yet none at all throughout the entire summer of 1996. Moreover, one batch of nest boxes appears not to have been used at all except in ‘major breeding years’, when unusually large numbers of animals were present in the wood. These included individual dormice that were marked one year, then disappeared; only to return in breeding years, having been living somewhere else. The general impression given is that a proportion of the population live in the wood, but many others live elsewhere and re-invade the study site in certain years. This is unexpected as *Glis glis* is normally considered to be a rather sedentary species and previous studies suggested that they do not have extensive ranges. Immigration into woodland sites (possibly by animals previously living in houses and outbuildings) would account for anecdotes that suggest periodicity in levels of tree damage by *Glis* and would also make pest control more necessary in certain years. ‘Irruptions’ of *Glis* may be linked to beech masting, itself driven by climatic factors. Predicting when edible dormice might become unusually abundant may be possible using weather data (Burgess, personal communication), and this could be used as a way of targeting the animal in specific years.

The changing economics of forestry may reduce the financial incentives for planting softwoods to such an extent that damage by edible dormice becomes a minor problem nationally and not cost effective to control. However, illegal translocations to areas where softwoods and beech occur in close proximity (e.g. the New Forest) raise concerns about the spread of this species. It is becoming an increasing nuisance in domestic premises, where a different set of issues are involved, and transfer to other parts of the country could make the problem of *Glis glis* suddenly much more significant at the national level.

Recommendations

- Test the effectiveness of repeated removals, using batches of nest tubes.
- Abandon attempts to control *Glis* where beech trees grow in association with softwoods.
- Eliminate *Glis* immediately if they are found outside their established range in the Chilterns.
- Dissuade people from illegally translocating edible dormice as a 'humane alternative to killing them'.

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CHAPTER 11

Population increases, impacts and the need for management of deer in Britain

Robin Gill

Summary

Deer populations have been steadily increasing in Britain for a number of years. The reasons for the increase include introductions (of both native and non-native species), increased woodland cover, reductions in extensive livestock grazing and favourable arable farming practices. The current assemblage of woodland herbivores is now very different from those present in historical and pre-historical periods, and is now focussed on browsers with an absence of large grazers. The prospect of grazing livestock in woodlands is gaining increased interest amongst conservationists. There is an urgent need to improve co-operation and support for deer management, to help reduce deer numbers.

Increases in deer populations

Altogether, six species of deer occur in the wild in Britain today (red deer, *Cervus elaphus*; roe deer, *Capreolus capreolus;* sika deer, *Cervus nippon*; fallow deer *Dama dama*; muntjac *Muntiacus reevesi* and Chinese water deer, *Hydropotes inermis*). All have been increasing in distribution and numbers during the past few years (Figure 11.1). As a result of these increases, there are now very few woodland areas left in Britain which do not have either resident or transient deer populations, the most obvious examples being on the Isle of Wight and Isle of Man. The greatest increases have occurred in the Midlands, East Anglia and more recently in Wales. Muntjac appear to have spread at a greater rate than the other species.

Increases in deer populations have not been confined to Britain, nor is this simply a recent phenomenon. Deer populations have been increasing in much of Europe for almost 200 years and in North America and Russia for between 100–150 years. There are several reasons why this has been happening, although the same factors have not been influencing deer populations at all times and places.

The most obvious reason is that deer have been introduced and on occasions deliberately released into the wild. In Britain, introductions have been happening since at least the 11th century, when fallow deer were introduced for hunting. From time to time, deer escaped and became established in the wild. Historical accounts suggest that escaping deer were often killed or caught by keepers or poachers. However, during the 20th century, fewer keepers could be employed, particularly during wartime.

Deer control has been minimal and uncoordinated, with many landowners showing more interest in pheasant shooting or a modest level of trophy shooting than deer control.

A number of land-use changes have also been beneficial for deer. The area of forest and woodland has been increasing and winter cereals are now grown more extensively, providing an important winter food source. Since grazing early in the year has little effect on yield, farmers do not normally see it as a justification for control. There has also been a reduction in extensive livestock husbandry, especially in lowland woodlands. Deer typically avoid close contact with grazing livestock, hence their removal makes woodlands more suited for settlement by deer. Livestock grazing also reduces understorey vegetation in woodlands, reducing both food and hiding cover for deer.

Figure 11.1 *Changes in the distribution of deer in the British Isles. Dark grey represents the approximate distribution of each species in 1961. Light grey represents the additional range between 1961–2001. Un-shaded areas bounded by a black line represent contracted range between 1961–2001. Sources: Whitehead, 1964; Corbett, 1971; Mitchell-Jones* et al.*, 1999; Johnson, 2001; Rotherham, 2001; Anon, 2002; H. R. Arnold, The Environmental Information Centre, ITE, Monks Wood, Abbots Ripton, PE17 2LS; The Wales Deer Initiative, PO Box 39, Brecon, LD3 8WD.*

An additional factor has been the climate, which has been in a warming trend during the past 200 years. Warmer winter and spring weather has been correlated with increased recruitment and overwinter survival, particularly in populations at higher latitudes and altitudes. Finally, large predators have been virtually eliminated from Britain, resulting in the removal of a major mortality factor.

Contrasts between the current status and prehistoric woodland herbivore faunas

With as many as six species of deer, the current status of wild ungulates in Britain is unprecedented and somewhat atypical of ecosystems in the temperate region. Densities can exceed 50 per km^2, and the number of woodlands where 3–4 species co-exist has been increasing. The post-glacial assemblage of large herbivores in Britain included just two species of deer (red and roe deer), as well as aurochsen (wild cattle, *Bos primigenius*), beaver (*Castor fiber*) and wild boar (*Sus scrofa*). Elk (*Alces alces*) occurred briefly, but disappeared during the Atlantic period as conditions became more temperate. In nearby parts of continental Europe, bison (*Bison bonasus*) and possibly horses (*Equus ferus*) were present. Further back in time, during the interglacial periods, temperate woodlands had an even more varied fauna, including most of these species (or their close relatives) as well as elephants (*Palaeoloxodon antiquus*), rhino (*Stephanorhinus* sp.), and hippos (*Hippopotamus amphibius*) in addition to deer (Yalden, 1999 and 2003).

The extinction of large mammals at the end of the last glaciation coupled with the further loss of aurochsen, wild boar and beaver in historical times has left an impoverished fauna. The introduction of additional deer species in recent times has not restored the assemblage, but instead left only medium-sized browsers or intermediate browser-grazers. The large grazers (aurochsen and bison), omnivore (wild boar) and very large browsers (elephants and rhinos) have gone.

The effects of deer on woodland ecosystems

Deer have a variety of effects in woodland. The most obvious effect, and of most concern to woodland owners, is browsing on trees and shrubs. Deer reduce the density of young seedlings and may eliminate them entirely, severely restricting the rate of tree regeneration. If the seedlings survive, browsing will reduce their growth rate. If the leading shoot is lost it may also introduce stem deformities, such as a bend or fork in the stem. Deer will also browse on coppice shoots, and repeated browsing of the re-growth may kill a stool.

Deer are all selective feeders and will browse on some tree species more than others. In woodlands managed by natural regeneration this can result in a decrease in the diversity of tree seedling species. Typically, the most palatable seedling species such as oak, willow, hornbeam and ash are the most severely depleted. Seedlings of some tree species (beech and birch) are reported to be depleted by browsing in some areas but may be able to increase in others. There may be a combination of reasons for this, but the most likely is preferences are dependent on the abundance of some other plant species in the area. Trees that are relatively unpalatable to deer (Sitka spruce, *Picea sitchensis* and Corsican pine *Pinus nigra*) are often favoured by foresters because they can be grown without fencing or with only modest deer control.

Deer can remove bark from trees, providing an entry point for decay-forming fungi. Most deer species can damage saplings by antler rubbing, which is done either to remove velvet or for scent-marking. The larger species (red, sika, and fallow deer) will feed on bark, usually resulting in larger wounds and therefore causing more serious damage. The combination of browsing and bark stripping can result in severe economic losses to woodland if measures are not taken prevent it.

Apart from the effects on trees, deer also have a major effect on the composition of woodland vegetation. Browsing reduces the height and density of shrubs, such as heather (*Calluna vulgaris*), bramble (*Rubus fruticosus*), ivy (*Hedera helix*) and honeysuckle (*Lonicera periclymenum*). Some herbaceous species, particularly taller herbs, are also eaten by deer and decrease in abundance, in common with some of the rarer flowering plants. The effects of browsing lead to changes in woodland structure, which may be detrimental to a wide range of woodland birds, small mammals and some invertebrates (Fuller and Gill, 2001).

In view of differences in foraging behaviour and diet, the effects of herbivores during various pre-historic periods would not have been the same as today. Evidence from woodland sites grazed by cattle and horses suggests that they maintain open areas in woodland and may promote diversity in grassland (see Chapter 15). They avoid grazing thorny plants which can facilitate tree regeneration. Elephants browse at a higher level than ungulates and are more destructive towards trees and shrubs.

However vegetation closer to the ground benefits to some extent from reduced shade. Interest amongst conservationists in the role of large herbivores has been growing (Vera, 2000), and has resulted in several re-introductions of some extant species (bison, feral cattle and horses) in protected areas in Europe.

Deer management problems

The problems of deer damage can be prevented or reduced by using various methods of tree protection (fencing, guards, or chemical repellents) or by reducing numbers by deer control (Table 11.1). In practice, a combination of approaches is usually used. Fencing has the obvious advantage in protecting vegetation entirely from deer, at least as long as it remains intact, but it becomes an expensive option for small (<2 ha) or awkwardly shaped areas, and an impractical option on a large-scale because it forces deer onto other areas.

Table 11.1 *Summary of deer management and woodland protection options.*

	Advantages	Disadvantages
Fencing	• Protects trees as well as other vegetation entirely against deer and other herbivores if needed.	• Fails completely if the fence is breached. May prove too expensive for some areas. • Unsightly. Interferes with public access. • Can exclude deer from ideal habitat. • Forces deer onto other areas. • Impractical for small or awkwardly shaped areas.
Tree guards	• Suitable for small areas.	• Only protect trees.
Chemical repellents	• Suitable for small areas.	• Require repeated application.
Population control	• Reduces impact on all vegetation over an extensive area. • Yields revenue from the sale of venison or stalking leases. • Can help reduce traffic casualties involving deer. • May reduce deaths due to disease or malnutrition.	• Effort must be maintained – populations recover quickly if shooting effort relaxed. • Shooting is impractical in urban areas. • Population reduction may be limited by immigration. Co-operation between neighbours is essential. • May fail to protect sites preferred by deer.

To reduce deer impact problems over larger areas requires population reduction through deer management programs. Research on methods of fertility control has been progressing on deer but as yet does not offer the prospect of being as efficient as shooting. If carefully applied, shooting can be an effective and humane method of population control, and has the benefit of yielding revenue from the sale of venison or stalking lets. However, the steady increase in abundance of deer in Britain is an indication that the shooting pressure has not been sufficient to contain the increase. There are a number of reasons for this.

In Britain, landowners normally retain shooting rights and they have varying interests as well as divergent attitudes towards deer control. Shooting for trophies does not require population reduction and is usually easier when numbers are high. Damage (even when it occurs) does not appear to be a serious enough concern to justify control efforts by some landowners. In urban areas, shooting can be unsafe and some landowners object to shooting in principle.

Finally when shooting is carried out, landowners are under no obligation to record or declare the numbers shot, and situations can arise where neighbours pursue quite different management strategies and be unclear about each other’s activities. The outcome is that deer control is at best incoherent, and in many cases completely ineffective.

Where deer control has been carried out, problems have been encountered with deer moving between properties. The larger species of deer range over areas of 10^2–10^4 ha and young roe deer disperse over distances of 1–10 km, sometimes further, to establish territories. To minimise the influence of these movements on deer numbers, population management needs to be approached over areas of 10^5–10^7 ha, areas vastly greater than most land holdings.

Many of these problems have been appreciated now for some time and resulted in the formation of the Deer Commission for Scotland (see Chapter 12) and the Deer Initiative for England and Wales. The purpose of the Deer Initiative was to foster more co-operation between landowners in deer management. So far, this has led to the formation of a number of deer management groups. However, the work involved in supporting deer management groups is considerable, involving liaison with landowners, information gathering and technical support for methods of population management. An organisation like the Deer Initiative would need more resources, as well as statutory powers, to do this effectively. However, without more effective deer management it is difficult to carry out woodland conservation and management adequately, and also difficult to justify the re-introduction of other large mammals.

Recommendations

The legislative, social and institutional framework for deer management needs to reviewed, particularly in England and Wales. To achieve effective population control, organisations responsible for deer management may need statutory powers to carry out control or other management tasks where landowners or stalkers are unable to achieve this themselves.

The work of building deer management groups needs to be continued and extended. More professional and technical support is needed for management groups to support or carry out key deer management tasks. These range from coordination of landowners and stalkers, estimating abundance, impact monitoring and the collection of cull data.

More information is required on the economic and ecological implications of deer damage. Schemes for monitoring the impact of deer at a local level should be considered.

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CHAPTER 12

Deer management in the uplands: the approach of the Deer Commission for Scotland

Andrew Raven

Summary

The Deer Commission for Scotland (DCS) is a statutory body that advises Scottish ministers on all matters relating to wild deer. The paper describes the remit and strategy of the DCS, and summarises some key resulting issues in relation to deer management in woodlands in Scotland.

Introduction

The Deer Commission for Scotland (DCS) is a statutory body that advises Scottish ministers on all matters relating to wild deer. Its overall function is to further the conservation, control and sustainable management of deer in Scotland, and to keep under review all matters relating to deer, including their welfare. The Deer (Scotland) Act 1996 gives the DCS a range of duties and powers, including measures to prevent serious damage by deer to agriculture, woodland and the natural heritage, and to prevent deer from posing a danger to public safety. In October 2000 the DCS published *Wild deer in Scotland – a long term vision* as a short set of principles to guide deer management in Scotland over the next 15–20 years. A strategy launched in November 2001 sets out how the DCS intends to realise that vision.

In working towards the vision, the strategy takes account of a wide variety of factors and influences. Five key elements collectively provide the framework on which it is based:

- *Public policy*, including the environmental, forestry, land-use and other strategies for rural development adopted by Scottish ministers.
- *Local deer populations* and the interaction between them and the land that they occupy.
- *Local management* of those populations in order to meet agreed objectives that are integrated with local land-use objectives through a collaborative and inclusive process.
- *Annual deer culls* that are set, carried out and verified in each locality in order to ensure that local deer populations are compatible with management objectives.
- *Compulsory powers* that can be deployed by the DCS where there is not adequate local deer control.

The strategy must deliver change in two key, and complementary, areas. On the one hand there is a need to fully integrate deer management into the wider policy context, while on the other it is essential to develop practical arrangements for local deer management that are sustainable and effective in delivering policy objectives. To lead this change, the DCS must adapt what it does and ensure that its resource deployment is fully aligned with its strategic priorities. These core themes translate into six broad strategic aims, and on an annual basis the DCS will publish a rolling three-year corporate plan setting out its detailed operational intentions against these:

- To contribute to development of a *public policy framework* that will enable the vision to be realised.
- To secure an effective system of *local deer management* across the deer range.
- To make effective use of the *regulatory provisions* that can be applied by the DCS.

- To ensure the provision of adequate *information, advice and training* to promote sustainable deer management.
- To undertake and support adequate relevant *deer-related research*.
- To deliver effective and efficient *administration of the DCS*.

The DCS will be implementing this strategy over a number of years, with modifications from time to time to reflect changing circumstances. Specific actions required by the strategy will be developed through the DCS's 3-year rolling corporate planning process, and will evolve as progress is made. The overriding short-term requirement is to ensure that the DCS fulfils its duties under the 1996 Act. The initial commitments are therefore:

- To *identify areas* where serious damage by deer is occurring or public safety is being threatened.
- To *stimulate effective local deer management in those areas* and ensure further serious damage or danger to public safety is prevented.
- To *ensure that the DCS has the necessary capacity and powers* to achieve this.

The DCS envisages a process of delivery that will form part of a progressively more integrated approach to land-use and rural policy in Scotland, and three key elements will be central to this:

- *The leading role of the DCS* in driving forward progress against the aims identified in the strategy.
- *Effective partnerships* between all relevant interests involved with or affected by deer management.
- *Rigorous and comprehensive monitoring of progress* feeding back into ongoing planning and decision making at all levels.

In the uplands, distinction between deer management in woodlands or on the open range is increasingly artificial, with similar principles applying to both, though the practicalities differ significantly.

Particular issues related to woodlands

With respect to woodland policy and practice, key issues include:

- A policy of further expansion of woodland area and improved management of existing woodland needs to take deer into account, as anticipated in the Scottish forestry strategy. Native woodland expansion, particularly by natural regeneration is threatened by deer browsing pressure.
- The increase and spread of deer populations raises questions as to whether initially deer-free forests can remain so.
- Forest design planning should take account of, and make provision for, cost-effective deer control.
- Habitat improvement can lead to increasing deer carrying capacity and hence numbers. Changing woodland management practice – for example, the increase in continuous cover forestry, may lead to increasing deer numbers and reduced efficiency in culling.
- Some deer management options need to be assessed carefully against other management objectives – for example, consideration of fencing must take account of public access and the potential impact on woodland grouse mortality.

Current DCS priorities relating to woodland are:

- Acting as consultee in assessment of woodland grant scheme applications, forest plans, and woodland certification.
- Developing and publishing best practice guidance.
- Developing population assessment, in both practice and improved research methodology.
- Encouraging collaborative deer management (deer management groups).

- Encouraging a hierarchy of deer management plans, e.g. Forest Enterprise's Deer Management Package.
- Taking action in priority areas, where there is damage by deer to trees and to woodland biodiversity.
- Performing regulatory duties, for example providing authorisations for out-of-season/night shooting.
- Assisting policy and management developments at the woodland/open range interface.

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Further information is also available at www.dcs.gov.uk

CHAPTER 13

Muntjac and conservation woodlands

Arnold Cooke

Summary

Muntjac deer (*Muntiacus reevesi*) can detrimentally affect a range of vegetation in conservation woodlands, and indirect effects on fauna also occur. Different effects begin at different deer densities. Small, discrete areas of sensitive vegetation can be fenced to exclude deer. Effects that are widespread in a wood may require a reduction in deer density. Remedial management can lead to recovery, but some changes may be irreversible. From a practical point of view, eradication of muntjac is unlikely to be feasible, and they will continue to contribute to grazing and browsing in many conservation woodlands. Providing levels of grazing and browsing remain low and irregular, they may be beneficial for biodiversity.

Muntjac and conservation woodlands

The Chinese or Reeves' muntjac (*Muntiacus reevesi*) is a native of southeast China and Taiwan. It was first released in Britain in 1901 near Woburn in Bedfordshire. During the rest of the 20th century, muntjac were transported to different parts of the country, and were often deliberately released or escaped from captivity. This process, together with natural spread from the many loci, has resulted in the species being recorded in virtually every English county. The muntjac can now be regarded as widespread and abundant over much of central, eastern and southern England (see also Chapter 11).

Conservation woodlands are those where wildlife conservation is at least one of the main management aims. The preferred habitat for muntjac in this country is broadleaved woodland with diverse ground and shrub layers, so the species is likely to be present in most semi-natural deciduous woods within its main range. There seem to be several factors underlying why this small deer can be a problem for managers in conservation woodlands. First, it is selective when feeding because its relatively simple digestive system demands an intake of readily digestible items (Putman, 1988). There may be direct conflict if it concentrates on vegetation of conservation importance. Second, radio-tracking studies have revealed individuals to have small, fixed home ranges of roughly 10–15 ha (Chapman *et al.,* 1993; Staines *et al.,* 1998), so muntjac may live throughout the year in the same sensitive woodland. Third, such studies also show that considerable overlap can occur between individuals' home ranges, thereby providing the potential for high densities to result. In conservation woodlands, where population growth is unlikely to have been controlled by man, the potential to reach high densities may have been realised. Moreover, large sites tend to have extensive, quiet blocks of woodland where even higher densities can build up.

The effects of grazing and browsing

Some grazing and browsing is beneficial to the biodiversity of conservation woodland, especially if irregular in time and space. In the past, muntjac were often viewed as a benign addition to the local fauna, until they surprised woodland managers by becoming a problem for certain sensitive conservation features. To illustrate the wide range of documented effects in conservation woods, examples of changes induced by muntjac are described for Monks Wood National Nature Reserve (see also the review by Cooke and Farrell, 2001). Monks Wood is 157 ha in area and is the largest wood in Cambridgeshire. The forest canopy is predominantly ash (*Fraxinus excelsior*) with pedunculate oak (*Quercus robur*), and the understorey includes much hazel (*Corylus avellana*).

Muntjac were first recorded in the wood in about 1970 and built up until, during the period 1985–1998, their density in the wood was at least 1 ha^{-1} with a total population of about 200. Other species of deer have been only relatively rarely reported; rabbits (*Oryctolagus cuniculus*) and brown hares (*Lepus europaeus*) are resident in and around the wood, but have been less frequently recorded than muntjac.

The most obvious effect in Monks Wood, first noticed in 1985, was on blocks of coppice. Regrowth stems were browsed by muntjac sufficiently severely and persistently that the coppice canopy failed and stools died, either because of the high level of browsing or because they were overgrown by other vegetation. Lack of shade from a canopy allowed pendulous sedge (*Carex pendula*) to dominate the ground layer and out-compete more interesting species. Thus, there were direct effects on the coppiced species and indirect effects on components of the ground flora, most of which were negative. Because of these problems, coppicing operations were suspended after 1994.

Other effects in Monks Wood, although obvious with hindsight, were not appreciated until they were investigated in the early 1990s. Bramble (*Rubus fruticosus*) is heavily browsed to a height of about 1 m, resulting in marked browselines. Persistent browsing ultimately kills the thickets. This, combined with growth of bramble seedlings being inhibited by browsing, has resulted in bramble becoming much less abundant in the wood. Leaves of bluebell (*Hyacinthoides non-scripta*) are grazed in February and March; this reduces the vigour of plants, resulting in smaller specimens that are less likely to flower. Those bluebells that do flower are likely to have their inflorescences bitten off in the spring. Despite a large decrease in numbers flowering successfully, the range of the bluebell in the wood does not seem to have been markedly affected.

With such effects on vegetation, indirect effects on fauna would also be anticipated. Invertebrates dependent on specific plants seem to be at particular risk (for example, see Pollard and Cooke, 1994). However, some plant species, especially grasses and sedges, have become more abundant, as have their dependent invertebrates (Pollard *et al.,* 1998). Amongst vertebrates, the nightingale (*Luscinia megarhynchos*) has decreased in numbers and the Chinese water deer (*Hydropotes inermis*) has virtually disappeared from Monks Wood since the 1980s, with muntjac being implicated in these declines (Cooke and Farrell, 2001).

In order to help to ascertain how frequently effects on vegetation might be encountered in sites in the main range of the muntjac in Britain, information on 'muntjac scores' has been collated for a sample of 60 sites. The score provides a simple measure of muntjac density. Each site was visited for 1–2 hr, and the following four variables were each scored 0, 1, 2 or 3: encounters with muntjac, and seeing their droppings, slots and paths. These sub-scores were summed to give a muntjac score that will lie between 0 (no signs recorded) and 12 (all variables recorded at high frequency). While the technique is subjective, all scoring has been undertaken by the author, and scores are used here in a comparative manner.

Numbers of sites with each score are shown in Figure 13.1. No sites scored 11 or 12. Sites are divided into those with or without a significant population of other deer species; in 47% of sites, only muntjac populations were present. Sites covered are those in which the author worked during 1993–2002 and are not necessarily representative of all sites in the main range of the muntjac. All are woodlands or wooded areas of fen; each is in the eastern half of England from southern Lincolnshire in the north to the Thames in the south, and most are nature reserves.

The distribution pattern in Figure 13.1 shows 78% of sites with scores in the range 2–6, with a peak at a score of 4. Monks Wood scored 9, so had an atypically high score. In addition to deriving the muntjac score, varying amounts of damage assessment were done by the author at each site. It is possible to determine whether effects on vegetation are likely at different muntjac scores:

- scores of 1–3: obvious and severe effects occur in a few woods only;
- scores of 4 and above: severe damage to unprotected coppice regrowth is likely;
- scores of 6 and above: obvious browse lines on bramble are likely;
- scores of 7 and above: severe effects may occur on bluebells;
- scores of 8 and above: severe reduction of bramble is likely.

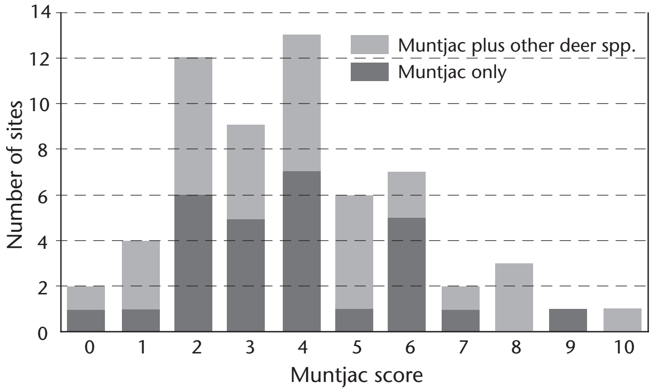

Figure 13.1

Muntjac score assessed in 60 sites in eastern England.

As the score is a simple measure of density, different effects on vegetation occur at different muntjac densities, with a greater range of effects being seen at high densities. From the numbers of woods with different scores, approximately half would be expected to have severe damage in unprotected blocks of coppice, but very few would have serious effects on bluebells. While this statement raises concern, the real situation is likely to be even worse for several reasons. First, other effects may occur at low muntjac densities; for instance, other species of ground flora may be more sensitive than bluebells. Second, the scores depicted in Figure 13.1 are the first recorded for each wood; when scoring was repeated in later years, it often increased as muntjac populations built up. Also about half of the woods held populations of other species of deer, which might cause additional damage, such as browsing foliage too high for muntjac to reach.

Management

So far in this account, changes due to browsing and grazing have been referred to as effects or damage, but when do they become real impact? At what point does the situation become unacceptable to conservation managers and remedial management become necessary? Decisions ought to be objectively based on information such as which species are declining and how rapidly, and whether biodiversity is being adversely affected. It is also crucial to be reasonably sure that muntjac are responsible for any unacceptable impacts before management is undertaken. In reality, however, decisions often reflect the perspectives of those people making them. Furthermore, translating concerns into action on the ground is a multi-stage process, and precisely what is done and when will depend on the policies and resources of the organisation responsible for managing a wood.

Once a decision is made to undertake remedial management, the choice usually comes down to protecting the vegetation or reducing deer numbers (or both). Small-scale impacts may need only small-scale solutions. Thus specimens of rare ground flora can be protected by small wire cages. Such protection may not be elegant, but it does work – at least in the short term. Similarly, relatively small areas of coppice of, for example, up to 1 ha can be satisfactorily protected by wire fencing providing it is flush with, or buried in, the ground; is high enough to stop muntjac jumping over and its mesh size is small enough to prevent them getting through. The Forestry Commission recommends a height of at least 1.5 m and a maximum mesh size of 7.5 cm. Electric fences, chestnut paling and dead hedging are often used, but are less reliable than wire fences at excluding muntjac. With larger plots and consequently longer fences, the chances increase of muntjac gaining access, perhaps via damage to wire fencing or through holes underneath made by other animals. Fencing whole woods is almost certain to enclose muntjac, and, unless there is also management of the deer themselves, such action is likely to worsen problems as they breed and their density increases.

Recent events in Monks Wood are reviewed here as a case study. In 1999, two wire fences were erected that enclose more than 10% of the wood in total. This management has been accompanied by stalking. Shooting has taken place on land surrounding the wood for many years, including in small woods near to Monks Wood, so deer have been managed across the local landscape. Since 1998, English Nature has permitted stalking within the wood itself. Overall, during the four winters 1998/9–2001/2, about 300 muntjac were shot inside the fences, in the rest of the wood and just outside it. The effect of this action has been to reduce the deer score in the wood from 9 to about 6 and deer density from 1.0 ha^{-1} to roughly 0.5 ha^{-1}. With this reduction, improvement might be expected in vegetation effects that become apparent at high scores and densities.

Three bluebell transects were monitored annually. Between 1993 and 1998, the period before stalking was allowed inside the wood, there was no sign of improvement in grazing levels. However, grazing on both inflorescences and leaves decreased steadily from 1998 to 2002, the period in which deer density was reduced. With less grazing of leaves, bluebell vigour recovered such that numbers attempting to flower increased 2–3 fold during this time. This, combined with less grazing of the inflorescences themselves, resulted in an increase in intact inflorescences of about five fold. There has been little sign of plant size recovering, however, and some changes induced by deer grazing may not be reversible. Bramble has started to recover, there being significant increases in height between 2000 and 2002.

Formal coppicing has not yet resumed in the wood. Nevertheless, hazel stools are cut each winter for a variety of reasons. In the late summer of 2002, regrowth success was assessed for >60 stools cut during the previous winter in unfenced situations. None of the stools had any regrowth >1 m in height mainly because of browsing by deer. Deer density outside the two main fences was still sufficient to cause problems for such operations. However, this is of little consequence as formal blocks of coppice can be protected within fences erected specifically for this purpose.

Therefore, one strategy for counteracting a problem of the magnitude experienced in Monks Wood is to control muntjac numbers down to a level at which the widespread effects caused at high deer densities are eliminated, then protect with fencing the more sensitive, discrete features, such as coppice. Outside these fences, a certain amount of browsing and grazing is likely to be beneficial for diversity, and appropriate populations of native browsers and grazers should be encouraged. However, eradication of muntjac is probably not a feasible option, and conservationists may have to learn to accept their presence, which will contribute to the overall levels of browsing and grazing. Management should remain flexible within a predetermined framework, with reviews of progress, based on targeted monitoring, being made at frequent and regular intervals. These principles can be adapted to the management of muntjac in other woodland situations.

Recommendations

In a conservation woodland with muntjac:

- monitor the deer population and look for and monitor changes in species of conservation importance, e.g. plant grazing/browsing levels and abundance.
- determine objectively whether effects are unacceptable, whether muntjac are responsible and what management may be required.
- undertake management of vegetation and/or deer, and continue to monitor.
- review progress regularly and continue with or amend management and monitoring as necessary.

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SECTION FOUR

Complex interactions and potential conflicts in woodland use

CHAPTER 14

Managing mammals in the Forest of Dean

Rob Guest

Summary

Managing mammals in the Forest of Dean has a long history. For centuries, deer were managed for venison, and the challenge now is maintaining a herd of deer at a level which does not compromise the sustainability or biodiversity values of the forest environment. There is a long tradition of grazing of free-roaming sheep in the Forest which is increasingly challenging as society becomes less tolerant of wandering animals on the roads and throughout the communities. A more recent problem is the huge grey squirrel population which poses a threat to forest sustainability. The management of these issues is complex due to the influence of another significant mammal – the many inhabitants of, and visitors to, the Dean.

Introduction

To understand managing mammals in the Forest of Dean one must consider the historical context. So, to start with a gem from the history books: a visit by Nelson to the area in 1802, when he was concerned about future supplies of timber for warships, led to a report for parliament and subsequent legislation in 1808. A tree planting boom followed that lasted until the 1840s. 1814 was the year of the mouse ... A plague of woodmice attacked the acorns which had been sown and ate the young trees. Different control techniques were tried until a freeminer hit upon the successful idea of a small drop pit with sloping sides. So management of mammals in 1814 consisted of a bounty scheme with the Deputy Surveyor at the time paying out on >100 000 mouse tails!! now to bigger things....

Deer

Status

Management of the Forest was traditionally associated with management of deer. The Dean became a Royal hunting forest in Norman times. The last record of the monarch actually hunting in the Forest was 1256, but it continued to be an important source of venison for the Royal table for centuries. There was an intricate management structure associated with this – today, the only part which survives is the Court of Verderers. The first record of Verderers in the Dean is 1216, but they still meet quarterly to address issues affecting the vert and venison within the Statutory Forest (i.e. they are charged with protecting the vegetation and the animals dependent on it within the area formally defined as the Forest of Dean by the latest perambulation in 1833).

Since Norman times, the Dean has had principally one species of deer – a herd of fallow. In the 13th century, the ratio from venison records was 25 fallow:6 red:1 roe but red and roe deer were absent by the 16th century.

An analysis of the archives reported by Dr C Hart OBE have shown that deer numbers have fluctuated over the centuries (Figure 14.1).

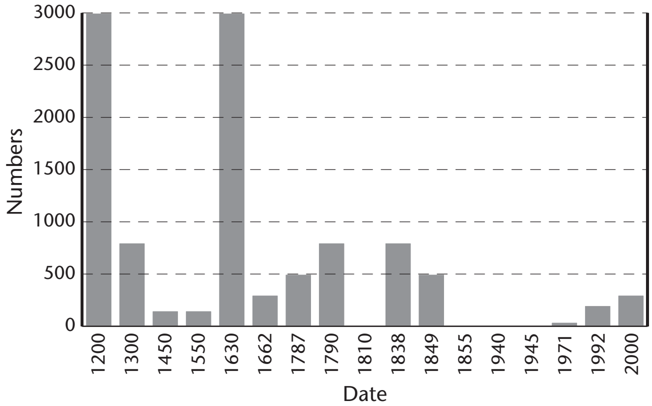

Figure 14.1

Deer numbers in the Forest of Dean 1200–2000.

In summary, deer were:

- numerous in 12th century and 13th century when the herd was safeguarded and offenders heavily punished;
- dwindling in 14th century due to poaching;
- scarce in 15th century and early 16th century;
- increasing in late 16th century when Elizabeth I introduced woodland management for ship timber;
- reduced dramatically to 300 in the 1630s before the Civil War by mass poaching;
- legally restricted to 800 in 1668 Dean Forest (Reafforestation) Act;
- reduced to about 10 by 1800 with decrease in cover and much poaching;
- numbered at 800 in 1840 following enclosure and replanting;
- reduced to 150 bucks and 300 does in 1850 with poaching;
- all gone by 1855 due to the 1851 Deer Removal Act (enacted to counteract poaching);
- re-established in the Speech House area during WWII;
- reported to number about 40 animals in 1971;
- assessed at 200 in number in 1992;
- censused in 2000 after culling at the end of the winter with night vision equipment and numbered 300.

Before foot and mouth disease in 2001, the deer population tended to be more dense in the enclosures which were not subject to sheep browse, but after the removal of all the sheep, the deer spread throughout most of the Dean. The period when there was no public access and the increased food availability because of the lack of sheep probably combined to cause an observed increase in numbers.

A few red deer were introduced in 1842, but the last was killed in 1848. At least four semi-tame red deer were dumped in the Forest in 1999. They have since moved to the southwest of the Forest.

Muntjac and roe deer continue to advance towards the Forest from the east, with known populations now 5 km east of the Forest, and some Muntjac observed to the south.

Evaluation of issues

Deer have an impact on the restocking of clearfelled areas, particularly when using broadleaved species. The impact of forest design planning, where small areas of clearfelling are located throughout the Dean, makes control of deer within these areas very difficult. In some areas, taller tree shelters for protecting young trees have been used, but this is only feasible on a small scale and it offers no protection for the ground flora. In 2002 deer fencing was tried in the Nagshead reserve to protect broadleaved restocking, and has been effective with so far no adverse comment from the public. In addition to restocking, the implementation of regeneration strategies for the old broadleaved stands is also made difficult because of deer browse on young seedlings. The vision of a healthy forest which is sustainable is dependent on deer populations being kept at an appropriate level.

There is an increasing problem of road traffic accidents involving deer. The main problem area used to be the road at the heart of the home range, but the incidence of accidents has increased significantly since the herd spread after foot and mouth disease. In 2001–02, 37 deer were killed within the Statutory Forest.

The fallow deer have traditionally been of high quality; healthy animals with bucks boasting good heads. Trophy stalking continues to generate local opposition because most of the demand for trophy stalking is from mainland Europeans, but it does offer scope for revenue generation. Safety of the operation is paramount, and limited stalking is only permitted on a one-to-one basis under close supervision of a wildlife ranger. On average, three trophy heads are obtained each year in the Forest.

Management strategy

The Verderers agreed in 1999 that:

- Deer should be managed so as to maintain the herd size at its level at that time, recognising that this will not prevent all detrimental effects on restocked sites.
- Deer control efforts are focussed on those areas where lower densities are required to assist establishment of tree crops or to facilitate the natural regeneration of woodland. Where it is deemed necessary, deer fencing or tubes will be used.
- A regular estimate of the population size of fallow deer should be undertaken using night vision equipment on sample transects throughout the Forest every two or three years.
- Trophy stalking should continue at a low level as an integral part of the culling programme. It is envisaged that 10 man-days stalking will be offered per annum.
- Means of reducing the incidence of road traffic accidents are currently being investigated by the Verderers.
- The spread of red, roe and muntjac deer into the Forest must be monitored, and steps should be taken to restrict the spread.

Sheep

Status

The mammals that have most exercised the wits and patience of forest managers in the Dean for centuries are sheep. There is a long tradition of free-roaming sheep grazing in the Forest – whether there are rights to do so is a moot point. On a number of occasions in the past 200 years, the issue has almost been tested in law, but the issue has never been satisfactorily resolved. Currently the authorities suffer the practice as a long-standing tradition.

It is a tradition which has shaped the Dean. Much of the forest was used for grazing with a range of stock which had an adverse effect on tree growth. It was only with the determination to secure timber supplies for ship building that the issue was addressed – this resulted in the Act of 1668 which recognised that the wood and timber had become totally destroyed except for one small area, and allowed the enclosing at any one time of 11 000 out of 23 000 acres for the growth of timber. This was not a popular move, and there was much damage to the fences. A little later in 1688 there were riots in which fences were torn down, official lodges destroyed, and the Speech House harbouring the Verderers Court attacked. For a further 100 years or so there was an uneasy truce, with the situation as described by Nelson in 1803 still unsatisfactory. Further legislation in 1808 gave enclosure for tree planting a further boost, with its associated backlash in 1831 with more riots. Special constabulary were called in to quell the trouble, and the ringleader deported to the Antipodes. Today, the situation is somewhat less antagonistic, but many of the issues remain. In 2001, foot and mouth disease (FMD) spread into the roaming flock and necessitated the destruction of all 5500 sheep on the Forest. The subsequent rank growth and expansion of bracken has been detrimental to the amenity of the area, to conservation values and to future grazing. In 2002, there are some 2000 hectares (out of a total of around 9000 ha) enclosed to exclude stock and sheep are once again starting to roam the remainder of the Forest.

Evaluation of issues

This long history of grazing has had an impact on the ecology of the Forest – there are biodiversity gains associated with the practice of grazing stock between widely spaced trees (e.g. rich lichen and fungal assemblages), and there are possibly links between the droppings of sheep and the important horseshoe bat populations. A number of butterfly species are dependent on the open conditions created in the woodland environment. But there are also problems. The major and long-standing issue is the potential for trespass by sheep into the enclosures causing damage to trees. Other matters which have gained in importance in recent years are the nuisance in the communities, road traffic accidents, and animal welfare standards. Free roaming sheep continue to be contentious, and the local population is split on the issue. Some are vehemently committed to retaining the tradition, others (particularly in communities where sheep roam) consider the practice anachronistic. Forest Enterprise (part of Forestry Commission England), as managers of most of the land in the Statutory Forest are seen to have a responsibility to resolve the issues on behalf of the community.

Management strategy

The potential return of the sheep following FMD was the cause of intensive negotiation – with all parties with an interest involved – this included Forest Enterprise as managers, the 'Commoners' Association (CA)' representing the graziers, the Verderers, MAFF (now Defra), the District Council, the Police, and the County Trading Standards and Highways Departments.

The outcome was an agreement that:

- Confirmed the tradition of free roaming grazing in the Statutory Forest, but outside the enclosures.
- Agreed a restriction on numbers of sheep allowed and required that all sheep be clearly marked.
- Allowed grazing by members of the CA, with qualifications for membership made clear (over 18 and resident in the hundred of St Briavels – which is a defined area surrounding the Forest of Dean from which it is said a hundred bowmen could be raised), but with the proviso that any non-members would have to agree to the same terms and conditions as members.
- All graziers would have suitable and sufficient grazing land other than the Forest, and that sheep would be shepherded and managed in accordance with the code of practice on welfare standards produced by MAFF (now Defra).
- The graziers would endeavour to heft the sheep in appropriate locations to minimise nuisance in the Forest communities, including road traffic accidents.
- Determined that carcasses would be cleared promptly to appropriate disposal points.
- The CA would hold public liability insurance to cover its members and that vehicles used by CA members in the Forest away from public roads would be under permit.
- Decided the number, location and longevity of winter feeding stations would be agreed.

The agreement was negotiated as a means of ensuring that the tradition continued, but in a manner more fitting to society's expectations in the 21st century. Restocking of the Forest with sheep has now been underway for six months, (but not without some problems associated with sheep being introduced into the communities).

Squirrels

Status

One of the biggest problems facing the Forest of Dean is the explosion in the population of introduced grey squirrels. Native red squirrels have not been present for many years – the population crashed about the time of the First World War and numbers are known to have hung on till the Second World War. The first authenticated records of greys were in the 1930s, numbers then increased rapidly.

Damage occurs through bark stripping during the late spring and early summer and appears to be related to density – little damage occurring when population density is lower than about 4 squirrels per hectare (see also Chapter 9). Significant damage has been associated with periods of peak populations – up to 15 squirrels per hectare have been recorded. Damage is cumulative, and the impact may not be apparent for some years. There are various forms of damage:

- Outright death of trees caused by the bark stripping. This is not common – usually less than 5% of damaged trees die.
- Destruction of form, ruling out formation of final crop trees. This is frequently observed in pole stage oak, as the damage to this species is frequently located in the live crown.
- Top snap leading to loss of increment, and effective death. This often occurs some time after stripping – the weakened section then being lower down in the crown – this type of damage has been significant in Norway spruce stands.
- Degrade of timber through abnormal growth around the wound – this is particularly common on the butts of beech, degrading the most valuable section of the tree
- Degrade of timber through fungal invasion of the wound. This occurs, but is probably of less importance than the other gross degrade described above.
- Reduction in seed crop, affecting regeneration of trees.

There are variations with species; of broadleaves, beech and sycamore are particularly vulnerable, oak and sweet chestnut moderately so, and ash and cherry are not so vulnerable. In the conifers, Scots pine, larch and Norway spruce seem more vulnerable than the other species. There is also an age effect, variously described as being influenced by thickness of bark, vigour, and phloem thickness. Mid rotation crops of 10–40 years age are generally more vulnerable than younger or older trees.

Evaluation of issues

The economic damage is significant for broadleaved crops with the potential for timber production, and in circumstances where squirrel populations become high, susceptible conifers can also be damaged significantly. But the significance of grey squirrel damage is not simply economic.

The impact of grey squirrels on the amenity values of our woodlands is increasingly apparent – in particular broadleaved woodlands of sycamore and beech, but increasingly stands of other species including oak, sweet chestnut, larch and Scots pine. In mid-summer, branches and limbs affected by squirrel damage become obvious as the foliage wilts, which is generally interpreted as unhealthy, dangerous, a measure of poor management, or simply unattractive. The most significant impacts are not necessarily associated with current damage – tree or branch death may follow the incidence of bark stripping by some years.

In places, however, there is a yet more significant and serious impact. An increasing area of our forest estate has the enhancement of biodiversity as a primary objective of management. Within broadleaved woodlands, this is often associated with the mature canopy trees, and there is also increasing recognition of the value and the contribution of over-mature veteran trees. These are at risk. When extremely high populations of grey squirrels occur, the damage to pole stage broadleaved species means that many saplings will never attain the stature to contribute to the canopy. Significant damage may only occur every few years when a combination of surplus food over the winter and mild conditions allow high populations to be sustained into the spring damage period. Over the lifetime of a broadleaved tree, however, this may be often enough to ensure that in some places only species such as ash can be guaranteed to mature into effective canopy trees. The impacts on future large veteran trees may not be fully apparent for decades.

The implications of this situation in the Forest of Dean are dire. The Forest seems to provide the ideal habitat for grey squirrels – there is a varied mixture of broadleaved and conifer trees offering a range of food and structure and shelter, there is a wide range of age classes intimately located throughout the Forest, and the size of each stand is relatively small increasing this diversity. Note that it represents the type of woodland being envisaged in much new planting and restructuring of existing woodlands elsewhere. Can such woodlands be sustained with periodic extremely high grey squirrel populations?

The significance of grey squirrels for biodiversity may not be restricted simply to impacts on the habitat, but they are less easy to document, and rely on circumstantial evidence. These are the consequences of grey squirrel presence for other fauna. Possibly the best documented is the relationship between the grey squirrel and the native red squirrel, where the increasing distribution of the grey has been followed by contracting distribution of the red. Less obvious, but increasingly considered, are the declines in other woodland species such as hawfinch and dormouse, where competition for food and direct disturbance and/or predation by grey squirrels may be a significant contributing factor in their demise.

The foregoing is a grim tale, and I suggest one of the biggest problems facing British woodlands in the 21st century.

Management strategy

In the Dean, management has to date concentrated on trying to control the population in the most vulnerable areas just prior to and during the main damage season. A bounty scheme was tried in the past, and trapping, shooting and poisoning with warfarin are all now deployed. The strategy costs tens of thousands of pounds each year. The reality is that the results are less than satisfactory. It is simply not possible to target vulnerable stands in the surrounding forest matrix. In addition, the allocation of budgets and the availability of skilled control staff are not able to respond to a task which fluctuates in scale from year to year. Squirrel densities during peak years have been far too high for standard control to reduce below the damage threshold. The current strategy in the Dean is to target control as best possible, but to highlight the seriousness of the situation wherever possible in the hope that further research may assist our plight.

Man

Management of the Dean involves significant commitment to issues associated with human use of the Forest, with many complexities. Over 30 000 people live in and around the Forest of Dean and for many the Forest literally comes to their doorstep. The Forest is closely intertwined with the community. Public awareness of the management is both high and generally well informed. 'Foresters' – natives of the Dean, are intensely proud of their Forest and have an inherent understanding of it as a working forest and are more than ready to involve themselves in issues of forest management.

The Forest is also a tourist destination. This was recognised years ago when the Dean was declared the first National Forest Park in England in 1938. About one and a half million visits are made to the woodlands of the Dean each year, and increasingly visitors demand a risk-free experience with increasingly sophisticated facilities. Walking, cycling and horse-riding are the most popular activities, but there are a huge range of specialist activities catered for, including rally driving, angling, athletics and nature study. The degree of use is beneficial in many respects – for example, fires are promptly reported and we gain some practical help. A high level of co-operation with parish and district councils and the county council and a wide range of other bodies, including Gloucestershire Wildlife Trust, Royal Society for the Protection of Birds, the Ramblers Association and statutory bodies such as the Countryside Agency, English Heritage and English Nature, is a cornerstone of our multipurpose management.

In the Forest of Dean we have in place the Forest of Dean Forum with representatives from many interest groups and elected councils; a Conservation Advisory Panel which includes experts in a range of environmental areas; an Archaeological Advisory Panel with a range of expertise in the area, and a Recreation Advisory panel bringing together representatives of forest users, all of which help us to develop and hone our policies and practices.

The close proximity of the forest to areas of habitation unfortunately brings with it antisocial behaviour. Problems with dumping of household waste and the use of unauthorised vehicles throughout the woodlands are common. With such a complex land holding intimately associated with the community, encroachment of land is also a common problem, and over a hundred cases are investigated and resolved each year.

Evaluation of issues

The overriding issue with management of an area as complex as the Dean is balancing the demands and aspirations of the various interest groups alongside the requirements to sustainably manage the forest environment and conserve its biodiversity. The various consultation procedures described above go some way towards this, but there are no simple solutions. Matters become complex when not all groups are willing to compromise with others. A particular issue in the Dean is balancing the local perspective against the visitors' perspective – despite having been a tourist destination for decades, there is still considerable opposition from locals towards tourism.

With the disappearance of the old heavy industries, the forest has become a more attractive place to live, and now many residents commute to work in cities such as Bristol, Gloucester, Birmingham, Newport, and Cardiff. The incomers may not have a perspective of rural management, and so do not necessarily have the same acceptance of forestry practices and can be more critical of any change to the environment.

Finance to manage these activities is also a recurrent challenge. Some returns are possible – for example, from car parking at the most popular sites, from letting of franchises for catering and from charging specialist user groups. But many activities do not generate, but absorb revenue. For example, collection of dumped rubbish throughout the Forest costs tens of thousands of pounds each year.

Management strategy

The overriding focus is on communication, and the various consultation routes described above require great commitment from all staff over and above their standard forest management duties. The core document for guiding forest management is the forest design plan, and procedures are in place for input from relevant organisations and the general public during the preparation phase of the plans.

In addition to this, the particular focus on human activity in the Dean is addressed through having a dedicated public affairs department. This includes a Forest Warden, (a post unique to the Forest of Dean) who patrols the Forest checking for bylaw infringements including encroachment. The team actively works with other agencies such as the police and the local Council to ensure activities are co-ordinated and appropriate to the circumstances. There is also significant input into information dissemination – with much contact with the media. At active work sites throughout the Forest, temporary information boards give details of ongoing operations to explain management procedures to the general public.

The network of facilities provided for public use (which in the Dean includes visitor centres, walking and cycling trails, an arboretum, play areas, viewpoints, picnic areas and a sculpture trail) are supplemented by a team of rangers who ensure that the facilities are safe and up to standard, and also offer a range of organised activities for the public.

Conclusions

The Forest of Dean is a complex area with a rich history. Management of mammals, including deer, free-roaming sheep, pests such as grey squirrels, and man is an important aspect of the overall administration of the Forest. Deriving appropriate management strategies which satisfy society's needs is a constant challenge – the environment, as well as the mammals within it are always in a state of flux. Adapting policy and practices which balance the various issues is constantly necessary to deal with the ever changing issues facing the managers of the Forest.

Further reading

The Senior Verderer for the Forest of Dean, Dr C Hart OBE, has written extensively about the history of the Forest, its woodland management and associated issues including mammals, and its traditions and much of the historical information has been sourced from the following books:

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In addition, Dr Hart edited 'The Forest of Dean, an Historical and Descriptive Account' written in 1858 by H G Nicholls and re-published in 1966 (David and Charles, Newton Abbot).

CHAPTER 15

Stock grazing in woodland

Helen Armstrong and David Bullock

Summary

British woodlands have evolved with large herbivores and recent thinking suggests that large, grazing animals may have been able to create woodland with a very open structure. Many, but by no means all, plant and animal species require this type of woodland. When deciding on a suitable stocking regime it is therefore important to set clear management objectives. Low stock grazing is generally good for biodiversity but particular species may need heavy, or no, grazing. Trees **can** regenerate in the presence of grazing animals especially if there are abundant regeneration niches and if there is some 'natural' protection from thorn bushes or dead wood. Our ability to predict the effect of a given stock grazing regime on a particular woodland is very limited. Additional monitoring, research and computer modelling are needed to improve our ability to provide management advice.

The problem

Stock animals are capable of having a range of effects on woodlands that are generally seen as deleterious. At high densities they can cause damage to field and ground layer plant species and prevent flowering and seeding. At even higher densities, for example around feeding points, or where a woodland is used as a holding area rather than to supply forage, the ground can become completely denuded of vegetation. Browsing by stock on young trees can completely prevent regeneration and bark stripping of older trees can damage, or kill, them. For these reasons, the exclusion of stock is often one of the first measures taken by a manager wanting to improve the nature conservation value of a woodland. However, this is not always the best action to take for two reasons.

First, some species thrive in heavily-grazed woodland. An extreme example is the netted carpet moth (*Eustroma reticulata)*, a rare species found, in Britain, only in parts of Cumbria and Wales. The caterpillars of this moth rely on the annual plant touch-me-not balsam (*Impatiens noli-tangere*) for food (Hatcher, 2003). The balsam cannot compete successfully with perennial vegetation and requires ground that has been heavily poached by cattle over winter. Hence conservation of this UK Biodiversity Action Plan Priority Species requires a form of management that would normally be seen as damaging.

Second, no grazing at all can be just as damaging to biodiversity as can heavy grazing. With no grazing, tall grasses and shrubs dominate, out-competing smaller herb and bryophyte species and generally leading to a decline in plant diversity. Although some bird and animal species may benefit from this dense shrub layer, many others will not. Invertebrates, in particular, often require a mosaic of open glades and closed woodland in close proximity to each other. This varied and relatively open woodland structure is not provided by either no grazing or heavy grazing, rather it requires light to moderate, or temporally variable, grazing levels.

Why is it that many species seem to require this sort of open, and varied, woodland structure? Were these species really confined to small, and temporary, gaps in the woodland or perhaps to exposed areas with high rates of windthrow or to areas of shallow soil where trees could not take root? Or was the intimate mix of habitats that they need actually quite common?

The history of grazing in British woodlands

After the last glaciation there were six species of grazing mammal present in mainland Britain which had the potential to affect woodland dynamics. These were aurochs or wild cattle (*Bos primigenius*), red deer (*Cervus elaphus*), roe deer (*Capreolus capreolus*), elk or moose (*Alces alces*), wild boar (*Sus scrofa*) and beaver (*Castor fiber*). Of these, only the red and roe deer remain in Britain today. But what sort of effect did these species have on the woodlands of the time? This is a very topical question because the Dutch ecologist Frans Vera recently published a book in which he challenged the traditional view that our 'natural' woodlands were largely closed-canopy forest in which the only gaps were caused by fallen trees (Vera, 2000). His view is that grazing and browsing animals drove woodland dynamics by creating mosaics of closed forest and savannah woodland alongside scrubby and open areas, much like the large herbivores of East Africa drive woodland succession today. Vera's vision certainly fits with the habitat preferences of many woodland invertebrates, lichens and birds. But how would it have been achieved? Figure 15.1 shows the cyclical succession, driven by large herbivores, that woodlands might have gone through before humans started to have a major impact. Grasslands were invaded by scrub species such as hawthorn (*Crataegus monogyna*) and blackthorn (*Prunus spinosa*) whose thorns protected them, to some extent, from browsing. Protected within the patches of thorn scrub, young trees, especially of light-demanding species such as hazel (*Corylus avellana*) and oak (*Quercus* spp.), could regenerate. Jays (*Garrulus glandarius)* are likely to have had an important role in planting acorns and they preferentially cache acorns in open areas. Over time, the other tree species overtopped the thorns and shaded them out, forming a stand of mature trees with no thorn understorey. The grazing animals could then graze under the canopy, preventing further tree regeneration. Eventually the mature stand died, allowing enough light to reach the ground for a grass sward to develop once again. Grazing pressure would have been high enough to prevent tree regeneration until the thorn species once again gained a foothold.

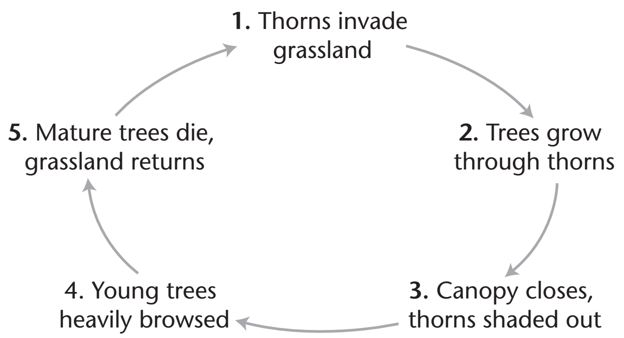

Figure 15.1

Schematic diagram of cyclical woodland succession driven by large herbivores (after Vera, 2000).

There is much debate currently about whether these processes really did occur and, if so, over what spatial and temporal scales. We will probably never know for sure but Vera's ideas do raise the possibility that, at least in lowland Britain, open woodlands may have been more common than was supposed. This may help to explain why many of today's woodland species need just this sort of woodland to survive. Vera's theory has also helped us to accept that large herbivores are integral to woodland processes, rather than just management tools to achieve a product.

But what about more recently? How have the species that need open woodlands survived to the present day? Along with Vera's theories has come a better understanding of the way in which our forebears managed woodlands. Increasingly there is evidence that many semi-natural woodlands owe their most valuable characteristics to intensive past management. This included grazing by cattle, and/or sheep, probably often at quite high densities. It might also have included putting pigs into the woodland in autumn to feed on acorns and other mast. As well as providing valuable forage and shelter for the animals, the woods were harvested for timber. Often this was done by coppicing young stems or pollarding older trees. When trees were coppiced, they had to be protected from browsing for at least five years using walls or fences. Coppicing results in quite dense stands of young stems with open areas where the stems have recently been cut. On pollarded trees, the new growth is above the height of grazing stock. Pollarding extends the lives of trees and this practice would have produced a forest with a low density of open-grown trees, including some veterans, little scrub cover and little, if any, natural tree regeneration. The relative importance of different outputs from the woodland, such as firewood, timber, charcoal and livestock, would have depended on the needs of the locals and the type of woodland. But, in general, what resulted from much historical grazing in woodlands may have resembled the open, varied habitats envisaged as common in pre-historic times and, in which many of our wood-pasture species evolved.

Latterly, these labour-intensive methods have ceased and woodlands have increasingly been used merely as 'holding yards' for stock. This has led to some of the negative effects discussed above. More recently, however, there has been an increase in interest in managing woodlands for biodiversity. There is a realisation that a lack of grazing can lead to a loss of both biodiversity and of bare patches of ground which act as seed beds for young trees. Interest in using stock as a management tool in woodland is now growing.

Using stock to manage for biodiversity

It will always be impossible to provide a grazing regime that is ideal for all species so the first, and most important, management action is to decide on the objectives for the wood. They might be very general, for example to increase overall biodiversity, or they might relate instead to a particular species or group. Either way, the objectives must be clear and the manager must have some understanding of what sort of woodland conditions are required. Most conservation managers use stock in woodlands either to open up the woodland, keep the ground flora short and reduce tree regeneration or to encourage trees to regenerate by creating suitable niches. This will usually require light to moderate grazing levels; however, there are occasions where a very heavily grazed (see the example above of the netted carpet moth), or an ungrazed woodland is required. Some woodland bat species need a dense understorey that retains humidity, and others require a high biomass of insects. To achieve these, very low, or no, stock grazing may be required.

Once the need for stock grazing has been determined, the next question is usually to choose between sheep and cattle. Aside from practical considerations, the ecological characteristics of the two species need to be considered. Sheep and cattle graze in different ways and both can have their uses. Cattle are thought generally to benefit biodiversity more than sheep for several reasons: they are less selective grazers than sheep and remove more coarse vegetation, such as purple moor grass (*Molinia caerulea*). Their trampling can also help to break up stands of bracken. This reduction in the cover of dominant plants means that smaller plant species are not out-competed and this can lead to a higher diversity in the sward. Cattle, like sheep, browse tree and shrub seedlings and saplings and so reduce densities of regenerating trees and shrubs; however, their hoof prints increase the number of potential tree regeneration niches. Whether cattle reduce or increase the density of regenerating trees at a particular site depends on the stocking rate and season as well as on other environmental conditions. Cattle also make tracks through woodland. It is thought that these benefit woodland grouse chicks by reducing their chances of getting wet from overhanging vegetation. This may be significant for the capercaillie (*Tetrao urogallus*) for which high spring rainfall has been implicated in low reproductive success (Moss *et al.,* 2001). Finally, compared with sheep, since cattle eat large amounts of relatively coarse vegetation, they produce copious amounts of dung, a valuable habitat for coprophagous invertebrates and fungi. Their large body size and high intake requirements mean they cannot thrive in woodlands with little ground layer vegetation, especially where graminoids are scarce. They may also have an effect on local deer populations either encouraging them by increasing the cover of the more nutritious plant species or discouraging them by creating a browse line on the trees which is above the height to which small deer can reach.

Sheep are particularly useful grazers in woodlands on slopes too steep for cattle, where forage availability is low or where it is desirable not to disturb the ground too much. Johnny's Wood in Borrowdale, Cumbria, is nationally important for its rich bryophyte flora. Thomason (1995) found that the bryophyte community does best when grazed in winter by 1.0 sheep ha^{-1} in open wood and 0.5 sheep ha^{-1} in closed wood. This stops the bryophytes being overgrown and allows some tree regeneration. Johnny's Wood is also good for wood warblers *(Phylloscopus sibilatrix)* which tend to breed in high canopy woods with little understorey.

The breed of stock needs to be suited to the site conditions (see Tolhurst and Oates, 2001 for breed characteristics) but, no matter which breed is used, the individual animals must be pre-adapted to the conditions on the site. Animals cannot switch from good quality forage to poor quality, or *vice versa*, without being given time to adapt physiologically and behaviourally to both the changed quality, and distribution, of forage and the availability of shelter and water.

Seasonality of grazing, number of animals and the length of time they are in the woodland will all influence the resulting structure and composition of woods. Johnny's Wood is one example where trial and error experimentation resulted in useful recommendations (Thomason, 1995). However, in general, there has been very little research on herbivore impacts on woodlands and it has rarely been possible to predict the outcome in terms of biodiversity in anything other than a general way. Current advice is often to maintain existing grazing if it seems to be working or to graze 'lightly', then increase the grazing pressure, if it does not appear to be achieving the desired outcome. However, with so many animal, and site, variables to take into account it can take many years to find the right grazing regime for a particular site. We would stress, however, that the objective may not be to achieve a particular product or pattern by introducing stock grazing. The objective may be to restore woodland processes in which the role of large herbivores is integral.

A survey of cattle-grazed woodlands in Britain

In 2000, Forest Research, the research arm of the Forestry Commission, started a programme of work on the impact of cattle on woodlands. Because of the lack of quantitative information, and the wide range of situations for which stocking regimes were required, we decided, initially, not to set up experiments. Instead, we gathered information from sites where cattle were already being grazed in woodlands. The aims were, firstly, to see who was doing what with cattle in British woodlands and, secondly, to see if we could use anecdotal information to draw out general lessons about successful cattle management regimes. The study was at two levels: initially we collated information from managers of sites across Britain. We then visited a proportion of these to gather more detailed information on woodland type, stocking regime and habitat areas as well as densities of, and browsing rates on, young trees. The site visits relied on systematic observations and categorisation of tree densities and browsing rates as, for example, high, medium or low; 105 sites were recorded of which 33 were visited. The sites were spread over most of Britain but there were particularly high numbers in Argyll and Cumbria. This is by no means a complete inventory of cattle-grazed woodlands in Britain but is likely to be a representative sample (Armstrong *et al*., 2003).

In England and Wales, sites tended to be owned by public bodies or by conservation charities whereas, in Scotland, most sites were privately owned. We did not exclude commercial plantations from the survey but most sites were semi-natural. In England and Wales the woodlands were largely dominated by oak (*Quercus* spp.), birch (*Betula* spp.), or ash (*Fraxinus excelsior*) whereas those in Scotland were dominated largely by birch, oak and Scots pine (*Pinus sylvestris*). Sites were generally small (5–50 ha) though a few were much larger (up to 5000 ha). Almost all the sites were grazed by other herbivores: most commonly roe deer, sheep and rabbits (*Oryctolagus cuniculus*). At most sites the cattle had access to open habitats as well as woodland. In England and Wales 23 breeds of cattle, as well as crosses, were being used and none was particularly common. In Scotland most sites were grazed by Highland, Luing or Aberdeen Angus.

The main objective of grazing with cattle in England and Wales was for nature conservation and in Scotland for cattle production. The nature conservation objectives fell into three main areas:

1. To benefit biodiversity generally by:
   - Reducing tree/scrub regeneration.
   - Reducing the existing shrub layer.
   - Maintaining open habitats.
   - Reducing dominant plant species.
2. To benefit individual species or groups.
3. To encourage tree regeneration.

For the first two objectives, most of these schemes were initiated too recently to judge whether they had achieved their objectives. However, most managers were happy with results to date. A preliminary analysis of factors affecting tree regeneration was possible using data from both the general survey and the site visits. Even though other herbivore species were present at most sites, there was a high (70–80%) chance of getting some tree regeneration at all cattle grazing pressures up to about 10 cattle months ha^{-1} (a little less than 1 cow ha^{-1} if grazed all year). But we only had five sites with more than 10 cattle months ha^{-1} so this conclusion is somewhat tentative. Note also that we have no information on what would have happened in the sample woodlands without cattle. It is possible that cattle encourage tree regeneration, even in the presence of quite high densities of other herbivores. This possibility deserves more investigation.

We used the data from the site visits to look for relationships between cattle grazing pressure and the density, or species, of regenerating trees. There was no relationship between cattle grazing pressure and the species of saplings present but there was an indication that the density of saplings declined as cattle grazing pressure increased. However, the large amount of variance around this relationship suggests that many other factors also affect sapling density. These might include available food for herbivores, seasonality of grazing, how long the present grazing regime had been in operation and the species composition of the saplings. Other factors such as weather, ground vegetation, invertebrates or soils may also be having an effect. The large number of factors that can affect densities, and species, of tree regeneration means that it is difficult to predict the impact of a given cattle grazing regime. One approach to making such predictions is to incorporate information about the effects of the various factors into a computer model. Jorritsma (1999) provides an example of just such a model for a pine forest on sandy soils in the Netherlands.

Recommendations for managers and researchers

- It is important to set clear management objectives before deciding on what sort of stocking regime might be suitable. Low stock grazing is generally good for biodiversity but heavy, or no, grazing can be needed for particular species.

- Trees **can** regenerate in the presence of grazing animals. Cattle, by creating regeneration niches with their hooves, can increase numbers of seedlings. This can counteract the effects of browsing. Thorn trees, or deadwood, can protect young trees from browsing.

- Systematic monitoring of woodlands grazed by stock, even if done qualitatively, can provide very useful information that could help improve our ability to provide advice on suitable stocking regimes.

- Further research is needed on the impacts of different stock grazing regimes on woodland structure and composition generally. In particular, it would be useful to know whether cattle can stimulate tree regeneration even in the presence of densities of other grazing animals that would otherwise be high enough to prevent regeneration.

- Computer modelling could help to take into account the many variables that affect both tree regeneration and the composition of the ground layer vegetation in grazed woodlands. Such models could provide the core of decision-support tools that could be used by land managers. This would reduce the need to determine suitable stocking regimes by trial and error.

Acknowledgements

We would like to thank the many site managers and others who provided information on cattle grazed woodlands. We are also grateful to several staff from the Forest Research Technical Support Unit who carried out most of the site visits and to Liz Poulsom of Forest Research who assisted with data collection.

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CHAPTER 16

Damaging agents or biodiversity: small mammals and young trees

Roger Trout, Niall Moore and Richard Jinks

Summary

A healthy small mammal population is one sign of a healthy habitat. However, young plantations, especially those in young farm woodlands, can be severely damaged by rabbits and small rodents. Planting grants include resource for reducing this risk, either by fencing or by enabling bare earth around trees to discourage rodent attack. The most extreme form, which also assists tree growth most markedly, involves a total herbicide application. While this effectively removes the potential for habitat for small mammals in the short term, the successional aspect of small mammal communities is merely delayed.

Introduction

Rabbits and some species of rodents can cause severe damage to seedlings and planted young trees. Rabbits have been shown to be the largest biomass of any species of mammals in England (Harris *et al.,* 1995) and their damage to a wide range of species is well known (Thompson and Wordon, 1956; Trout, 2003). The proportion of damaged trees in young woodlands, such as farm woodlands has been shown to be directly related to rabbit density but even low numbers can cause unacceptable damage (Hodge and Pepper, 1998; Trout, 2003). Grants to promote the planting of woods on farms (Farm Woodland Premium Scheme) FWPS include a substantial element (at least a third) towards protecting trees from damaging mammals, particularly rabbits and more recently deer. Typical management includes fencing or individual tree protection according to the size of plantation.

The conflict

Even if rabbits are excluded, small rodents and insectivores often thrive in newly-planted woodland sites. Their presence may have a strong biodiversity value, typically as a food resource for a wide variety of avian and mammalian predators. However, they can also cause unacceptable levels of damage, typically to tree seeds, emerging seedlings and to transplanted trees. Woodmice, *Apodemus* species principally eat the seeds and seedlings; Short tailed field vole, *Microtus agrestis* and bank vole *Clethrionomys glareolus* remove the bark from the base of young trees (the latter sometimes also climbs to remove bark near branches). There is a change in relative abundance of small mammals in typical broadleaved woodland over time (Gurnell, 1985). When the field layer declines as trees reach the thicket stage (branches touching) field voles become scarce while bank voles and woodmice increase. A similar pattern emerges for ageing coppice (Gurnell *et al.*, 1992). A recent series of studies has shown that in FWPS plantings, where trees are planted in highly productive ex-arable land the ground vegetation may be extremely luxuriant and small mammal numbers can reach very high numbers of hundreds per hectare (MacVicker and Trout, 1994; Moore *et al.*, 2003). In the north of England and Scotland, ‘cyclical’ field vole populations occur in these grassy young plantations (Lambin *et al.*, 2000; also Chapter 17 this volume). In years with high numbers of voles a considerable proportion of small trees can be damaged. In a sample of FWPS sites in Yorkshire, potentially damaging small mammal species occurred in 11 out of 12 sites (Moore *et al.,* 2003).

One aim of the *English forestry strategy* is to increase the area of woodland cover (Forestry Commission, 1998). Changes in farmland and forestry policies suggest a move towards doubling the woodland cover in the UK, perhaps by 2040. This presents an enormous opportunity for mammalogists, to study the dynamics of wide range of small mammal species during the woodland establishment phase. However, for the forester there is an equally great challenge to ensure that the planned woods actually survive and grow.

Woodland expansion can occur by several means. Natural regeneration of felled or thinned woodland can occur from suckers or by seeds from the surrounding trees falling onto the woodland floor, being transported by animals or moved by natural actions e.g. wind/water. This can be more successful where there is little grazing pressure. Alternatively, man can determine precisely where trees occur through either direct seeding or by planting small trees.

Natural seeding success varies considerably from year to year and site to site, principally dependent on ground vegetation, the overstorey, soil characteristics but also the annual variability of the seed crop. Oak, beech and conifer seed releases can vary by many-fold, e.g. 100–10 000 KJ/m^2 (Gill *et al.*, 1995). This not only affects the number of potential new trees but also the amount of food available for small mammals. Woodmice and bank voles survive better in good mast years; woodmice apparently bred more successfully in the spring following a good mast year (Mallory and Flowerdew, 1994).

Small mammals also feed on seed in nurseries and direct-seeded sites. Direct seeding is the establishment of new woodland by sowing tree and shrub seed directly onto a prepared site. Compared to conventional planting, direct seeding has the potential to increase the speed of woodland establishment on ex-farmland, reduce establishment costs and produce varied stands with a more natural appearance. However, even more so than with using plants, weed control and protection from browsing mammals are essential to ensure the survival of seeds and emerging seedlings. However, establishment is less predictable, and complete failure is common due to the vulnerability of the seed and young seedlings to extreme soil conditions, climate, fungal, mammal and insect attack. For example, seedling emergence of ash in an experiment sown at three sites varied between 68% (ex-pasture, surface water gley), 32% (ex-arable, greensand) and 0% (windblown beech woodland) (Willoughby *et al.*, 2004). Seed predation by mammals was responsible for the complete failure at the woodland site. In such habitats seeds are often rapidly removed within a few days of sowing. Current research is concerned with identifying the characteristics of sites where direct seeding is likely to succeed, and developing robust silvicultural methods to increase establishment success. One particular situation currently appears to be ex-agricultural ground (e.g. FWPS) where the herbicide programme needed to ensure seedling emergence and survival keeps the ground clear of vegetation for up to two years. However this herbicide policy can result in a temporary paucity of mammalian fauna in comparison to 'normal' farm woodlands which usually quickly have a thick grass component.

The normal method of increasing woodland cover under the FWPS involves planting small trees, usually grown for 2–3 years in nurseries and prepared by root pruning and lifting in autumn (bare rooted) or using seedlings grown in small specially designed containers (cell-grown). The ground is often previously highly productive farmland and competition from grasses and weeds can easily kill the planted trees if not controlled. In order to establish whether there was a link between vegetation and vole populations on new farm woodland, the amount of vegetation in experimental paddocks in Hampshire was manipulated by rabbit grazing or by mowing (MacVicker and Trout, 1994). Vole density and sign indices reduced in line with vegetation biomass. In each case damage by voles was highest when vegetation was thickest (Figure 16.1.) despite the presence of herbicide sprayed patches round the trees. In the mowing study the index of vole activity fell by almost 90% after mowing. During a similar study in Yorkshire, mowing in July also severely affected the density of shrews but after 12 months the species composition and density had returned to previous levels (Moore *et al.,* 2003).

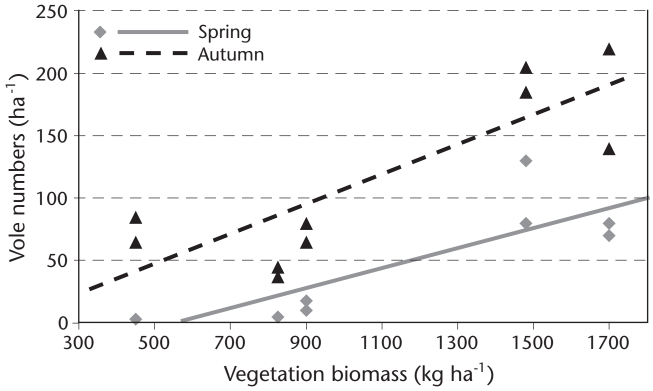

Figure 16.1

Relationship between vegetation biomass and field vole density in farm woodland plantings in southern England.

Management to reduce damage

If management of the young plantations needs to include reduction of the impact by small mammals, then there are several options, based on three different principles for managing damage:

- Prevent the mammal from reaching the vulnerable crop by physical or chemical means.
- Make the habitat less suitable for the mammal so they do not live or remain there in large numbers.
- Remove the animals concerned to a density where there is only limited or acceptable levels of damage.

In well-planned management, several options are often used concurrently.

For rabbits, individual protection using guards, the use of repellents, or perimeter fencing of larger plantings are used (Hodge and Pepper, 1998). Ensuring a lack of harbourage within or nearby (e.g. brash: the waste after felling; windrows: piles of removed tree roots) prevents the local population density rising. Trapping, shooting, fumigating and ferretting are the most commonly used methods for direct population control (Trout, 2003), but are not always effective.

For small rodents the options are different. Small coiled 'vole tubes' 25–30 cm in length can provide protection so long as they are pushed partly into the ground and any surrounding vegetation does not provide a bridge. A problem can occur with the larger individual protective tree shelters normally used against rabbits and deer, because rodents may live inside the tubes (protected from predators and poor weather) unless the tubes are pushed well into the ground. In one study all unguarded trees and 60% of those in tree shelters were eaten after 10 months (MacVicker and Trout, 1994). The problem is most acute on soils that crack in summer.

A common form of management by foresters is the use of herbicide to reduce competition from vegetation to establishing trees; this has a secondary impact on voles, because they tend not to cross bare ground. The management operation can vary from spraying the entire plot prior to or after planting in winter, spraying strips where the trees are planted, or reducing the weed cover round each individual tree by circular spray spots 1 m in diameter. As an alternative, there are a variety of small sheets of material (mulch mats) that can be placed round the base of trees to reduce weed competition without the use of herbicide. This is an attractive option under the UK Woodland Assurance Scheme (Anon, 2000) which seeks to reduce pesticide use. However, research indicates that the voles live under the mats and ringbark the tree or eat the roots; in one trial 93% of mats had vole signs after 8 months (Moore *et al.*, 1996). Year-long trials in Kielder Forest tested five potential repellents painted on the underside of these mats in an attempt to lower the attractiveness. Natural predator odour and individual chemicals failed to show a long-term effect (Moore *et al.*, 1996). The two commercial products tested (Aaprotect and Renardine) were reasonably effective (40% infested) after 8 months; however neither is cleared for use against voles. An experimental trial of a material called Woebra resulted in only 8% damaged in 10 months. However, in practice there are no repellents against voles currently cleared for use in the UK.

Killing rodents is not a viable option for species that can reach several hundred per ha, though indiscriminate large pitfall traps have apparently been used overseas. Perches for predatory birds can be provided but there is no evidence that this action, which may favour an increased predator pressure, actually suppresses high vole populations or prevents damage to trees.

Conclusion

While small mammals can be a considerable problem to young trees and benign management may be very costly, the presence of small rodents, and other small mammal species, are an important biological resource and are an indicator of healthy habitat. As such they should be encouraged in future land management or at least not discouraged where practicable. The policy to increase the woodland area in England presents an important opportunity for foresters and mammalogists to consider and understand the cross implications of each other's aims and to plan to achieve a balance in time and space where possible. Where heavy vegetation is expected, spraying and maintaining large spots or strips rather than overall spraying may allow both objectives to be achieved.

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CHAPTER 17

Relationships between clearcutting, field vole abundance and the breeding performance of tawny owls in upland spruce forests in Britain

Steve Petty and Xavier Lambin

Summary

This paper summarises a comprehensive set of investigations into the ecology of raptors and field voles within upland spruce forests. The relationship between forest management, field vole (*Microtus agrestis*) populations and populations of tawny owls (*Strix aluco*) in northeast England and west Scotland is briefly described and references provided.

Introduction

Clearcutting (or clearfelling) is a widespread method of harvesting timber in upland conifer forests in Britain. Clear cuts provide extensive areas of early seral stage vegetation within otherwise closed-canopy conifer forests, and thus provide opportunities for wildlife adapted to such habitats. Here we introduce comparable data collected during 1984–1990 on the small mammal communities present on clear cuts in one conifer forest in northern England (Kielder) and another in western Scotland (Glenbranter), and show how variation in the abundance field voles *Microtus agrestis* influenced the breeding performance of tawny owls *Strix aluco*.

Field voles were by far the most abundant small mammal species present on clear cuts dominated by grassy vegetation, and while vole populations exhibited large multi-annual variations in abundance, these were not synchronised between forests. In both forests, tawny owls were the most abundant raptor and field voles their most important prey.

Breeding performance and vole abundance

Vole abundance influenced the breeding performance of tawny owls in a similar way in both forests, with breeding being earlier, a higher proportion of pairs laying eggs and clutch and brood sizes being larger when voles were abundant. For example, in Kielder the mean start of incubation commenced on the 16 March in the best vole year and on the 10 April in the poorest vole year, 94% of territorial pairs laid eggs in the best vole year but only 8% in the poorest vole year, and mean clutch and brood sizes per territorial pair were 3.22 and 2.94 respectively in the best vole year but only 0.21 and 0.14 respectively in the poorest year.

Over the six year period, there was greater temporal variation in the breeding performance of tawny owls in Kielder (proportion of pairs laying eggs Coefficient of Variation (CV) = 51.33, start of incubation CV = 40.11 and mean brood size CV = 64.06) than in Glenbranter (proportion of pairs laying eggs CV = 16.85, start of incubation CV = 16.97 and mean brood size CV = 37.53), which partly reflected the greater temporal variation in vole abundance in Kielder (CV for April/May vole abundance was 67.12 in Kielder and 46.85 in Glenbranter). Between the best and worst years, the number of chicks produced per territorial pair of owls varied 21-fold in Kielder (0.14–2.94 chicks), but only 4-fold (0.45–1.75 chicks) in Glenbranter.

Conclusion

The work has highlighted:

- The importance of clear cuts as a habitat for small mammals in upland conifer forests. In particular, field voles abound wherever grassy vegetation develops. In contrast, older (closed-canopy) spruce forests have low densities of small mammals.

- Field voles are an important source of food for a wide range of avian and mammalian predators in upland conifer forests.

- Field vole populations exhibit large multi-annual variations in abundance that have important consequences for the demography of their predators.

- Over time, patch clearcutting creates habitat mosaics of different-aged stands of trees. The optimum size of clear cuts will differ among species. For example, tawny owls appear to benefit from small clear cuts (up to 25 ha) that create fine-grained mosaics whereas short-eared owls are only present in coarser-grained habitat mosaics with clear cuts larger than 60 ha.

A fuller account of the work will be published elsewhere.

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CHAPTER 18

Diseases of woodland mammals

Paul Duff

Summary

This paper aims to provide woodland managers and mammalogists with an overview of a selection of diseases that affect woodland mammals. Zoonotic diseases (capable of transmission to humans), as well as diseases and causes of death in woodland mammals are covered. Advice is given on field investigation and monitoring, and expert sources are identified.

Introduction and concepts

Assessing the significance of disease in woodland mammals is important for the following reasons:

- Wildlife may be vectors for diseases transmissible to humans or domesticated stock.
- Disease may have a significant effect on wild populations.
- Disease may reflect a change in the woodland ecology, for example, through pollution.

Simpson (2002) provides one of the few summaries of important infectious diseases of wild animals (and birds) in the UK, while Duff (2003) lists organisations that may be able to assist in diagnostic investigations. The Handbook of British Mammals has notes on disease, but there are now significant gaps in the information provided. The Veterinary Laboratory Agency (VLA) Wildlife Quarterly Reports provide notes on diseases of current interest and these can be found on the VLA Website, the address of which is given in the Contacts section.

'Disease' in its widest sense includes toxicities, neoplasias and malnutrition, in addition to infectious disease and those conditions caused by human agency (e.g. shooting, hunting, trapping and road traffic accidents). The diagnostic investigation has to consider all possible causes, because in order to assess the significance of a problem the cause must first be found. The investigation requires specialist examinations and diagnostic tests. However, woodland managers can make an early assessment by taking a good clinical history. This involves recording the clinical signs and evaluating the problem *in situ*; in the light of what is known of the ecology of the species in the particular woodland.

On the other hand, managers may suspect disease in a wild population, but observe no clinical signs. In these cases the disease may be *sub-clinical,* that is, involve animals that show no outward signs of disease but are still carriers of infection. Testing for the presence of a sub-clinical infection usually requires taking samples, often under license, and performing laboratory tests on these samples. It is also important to point out that signs of infection may be found, such as white tapeworm segments (that look like grains of rice in faeces), or parasites such as fleas, that are of little significance to the health of their mammalian hosts.

When wild mammals occur at high densities then transmission of infection within the population may be frequent. As a consequence the frequency and severity of the disease caused by the infection may also be high. Disease may appear to act as a natural selector curbing the growth of a burgeoning population, but in wild populations cause and effect are far from clear, and one can only conclude that this is an interesting but usually unproven concept.

Zoonotic disease

Investigators must be aware of zoonotic disease (diseases transmissible from animals to humans) for their own health and for the health of others e.g. users of the woods and for those who receive their diagnostic specimens. All UK mammals have the potential to carry zoonotic disease so be wary and take common sense precautions. Making a risk assessment for known zoonotic diseases can be difficult; however, these conditions in the UK are usually not severe and human fatality is very rare, so a common-sense approach is required.

Prevention is the best method of zoonotic disease control; in particular using protective clothing (disposable plastic gloves) and exercising good personal hygiene. Do not place your fingers near your mouth, or scratch your chin, for example, when pondering what killed the dead fox you have just turned over. Remember always to wash your hands after handling animals and their products (household soap is fully effective), even when gloves were used. Infectious zoonotic agents survive in cadavers and infection can spread from a dead animal to humans, consequently post-mortem examinations should be done by trained personnel in suitable facilities.

Zoonotic diseases here are arbitrarily split into three categories:

- Those where wildlife are a significant reservoir for disease in humans, e.g. Lyme disease, and Wiel's disease (leptospirosis).

- Significant zoonotic diseases present in northern Europe but probably not found in the UK, e.g. tularaemia, hantavirus, rabies virus. The English Channel effectively protects us from a host of zoonotic diseases; the three mentioned are present in countries bordering the Channel and North Sea. We need to ensure that these are not introduced and, when visiting the Continent, that we are aware of them.

- 'The other zoonotic diseases'. These occur infrequently in humans in this country. They include – hepatic capillariasis (mainly rodents and lagomorphs), salmonellosis (all species; some like hedgehogs and badgers have host adapted strains), ringworm (many species), toxocariasis (foxes), tuberculosis (badgers and deer but exceptionally present in other species), cowpox (rodents, in particular voles), toxoplasmosis (many species), *E. coli* 0157 (rabbits), campylobacteriosis, yersiniosis, cryptosporidiosis (most species). Diseased and 'found dead' wild animals must not be consumed by humans and wild game must be cooked thoroughly.

From group 2, the epidemiology of Hantavirus infection is considered. This is a viral infection of rodents, in particular bank voles, in which infection is sub-clinical. The epidemiology of this disease has been studied by Marc Artois, in the Ardennes where it is common. When bank vole populations increase, the number of human hantavirus cases simultaneously increases. The simultaneous occurrence suggests that humans are not infected by *direct* contact with the voles but by indirect transmission arising from contact with materials (e.g. wood, dust and vegetables particularly in wooden sheds) contaminated with infected vole urine. In Finland, human infection arises from sawdust present in sauna huts that becomes contaminated by vole urine.

Diseases and causes of death in woodland mammals

The list of diseases recorded in British mammals has increased in recent decades due to a wider interest in the subject, more cadaver examinations and the wider use of diagnostic tests, in particular serological tests. Limited space allows for only a selection here. These have been considered under the headings **Viral**, **Fungal Bacterial** and **Parasitic** infections together with some **Non-Infectious** conditions. Traffic trauma, shooting, trapping and poisoning incidents are found in most species, and when not fatal, these anthropogenic injuries may lead to animals that appear sick. Traffic accident victims, for example, may eventually die some distance from a road.

Fox

Viral – *Rabies*, sylvatic or fox rabies has never occurred in the UK and this without doubt has had a significant influence on how we as a country regard our wildlife as a wholly beneficial resource. *Canine Distemper Virus* infection has been suspected but not confirmed in UK foxes. **Bacterial** – Foxes are prone to opportunistic bacterial infections, one example is septicaemic bacteria introduced by *bite wounds* from intra-specific aggression particularly at the time of the vulpine rut. *Leptospirosis*, probably not uncommon in foxes. **Parasitic** – *Mange*, common in some populations and the subject of study but the causes of the waxing and waning incidence are unclear. *Heartworm* – first seen in Cornwall several years ago now more widespread, can be fatal. **Non-Infectious** – *Poisoning* – deliberate or accidental is not infrequent. Some poisons e.g. metaldehyde and carbamate may be manufactured with a dye, obvious in stomach content. These agents often retain their high toxicity in cadavers, which should be examined by trained pathologists.

Badgers

Viral – *Badger Herpes virus*, thought to be non-clinical. *Distemper* – probably not in the UK. **Bacterial** – opportunistic infections (as in foxes), including *bite wounds* associated with intra-specific aggression. *Bovine TB (tuberculosis)* – an enormous field and the subject of intensive, multi-disciplinary research **Non-Infectious** – traumatic deaths (*road traffic accidents*).

Other Mustelids

Viral – *Aleutian disease* is carried (sub-clinical) in Mink, but it is unclear whether infection gives rise to disease. **Bacterial** – *bite wounds* in otters particularly to the external sex organs, due to intra-specific aggression. **Non-Infectious** – *Road traffic accidents*, gamekeeper killing *traps*.

Deer

Viral – *Foot and Mouth Disease (FMD)*, exotic (does not normally occur in the UK), and if cases did occur in the 2001 FMD outbreak (none were confirmed by Pirbright Laboratory in wild or farmed deer) these did not prolong the epidemic. *Malignant catarrhal fever*, *Mucosal disease* – both occasional. **Bacterial** – *Tuberculosis, Johnes disease*; both occasional and both may reflect 'spill-over' of infection from domestic animals but this is unclear. **Parasitic** – *liver fluke* in roe, *lungworm* in red deer, *gastro-intestinal helminthiasis*. **Non-Infectious** – *OO rape poisoning* (roe), *'roe die-off'*, it is unclear whether either disease occurs in UK roe. *Road traffic accidents* are a major cause of injury, estimated at 30 000 deer annually. In recent years between 10 and 20 people are also known to have died in deer-related RTAs. Deer-related RTAs can be reported on a national register by visiting (www.deercollisions.co.uk).

Boar

Viral – *Swine Fever, FMD, Aujesky's disease* (all exotic, but important foci of the highly infectious Swine Fever occur in boar in Northern Europe). **Bacterial** – *Tuberculosis, Brucella suis* (both exotic). **Parasitic** – *Trichinellosis* (exotic). All the diseases mentioned are currently exotic in the UK feral boar population.

Insectivores/Hedgehogs

Fungal ringworm, **Bacterial** – *Salmonellosis – Salmonella enteritidis pt 12* in hedgehogs **Parasitic** – *Crenosoma* sp. lungworms.

Bats

Viral – *European bat lyssavirus (EBL)* [Please note: send all dead bats to the Rabies Unit, VLA Weybridge, KT15 3NB, for EBL monitoring]. All bat workers should be rabies vaccinated. Recently a small study in Scotland has detected EBL antibodies in approximately 10% of the Daubenton's bats tested, this further strengthens the probability that EBL is endemic in British populations of this bat. Two other common bat species examined in this study did not have antibodies. **Non-Infectious** – poisoning and predation – groups of dead bats outdoors may indicate *cat predation*, while if found indoors they may be due to timber treatment *poisoning*.

Rodents

Viral – *Cowpox.* Cowpox is a zoonotic disease. Studies by Malcolm Bennett indicate that in the reservoir rodent species, cowpox is usually sub-clinical, but skin lesions of varying severity occur when humans and cats are infected. There were 51 human cases between 1969 and 1993. Seasonal increases in both human and cat cases occur in the autumn months and are probably associated with increases in the main reservoir population, the bank vole. **Bacterial** – *Mycobacterium microti* ('rodent tuberculosis')*, leptospirosis, yersiniosis, campylobacteriosis* (zoonotic bacterial diseases).

Squirrels

Viral – *Squirrel parapox.* There is increasing evidence that parapox virus infection is an important factor involved in the extirpation and replacement of the native red squirrel population by the introduced grey squirrel in the south of mainland England. The grey squirrels are sub-clinical carriers of the parapox virus, but infection is invariably fatal in reds, which develop severe skin lesions. This is a rare example in the UK of an infectious disease significantly reducing a metapopulation. **Parasitic** – *Toxoplasmosis*.

Hares

Viral – *European Brown Hare Syndrome (EBHS).* **Bacterial** – *Yersiniosis*. **Parasitic** – *Coccidiosis*. *Dysautonomia* (cause not known). All these conditions are more prevalent in the autumn months, probably related to increased hare densities at this time of year, and all have been recorded in outbreak form.

Rabbit

Viral – *Myxomatosis, Rabbit Haemorrhagic Disease (RHD)*. RHD first occurred in domestic rabbits, probably introduced from the continent, in the early 1990s, and shortly afterwards the disease appeared in wild rabbits. Local epidemics in wild rabbits occurred and still occur generally in the autumn months in an unpredictable pattern. The disease could have produced losses of greater severity but it appears that populations may derive immunity from a virulent RHD-like virus. **Bacterial** – bites/scratches causing large bacterial *skin abscesses*. **Parasitic** – *Hepatic coccidiosis*, producing a friable, sawdust-like liver. This is a frequent cause of mortality in young rabbits. Affected individuals, dull with rough coat hair, may be obvious in observed colonies compared to the older immune, healthy rabbits.

Field investigation practice

On finding the body of a dead mammal, it is important to establish if this is part of a larger mortality incident. This can obviously pose problems in woodlands where vegetation may conceal bodies and even terminally ill animals can easily find cover in which to die quietly. The next consideration is to assess the risk of zoonotic disease, based on a knowledge of the diseases recorded as occurring in that species. With regard to equipment for safe handling, the basic tools are disposable plastic gloves and rat-toothed forceps (with opposing teeth at the end of the forceps for gripping tissue). With these, almost in the manner of a scavenging bird, a cadaver can be efficiently examined, and the body turned over with minimal risk to the operator. The forceps can also be used to place smaller bodies in seal-able bags. These bags and sampling tubes are also required for faecal samples or hair and skin lesions from dead animals that cannot be moved to a post-mortem facility. If cadavers have to be stored they should be placed in a sealed container or bag, and refrigerated in preference to freezing, although either is acceptable. It is still possible in the UK to send small pathological samples including cadavers through the post, provided that the Post Office regulations for this purpose are followed to the letter. It is essential that specimens should not pose a risk to specimen handlers. Where live affected animals are observed, a careful description of the clinical signs is required.

The next step in an investigation is to consider the possible causes of death and for this the mnemonic THINPET has been devised in order to list the seven commonest causes of wild animal death. The initials stand for the following causes:

T – Trauma	e.g. road traffic accident
H – Human	e.g. shot/trapped – death directly caused by human activity
I – Infectious	viral, bacterial and parasitic diseases
N – Nutrition	including starvation and malnutrition
P – Predation	important to assess whether it occurred before or after death
E – Environment	e.g. storm deaths, hypothermia
T – Toxic	accidental or deliberate poisoning

There are several methods of making a diagnosis but the commonest pathway involves the following steps, (1) history (epidemiology), (2) clinical signs (if observed) (3) pathology (post-mortem) and (4) laboratory tests. The information from each of these steps is considered with the possible causes, prompted by THINPET, to hopefully allow a diagnosis as step (5). A veterinary pathologist should undertake the diagnostic post mortem examination. The process can be summarised as:

(THINPET) History ➤ clinical signs ➤ pathology ➤ laboratory tests ➤ diagnosis

Monitoring of mammalian diseases

Monitoring diseases of wildlife on a local, national and international basis are increasingly important for the reasons listed in the Introduction. National monitoring of wildlife diseases has three requisites:

- Regional pathology – dead and dying wildlife does not travel well!
- Expertise in pathology, diagnostic tests and disease investigation.
- Knowledge of the species and its ecology.

Since 1998, the VLA, in the first government-funded project of its kind, has been sponsored by Defra to provide surveillance for wildlife disease in England and Wales. This has resulted in the VLA Diseases of Wildlife Scheme (VLADoWS). The Scheme has several objectives but is particularly focused on investigating wildlife diseases where there may be risks to human health and the health of domesticated stock. If submission criteria are met, then necropsy and diagnostic examinations, to a high scientific standard, are undertaken, with the person making the submission receiving a copy of the post-mortem reports.

The VLADoW Scheme has important links with other wildlife disease projects including the sharing of a database with a similar scheme in Scotland. The VLA has 14 Regional Laboratories and 2 university surveillance centres (see Contacts section below) with extensive diagnostic tests, and it offers at least 2 of the 3 essential points listed above. The Agency also has unrivalled experience in investigating wildlife disease and the first reports of many new diseases of wildlife in the UK have come from the VLA and its predecessor organisation.

In addition, there is also a range of other government and non-government projects that make a valuable contribution to wildlife disease investigation.

Conclusions

The word 'disease' has negative connotations that run counter to those feelings of vigour and natural beauty that inspire our interest in wildlife and wild places. However, disease and death are essential functions of the most pristine ecosystems, while in the unbalanced or polluted ecosystem, abnormal rates of disease may be the first observed sign of a developing problem. Disease may also be an important factor in the conservation of our wildlife resources, and for our own health and the health of our farmed stock. For these reasons, wildlife disease should be both investigated and monitored. In woodland habitats the profusion of cover makes the investigation of wildlife disease more difficult.

Recommendations for managers and researchers

- Be aware of the diseases and causes of mortality that are common in the mammals in the woods that you manage or study.
- Be aware of the recognised zoonotic diseases in these species and produce local risk assessments for you, your staff and people using the woods.
- Be aware of the natural patterns of diseases in your ecosystem; if new diseases or new patterns of mortality emerge, have a simple investigation plan.
- Identify sources of help (local and national), should diagnostic investigation be necessary.

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Contacts

VLA Diseases of Wildlife Project contacts:

- North and general enquiries:
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VLA Diseases of Wildlife Project website for Wildlife Quarterly Reports and project information: www.defra.gov.uk/corporate/vla/science/science-end-survrep.htm

SECTION FIVE

Woodlands and their mammals in the future

CHAPTER 19

How will our woodlands be managed in the future and what will be the implications for mammals? Crystal ball-gazing focussed on Welsh forests

Ruth Jenkins

Summary

This paper seeks to give some insights into the way that future changes in forest management, and the resultant character of forests, may interact with the mammal populations.

The impact of current and future forest policies

I have had to do a certain amount of crystal ball-gazing to tackle this topic. Where will current policies take us in the next 50 years? It would be difficult enough to describe exactly how **current** policy might impact on mammals, but no doubt such predictions must take into account further changes to government and its policy priorities. There will of course be woodlands and forestry in 50 years time. However, society's understanding of what constitutes a forest, and what functions are provided by forests, may have to change to meet the challenges of sustainable development.

Change is constant, and through the past 80 years forestry practice (and the Forestry Commission) has had to adapt to survive. Although the refinement through evolution is not so advanced as demonstrated by the mammals that are the focus of other papers, a winning design is emerging! Although sometimes reluctantly, forestry has managed to widen its aims. In Wales, devolution is allowing us to develop and measure forestry's profile against local and global needs and is giving us the tools to enable this fantastic resource to fit the needs of society. Other papers in this publication indicate the degree to which practices have changed, and mammals have benefited. We are using ecological information (such as that illustrated in the earlier chapters) to help create policies and then seeking to ensure woodland management operations and incentives reflect these policies.

Is knowledge sufficient and accessible for future policy development?

Do we know enough to make further improvements to policy and practice? The bane of conservation is often lack of information. Although some may consider that there is enough information for foresters to make informed decisions about woodland management, such an assumption is wrong. Information requires work to collect but good communication to make it available. Despite study, research and technology and the excellent work by the voluntary sector and other agencies, land managers are still not well enough informed. Access to existing information is not yet widely available. The National Biodiversity Network (NBN) and local record centres may hold the key to improving this situation. In Wales we hope to have local record centres covering the whole country within the next few years, and many partners and volunteers are supporting this development. However, making this information available is still not enough, we need to make sure that it is central to, and demanded by, the grant aid and associated regulatory schemes that are developed for 21st century forestry. Forestry Commission Wales is currently building a woodland management planning system, which will provide a route to grants and permissions. This route will direct land managers past this information, ensuring that the environment is treated as an essential feature of management.

A broader view of woodlands and forests

Managing woodlands for the environment today is not just about preventing damage. A common direction needs to be identified for the forest economy, people's interaction with woodlands, and respect for the forest environment. As people have become disconnected with woodlands over time the value placed on this resource has decreased. The Tree Council recently presented some interesting figures at an FC/National Park conference – 98% of survey respondents can name a British tree species and 98% say they want more trees but, alarmingly, only 27% want them in forests. This indicates a need to reconnect people with forests and one way to do this is to change the concept of what constitutes a forest. This will require a wider debate on land-use, not just a focus on where we expand, how we manage and when we fell trees in our forests. Forestry cannot (and indeed does not) continue to make decisions in isolation from other land-uses and owners. This will require an integrated and collective vision for the environment and decision-making on a much larger scale.

It is important to remember that woods are not just ecological habitats. Without other purposes (and values) it is unlikely that they will survive to support our wildlife. Forests need to sustain people as well as wildlife. Owners still want an economic return but are increasingly looking for revenue from sources other than timber. Finding new markets for woodland will require us to re-evaluate their role to society in order to see how we can use them to improve our lot and the habitats for our wildlife.

Wildlife is clearly part of this woodland future, but there are some mammals (such as grey squirrel, vole and deer) which are likely to continue to cause specific problems in the achievement of the objectives of woodland management.

Future management for woodland mammals

Mammals are already changing the way woodlands and forests are managed in Britain. Previous chapters have already discussed the management of woodlands for red squirrel, dormice and bats. Such species are certainly the focus of management in Welsh forests, with Forest Enterprise and Forest Research taking a lead at Clocaenog forest in north Wales. We are now looking at what the findings of this work will mean for other forests in Wales. We still have many lessons to learn. Research and science are often not enough. We must not ignore the energy of local individuals and groups in making a success of conservation projects.

Habitat networks

Limiting factors for species are often related to patch size and habitat isolation. For example, the pine marten has declined due to woodland fragmentation and changes to the management of this resource. The scale of these limitations probably varies depending upon species characteristics and whether the focus is mammals, plants or insects. Work on forest habitat networks elsewhere in Britain is improving our understanding of how de-fragmentation of woodland can reduce the ecological isolation of woodland species, thereby making them more robust against change. Such defragmentation should not be so extensive that woodland would start to affect the open ground species of conservation concern (or the aesthetics of valued landscapes).

Habitat quality is also important – in George Peterken's phrase 'the matrix matters'. High quality matrix habitats often survive intensive land-use practices when they occur within woodland. This obviously requires some open ground. Small woodlands of less than 3 ha usually have no gaps, in addition they are less likely to have continuity of management. Therefore, an aim in the future should be to increase the size of small woodlands and to target new woodland cover in areas where there is more chance of linking important habitats. In England this has been put into practice with the Forestry Commission's JIGSAW project and in Wales grants have also been targeted to build on woodland within existing natural networks such as riparian habitats where there are more likely to be existing stepping stones.

However, more information is needed on the interaction between mammals and internal habitat networks. Networks within networks may be equally important for connectivity of species. For example, it may be important to link wet woodland with more wet woodlands as the species associated with one habitat require certain patterns and processes only associated with certain woodland types or attributes such as deadwood. Some linkages may be problematic, allowing the spread of alien and invasive species and perhaps creating population sinks as individuals move and become more vulnerable. Forestry Commission Wales together with Countryside Council for Wales are hoping to build on this work to see how forest networks might improve biodiversity in Wales and contribute to sustainable development. Defragmentation might also provide improved economic and social opportunities.

Woodland management for the future

An earlier paper (Chapter 3) has already given an overview of contemporary woodland management. Political developments such as devolution, the impact of global markets on timber prices, increased knowledge about how woodlands impact on and benefit the environment are all changing the way we value woodlands and consequently changing the way we manage them. The long-term nature of forestry means that the decisions we make now will of course impact for a very long time. We have already seen, quite clearly, the impact of past decisions such as tax incentives and conversion of ancient woodland sites. Making changes that actually deliver tangible outcomes for the environment will often take a long time.

Some of these changes have already started. More than 50% of woodland in Wales is certified as sustainable. The forestry sector has already made a positive response to restoration of ancient woodlands and the country woodland strategies have put a high-quality environment at the heart of woodland management. The Welsh Assembly Government has set a target to increase the area managed using continuous cover techniques, thereby resulting in less clearfelling. The devil is, of course, in the detail and the value of continuous cover to wildlife will require decisions to be made beyond the stand scale. Continuous cover forestry is often equated with reduced disturbance and greater continuity and while there are obvious advantages, the reality of this will require careful monitoring. It may result in more natural regeneration of trees, smaller-scale operations, more frequent interventions, and perhaps more deer and more squirrels (but will these be red or grey?).

What about future mammals and their role in woodland management?

There has been recent interest in reintroducing beaver in Wales, and developments in Scotland are being watched with interest. Such species could play a role in re-naturalising some wetland and riparian areas and may provide a tool for management at a landscape scale. Large grazing mammals such as deer and domestic cattle also have a role and the reform of Common Agricultural Policy (CAP) may provide opportunities to manage the forest and the open ground matrix in a more integrated fashion.

Conclusions

I have tried to consider the future of woodland management, what future woodlands will look like, what use will we make of them, and what value they will have for mammals.

Let me summarise the view of Wales in my crystal ball. Forests of the future will be larger, but even with CAP reform we will not get the oft-quoted 30% cover that may reflect widespread habitat connectivity. The forests will be more diverse in structure and will include more open space. Forests will not just be trees but woodland in a matrix of other habitats and other land-uses including development. There will be less contrast between woodland and other land and more connectivity. However, some woodland will remain deliberately isolated to help retain species (such as the red squirrel), which would be harmed by invasive species utilising the habitat networks. Decisions about where we put and how we manage these woodlands will be based on better information about the environment and local people will be better consulted.

Larger areas will allow greater flexibility of management. There will be less distinction between foresters and farmers, and woodlands will be seen more as a solution than a threat; as areas to buffer pollution, slow flood water, improve soil condition and provide a renewable and local source of heat and other non-timber benefits. More people will do work associated with woodlands – be it recreation or eco-tourism or land management or making wood fuel. The local rural economy will be more in tune with the environmental value of the species that live in these forest landscapes. Landowners and managers will work more collectively encouraged by incentives and local woodland based economies enabling an integrated approach to woodland management.

Active management costs money – we are already seeing how the cost of restocking uneconomic forests is changing the focus on how and with what tree species we restock. Foresters are becoming more relaxed about allowing natural processes to take over. This is likely to result in more diverse species composition in many upland uneconomic forests. It is also likely that some areas will remain unfelled allowing non-native plantations to reach old growth. Who knows, these future natural forests may become highly valued for their contribution to wildlife. There will also be areas of existing forest, which are best replaced by other habitat or transformed to a different form of woodland. One thing for sure there will be more regional distinctions – doing it differently in different places.

However, we cannot afford to make landscape-scale decisions based on ecology alone. We need to develop sustainable landscapes which have a meaning and purpose to modern society, and by being valued provide benefits to our ecology. Woodland is a resource and the objectives of its management and what it looks like will depend on how we (21st century society) want to use it.

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