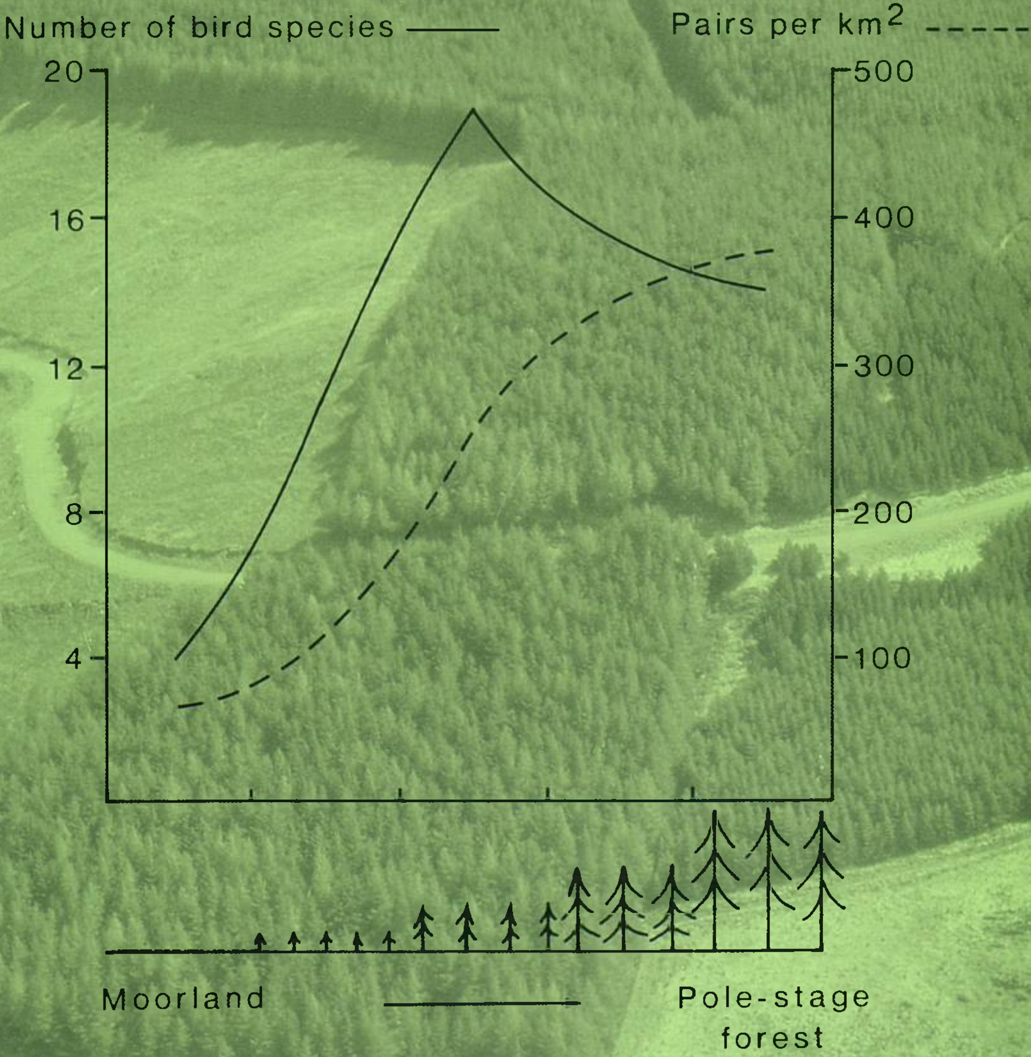
Forestry Commission

Occasional Paper 26

Forest Bird Communities

S J Petty
M I Avery



FORESTRY COMMISSION OCCASIONAL PAPER 26

Forest Bird Communities

A review of the ecology and management of forest bird communities in relation to silvicultural practices in the British uplands

---

S.J. Petty
Bird Ecologist, Forestry Commission
and
M.I. Avery
Senior Research Biologist, Royal Society for the Protection of Birds,
formerly Research Fellow, Edward Grey Institute of Ornithology, Oxford

Forestry Commission, Edinburgh

First published 1990

ISBN 0 85538 237 6
ODC 148.2 : 152 : 2 : (048.1)

KEYWORDS: Birds, Forestry

Enquiries relating to this publication
should be addressed to
The Technical Publications Officer,
Forestry Commission, Forest Research Station,
Alice Holt Lodge, Wrecclesham,
Farnham, Surrey GU10 4LH.

Contents

Chapter 1

Introduction

The need for a review

By the time of the formation of the Forestry Commission in 1919 most native forests had been stripped from the British uplands, primarily by man, over a period of many centuries. This process accelerated towards the time of the industrial revolution and once devoid of trees, the development of sporting interests and sheep farming were essential to perpetuate the treeless character of many moorlands. This was achieved by a combination of heavy grazing and regular burning which effectively prevented any natural regeneration. By the turn of the 20th century, less than 5 per cent of the land surface of Britain was forested, less than any other country in mainland Europe.

Afforestation, mainly with exotic conifers grown on relatively short rotations, has had a dramatic effect on the landscape of many upland areas over the last 50 years. Continued expansion of forestry in the uplands (Centre for Agricultural Strategy, 1980; Forestry Commission, 1977) has led to concern being voiced for the characteristic plant and animal communities associated with upland areas (Nature Conservancy Council, 1986) together with aesthetic and landscape objections. Much of this concern has focused on the effects of afforestation on bird communities. A widely held view is that moorland birds with restricted range are replaced by more widely distributed forest birds (Nature Conservancy Council, 1986; Thompson *et al.*, 1988).

It is therefore timely to review what has been published on the ecology and management of forest bird communities and to relate this to silvicultural practices in the British uplands. The review is aimed mainly at two groups. First, it will provide forest managers with information on how bird communities function in a dynamic forest environment and what foresters can do to achieve a richer, better balanced avifauna within their forest. Second, it will provide the various conservation bodies with a background to current forest practices in the uplands and how these are likely to affect birds.

The review has used literature from different geographical regions. Chapters 2, 5, 6 and 7 are based mainly on British studies and practices while Chapters 3 and 4 use literature from the much larger Holarctic region (Figure 1.1). The literature from these regions has been searched up to the end of 1986 although some studies published in 1987-88 and in press have been used. The review is not intended to be a comprehensive account of all the published work in a particular field. Chapters 2 and 6 are 'scene setters' where the aim is to explain why our natural vegetation/wildlife communities and forests are the way they are. The objective of Chapters 3, 4, 5 and 7 was to be selective and critical and to use the most important papers published in scientific journals and books. Throughout we have tried to be objective and unbiased. Chapters 3, 4 and 5 were written by MIA and Chapters 2, 6 and 7 by SJP.

Acknowledgements

Many people have helped with the preparation of this review and to these we are very grateful. Anabel Seddon helped by collating and keywording many of the earlier references, John Williams prepared the figures; Winifred Cooper, Diane Chadwick and Linda Petty undertook the

typing including numerous revisions. A number of people commented on, and greatly improved draft chapters: they were Ian Newton, Colin Bibby, Roderick Leslie, Phil Ratcliffe and Julian Evans.

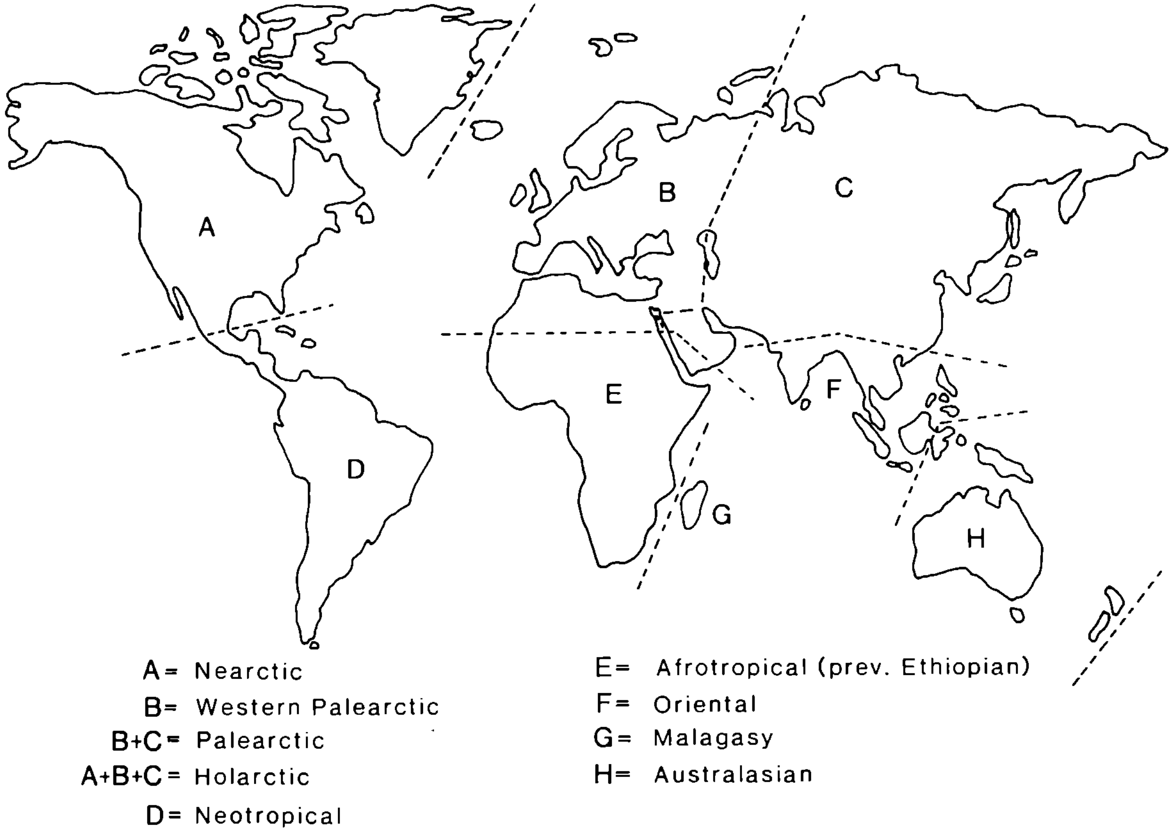


Figure 1.1 Zoogeographical regions of the world (redrawn from Pemberton, 1981). Regions are often separated by transitional zones and therefore the boundaries are approximations.

Chapter 2

Historical aspects of forests and bird populations in the British uplands

Introduction

The development of a number of techniques has enabled paleoecologists to trace the development of the British flora not only since the last ice age but also in the previous interglacial periods. These methods have relied largely upon the preservation of both macroscopic plant remains (for example wood, seed and leaves) or microscopic (pollen) plant remains in rapidly accumulating organic or mineral deposits such as mires, moor humus and lake sediments. The age of these remains has been determined either by radiocarbon dating or by relating the material to the layers in which they were found. Godwin (1975) summarises most of the techniques that have been used. Radiocarbon dating enables an estimation of the age of material up to approximately 50 000 years to be made. Radiocarbon dates have been calibrated by using tree rings in long lived species such as coast redwood and bristle-cone pine (Suess, 1967; 1970). In this chapter all dates refer to radiocarbon years before the present (BP).

Pollen analysis has led to a great advance in our knowledge of past plant communities. There are a number of potential problems associated with the method, the major one is that pollen can travel a great distance in certain weather conditions. Despite this and other analytical difficulties, pollen analysis is now accepted as a valid method for determining the composition of former plant communities. Godwin (1975) quoting von Post, a pioneer of the system, stated that "by establishing these frequency figures layer by layer through the pollen bearing strata it becomes possible to follow former plant geographic changes from place to place and from one time period to another". West (1968) describes methods for taking samples in the field, while both he and Faegri and Iversen (1964) give accounts of the microscopic identification and counting of pollen.

Compared with what has been discovered about past plant communities, very little is known about the species of birds previously present let alone details of bird communities. Compared with mammal remains, bird bones are very fragile and there are few fossil remains. Most bird remains have been found in cave deposits and archaeological sites. Bird remains are also biased towards the larger species with more substantial skeletons. From the few sporadic records of birds and a knowledge of plant communities, it is only possible to speculate on the bird communities present in the British Isles prior to recent history. From AD 1600 the literature becomes increasingly more valuable.

The development of plant communities in the Pleistocene period

Within the British Isles the Pleistocene period has produced a number of glacial stages interspersed with periods of climatic amelioration. The last three interglacials were the Hoxnian, Ipswichian and the present-day Flandrian (Godwin, 1975). The latter commenced approximately 10 000 years BP as the ice sheets from the Weichselian glaciation finally retreated northwards. Turner and West (1968) have suggested that these interglacials showed a characteristic succession of plant communities which they divided into four periods.

1. *Pre-temperate forest.* Comprising boreal pioneers such as *Betula* and *Pinus* which developed gradually after the last ice age. Light demanding species and herbs are also a significant element of the vegetation.
2. *Early-temperate forest.* A gradual improvement of forest soils leading to the development of mixed broadleaved forest with *Quercus, Ulmus, Fraxinus* and *Corylus* predominating.
3. *Late-temperate forest.* A slow degradation of soils due to a change in forest structure leading to a decline in oak and the expansion of late migrants such as *Picea, Abies* (but neither in the Flandrian), *Carpinus* and *Fagus.*
4. *Post-temperate forest.* Climatic deterioration leading to the re-appearance of *Pinus* and *Betula.* Soils becoming more acid and with an opening of the forest canopy leading to the development of heath and bog communities prior to the next glaciation.

These of course are very broad divisions in a large timescale, for example the Hoxnian interglacial lasted for approximately 40 000 years (Shackleton and Turner, 1967). Variation in geology coupled with climatic fluctuations resulted in deviation from this development. The development of plant communities in the last three interglacials have also shown differences. For example neither spruce nor fir managed to reach Britain in the present Flandrian, as the

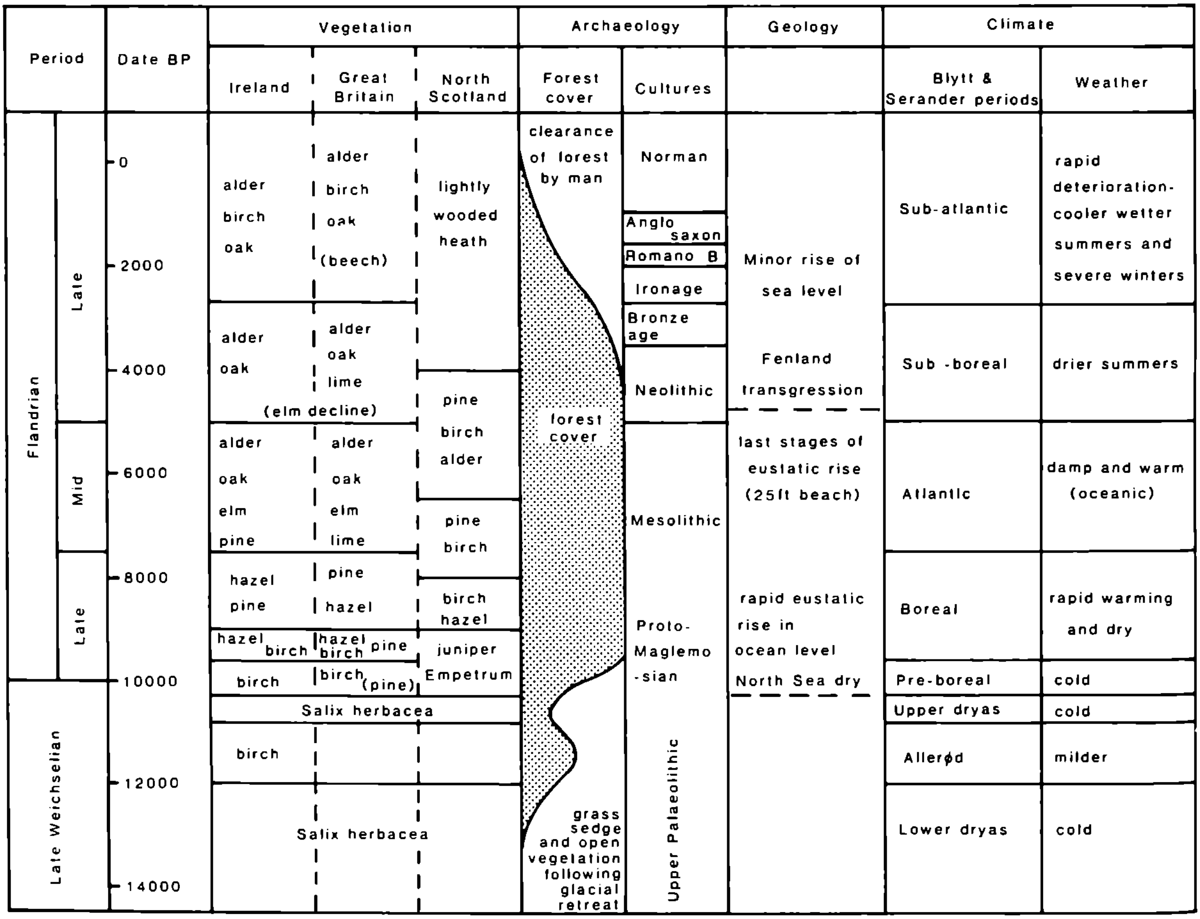


Figure 2.1 Correlation table showing the main events which have influenced vegetational development in Britain since the latter stages of the last ice-age (Weichselian) and through the present Flandrian period (adapted from Godwin, 1975).

land bridge to Europe disappeared before they reached the Channel coast. Spruce, which is assumed to be *Picea abies*, was present in the Hoxnian and Ipswichian interglacials, but it seems to have disappeared from the British Isles in the most severe phase of the last glaciation around 60 000 years ago. *Abies* was last present in the Hoxnian only.

Forests and birds in the Flandrian period

The Flandrian, although referred to as a post-glacial period, is considered by many to be an incomplete interglacial. Figure 2.1 shows the main archaeological, geological and climatic factors that have been responsible for our present vegetation. The Flandrian commenced about 10 000 years BP in the wake of a gradual warming over the preceding 5000 years, leading to the retreat of the Weichselian ice sheets, although a number of climatic fluctuations led to temporary glacial readvances. For our purpose the Flandrian can conveniently be divided into four periods; early, mid, late and recent (Figure 2.1).

The early Flandrian, 10 000 – 8500 years BP

A boreal period which saw the establishment of arctic alpine communities and the gradual expansion of birch, a species with wind dispersed seeds which was ideally adapted to colonise quickly once the habitat was suitable. Throughout this period a land bridge existed between Britain and the European mainland, across the southern North Sea; Ireland was, however, cut off from Britain at the beginning of the Flandrian. Parts of southern Britain and southern Ireland which were never glaciated, undoubtedly provided a pool of some of the more hardy plant species which then extended their range once conditions improved. The land bridge also allowed migration of other less hardy plants and trees from Europe. Towards the end of this period, birch and hazel extended to northern Scotland, while Scots pine had become a prominent part of the vegetation in parts of England but had only just started to colonise Scotland (O'Sullivan, 1977) (Figure 2.2). The melting of the Weichselian ice sheets had two main effects; a eustatic rise in sea level and an isostatic elevation of the northern land mass which had previously been covered by extensive ice sheets.

Bird records for this period are scarce. The tundra-like conditions that existed prior to the start of the Flandrian are reflected in the occurrence of ptarmigan from a number of sites from Inchnadamph in west Sutherland to Staffordshire (Ritchie, 1920; Bramwell, 1960). Snowy owl was recorded from a cavern in Kent and the occurrence of northern vole, arctic lemming and arctic fox suggested that wildlife communities may not have been too different to those which occur in the northern European tundra and taiga today. Early Flandrian remains from Mesolithic hunters at Thatcham, Berkshire and Scarr Carr, Yorkshire included red-throated diver, great crested and little grebe, goldeneye, mallard and red-breasted merganser. At that time these areas would have consisted of birch and pine forest with Mesolithic settlements along the side of lakes. Crane and white stork were also recorded and reflect the more continental climate that existed in Britain.

The mid-Flandrian, 8500 – 5000 years BP

The sea level continued to rise leading to the flooding of the North Sea and the separation of Britain from mainland Europe from around 7800 BP. This prevented the further migration of plants and sedentary birds, and the climate became gradually more oceanic in character. The vegetation of Britain developed from the early successional forests of birch, hazel and pine to thermophilous broadleaved forest, particularly south of the Central Lowlands in Scotland (Figure 2.2). North of this, Scots pine increased to its maximum extent (O'Sullivan, 1977). Birks *et al.* (1975) used principal component analysis to describe regional differences in pollen frequency at 5000 BP towards the end of the mid-Flandrian, when forest development had probably reached its maximum in the British Isles but had been little affected by Mesolithic communities. They not only provided pollen frequency distribution maps for *Pinus, Betula,*

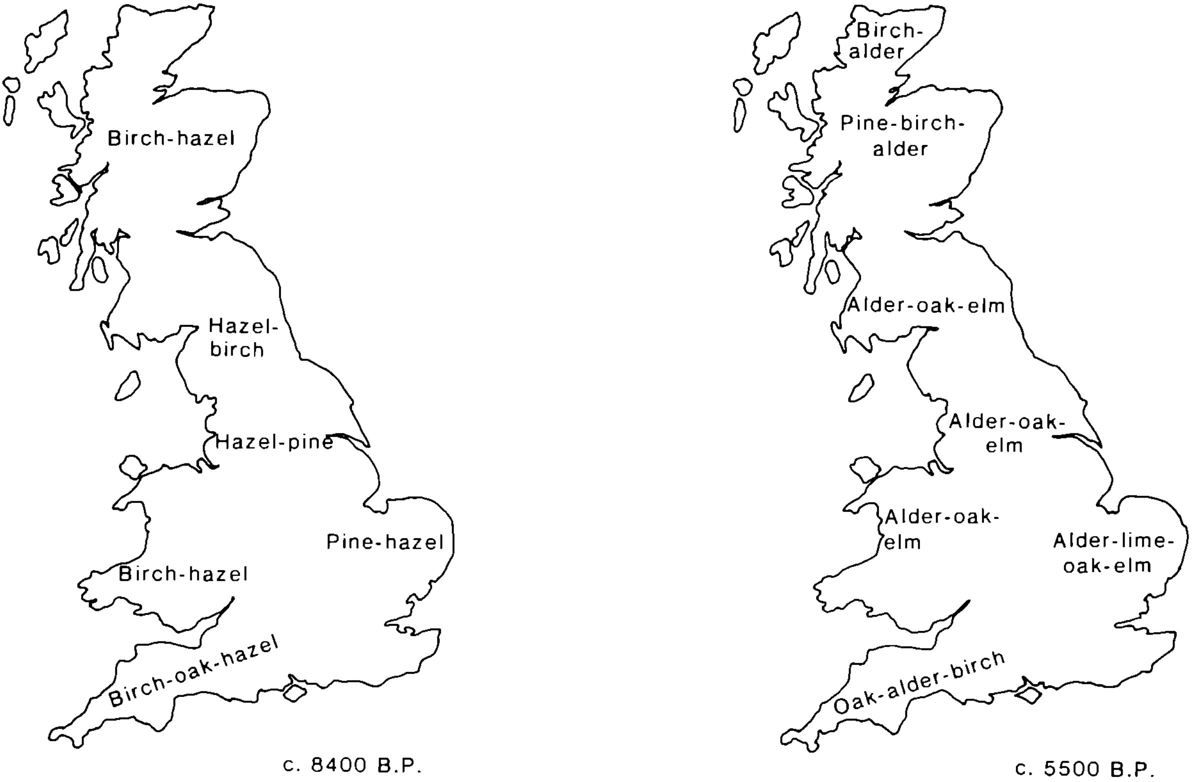


Figure 2.2 The distribution of the main forest types in Britain at the end of the Early Flandrian (8400 BP) and at the beginning of the Late Flandrian (5500 BP). At 8400 BP Britain was still attached to Europe, while at 5500 BP the sea level was higher than at present, and forest development has reached its greatest extent without being greatly influenced by man (re-drawn from Godwin, 1975).

Quercus, Ulmus, Alnus, Corylus and *Tilia* but described four main gradients which comprised abundant pine and birch in northern and eastern Scotland and western Ireland; of high elm and hazel in central and eastern Ireland; of abundant alder and hazel in England and lowland Wales, with oak more prominent in the north and west and lime more frequent in south and east England. These broad regional trends were further modified by local variations in the abundance of many plants due to differences in the speed of migration and colonisation, and also due to topographical and geological differences (Turner and Hodgson, 1979, 1983; Tallis and Switsur, 1983). Regional variations were demonstrated by Turner and Hodgson (1983) for the northern Pennines. Here small populations of pine existed on the Millstone Grits in the north east of the region and showed similarities with forest composition in parts of the Cairngorms, rather than the rest of northern England.

There is very little information about bird communities in the mid-Flandrian. Towards the end of this period, oceanic influences and the extensive nature of the forests would have seen many of the animals established which form our present-day wildlife communities. It is difficult to comprehend the scale of the forests which must have existed. Mesolithic cultures were primarily hunter-gatherers and were unlikely to have had a major effect on the vegetation. Some of the larger mammals may have been locally affected by man, but there is enough evidence to suggest that wolf, brown bear, wild boar and red deer for instance were still widespread. Bird communities would have developed to their fullest extent, with some of the larger and more spectacular species, such as white-tailed eagle and capercaillie recorded from a number of sites in England. Northcote (1980) looked at nine Cambridgeshire lake deposits (7000-3000 years BP) and suggested that the commonest wetland birds were mute swan, bittern, mallard and crane in that order. She also recorded Dalmatian pelican and white-tailed eagle. The eagles would then have bred in the extensive forests surrounding such rich fenland feeding areas.

The late Flandrian, 5000 years BP – AD 1600

The Blytt and Serander classification (Figure 2.1) indicates a drier, sub-boreal period from around 5000 to 3000 BP. This dry period is not altogether confirmed by other studies, but most authors agree that from around 4000 BP the climate again became progressively wetter (more oceanic). Man also started to have an impact on the vegetation (Figure 2.1). These two influences (climatic and human) had the greatest effect on shaping our present day communities from the extensive forests that existed at the end of the mid-Flandrian.

O'Sullivan (1977) reviewed the extent and history of Scots pine in northern Scotland. He concluded that pine retreated from the western Highlands from about 4000 BP and was gradually replaced by widespread blanket bog formation. Many of these bogs will have persisted to the present with their unique range of birds, particularly waders (Stroud *et al.*, 1987). In contrast, in the eastern Highlands some areas of pine forest have survived until the present although the composition of the forest has been much changed by man, for example the present day forests appear to contain less birch and more *Calluna* (O'Sullivan, 1977).

The fate of pine forests in the eastern Highlands was highly variable and this appears to have been largely influenced by man. There is evidence of forest clearances occurring just prior to the start of the late Flandrian period. On some sites this was followed by the re-establishment of pine, while in other areas a plagioclimax of *Calluna* was maintained and slowly extended. Most of the eastern Highlands appear to have remained climatically suitable for Scots pine since the time when major pine immigration and expansion commenced around 8000 BP. However, there have been periodic fluctuations which have affected altitudinal tree limits; around 4000 BP pines grew at nearly 800 m in the Cairngorms, and there also appears to have been east–west altitudinal gradients in the tree line (Pears, 1969, 1975).

In England and southern Scotland significant forest clearances appear to have been initiated by Neolithic people from around 5500 BP. This is shown most dramatically in the abundance of elm in numerous pollen analyses from around this period. It had been speculated that elm was a favoured feed for domestic stock, particularly cattle, and that elms were selectively removed for this purpose (Godwin, 1975). Dimbleby (1984) though, has stressed that we know relatively little about the effects of Neolithic people on forest cover.

Forest clearances and agricultural use of the land increased and accelerated throughout the Bronze Age, Iron Age and into historical times, as it became necessary to feed an ever increasing human population (Dimbleby, 1984). In Roman times, pollen analysis suggests that in the south of England arable farmland dominated the landscape, although in the north and west much woodland remained. However, the effect on both Roman and Anglo-Saxon cultures on the vegetation of Britain is poorly documented. The Medieval period saw some of the more remote areas in the uplands being settled, particularly by the Cistercians, with a great expansion in sheep farming and a corresponding permanent loss of woodland.

There is little evidence to show how these anthropogenic changes affected bird communities during this period, although they must have been dramatic. The massive forest clearances, particularly in the south of Britain, but also increasingly in the north, will have reduced the population and ranges of many forest birds such as honey buzzard and capercaillie. Some of the larger predatory mammals were exterminated during this period. The brown bear may have survived until the 9th or 10th century, while wolves were reduced to low numbers and finally became extinct in England by 1500 (Richie, 1920; Corbet and Southern, 1977). The increasingly open nature of the countryside encouraged birds such as crow, rook, woodpigeon and also red kite, raven and buzzard which were able to benefit from the increase in farming and Medieval settlements in the lowlands. Bramwell (1960) lists a range of birds present in the early Iron Age, many of which are still present today, but some are not. From two sites in Somerset he notes the presence of goshawk, white-tailed eagle, kite, Dalmatian pelican, crane and capercaillie. He also records the prey remains from a golden eagle eyrie in the Peak District in Saxon times. Here the eagles fed largely on black grouse, and also on young deer, lambs, piglets, polecats, squirrel and ducks. Presumably the predominance of black grouse in the eagles' diet reflected the abundance of open forest or forest edge habitats in the area. More red grouse would have featured in their diet if open moorland had been the predominant habitat surrounding the eyrie.

Recent changes, AD 1600 – present

Although climatic fluctuations during this period will have affected what was left of Britain's natural vegetation and the distribution of birds, anthropogenic changes had by far the biggest influence (Richie, 1920; Ratcliffe, 1984). We will look briefly at three aspects which greatly affected birds during this period, namely the continuing forest clearances, persecution and pollution.

It has been estimated that by 1700 forest comprised around 12 per cent of the total land surface (Holmes, 1975). This declined further, so that by the early 1900s less than 5 per cent of Britain was forested, less than any other mainland European country. Much of this loss of forest occurred during periods of war. The Seven Years' War of 1756-63 left England stripped of oak suitable for naval timber, while the Stuart rising of 1745 led to large-scale clearances of many of the remaining native pine and broadleaved forests (Holmes, 1975). The populations of the unique group of birds associated with native pinewoods in Scotland must have been at an extremely low ebb at this time. The capercaillie was regarded as rare throughout the 1700s, and it finally became extinct in Scotland in the 1780s and Ireland in the 1760s (Harvey-Brown, 1879; Richie, 1920; Pennie, 1950-51; Lever, 1977). The last wolves were also killed in both Scotland and Ireland at about the same time. A further loss of forest occurred in the Industrial Revolution and during the two wars in the present century, when 180 000 ha and 150 000 ha respectively of the best timber were cut. Local industries which used charcoal such as iron works, tanneries and glass works did however help to maintain woodland (especially coppice) in some areas.

As well as these negative effects on forests, there were also some positive actions which were spurred on by worries about Britain's dwindling timber stocks. It was during this period that many of the earlier forestry techniques were developed. The publication in 1664 of John Evelyn's *Sylva: a discourse on forest trees and the propagation of timber in His Majesty's Dominions* encouraged many landed gentry into planting oaks during the Restoration period. Another period when large areas were planted, particularly in Scotland, was between 1750 and 1850. The Duke of Atholl created 4000 ha of new forest in Perthshire, much of which was larch. In the valleys of the Spey, Don and Dee new forests were also planted, largely with native Scots pine. In all, over 200 000 ha were planted on these large Scottish estates during this period (Holmes, 1975). The first successful reintroduction of capercaillie from Sweden at Taymouth Castle, Perthshire in 1837-38 was undoubtedly due largely to the creation of these new forests (Harvey-Brown, 1879; Richie, 1920; Pennie, 1950-51). The fortunes of crested tit and Scottish crossbill must also have improved at this time. However, this area appears to be unique in the British uplands, as generally the trend was towards deforestation as the area devoted to sheep farming progressively increased.

While the loss of forests led directly to the decline of many birds dependent on this habitat, man also began to have an increasingly important impact through direct persecution, particularly on the larger and more spectacular raptors. Their low density, low rate of reproduction, and the change to breeding in small woods rather than large forests, made them highly vulnerable. This pressure came largely from gamekeepers, whose density in the countryside increased dramatically as gamebird shooting (pheasant and partridge in the lowlands, and red grouse in the uplands) became more and more popular with wealthy landowners. More efficient guns, traps and poisons were used ruthlessly. Golden and white-tailed eagles were also persecuted because of their predation on lambs. By this time white-tailed eagle populations were largely concentrated around the north-west sea coast of Britain and Ireland, where the sheep populations were greatest, so they were relatively easy to remove and finally disappeared as a British bird in the early 1900s (Love, 1983). In comparison, golden eagles were less approachable and managed to survive, largely because pairs in the more remote inland areas of Scotland would have been little affected by man, although they were eliminated from England, Wales and Ireland.

Ticehurst (1920), examining the churchwarden's accounts, showed the relative abundance of red kite, buzzard and raven at Tenterden in the Weald of Kent in the 17th century. Around 1677 an intensive campaign began and over a 14-year period 380 kites were paid for, including 100 during 1684-85. The numbers dropped significantly after this, perhaps indicating that the local population had been substantially reduced. An increase in the number of buzzards and ravens killed during the same period is also documented. This early persecution was probably instigated because of predation on poultry. It indicates the past abundance of these three birds in lowland Britain, from which all are now virtually absent. Moore (1957) showed that the status of buzzards has fluctuated since 1800 with the intensity of game preservation. Buzzards are now largely confined to the west of Britain.

Bijleveld (1974) and Newton (1972b, 1979a and b) document examples of the persecution of raptors by gamekeepers, and these give an insight into what must have been a widespread practice. In Glen Garry during 1837-1840, 98 peregrines, 78 merlins, 462 kestrels, 285 buzzards, 3 honey buzzards, 15 golden eagles, 27 white-tailed eagles, 18 ospreys, 63 goshawks, 275 red kites and 68 harriers (a total of 1372 raptors) were paid for. In the Langwell and Sandhide estates in Sutherland during 1819-1826, 295 adult and 60 juvenile eagles, and 1115 other raptors were killed. There are many other accounts of such slaughter. The accuracy of such figures has often been queried but in most instances payment was only made on the production of a corpse, head or feet.

It was not surprising that such heavy killing eventually led to the demise of some species; goshawks last bred in the 1880s; red kite ceased to breed in England and Scotland, but a small population survived in a part of central Wales where there was little history of keepering; hen harriers almost ceased to breed on mainland Britain; osprey did stop breeding in Britain; and the ranges of golden eagle, buzzard, peregrine and sparrowhawk contracted.

Pollution of the environment from man's activities also affected birds, but in less direct ways than by killing. Since the Industrial Revolution, acid deposition has resulted in the acidification of some upland lakes and watercourses. Flower *et al.* (1987) used diatom analysis of sediment cores from lakes in Galloway to show that acidification appeared to be more pronounced in catchments with acidic parent material and at high elevation. The decline in pH of six lochs started between 1840 and 1920, commencing first in the most acid catchments. Acid deposition was the most probable cause of acidification at all sites. Afforestation in three of the catchments may be associated with small declines in pH, but the main trend was established well before afforestation commenced. Increasing acidification of watercourses will affect birds that are dependent on these systems for food. Ormerod *et al.* (1985) in Wales showed that there were fewer dippers on the more acidic streams.

Since the last war, the insidious effects of agricultural pesticides have had dramatic effects on some of our more important avian predators. Organochlorines were used as seed dressings and for a number of other agricultural uses; birds and mammals which ate the treated seed accumulated these compounds in their body tissues which were then passed on to raptors as they ate contaminated prey. DDT came into use after the Second World War and was in widespread use by about 1947. This resulted in many bird-eating raptors such as sparrowhawk, merlin and peregrine, laying thin-shelled eggs, some of which broke during incubation or failed to hatch, so leading to reduced productivity (Newton, 1979a). Restrictions on the use of these cyclodiene compounds in the 1960s and 1970s led to dramatic increases in both sparrowhawk and peregrine numbers in Britain (Ratcliffe, 1980; Newton, 1986). Raptors were not the only group affected, and other predators at the top of food chains, such as heron, also showed considerable egg-shell thinning (Cooke *et al.*, 1982).

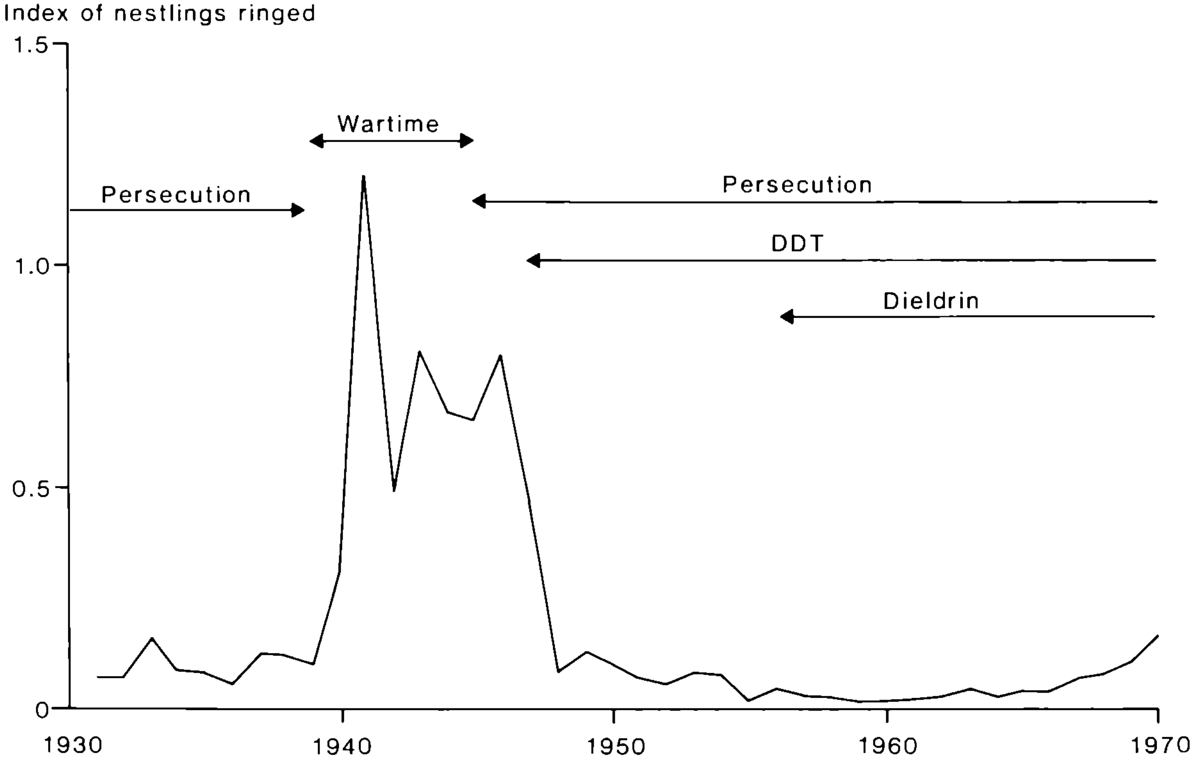


Figure 2.3 An index of sparrowhawk populations (young sparrowhawks as a proportion of the total nestling birds ringed in Britain) demonstrating the major impact of man. The increase in sparrowhawks during the Second World War is presumably due to less gamekeeping during this period (adapted from Newton, 1972b).

ton, 1979a; Ratcliffe, 1980; Cooke *et al.*, 1982; Newton, 1986b). However the major population declines in birds of prey occurred later with the introduction in 1955 of the more toxic cyclodiene insecticides, such as aldrin and dieldrin (Figure 2.3). These caused mortality among adult raptors, and led to widespread population declines. The most notable were in peregrine populations in Britain, northern Europe and North America, but many other species were also affected (Newton, 1979a). Restrictions on the use of these cyclodiene compounds in the 1960s and 1970s led to dramatic increases in both sparrowhawk and peregrine numbers in Britain (Ratcliffe, 1980; Newton, 1986). Raptors were not the only group affected, and other predators at the top of food chains, such as heron, also showed considerable egg-shell thinning (Cooke *et al.*, 1982). Other chemicals such as PCBs and mercury may also affect predators (Newton, 1979a). The illegal use of poisons (Cadbury, 1980) is also having a major impact on the distribution of raptors.

The last 10–15 years have seen the fortune of many British raptors substantially improved. Buzzards have recolonised some of their former range; sparrowhawk, peregrine and other raptors have increased as a result of restrictions on the use of pesticides; osprey has naturally recolonised Scotland and goshawk and white-tailed eagle have been successfully re-introduced; while a number of species have benefited from afforestation which has provided a suitable habitat with little or no persecution for species such as sparrowhawk, goshawk, hen harrier, red kite, buzzard and most of the owls.

Summary

The last glacial period came to an end around 10 000 BP and over the next 5000 years the vegetation of the British Isles changed largely in response to climatic fluctuations, although edaphic factors were also involved. By around 5000 BP forest cover had reached its greatest extent, covering most of the land surface and comprising species of which our present-day ancient woodlands are a mere fragment. Little information exists about bird communities, but these evidently changed from a tundra/taiga community at the start of the Flandrian period, to a largely broadleaved forest community by 5000 BP except in northern Scotland where pine/birch forests were extensive. Many of the large avian and mammalian predators were widespread even in the lowlands.

From 5000 BP man increasingly became the dominant influence affecting vegetation and wildlife, although climatic fluctuations continued to occur. Up to AD 1600 adverse effects on wildlife will have been largely through forest clearances. From AD 1600 persecution and hunting gradually reduced or removed many of the larger mammals and birds particularly from the lowlands, but also increasingly from the uplands. By 1900 less than 5 per cent of Britain was forested, and bird populations have been dramatically changed from a predominantly forest community, to species which could adapt to the open conditions created by man.

Chapter 3

Bird census methods and techniques

Introduction

Describing the birdlife of an area entails two separate activities. First the data have to be collected and then they have to be expressed in an intelligible form. It is almost certain that no two studies have chosen precisely the same two ways of tackling these problems. In this chapter we deal briefly with the choice of study areas, the range of field techniques that have been used to collect data, and the methods used to summarise the results.

Choice of study areas

Many of the published studies of bird communities in Britain have been based on small sample sizes of unrepresentative woods. Not surprisingly the first woods to be surveyed were sites which were particularly good for birds, often nature reserves. These studies provided a database of descriptive material for gross comparisons between areas and tree species. Other studies have looked at woods covering very large geographical areas. Unless sites are chosen carefully, so that samples from different parts of the country are carefully matched for all relevant parameters, any results will be confounded by the effects of geographical area. For more refined analysis, much larger samples are necessary from woods of representative sizes, management regimes and different tree species within the same general area.

Census techniques

Two main census techniques have been used in the majority of studies of woodland birds; territory mapping and line transect methods. A cursory analysis of published papers shows that the territory mapping method is about twice as popular as the line transect method. In recent years a third method has grown in popularity; the point-count method.

Territory mapping

The rationale behind this method is that by repeated visits to an area the actual territories of its bird community can be mapped. On each visit all birds seen or heard are recorded on maps and then standard methods are used to describe the territory boundaries (Enemar, 1959; Williamson, 1964). This method is used in the British Trust for Ornithology's Common Birds Censuses. This method collects more information than the others but the information on actual territory boundaries is rarely used except in producing the estimate of territory numbers, although in principle territory boundaries could be related to habitat features. This method is clearly not suitable for non-territorial species nor for those times of year when territorial activity is low. In addition, the necessity for repeated visiting constrains the area which can be surveyed.

Transects

Here the observer follows a route through the habitat and records all species heard or seen within a set distance on either side of the path. This method can be used in all seasons and greater areas can be covered than with the former method. There is always the danger that the route taken will make concessions to the observer's ease of travel and therefore be a biased sample of habitat. Because of the differences in detectability between different bird species, it is not usually possible to collect accurate data on population densities using this method. However, assuming that the detectability of species does not differ between habitats (an assumption which may be reasonable for some pairs of habitats), relative densities can be compared between habitats; this is often all that is required.

Point-counts

In this method the observer stands at a spot for a set time (usually 10 minutes or less) and records all birds detected. The observer records either the distance to each bird or the number of birds seen both inside and outside a circle centred on the observer. The radius of the circle will vary depending on the habitat. The great advantage of this method is that it is quick, which means that many more sites can be sampled than by using other methods. By randomising the position of the plots a representative sample of habitat is obtained. This means that the different point-counts are statistically independent of each other. It is possible to compute approximate densities if one is willing to make various assumptions about how the detection probability of different species changes with distance (Bibby *et al.,* 1985).

Each of these three methods has its keen proponents and each technique has its own advantages and disadvantages (Ralph and Scott, 1981). The method to be chosen really depends on the use to which the data will be put. However, it is worth noting that use of the point-count method, the newest and most unfamiliar, enables sampling of many more sites than the other two methods, for the same amount of manpower. It therefore sacrifices detail in exchange for replication. This trade-off is often well worth taking, particularly if the aim of the study is to make generalisations; these will be more worthwhile if based on a large number of sites which are quickly sampled than a very small number of sites known in great detail.

Diversity indices

An aim of many of the studies with which we shall deal, has been to compare the bird communities in two habitats or a number of successional stages. A concept which appears again and again is that of diversity. Unfortunately there is a diversity of diversity indices (see Peet, 1974; Southwood, 1978; Magurran, 1988). The vast majority of papers use one diversity index, the Shannon-Weaver (or Shannon, or Shannon-Wiener) index which is based on information theory (Shannon and Weaver, 1949), and so this will be described here. For an introduction to the use and abuse of this index see Peet (1974).

All diversity indices attempt to combine the effects of two factors which are intuitively related to what most people would be happy to regard as diversity; number and evenness. Thus a community of ten species will be more diverse than a community which has only five species. And, of two five-species communities, the one with the more even distribution of species is the more diverse, because it departs farthest from a community dominated by a single species.

Pielou (1966) explains the link between information theory and diversity. The diversity can be thought of as determining the amount of doubt about the identity of an individual drawn at random from the community. Since information theory deals with the measurement of uncertainty, it can be used as a measure of diversity. The expression for the information content per individual within an infinite population is given by the Shannon-Weaver formula (H) as:

$$H = -\Sigma p_i \ln p_i$$

H can be used to describe the diversity of bird community, where p_i is the proportion represented by an individual species in a community. Thus, for a community with ten individuals; if all were of the same species H would be zero ($-1 \ln 1 = 0$); if there were nine of one species and one of another then H would be 0.32 ($-0.9 \ln 0.9 + -0.1 \ln 0.1 = 0.09 + 0.23$); and if each of the ten birds were a different species, H would be 2.30 ($-0.1 \ln 0.1 \times 10 = 0.23 \times 10$). Since the diversity index depends on both the number of species in the community and the way in which individuals are distributed between species, it is possible for many different communities to have the same score on the diversity index

(Figure 3.1). Another difficulty in interpreting diversity indices, is that the range of possible values varies with the number of species in the community (Figure 3.1).

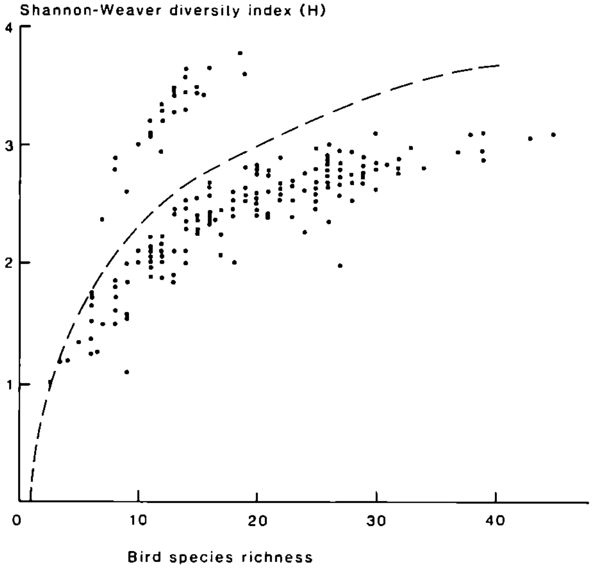


Figure 3.1 The relationship between the number of species in a community and the Shannon-Weaver diversity index (*H*) taken from a number of bird community studies (Table 3.1). The X axis is the minimum possible value of *H*, while the broken line shows the maximum value. The points above this line are from a study where *H* was incorrectly calculated.

The correlation between species number (species richness) and diversity is so high, both within and between studies (Figure 3.1; Table 3.1), that unless the evenness of the community is of particular interest, then species richness is preferable to species diversity for most purposes, since it is far easier to compute and to understand. In most studies that attempt to assess the conservation value of different sites, bird species diversity and the bird species richness are of limited importance since different individual species are regarded as being of differing value and interest. Few people would regard a community of golden eagles that preyed on ptarmigan as being of less interest than a community of many common bird species, although it would undoubtedly have a lower diversity index.

Table 3.1 Correlation between species richness and species diversity of woodland birds

Study	*r*	*n*	*p*
Adams, 1974	0.86	7	<0.05
Anderson *et al.*, 1983	0.82	22	<0.01
Beedy, 1981	0.85	8	<0.01
French *et al.*, 1986	0.91	19	<0.01
Haila *et al.*, 1980	0.96	15	<0.01
Hino, 1985	0.89	7	<0.01
Holmes and Sturges, 1975	0.77	57	<0.01
Johnson and Landers, 1982	0.94	25	<0.01
Morrison and Meslow, 1983	0.76	17	<0.01
Moss, 1978a	0.65	8	<0.05
Szaro and Balda, 1979	0.99	15	<0.01

In practice very few of the studies which calculate and tabulate values of species diversity use this information. For these reasons we largely confine ourselves to discussions of species richness rather than species diversity. However, there are also problems with using this measure as pointed out by Usher (1983), because the total number of species recorded is dependent on the area searched, the size of the sample and the time spent searching.

Expressions of diversity are complex, and it appears almost impossible to describe and compare bird communities using one meaningful numerical value. In practice, visual or tabulated displays are often more valuable.

Vegetation description

Woodlands are complicated three-dimensional habitats, so describing them is difficult. A large variety of methods have been used, these range from broad categorisations of habitats, through fairly simple indices to complex multivariate techniques. The method to be used depends largely, or should do, on the use to which the results will be put. Other constraints usually include the ease with which the vegetation parameter is measured; some of them are extremely complex.

Broad categories

One of the simplest and most obvious ways of describing a habitat is by identifying its dominant species. Most people would be able to conjure up a picture of an oak wood or a spruce plantation and such a simple description is sufficient for many purposes, particularly for comparisons at a gross level. Clearly a recently planted spruce forest is very different from a mature one, and so some indication of the point in the succession should be given. This may take the form of a simple description such as 'mature' or 'newly planted'. The age of the trees gives some indication of the stage which they have reached, but because of different growth rates of trees on different soils and under different climatic conditions, it may be better to provide information on the top height (see Chapter 6) of the trees. Yield models have been constructed for all the main species of commercially grown forest trees; these yield models are based on the top height/age relationships (Hamilton and Christie, 1971). Top height is relatively easily measured and can be used to define the division between the various growth stages of a crop, this interface may differ depending on the species or community under investigation (Ratcliffe and Petty, 1986). Diameter at breast height (DBH) or the density of trees above a critical DBH are other commonly used measures.

Although trees are the most obvious components of a woodland they may not always be the most important component for birds. A complete botanical survey of a site is usually beyond the ability of most ornithologists and is a highly time-consuming activity. Instead, a common method of vegetation description records the presence or absence of major species in characteristic layers of vegetation; ground and shrub for instance.

The number of measurements that can be made of vegetation parameters is practically unlimited, so that the choice of which to use should be determined by the use to which they will be put. However, one point to bear in mind is that extra information, unused in the planned study, may be of use to future workers. Another general rule, which has widespread applicability, is that data that are collected need not be used but those that are not collected cannot be used! Since the speed of change in some woodlands is not rapid, information on vegetation features can sometimes be collected retrospectively.

Simple indices

Foliage height diversity

One of the most widely used indices is that of foliage height diversity (FHD). This index was used by the influential theoretical ecologist Robert MacArthur and has been used by many subsequent researchers (see Chapter 4). In essence it measures the diversity of the foliage profile, and uses the Shannon-Weaver index (see above). The use of the Shannon-Weaver formula in this case, can be thought of as more artificial than in calculating the bird species diversity, since the foliage profile is a more nebulous thing than the species list for an area. In order to calculate the FHD it is necessary to define a number of layers of vegetation and measure the amount of foliage in each.

Methods used to measure the vegetation cover at different heights vary greatly (Erdelen, 1984) and include: measuring the distance at which 50 per cent of a chequered board is eclipsed, recording the vertical distance to the nearest leaf, recording the presence or absence of vegetation in different height zones above a series of points. It is by no means clear how these measures relate to each other.

Also problematical is the subdivision of the vegetation into layers in order to calculate the diversity of the foliage profile. Different studies have used different numbers of layers and also different heights to form the boundaries of the layers. In many cases the divisions may have been chosen to mirror the author's subjective feelings about the layering of the vegetation, in others on some less flexible mathematical basis. Again this means that comparisons between studies are difficult if not impossible.

Interpreting the bare FHD indices is also difficult. Many different vegetation structures can generate the same FHD index. For example, three profiles with 60, 30 and 10 per cent of the vegetation in their layers, would all produce the same FHD index regardless of which layer contained each of these amounts. Thus there would be six ways of producing the same FHD index with these same values of vegetation abundance, and many more using combinations of different percentages.

The interpretation of correlations between FHD and other measures, most commonly bird species diversity (BSD), is further complicated by possible correlations between FHD and other habitat features such as tree species and successional stage. Erdelen (1984) provides a useful introduction to the complexities of diversity indices.

Other indices

FHD has probably been an over-used measure of vegetation. Erdelen (1984) compares many other indices, including plant species diversity, percentage vegetation cover (either within foliage layers or combined across them), diversity of stratification (Blondel *et al.,* 1973), horizontal, vertical and total diversity (Blondel and Cuvillier, 1977) and finds that there are numerous intercorrelations between them, all of which are positive. Erdelen suggests that the index DT (total diversity) may be a useful one since it includes components of horizontal and vertical diversity. However, this means that the index can take high values if either one of these components is high, which will make interpretation of correlations between DT and other measures difficult.

Principal component analysis (PCA)

A problem with using one simple index of vegetation structure comes when the results of analysis are to be interpreted. For example, a correlation between FHD and BSD might suggest that management techniques which increase FHD might give rise to an increase in BSD. However, since many vegetation measures are intercorrelated (Erdelen, 1984) it might be that some other vegetation measure which happens to be correlated with FHD could be the real causal factor. One way of taking account of many indices at once is by combining several measures in one.

Principal component analysis reduces a large number of related measures into a smaller number of measures. For example, if one had measured 20 components of vegetation structure in a variety of woods, the use of principal component analysis would allow those measures which were closely correlated to be combined into one single measure. In essence the way this works is that all variables are plotted against all others, and a line which 'explains' the greatest proportion of the variance of this n-dimensional cloud is the first principal component. The contributions of the different vegetation variables to this principal component is shown by a weighting factor, the magnitude of which indicates how closely correlated the measure is with the principal component, and the sign of the weighting factor indicating the direction of the relationship. After identifying the first principal component, the residual variance in the data cloud can be used in a similar way to find the second, and further principal components.

Bibby *et al.* (1985) used principal component analysis to simplify 16 vegetation measures made in restocked conifer plantations. The first principal component axis appeared to be related to successional features. The three most highly weighted factors were the amount of foliage between 0.5 and 2 m, the height of the trees and the age of the trees, all of which will rise with succession. The second axis was clearly related to the gradient between largely broadleaf and largely conifer woodland. In this case the first two principal components had obvious interpretations. If this is not the case, then using the principal components in further analysis is not likely to shed much light on the matter. For examples of studies which have used this technique see James (1971), James and Wamer (1982), Anderson, B. W., *et al.* (1983) and Bibby *et al.* (1985).

Chapter 4

Factors determining forest bird communities

Introduction

Several levels can be used to determine the composition of bird communities in temperate forests. The approach taken here is to try to identify useful predictors of community composition and to put less emphasis on the underlying biological forces driving the observed patterns. For example, we have considered it more important to try to describe the apparent effects of forest area on species number rather than to get involved in whether this pattern is a result of lower extinction rates in large forests, large forests containing more micro-habitats or whether it simply arises because large forests contain more individuals and therefore might be expected to have more species.

The ideal approach

Is there an ideal method of identifying the important factors affecting community composition? Yes there is – but it cannot be used practically to answer the questions in which we are interested. The ideal method to use is that of the controlled experiment. Ideally we would be able to design experiments in which all relevant factors are held constant except the one factor whose effect we were studying, which could be manipulated by us at will. For example, to look at the effect of tree species on community composition we would plant equal-sized blocks of trees at equal distances apart in a randomised order on a large grid and then monitor their bird communities through time. The experiment would be repeated in different places and at different times in order to check its validity. At the end of the experiment we could be sure that any differences in bird community were due to differences in tree species between the plots since all other factors had been randomised. Not surprisingly this experiment has not been performed and is not likely to be.

In the absence of such ideal controlled experiments the researcher has to use the variation that exists in the real world. For example, a study of the bird communities in woods of different species can be carried out by visiting samples of different woods. It is important to realise why this approach, although the best available, is not ideal. Surveys of all British semi-natural oak and pine woodlands might well find that oak woodlands supported more species of birds than did pinewoods on average. What would this mean? Not that pinewoods everywhere in Britain contain fewer species than oakwoods, because it may be that in northern Britain pinewoods contain more species than oakwoods, but in southern Britain the reverse is true and in the sample there were simply more southern than northern woods. If this were true, then we would have identified an important confounding variable, latitude, which is influencing the results. The correct design of experiments removes this worry, but it is an ever-present one when we are using natural variation. In our pine/oak example other potentially worrying confounding variables would be altitude, forest area, forest age, isolation from other woods, soil type and a host of others.

What can be done about confounding variables? First of all the probability that a confounding variable will influence our results by chance can be minimised by having a large sample of sites. In small samples there is a very high chance that confounding variables will exist. By sampling a large number of sites the probability that the sites will differ by chance is rapidly reduced. Too many studies have relied on a very small number of sites for their conclusions to be regarded as better than anecdotal. Increasing the sample size does not necessarily totally remove the problem of confounding effects. There may be systematic differences in important respects between the categories to be compared. It may be, in our example, that pinewoods (for both silvicultural and historical reasons) are usually planted at higher altitudes than oakwoods. In this case we can only recognise that the possibility exists that altitude may be affecting our results. Therefore, predicting the bird communities of new oakwoods at higher altitudes or new pinewoods at lower altitudes would be unwise using our dataset.

Another way of trying to remove the effects of confounding variables is by using multivariate techniques. In our example of the bird communities in pine and oakwoods this would mean that as well as recording the tree species and bird species of woods we would also note other features of the woods which we thought might be relevant such as altitude, latitude, size and age. Then using multiple regression analysis we could investigate which of the potential independent variables was the best predictor of the number of species in the bird communities and which others had significant independent effects. Multivariate statistics do allow an investigator to examine the effects of many factors at once but they still remain a long way behind experimental treatments in power. One reason for this is that it may be impossible to disentangle the effects of several variables which are always highly correlated in nature. Confounding variables complicate the interpretation of many studies in the following discussion of factors affecting bird communities in forests.

Regional differences

The potential pool of woodland species varies from place to place. In Europe as a whole the number of bird species found in woodlands decreases from south to north and east to west (Newton, 1986a). These trends can be seen within the British Isles. Many woodland species are absent from Ireland, for example nuthatch, tawny owl, willow tit, marsh tit, lesser spotted woodpecker, great spotted woodpecker, green woodpecker; and from Scotland, firecrest, nightingale and lesser spotted woodpecker. A few species are found only in Scottish woods (crested tit, capercaillie and Scottish crossbill). Fuller (1982), using data from the British Trust for Ornithology's Register of Ornithological Sites, shows that there is a general trend of decreasing number of breeding woodland species from south-east to north-west across the country (Figure 4.1). A wood in north-west Scotland

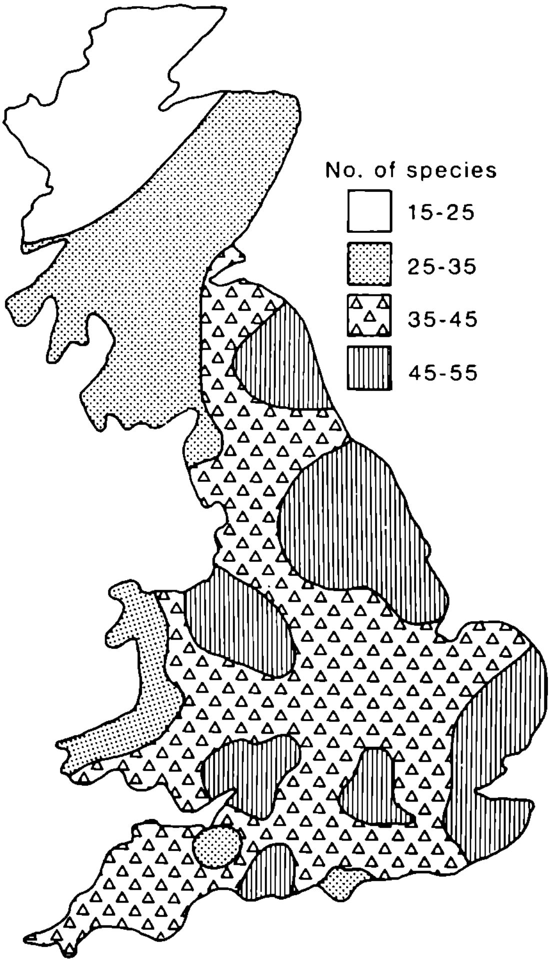


Figure 4.1 The numbers of breeding bird species in British woods. The numbers have been corrected to allow for woodland area and are those expected for a wood of 50 ha (from Fuller, 1982).

would be expected to contain 15–25 species whereas one in Kent might have 45–55 species. The latitudinal trend will result partly from many factors which vary with climate, such as the dominant tree species and the abundance of insects.

Altitude

In America (Able and Noon, 1976) and France (LeBreton and Broyer, 1981) it has been shown that the number of species in bird communities decreases with altitude. In north-eastern USA the number of species present at 200–800 m was about 22 whereas at 1200–1400 m the number of species had dropped to around eight. Fuller (1982) shows that species richness decreases with altitude in British uplands. Newton *et al.* (1986b) showed that woodland songbird numbers decrease with altitude (and increase with soil productivity) but are apparently not affected by latitude. In this as in many other cases the effects of altitude mirror those of latitude, perhaps because a rise in altitude has a similar effect on the climate experienced by an animal as a move northwards (in the northern hemisphere).

Forest area

North American studies

Several authors have investigated the bird communities of forests of differing sizes in the eastern United States. The background to these studies is that approximately 80 per cent of the woodland area of states such as Maryland has been destroyed by human activity in the past 300 years. The remaining forest is considerably fragmented providing a mosaic of woods of different sizes (Whitcomb *et al.*, 1981). Several studies have shown that the number of species found in a woodland remnant increases with the area of the plot (Forman *et al.*, 1976; Galli *et al.*, 1976; MacClintock *et al.*, 1977; Whitcomb *et al.*, 1981; Ambuel and Temple, 1983; and Blake and Karr, 1984). An important extra finding is that the species composition of the forest community also changes in a predictable way with forest area. Some species are rarely found in forests below a particular size. For example, Galli *et al.* (1976) never found red-shouldered hawks in woods smaller than 10 ha and many insectivorous species were only found in the larger areas. Blake and Karr (1984) showed that many species of the forest interior were only rarely found in woods below a certain critical size of about 24 ha, but their sample included only 12 forest patches, and the largest of these (600 ha) covered a much greater area than the other 11 put together (352 ha). The woodland edge species are also less likely to be found in larger woods (Whitcomb *et al.*, 1981). These findings have led to recommendations that some large areas of woodland should be held intact since, in general, it is those species which appear to depend on large areas which are the most threatened (Robbins, 1979; Anderson and Robbins, 1981).

Although these findings seem consistent, in that large areas hold more species and different species than small woods, it is possible that this result is due to the confounding of area with another, unknown, more important factor (Lynch and Whigham, 1984). Robbins (1980) used multiple regression techniques to quantify the relative contributions of vegetation factors, forest isolation and area on bird community composition. He found that canopy height and isolation were more important factors than was forest area in predicting the abundance of 51 bird species. This suggests that other findings which have tried to examine the effect of area may have been confounded by the effects of isolation and vegetation characteristics.

Lynch and Whigham (1984) also used multiple regression in a study of forest fragmentation in Maryland. Their results support those of Robbins (1980) by suggesting that both forest isolation and floristic diversity were each more important than forest area in predicting the abundances of 30 forest-interior birds. In this study, forest area was highly correlated with measures of isolation and canopy density. Thus small forests tended to be isolated and have dense canopies. The multiple regression analysis suggested that it was these factors, rather than area, which were really affecting the birds. It would not be fair to say that the results of

Robbins (1980) and Lynch and Whigham (1984) discredit completely the idea that large areas are important for some species. Lynch and Whigham did find that area was an important predictor of the abundance of some species which the previous studies had suggested were highly area-sensitive. An example of one such species is the worm-eating warbler.

The more detailed multiple regression analyses may not refute the previous findings for another reason; the characteristics of the forests differed in many respects in the different studies. In the Piedmont region where Whitcomb *et al.* (1981) worked, the degree of forest isolation was much greater than that in the Maryland plain where Lynch and Whigham (1984) worked. This may account, to some extent, for differences in their findings. However, it is clear that future research must try to take account of all the factors which have been suggested as being important and not simply focus on one factor such as area.

A more serious objection to using species/area relationships to recommend minimum habitat requirements is as follows. Although large woods tend to have lower densities of birds, at least in Europe, they will contain more individuals. They can therefore be thought of as larger samples from the species pool. In this case one would expect to find more species in the large woods than in the small ones simply by chance. Rare species will be unlikely to be found in many of the smallest woods even if they have no preference for woods of a particular size.

European studies

There have been many fewer European than American studies so they will be treated in less detail here. Gromadzki (1970) found that the size of an afforested area (shelterbelts, shrub-belts and hedges) influences the number of species to be found nesting in it. Although large woods have more breeding species than small woods, small woods hold birds at higher densities.

A recent study (Opdam *et al.,* 1985) in the Netherlands used a multivariate approach to investigate area effects in 68 small woodlots. Area was found to be an important predictor of species number. This was true when all species were included in the analysis and also when only those species which are restricted to mature woodland were considered. Incidence functions, which relate woodland size to the probability that a particular species will occur in a wood, indicated that some species (for instance great spotted woodpecker, nuthatch, marsh tit and, surprisingly, blue tit) are much more likely to occur in large woods than in small ones. These results differ in some respects from those of Fuller (1982) in Britain but the differences are probably attributable to the very different sizes of woods in the two studies. For example, Fuller found no effect of wood size on incidence of blue tit or nuthatch but many of the woods in his smallest size class were bigger than the largest woods in the Dutch study. This study can be criticised because the effort expended per unit area in woods of different sizes was not constant (Woolhouse, 1983, see next page) and this can greatly affect the relationship between species number and area.

Helle (1984) investigated the effects of habitat area on the density and species richness of birds in different states of secondary succession. He found that bird density changed with successional stage and also with area. Early in the succession densities were constant across sites of different areas. At the later stages of the succession densities of birds were higher, but particularly so for the smallest sites. Oelke (1966) also found that small patches tended to have higher densities of birds than do large patches of woodland. This may be due to edge effects (small woods have a higher proportion of edge than do large woods of the same shape). Helle (1984) used rarefaction techniques (briefly, they correct for unequal sample sizes) to assess whether later stages of the succession were richer in species or whether this was just an artefact of their higher bird densities. He found that the late stages in succession remained richer in species even after their higher bird densities were taken into account, and that large patches contained more species than small patches for both early and late successional stages. However, here he did not use rarefaction to take out the effects of bird density. Although the smaller patches held birds at higher densities this effect was not great enough to swamp the effect of area itself. In other words, just as we would expect, large areas still held more birds than small areas. This means, again, that the species/area relationship may be partly an artefact of taking different sized samples from the same species pool.

British studies

There have been few studies of the effects of woodland area on forest bird communities in Britain. A much quoted study is that of Moore and Hooper (1975) which used data from 433 woods. Larger woods contained more species than small ones and the authors suggested that a useful rule of thumb would be that to double the number of species contained in a wood its area must be increased 10-fold. The applicability of this rule is questioned by the work of Woolhouse (1983) who used data collected by the British Trust for Ornithology's Common Birds Census to investigate species/area effects in 30 British woods. Woolhouse showed that sampling effort can have a big effect on the slope of the species/area relationship. In his own study he showed that the Trust's data demonstrated that large woods tended to be less thoroughly censused (in terms of hours/ha) than small ones. Less thorough censusing leads to smaller numbers of individual birds being seen and therefore fewer species are recorded. The apparent bias in sampling effort which Woolhouse found altered the factor by which the area of a wood must be increased, in order to double the species found in the wood, from being a 21-fold to a 7-fold increase. Moore and Hooper give only scant details of their sampling regime, stating that less time was spent in small woods than large woods. Considering the importance of the effects of sampling effort on the species/area relationship the 10-fold rule cannot be relied upon.

There are other problems relating to the applicability of the findings of Moore and Hooper. Their woods were a mixture of coniferous and deciduous ones mostly from southern Britain. The larger woods are mostly National Nature Reserves and therefore, although not created as reserves specifically because of their bird populations, might be expected to be richer in plant diversity or other relevant respects. In addition, woods in different parts of the country differed in their areas. Fuller (1982, quoting Dobson and Fuller, in preparation) claims that the species/area relationship differs regionally in Britain. Differences in species richness were more closely related to woodland area in southern Britain than in Scotland where there was little or no effect of area on species richness. This will partly be due to the fact that Scottish woods contain fewer species than English ones (see above).

Fuller (1982) produces incidence functions for British woodland species recorded in the BTO's Ornithological Site Register. Incidence functions (Diamond, 1975) show, for woods of a given size, the proportion of woods in which particular species are found. In Britain some species, such as wren, robin and blackbird, are found in almost all woods of whatever size. There are no species that appear to favour small woods and avoid large ones, although Sharrock (1976) suggests that the long-eared owl may be such a species simply because it is forced out of large woods by tawny owls.

There are several species whose incidence (proportion of woods occupied) is much bigger in large woods than in small ones. Examples of species include nightingale, turtle dove, garden warbler, tawny owl and tree pipit. These data suggest that, as in the American studies, the species/area relationship describes some predictable changes in bird community composition with area. Big woods hold more species than small ones, and this is because extra species are added to the community but no species avoids large woods. These results do not, however, by themselves elucidate the mechanisms underlying this phenomenon. It is likely that part of this effect is simply due to rare species being more likely to be found in large woods because of the larger sample sizes of birds which are involved. Ideally this effect would be removed from the data by using rarefaction techniques.

Incidence functions would be most informative in the following two cases where these confounding effects could be discounted. First, if there is a very clear-cut step-function in the incidence function then this would seem to be indicative of a real preference for large woods. Second, species showing an apparent preference for small woods cannot have been biased in this direction by sampling effects (unfortunately there do not appear to be any such species).

Bibby (Bibby *et al.*, 1985; Bibby *et al.*, 1989) has investigated the effects of habitat area on bird communities in two different contexts. In both studies the data were collected using the point-count method, so the relationship between species number and area is not a conventional species/area relationship since effort per unit area is constant in woods of different sizes.

In one study of birds occurring on restocked (clear felled and replanted) conifer plantations, more species were found on larger areas (Bibby *et al.*, 1985) (Figure 4.2). To investigate whether this was due to large plots having large numbers of individuals present on them, samples of birds were drawn from the overall species pool. It was found that observed and predicted numbers were highly correlated, suggesting that the effect of area is really an effect of bird numbers; large plots only have more species because they have more individuals in them.

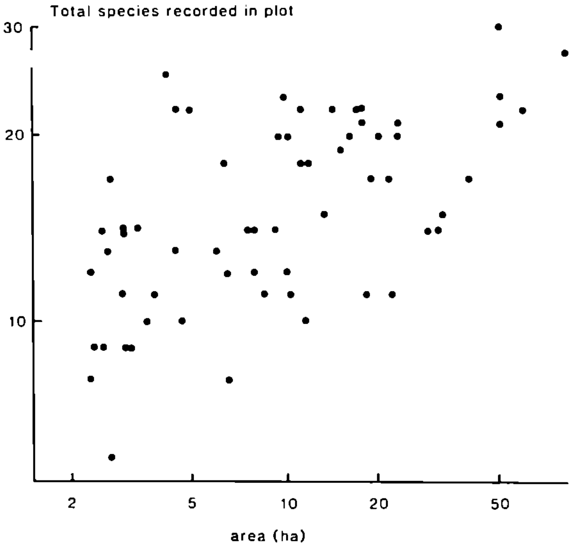


Figure 4.2 Number of bird species recorded in 62 restocked conifer plantations in Wales in relation to area. Both axes are logarithmic (redrawn from Bibby *et al.*, 1985).

The other study looked at the effect of the area of 'islands' of broadleaves in conifer plantations on bird communities (Bibby *et al.*, 1989). Here it was found that the species whose presence in the forest was closely associated with broadleaves, such as chiffchaff and pied flycatcher, increased in number linearly with the area of the patch of broadleaves. This result is important because the species which relied on broadleaves were generally of greater conservation interest than the other species which occurred in these stands. Bibby *et al.* (1989) referred to the density of these birds increasing with the area of the broadleaved patch.

This use of the term density is confusing since at face value it suggests exactly the opposite effect to that actually found. The confusion arises because density is here used to mean the number of birds within the area (30 m radius) censused by a single point-count. This area may be larger or smaller than the area of broadleaf and therefore has to be multiplied by an appropriate factor to reach the number of birds found in and around broadleaved patches of different sizes. The calculation of an appropriate correction factor is not completely straightforward, but using reasonable estimates the data suggest that 10 × 0.1 ha patches of broadleaves would support six times more of these discriminating birds than a 1 ha block (C. J. Bibby, personal communication).

These results should be treated with caution, as the response of these discriminating species in this Welsh study, may well be influenced by their abundance in habitat adjacent to the conifer forests. In the northern uplands of Britain, many of the discriminating birds are either absent, patchily distributed or scarce.

Tree species

North American studies

There are a large number of American studies which have described the species to be found in woods of different dominant tree species; they are not reviewed in detail here. Comparisons between different studies are made difficult because of the differences between them in the intensity of sampling and in the area sampled. This means that different studies will have recorded different numbers of individual birds and different numbers of species. Ideally we would wish to compare the species richness of equal areas of different types of woodland. However, differences found between different types of woodland using such comparisons might be confounded by differences in density, so we would also like to compare different woodland types on the basis of equal numbers of individual birds.

A paper by James and Rathbun (1981) suggests the use of rarefaction curves to enable this to be done. Rarefaction enables the calculation of the expected number of species which would occur in a random sample taken from a given distribution. This means that given a real sample of birds observed in an area, it is possible to calculate how many species would have been seen if fewer individual birds had been recorded. This enables the comparison of datasets of different sample sizes.

Similarly rarefaction can be used to compare species lists from different sized areas. This is slightly more involved. James and Rathbun (1981) suggest that samples from large areas should be reduced proportionally down to the area to be compared (for instance, reduce the sample size by one half to compare an area with another which is half its size), and then use rarefaction to calculate the predicted number of species to be found in that reduced sample. This method enables the researcher to account for the confounding effects of sample size in his calculations.

James and Rathbun (1981) used rarefaction to examine 37 breeding bird censuses carried out by amateurs in the USA. They found that for a given area, deciduous forests contained more species than mixed forests, which were richer than coniferous ones. Deciduous forests also had densities of birds of about five times those of coniferous forests. The more widespread use of this method would greatly improve the rigour of comparing different bird communities.

British studies

There have been many studies in different parts of Britain which have described the bird communities in woods consisting of different tree species. Many of these studies are thorough descriptions of the birds of individual woods but give little insight into the general applicability of their findings. The most wide ranging studies have been those of Yapp (1962) and Simms (1971) who surveyed a large number of woods of different tree species in different parts of the country. The summaries of their work do not disclose the variability of their findings. However, it is notable that their findings closely parallel each other and are based on larger numbers of woods than other studies (see below).

Table 4.1 lists studies of bird communities in different types of wood. Since the studies were not carried out on plots of the same size, nor in woods of the same size, nor over the same period of time, nor using the same census techniques, nor necessarily at the same time of year and certainly not with the same personnel, comparisons between studies would be largely unjustified. Comparisons within studies are more worthwhile.

Table 4.1 Studies of bird communities in different types of British woods. N = number of species recorded in the census, 1,2,3 = first, second and third most abundant species. D = territories per 100 ha (1 km^2)

Author	*Main tree*	*Site*	*N*	*1*	*2*	*3*	*D*
Adams and Erdington, 1973	Oak/beech		15	WR	CH	GT	386
	Oak/beech		16	BT	GT	CT	214
	Douglas fir/Norway spruce		7	GO	WR	CH	329
	Douglas fir		6	GO	WR	CH	271

Author	*Main tree*	*Site*	*N*	*1*	*2*	*3*	*D*
Adams, 1974	Oak/beech/pine		14	WR	CH	GO	867
	Oak/beech		14	WR	BT	CH	616
	Birch/ash/alder/oak		14	WR	CH	RO	574
	Oak/birch		12	WR	WW	PF	475
	Corsican pine/Scots pine		6	WR	GO	CT	492
	Sitka and Norway spruce		6	WR	GO	CH	283
	Sitka spruce		6	GO	WR	CH	483
Batten and Pomeroy, 1969	Mixed	Rhum, Inner Hebrides	19	CH	WW	WR	567
Flegg and Bennet, 1974	Oak, coppice	Kent	23	SG	WP	CH	990
	Oak, standards	Kent	31	HE	WP	BT	3430
	Oak, gladed	Kent	39	TD	D	WP	5100
	Oak	SE England	32	RO	WR	BT	1163
	Oak	SE England	23	RO	GT	WR	1175
	Oak	SE England	29	BT	BB	WR	2736
	Oak	SE England	38	WR	RO	BT	1279
	Oak	SE England	52	D	WR	TD	3174
	Oak	S England	27	WR	BT	RO	1100
	Oak	N England	25	BT	RO	WR	913
	Oak	Wales	22	BB	RO	BT	453
	Oak	Wales	25	PF	CH	WR	454
	Oak	Scotland	30	CH	WR	RO	1154
	Oak	Eire	19	CH	RO	BT	1518
Fuller and Taylor, 1984	Lime	Lincolnshire	19	ST	WW	RO	nd
			22	WW	RO	BB	nd
			24	WW	RO	CH	nd
			29	WW	RO	ST	nd
			30	WW	WR	RO	nd
			30	CH	WW	WR	nd
			32	WW	GW	CH	nd
Hope Jones, 1972	Oak (grazed)	Wales	20	BT	PF	CH/WR	469
	Oak (not grazed)		31	WW	CH	TP/RE	908
	Birch		13	WW	CH	WR	400
	Oak + rhododendron		12	CT	CC	many	219
	Oak (degenerate)		4	TP	GT	PF	81
Irvine, 1977	Oak/beech	New Forest	–	RO	BT	CH	931
	Oak		–	CH	BT	GO	635
Lack and Lack, 1951	Pine	Breckland	10	CH	CT	WR	nd
Massey, 1974	Oak	Mid-Wales	33	WW	CH	TP	968
	Ash		26	RO	WW	RE	837
	Alder		31	RO	WR	CH	1005
	Mixed		21	WW	CH	RO	1035
Moss, 1978b	Sitka spruce	Ae Forest	9	GO	CH	WR	396
	Norway spruce	Ae Forest	8	GO	CH	WR	552
	Sitka and Norway spruce	Ae Forest	8	GO	CH	WR	498
	Sitka spruce	Ae Forest	8	GO	CH	WR	411
	Sitka spruce	Greskine	8	GO	CH	WR	520
	Norway spruce	Greskine	4	GO	CH	CT	597
	Larch	Ae Forest	8	WR	GO	CH	446

Author	Main tree	Site	N	1	2	3	D
	Scots pine	Annan Valley	6	WR	GO	CH	208
	Scots pine	Annan Valley	8	WR	CH	CT	340
	Birch/pine	Annan Valley	16	WW	WR	RO	490
	Birch/pine	Annan Valley	17	WW	WR	CH	880
	Mixed	Annan Valley	17	CH	WR	WW	1669
	Scots pine	Abernethy	9	CH	GO	WR	428
	Scots pine/lodgepole pine	Abernethy	8	CH	GO	CT	172
	Scots pine	Glenmore	6	WR	CT	CH	151
Parsons, 1976	Mixed	Somerset	35	WR	GO	RO	3311
			32	WR	GO	RO	3441
			31	WR	GO	BB	3519
Shaw, 1976	Oak	Glen Falloch	22	CH	BT	RO	516
		Craigrosten	23	RO	WR	WW	418
Simms, 1971	Oak, pedunculate		–	CH	RO	WR	nd
	Oak, sessile	England	–	CH	PF	WW	nd
		Scotland	–	CH	WR	BT	nd
		Ireland	–	CH	BB	RO	nd
	Alder	Scotland	–	CH	WR	RO	nd
	Beech	England	–	CH	BB	GT	nd
		Scotland	–	CH	BB	BT	nd
		Ireland	–	CH	BB	WO	nd
	Scots pine	Scotland	–	CH	CT	GO	nd
		Ireland	–	GO	BB	CH	nd
	Norway spruce	Scotland	–	WP	CH	GO	nd
	Sitka spruce	Scotland	–	WP	CH	WR	nd
	Pine/spruce/larch	Scotland	–	CH	WR	RO	nd
		Ireland	–	CH	MT	WW	nd
	Coniferous/deciduous	Ireland	–	WP	CH	BB	nd
	Ash	Scotland	–	CH	WR	RO	nd
		Ireland	–	BB	WW	CH	nd
		England	–	WW	CH	WP	nd
	Birch	Scotland	–	WW	CH	TP	nd
		England/Wales	–	CH	WW	TP	nd
		Ireland	–	WW	RO	CH	nd
Watson, 1969	Pine	Scotland	16	CH	WW	RO	nd
			12	CH	CT	WW	nd
			13	CH	CT	CrT	nd
			9	CH	WW	CT	nd
			9	CH	CT	GO	nd
			14	GO	CT	CH	nd
Williamson, 1969	Scots pine	Wester Ross	11	WR	CT	CH	217
	Birch	Wester Ross	20	WW	CH	WR	1100
	Oak	Wester Ross	19	CH	WW	GT/WO	1320
	Alder	Wester Ross	13	WW	CH	LR	550
Williamson, 1971	Redwoods	Wales	24	GO	WR	BB	1097
Williamson, 1972	Oak	Argyll	24	CH	WW	RO	583
			21	CH	RO	WW	930
			14	CH	WR	TP	330
			20	CH	RO	BT	835

Author	*Main tree*	*Site*	*N*	*1*	*2*	*3*	*D*
			19	CH	RO	WR	742
			15	WW	CH	RO	360
			15	WW	MP	WC	220
Williamson and Williamson, 1973	Yew/oak	Sussex	26	CH	WH	WR	nd
	Yew		10	CH	BB	RO	nd
	Yew		18	CH	RO	BB	nd
	Yew		20	RO	CH	BB	nd
Williamson, 1976	Oak/birch/hazel	Argyll	29	CH	CT	GT	826
		Galloway	29	CH	RO	WR	712
Williamson, 1974	Oak	Stirlingshire	31	CH	WR	WW	1600
			29	CH	WR	RO	1280
			20	CH	WW	WR	1300
			22	WR	CH	WW	1130
			19	WR	WW	CH	880
			30	CH	WR	WW	1160
			30	RO	WR	CH	1150
Williamson, 1975	Mixed	Rhum, Inner Hebrides	25	CH	RO	WW	1060
Williamson, 1968	Mixed	Yorkshire	11	WW	CH	many	nd
			20	WW	ST	BB	nd
			19	WW	ST/BB/MP		nd
			29	WW	CH	ST	nd
			12	WW	CH	MP	nd
Yapp, 1955a	Birch	Wales	15	WW	BB	RO	nd
	Birch/oak		17	WW	RO	TP	nd
	Oak/beech		34	CH	TP	RO	nd
Yapp, 1955b	Mixed	Lake District	8				nd
	Oak		17				nd
	Birch		11				nd
	Alder		14				nd
Yapp, 1962	Oak, pedunculate		–	CH	WW	RO	nd
	Oak, sessile		–	CH	WoW	PF	nd
	Birch	England/Wales	–	CH	WW	TP	nd
		Scotland	–	WW	CH	TP	nd
	Ash		–	WW	CH	RE	nd
	Beech		–	CH	GT	BB	nd
	Pine	Scotland	–	CH	WW	CT	nd
	Yew		–	CH	BB	RO	nd
Yapp, 1974	Birch	Northwest	42	WW	CH	TP	nd
		Scotland	29	WW	CH	TP	nd

BB	Blackbird
BT	Blue tit
CC	Chiffchaff
CH	Chaffinch
CrT	Crested tit
CT	Coal tit
D	Dunnock
GT	Great tit
GO	Goldcrest
GW	Garden warbler
HE	Heron
LR	Lesser redpoll
MO	Meadow pipit
PF	Pied flycatcher
RE	Redstart
RO	Robin
SG	Starling
ST	Song thrush
TD	Turtle dove
TP	Tree pipit
WC	Whinchat
WH	Whitethroat
WoW	Wood warbler
WP	Wood pigeon
WR	Wren
WW	Willow warbler

Particular species of birds are associated to some extent with particular species of trees but the data which exist at present are insufficient to show this convincingly. For example, although the highest densities of crested tits are largely confined to the remnants of the Caledonian forest in Scotland there appear to be no studies which show the extent to which this species relies wholly on Scots pine within this area. Similarly pied flycatchers and wood warblers are common in sessile oakwoods in parts of Wales but their use of other tree species within this area, and in other areas is not well documented.

To some extent this lack of data is understandable; every birdwatcher knows that hawfinches and nightingales are not abundant in coniferous plantations, that chaffinch and wren are the commonest birds in many woods of many different tree species, that goldcrests and coal tits often dominate the communities of conifers and that willow warblers are often abundant in young broadleaved woods. These prior expectations are borne out by the studies summarised in Table 4.1 but the lack of detailed information is unfortunate. What are missing are studies of many different woods in the same geographical area. This is probably because studies of woodlands have relied on the mapping method of censusing which is very labour intensive and has limited the number of woods surveyed. This has meant that many of the data are no better than anecdotal, and often relate to sites of known avian value and interest such as nature reserves. Such data cannot really be regarded as representative of extant British woods.

There have been two studies which have surveyed large numbers of woods in Britain, but in both of these the results were summarised in such a way as to make it impossible to assess the degree of constancy of the bird communities in different types of wood. These studies (Yapp, 1962; Simms, 1971) both used the transect method of sampling which allowed them to cover more woods than would have been possible using mapping. Although the transect method they used cannot give accurate information on densities of woodland birds and also is biased towards detecting conspicuous species (Newton and Moss, 1981), it can provide a useful method for covering large areas quickly (see Chapter 3) and provides a database for comparing different woods. An encouraging aspect of these two studies is the close level of agreement between them.

Pied flycatchers and wood warblers are both common in Yapp's sample of sessile oakwoods but not in pedunculate oaks. The differences between the bird communities of the two oak species are interesting and are fully confirmed by the larger sample of woods sampled by Simms (1971). In the oak woods studied by both Yapp and Simms the same five species were found to be the most common, and the other species maintained very similar ranking. Yapp suggests that different management practices have meant that pedunculate oakwoods have more luxuriant growths of shrubs in their understorey and that this discourages ground nesters such as wood warblers and tree pipits. This would suggest that the actual tree species is less important than the structure of the woodland although this remains to be tested by comparing woods of similar structure but different species.

Yapp shows that sessile oakwoods tend to be found further north and west in Britain than their pedunculate counterparts. Partly as a result of this sessile oakwoods also tend to occur at higher altitudes than pedunculate oakwoods. These factors may account for the smaller number of species recorded in sessile woods, and their slightly different identities. In other words the apparent differences between the bird communities of these two closely related tree species may (or may not) be due to confounding effects of geographical location, altitude and management practices rather than a real difference associated with the two species.

The similarity between the results of Yapp and Simms is continued in their studies of birches, even to the extent that geographical differences are confirmed between England/Wales and Scotland. These results are encouraging in that they suggest that there may be real and measurable differences between different areas and between different tree species. They also suggest that more research in this area would be fruitful.

Flegg and Bennett (1974) take issue with the findings of Yapp and Simms that there might be a "definite pattern to the distribution and frequency of oak woodland birds". They suggest that the use of the term 'typical' is inappropriate in this context. To illustrate the range of species and densities which occur in oakwoods they used data from 10 BTO Common Birds Census (CBC) plots and one Irish site. Their point of disagreement with Yapp and Simms, while not spelled out, appears to be that the dominant and rare species vary with geographical area and management; both points which appear to be well covered in Yapp's and Simm's works.

In fact the bird communities shown by Flegg and Bennett (1974) do not appear to vary greatly, although this is a somewhat subjective assessment. For the 11 sites only eight species are needed to occupy the 33 slots for commonest, second commonest and third commonest species; five more than the minimum possible but 28 less than the maximum possible. The species number and overall density in the Scottish plot is very similar to those of plots in south-east England of similar size. Given the fact that there may be small differences between individual woods, the general point that there may be overall similarities in bird communities between woods of the same dominant tree species is still a tenable one.

Succession

"Large oaks from little acorns grow", and, not surprisingly, different birds are associated with different stages of the succession. From planting until the end of a rotation may take up to a century or more depending on the tree species. This means that it is impractical to study the changes in bird communities through the whole of the succession in one place. In practice, then, most studies have included nearby woods of different ages, and assumed that these will give an accurate picture of the changes which would occur in an individual wood. There are obvious dangers in this approach, sensible though it is. It may be that the areas differ in some important respect, other than the age of their trees, which will influence their bird communities. This is especially likely to be true of planted woodlands since there are good reasons to suppose that certain types of land (lower, more fertile) will be planted first. Historical factors may affect the development of woods of similar ages in different ways. For example, differences in thinning regimes, or the incidence of storm or insect damage, may allow stands planted at the same time on similar sites to show great structural differences later on in the rotation.

The alternative to studying a number of woods at different growth stages, is to follow a single wood through time. This method suffers from the practical disadvantage of having to wait decades for any results. Also, it is not free from the possibility that special factors acting in particular years may give atypical results. In practice most studies have concentrated on spatial successions.

Non-British studies

There have been many North American studies of avian succession. We do not intend to discuss these in detail since they deal with unfamiliar species of both trees and birds, and are therefore of limited relevance to the British situation. Reviews of the American literature can be found in Hamilton and Noble (1975), Meslow and Wight (1975), Shugart *et al.* (1975) and Anderson (1979). During the course of succession there is a general trend for species richness and bird density to rise. Some authors have suggested that there should be a decline in species diversity in the last stages of succession (Margalef, 1968); others have claimed that there is in fact a decrease in diversity in mid-successional stages (Meslow and Wight, 1975). Since most studies only consider at most six stages of the succession it seems unlikely that convincing deviations from the general upward trend will be found. Notable studies of woodland succession include (Johnston, 1947; Odum, 1950; Johnston and Odum, 1956; Karr, 1971; Shugart and James, 1973; Dickson *et al.*, 1984; Engstrom *et al.*, 1984).

European studies confirm the American findings that species number and bird density rise through the succession (Turcek, 1957; Glowacinski, 1972, 1975a, 1979, 1981a, 1981b;

Glowacinski and Jarvinen, 1975; Glowacinski and Weiner, 1975). The work of Glowacinski is particularly detailed since he studied 21 different stages of succession in an oak–hornbeam forest. Bird species richness (and diversity and density) showed a bimodal rise with succession. Peaks occurred at the scrub and climax stages. The early stages of succession contained characteristic open country species such as shrikes, the last stage in the succession was dominated by collared flycatcher, chaffinch and blue tit. There was a tendency for the later stages of the succession to be dominated by resident species. Other important studies include those of Ferry and Frochot (1970) and Helle (1984, 1985a, 1985b).

British studies

Some of the earliest studies of succession were those of Lack (1933, 1939) and Lack and Lack (1951) in the Breckland pine plantations of Norfolk and Suffolk. The unplanted heaths were dominated by skylarks, meadow pipits, stonechats and whinchats. Between the age of 4 years and 15 years scrub birds dominate the bird community and after about 10 years none of the original heathland species remain. In particular willow warblers built up to a peak where they comprised just over half of the total bird community when the trees were 9 years old. From 7 years after planting woodland species began to dominate; chaffinches and coal tits grew in numbers and were the two commonest species in trees of 26–27 years. At this stage willow warblers comprised only about 1 per cent of the total bird population which numbered 99 breeding pairs. Lack and Lack (1951) showed that in one particular plantation the changes in bird communities over a period of 18 years mirrored very closely those found by looking at the spatial succession (Figure 4.3).

A recent 10-year study of the birds of sweet chestnut coppice investigated the effects of succession in coppices varying from 0 to more than 19 years of age (Fuller and Moreton, 1987). Some species (tree pipit, yellowhammer) reached their peaks in the first few years of the coppice cycle whereas others reached theirs in mid-cycle (willow warbler) or in the oldest coppices (robin, blackbird). Those species which peaked in numbers in young coppices were also found in the most restricted range of ages; whereas species such as robin which were commonest in the oldest stands were nevertheless present in coppices of all ages. Species richness declined with coppice age, particularly after canopy closure, and the migrant/resident ratio also declined with coppice age. Fuller and Moreton point out that in order to maximise species richness and the numbers of birds with most restricted amplitudes a short coppice cycle (e.g. 8-year) should be maintained.

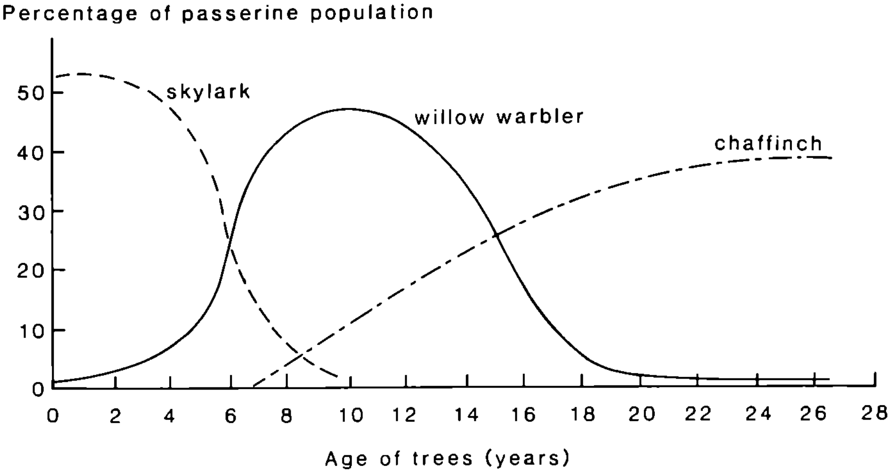


Figure 4.3 Changes in the population of three species as the trees grow (redrawn from Lack and Lack, 1951).

There have been several British studies which have investigated the changing bird communities associated with succession (Table 4.2 and Chapter 5). In all these studies, bird species richness (and BSD) and overall bird density were greater at the end than at the beginning of the succession although in some cases these measures peaked at an intermediate stage, often one which is basically scrub. Some studies have included only songbirds but this is unlikely to alter the results very much, although many authors point out that it is usually the scarcer passerines and non-passerines which are of the greatest conservation interest.

Vegetation structure

Woods are complicated places. The structure of the vegetation is a plausible factor which may determine the composition of a bird community. It has often been shown that different species of birds tend to feed in different parts and at different heights in the vegetation (Hartley, 1953; Gibbs, 1960; MacArthur, 1974) which suggests that the more varied the vegetation structure the more species of birds may be present. Many different measures of vegetation structure have been used by ornithologists, but the most common is the foliage height diversity (FHD) (Chapter 3).

Foliage height diversity

The American ecologist Robert MacArthur first used foliage height diversity (FHD) to investigate the bird species diversity in deciduous woods (MacArthur and MacArthur, 1961; MacArthur *et al.*, 1962; MacArthur *et al.*, 1966) and found that FHD was a good predictor of bird species diversity (BSD). Plant species diversity was shown to account for an insignificant proportion of the variation in BSD after the effect of FHD was removed. MacArthur does not state, though it is possible to calculate from the data, that FHD explained a slightly higher proportion of the variation in BSD than did plant species

Table 4.2 Studies of bird communities in relation to succession in British woods

Author	*Succession* *From*	*To*	*Study site*
Batten and Pomeroy, 1969; Williamson, 1975	Moorland	Mixed woodland	Rhum, Inner Hebrides
Bibby *et al.*, 1985	Clearfell	Mature conifer or broadleaf	North Wales
Dougall, unpublished	Moorland	Sitka spruce (mainly)	Craik Forest, S. Scotland
Fuller and Moreton, 1987	Felled chestnut coppice	Mature chestnut coppice	
Hope Jones, 1972	Open pasture	Mature oak	West Wales
Jessop, 1982	Moorland	Mature pine	Glen Affric, N. Scotland
Lack and Lack, 1951	Heathland	Mature pine	Breckland, East Anglia
Moss, 1978b; Moss *et al.*, 1979a	Moorland or grassland	Sitka and Norway spruce	South Scotland
Niles, 1971	Moorland	Mature spruce	Dartmoor, Devon
Sykes *et al.*, 1983	Moorland	Thicket stage Sitka spruce (mainly)	Cumbria
Yapp, 1955	Clearfell. Oak/birch scrub	Mature oak	Wyre Forest, W. Midlands

diversity. However, in mixtures of habitats (e.g. patches of deciduous and coniferous forest), measures of vertical diversity were insufficient to predict BSD.

Many studies (Willson, 1974; Rov, 1975; Ulfstrand, 1975; Moss, 1978a; Haila *et al.,* 1980; Beedy, 1981; Newton and Moss, 1981; Hino, 1985; French *et al.*, 1986) have demonstrated a significant positive relationship between BSD and FHD but others have not (Willson, 1974; Balda, 1975; Pearson, 1975; Hino, 1985). Erdelen (1984) has pointed out that many studies compare FHD and BSD across, rather than within, habitats and so it is doubtful whether the observed effect is solely due to differences in FHD. For example, in three studies (Willson, 1974; Erdelen, 1984; Hino, 1985) the BSD/FHD relationship is only significant between habitats not within them. Erdelen also points out that FHD has been defined and measured in slightly different ways in each study and that in some cases (MacArthur and MacArthur, 1961; MacArthur *et al.,* 1966; Moss, 1978a) it is explicitly stated that various subdivisions of the vegetation profile have been tried in calculating the FHD and the one giving the highest BSD/FHD correlation has been chosen.

The study of Moss (1978a) was the first important British investigation of the relationship between BSD/FHD. He measured BSD and FHD on 18 sites in the Spey Valley and in Dumfriesshire for between 1 and 3 years. The sites differed in their woodland type, consisting of plantations of spruce, pine and larch, and natural or semi-natural birch, pine or mixed woodland and some were upland, others lowland sites. Moss plotted the relationship between BSD and FHD for all 34 censuses and found a highly significant positive relationship. However, this treats all 34 points as statistically independent which they clearly are not since many are simply replicates from different years. Recalculating the regression equation for the 18 sites does not greatly affect the relationship (Newton and Moss, 1981) (Figure 4.4) but it is difficult to assess to what extent the relationship is due to differences between sites or tree species and therefore the effect of FHD itself is not easily discernible. More recently, French *et al.* (1986) demonstrated a significant correlation between FHD and BSD but again this result came from a small number of different woods.

Other studies

There are many studies which have described the bird communities of woods of different structures; two of these are described here.

Blondel *et al.* (1973) studied aspects of vegetation structure. They used the number of layers of vegetation and the percentage vegetation cover in the wood as their measures. Bird species richness rose with the number of vegetation layers which were present. Avian density showed a sigmoid rise with vegetation cover whereas bird species richness peaked at intermediate values of this factor. Avian species diversity showed a tight correlation with vegetation

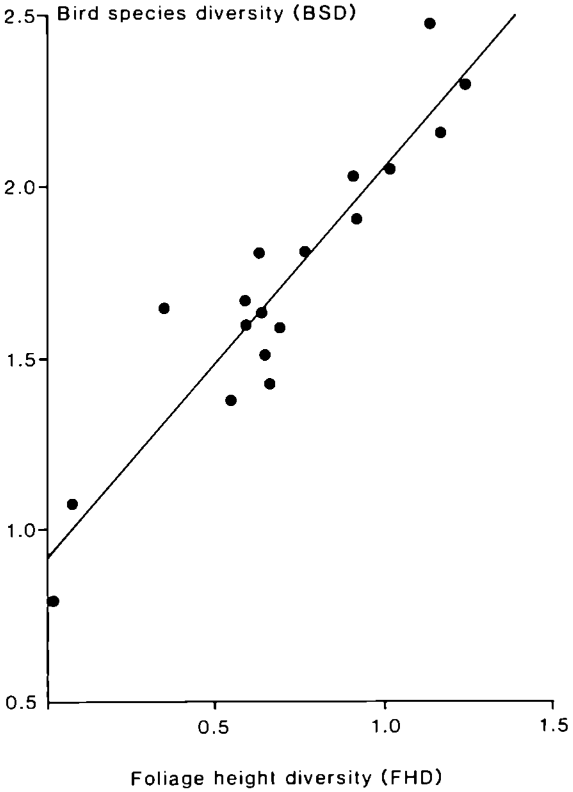


Figure 4.4 Relationship between bird species diversity and foliage height diversity from a variety of mature woods with broadleaved, coniferous or mixed crops (from Newton and Moss, 1981).

layer diversity. These results can be taken to indicate that species richness peaks before the climax vegetation community is reached but that bird density continues to increase throughout the succession.

James and Wamer (1982) have taken the analysis of the relationship between vegetation and bird communities to a level of sophistication untouched by other studies. They used rarefaction to arrive at standardised measures of species richness for many different bird censuses. The vegetation of the plots was then analysed by principal component analysis and three main components were identified: tree species richness/canopy cover, variation in canopy height, and tree density. A SYMAP computer program was then used to draw contours of either the number of bird species expected in 10 ha or the number of individual birds to be found in 10 ha in relation to vegetation structure. Bird species richness (per 10 ha) peaks in mature deciduous forest but not in those that have the highest tree species richness, canopy height or tree density. The number of birds (per 10 ha) shows a similar pattern except that it does peak at the highest values of tree species richness and canopy height.

Bird species richness and density are minimal in coniferous forests characterised by high tree density, low canopy and few species of trees. James and Wamer (1982) used careful statistical analysis to control important confounding effects which are often acknowledged and then ignored in other studies. Although rarefaction, principal component analysis and computer mapping may not yet be in the armoury of every scientist studying birds in forests, this study does seem to point the way ahead to a clearer understanding of the important processes acting on forest bird communities. In the absence of clear experiments, sophisticated statistical treatment is a sensible course to follow.

Edge effects

It is a general rule that the edges between vegetation types (ecotones) are more diverse in their flora and fauna than the separate habitats (Pianka, 1974; Burgess and Sharpe, 1981). This finding has important consequences for the management of habitats since it suggests that the maximum diversification of habitats within an area will maximise the number of species which occurs there. This suggestion runs counter to the suggestion that some species require minimum areas of their preferred habitats in order to exist in a particular area.

The forest edge may differ in many ways from the interior. More light is available so that the growth of shrubs and trees may be enhanced and lead to a higher primary productivity (Ranney *et al.*, 1981), which is likely to be translated into higher insect abundances (Hansson, 1983), and thus favour insectivorous birds. The open forest edge may also provide room for aerial displays and for using prey-catching techniques.

Haapanen (1965, 1966) reviewed studies which have suggested that some species prefer the edges of forests. Sammalisto (1957) stated that white wagtail and winchat were found on edges of forests where they join open mires but never inside the forest. Jalkanen (1960) identified species which showed a preference for forest edge, for different types of forest (dry sites, spruce stands, hardwood and mixed stands) in agricultural land. Some species were edge species in all habitats (e.g. yellowhammer) and most of the species exhibiting edge effects were warblers, thrushes and buntings. The edge species did differ between habitats. For example the great tit showed a preference for the edge of conifer plantations but not in broadleaves. This was probably because nest sites were rare in conifer stands whereas food was not. Therefore great tits could use some conifer stands providing that there were nest sites in the adjacent habitat (Bibby *et al.*, 1989). Both great tits and blue tits can often be encouraged to nest in low-elevation conifer stands by providing nestboxes (Lack, 1966; Perrins, 1979).

Many studies have demonstrated that the density of particular woodland species increased or decreased at the forest edge (Lay, 1938; Johnston, 1947; Hogstad, 1967; Gromadski, 1970; Odum, 1971; Galli *et al.*, 1976; Anderson *et al.*, 1977; Jarvinen *et al.*, 1977; McElveen, 1979; Strelke and Dickson, 1980; Whitcomb *et* *al.*, 1981; Helle and Helle, 1982; Kroodsma, 1982a; 1982b; 1984; Hansson, 1983; Fuller and Whittington, 1987). Some of these studies are based on man-made habitat boundaries such as the paths of power lines through forests. In these cases Niemi and Hanowski (1984) suggest that the control sites used to assess the effects of the habitat change may rarely be well matched with the treatment area. Presumably this is because man-made habitat alterations are rarely carried out at randomly chosen locations within a habitat; instead they may be affected by details of the local topography. This point does not change the fact that edge effects exist but it questions their cause.

Here we limit ourselves to a discussion of two particular recent studies which deal with typically British species (although one study was carried out in Sweden) of woodland birds because they appear to demonstrate some interesting effects.

Fuller and Whittington (1987) studied edge effects in Lincolnshire ash–lime woods. They used territory mapping to study edge effects along rides within the woods, and at the woodland edge. The territory maps were visually checked for edge effects. Interestingly there were no convincing edge effects at rides for any species; neither avoidance nor preference were indicated. In contrast, there were many striking cases of preference being shown for the edge of the wood. Quantitative comparisons of the density of registrations supported this subjective analysis. In matched pairs of plots, the density of nine species did not differ between the edges of rides and the interior of the wood. In contrast, many species showed an effect of distance from the edge of the wood on their densities. Species showing strong edge effects included blackbird, song thrush, chaffinch, garden warbler, blackcap and willow warbler. For some of these species the strength of the edge effect varied between different habitats within the wood (thinned coppice, unthinned coppice, high forest).

Fuller and Whittington carried out vegetation sampling at different distances from the ride and forest edge to relate this to the bird data. They found that for low, medium and high shrub layers at the edge of the wood the density of vegetation decreased as one passed away from the edge and into the interior of the wood. Thus it appeared that the bird species showing edge effects were responding to a factor correlated with the greater vegetation density. Comparing the density of vegetation at the woodland edge with that at the edge of rides showed that the woodland edge vegetation was more dense, particularly in the lower layers, than that at the edge of the ride. This suggests that the lack of an obvious ride edge effect was due to the less dense vegetation at the edge of rides compared with the forest edge. This might be associated with the narrowness of rides allowing less light to penetrate than is possible at the forest edge.

Hansson (1983) studied the edge effects across the transition between an old (around 100 years old) spruce/pine forest and a clear felled area by walking transects across the transition and noting the position of all birds. The transects penetrated 250 m into both the forest and the clearcut. Hansson found that many species showed very strong edge effects. Some forest species such as robin and goldcrest were much more abundant at the forest edge than at the forest interior. Two species, great spotted woodpecker and tree pipit, were found commonly at the edge of the forest and on the edge of the clear felled area. Their numbers were higher in the centre of the clear fell than in the forest interior but much higher at the junction of the two habitats.

The species found primarily on open ground also showed edge effects; avoidance of the open land near to the forest. These species included yellowhammer, red-backed shrike, woodlark and stonechat. Few details are given of the rarer non-passerines found on the clear felled areas. However, these results suggest that the size and shape of felling coupe, and thus the amount of edge, will affect the numbers of both forest and open country birds. If the finding here, that forest birds peak in numbers at the edge whereas open country birds avoid the forest edge, is generally true, then this would have implications for the best size of felling coupe which would maximise the conservation potential of an area (Chapter 7).

Summary

Many factors affect the composition of forest bird communities. Many of the studies discussed look at particular factors but their results may be confounded by others. For example, the wide-ranging investigation by Moore and Hooper of species/area relationship may be confounded in that their small woods tended to be found in the north of the country and/or tended to be at higher altitudes than their sample of large woods then it would mean that part of the apparent effect of wood size was due to wood location.

Despite these kinds of complications the following suggestions are supported by data.

1. Woodland bird species richness decreases towards the west and north of the country.
2. Woodland bird species richness decreases with altitude.
3. Woodland bird species richness is greater in large woods than small woods.
4. The typical bird community of different successional stages is different.
5. The typical bird community of different tree species is different.
6. The numbers of species and of individual birds, tend to be different at the junction of two habitats from those typical of the habitats.

Chapter 5

The effects of upland afforestation on birds in Britain

Introduction

Present forestry policy is to increase the area under trees. New afforestation is likely to continue in northern and western uplands of Britain (lowland farmland is not the concern of this review) where land prices are generally low, soils are poor, the climate is wet, and agriculture is not highly profitable. In these areas, which are some of the remotest in the country, afforestation leads to great ecological changes. In this chapter we assess the information that is available to discuss and predict the effects of afforestation on moorland bird communities.

Conservation and forestry organisations have come into conflict over upland afforestation, and extreme attitudes have been taken by both. As examples of unhelpful exaggerations which pervade the literature we take two, one from each side, to illustrate the extreme stances which are sometimes adopted, and which can be shown in both cases to be untenable.

Foresters sometimes claim that they are restoring tree cover to areas which were deforested by man but which otherwise would naturally be covered with trees, with the implication that this would result in the return of a more natural bird community. This is not a reasonable argument. Most plantations contain non-native trees which are grown in dense stands and felled long before biological maturity. They therefore do not present the variety of age-classes and tree species found in a natural forest and do not contain an appreciable number of very old trees with nest holes. While many scarce species have benefited from afforestation, some of which have substantially increased their range in Britain, there is nevertheless little reason to suppose that any new species are likely to be added to the list of British breeding birds by an increase in afforestation. In contrast, there are reasonable grounds for concern that the extent and location of new afforestation in some areas may lead to a reduction in the British population of some moorland bird species of high conservation value.

A contrasting, equally exaggerated claim is to assert that all afforestation is necessarily harmful to birdlife. This is too extreme a reaction since some species do indeed gain from afforestation and in some areas of Britain upland afforestation would do little harm to important bird species. Some of Britain's uplands support small numbers of few species of common birds and therefore cannot be considered as being greatly damaged by afforestation. In addition, it is not necessarily true that small amounts of afforestation are harmful; many moorland bird species which might indeed be adversely affected by large-scale afforestation may be unaffected or actually favoured by small-scale afforestation. Newton (1983) takes a more balanced view, ". . . the effects of afforestation on birdlife have so far been mainly beneficial. No species has yet been seriously threatened on a national scale from the effects of commercial forestry, but some have markedly declined on a regional scale. This has occurred where tree planting has been so widespread as to occupy the bulk of the productive upland. Presumably, with increasing afforestation, more and more areas could come under threat from blanket forestry".

In this chapter we consider the rather scant data which exist on the subject of upland afforestation. In particular we discuss: the changes that occur in bird communities as forests grow; which species will gain and lose from forestry; which upland areas will be worst affected by forestry; and how the form of afforestation will affect upland species.

Bird communities in relation to the growth stages of conifer forests

The process of planting open moorland with conifers brings about enormous ecological changes (Chapter 6). To assess the overall effects of afforestation it is necessary to know the response of birds to each of the stages in the silvicultural cycle. Some studies have looked at spatial successions on newly planted land, but understanding more about the clear felling stage is very important too, as this stage will occur repeatedly in the future. For instance, will moorland birds move on to the clear felled areas to breed? If they do, then this may mitigate any adverse effects of afforestation on upland species. How bird populations change in response to successional changes on newly planted ground is relatively well described for the British uplands but the population dynamics for birds of restocked areas are much less well known. Here we review each of these stages in turn.

Planting to mature forest

The changes in the bird community of an upland area were studied by Dorian Moss (Moss, 1978b, 1979; Moss *et al.*, 1979a; Newton and Moss, 1981) in Scotland. He restricted his study to a relatively small number of plots and used the mapping technique to assess the bird numbers, where he considered passerines only. Moss (1979) and Moss *et al.* (1979a) showed that the original passerine community of the grassland or moorland consisted of a very small number of species. Skylark and meadow pipit dominated the community with wheatears present in some sites. Meadow pipit showed a short-term increase after planting whereas skylark decreased almost immediately afterwards. During the early successional stages, birds which are typical of scrub moved in; willow warblers and wrens were the most abundant. After canopy closure (around 20 years) the community was dominated by species typical of woodland; goldcrest, coal tit, chaffinch, wren and robin were all common. Both bird species richness and density increased through the first rotation (Figure 5.1). Species richness peaked in the stage prior to canopy closure, when birds from both moorland, scrub and woodland communities were present, thereafter it declined (Figure 5.2). Presumably it would have increased again if old growth crops had also been studied by Moss *et al.* (Currie and Bamford, 1982a). Bird density increased in a sigmoid fashion, throughout the stages studied. Similar results to these have been found by other workers (Batten and Pomeroy, 1969; Niles, 1971; Williamson, 1975; Jessop, 1982; Sykes *et al.*, 1985).

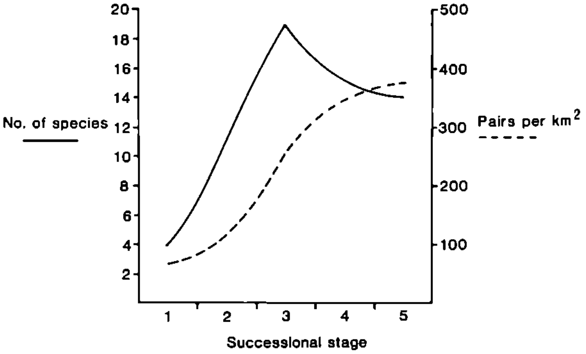


Figure 5.1 Changes in songbird density and species richness resulting from afforestation of moorland with spruce (adapted from Moss *et al.*, 1979). The forest growth stages (1-5) are the same as in Figure 5.2.

The effects which planting has on non-passerines and the scarcer passerines have been studied to a much smaller extent, which is unfortunate, since these birds include the most interesting species from the conservation point of view.

The removal of grazing animals from land about to be planted allows the ground vegetation to grow. This triggers a rapid and enormous rise in numbers of field voles in grassy habitats with population peaks every 4-6 years (Chitty, 1952). There are often two to three cycles in vole populations before canopy closure. This food bonanza is exploited by predatory birds, the

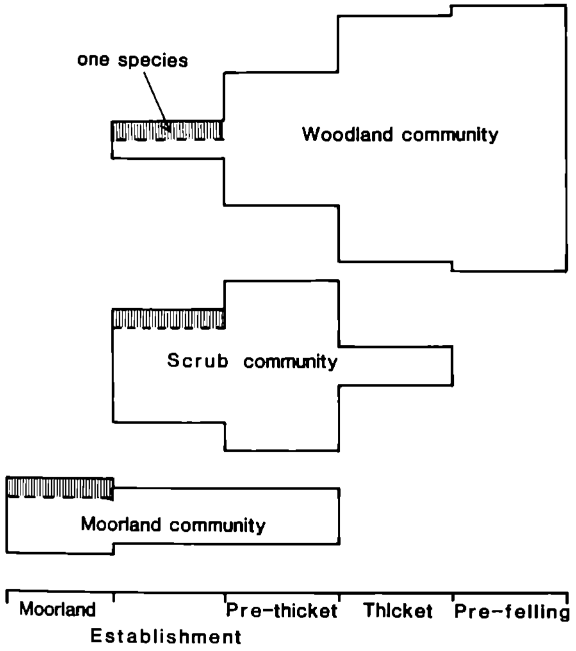


Figure 5.2 Changes in the moorland, scrub and woodland component of the bird community as a result of afforestation (from Ratcliffe and Petty, 1986).

foremost of which is the short-eared owl (Goddard, 1935; Lockie, 1955). Very occasionally vole numbers increase by a factor of 200 in plantations (Charles, 1956) but usually this increase is about 10-fold (Charles, 1981), and short-eared owls can reach densities of six pairs per km^2 (Lockie, 1955). During the peak in a vole cycle the owls' productivity is high, but this falls off as the trees grow and the numbers of voles decrease. Other vole predators such as kestrels (Village, 1980), long-eared owls (Village, 1981), and barn owls (Shawyer, 1987) also make use of the temporary abundance of food (Village, 1980, 1981), though their ability to do so is restricted by the availability of nest sites whereas short-eared owls are ground-nesters.

Hen harriers also have become charcteristic birds of the early stages of planting in the British uplands. Their spread from Orkney on to the mainland of Scotland and south to other parts of the country has been closely associated with younger conifer plantations. This may be only partly due to high vole numbers since their prey also include many of the birds which increase after afforestation (Watson, 1977; Cramp and Simmons, 1980). It is likely that a major factor favouring hen harriers in recently planted ground is the lack of persecution. It has been suggested (Watson, 1977) that hen harriers which inhabit new plantations were already present in those areas before the forests arrived. This may be true for some areas of Scotland but the spread of hen harriers into many new areas cannot be explained in this way.

Black grouse use young plantations from planting to the scrub stage (Borset and Krafft, 1973). In Britain this is up to about 20 years after planting but they also use open heather moorland, birch woods and natural pine woods (Johnstone, 1967). While they may be an example of a species favoured by a mosaic of different land uses, their diet consists mainly of ericaceous plants, so a reduction of this habitat by both the conversion of heather/moor, whether to grass or to trees, may inevitably lead to a reduction in black grouse numbers. Cadman (1949) suggested that local increases in black grouse numbers were associated with afforestation but Johnstone (1967) considered this to be unproven. The close association of black grouse with commercial forests in north Wales (Grove *et al.*, 1988), suggests that their future may be closely linked to whether the quality of forest habitats can be improved for black grouse. Storaas and Wegge (1987) found that female black grouse did use plantations for nesting, but in smaller proportions compared with other habitat.

Whereas black grouse are usually associated with the early successional stages of a forest, capercaillie are more often found in older stands, particularly Scots pine (Johnstone, 1967; Palmer, 1976; Jones, 1982a, b anc c). The best habitat seems to be dry open pine forest with a thick undergrowth of heather, blaeberry and juniper (Moss *et al.*, 1979b), although capercaillie will sometimes nest in broadleaved and mixed woods, particularly where Scots pine and larch are present. Moss *et al.* (1979b) recommend that to be suitable for capercaillie, sites should have open areas with ericaceous shrubs and some old trees. Capercaillie are similar to crested tits (see below) in having higher densities and breeding success in natural forests compared with plantations, but plantations probably contain an important proportion of the total British population. Capercaillie do eat pine foliage and a lot of *Vaccinium* and *Calluna* too (Linden, 1984; Zwickel, 1966), but within Europe they also occur widely in other forest types, as do crested tits, so for both species it is possible that a well developed ericaceous ground vegetation may be more important than the trees.

Cook (1982) estimated that about 40 per cent of the Scottish crested tit population inhabited plantations and the rest were in natural pinewoods. Crested tits were only found in pine plantations which were over 20 years old. This is probably partly because they need large dead stumps in which to excavate nest holes, although it may also be related to food availability. This species has spread this century and Cook (1982) suggested that this increase in range had continued since the last Breeding Bird Atlas was published (Sharrock, 1976). This spread is largely due to the birds moving into the developing pine plantations.

This illustrates an interesting point. Even though pine plantations seem to be less favoured than natural forests in that they hold lower densities of crested tits, the very large area of plantation suggests that before much longer there could be a higher proportion of the British population of crested tits living in plantations than in the more natural forests. Many apparently suitable woods, for example on Deeside, are not inhabited by crested tits and this may simply be because the species is highly sedentary. It is possible that as plantations mature this will enable a further spread of this species both by providing new habitats and by providing links between existing suitable habitats which may be beyond the species' current reach. The effect of forestry on crested tits is difficult to assess but appears to be beneficial.

Many common passerines gain from afforestation, but most are common woodland species in much of Britain, for instance willow warbler, coal tit, goldcrest, chaffinch. Large increases in the numbers of these species in an area, appear at first sight to be of little benefit to an area's avifauna. However, an increasing biomass of songbirds is readily exploited by sparrowhawks (Newton, 1986b). Bibby (1987) also showed that merlin in Wales were taking many woodland birds in the spring, presumably from forest edges or as these birds move between forest blocks over moorland. Hen harriers and tawny owls also feed on birds at times, but at different stages in the growth of a forest (Watson, 1977; Petty, 1987b), while some populations of goshawks have been shown to take many woodpigeon, particularly in the summer (Cooper and Petty, 1988). Therefore, large increases in the overall numbers of birds are likely to have beneficial effects on the numbers of some predatory species.

Conifer forests can be made more attractive for birds in a number of ways (Chapter 7). Bibby *et al.* (1989) suggest that the presence of relatively small areas of broadleaves in a coniferous forest can attract species of birds which are normally associated with broadleaved trees. These include such species as pied flycatcher, redstart, wood warbler and willow tit, which although not nationally rare may be of local interest. Currie and Bamford (1982a) showed that extending normal financial rotations encouraged most of these species, together with others more typical of old growth conditions.

Crossbill and siskin formerly had very restricted breeding ranges in Britain and both have greatly benefited from the scale of new conifer forest. Both species feed on seeds of conifers (Staines *et al.*, 1987), and in good cone years large breeding populations can be found in conifer forests throughout Scotland, in many parts of Wales and in northern England (Parslow, 1973; Sharrock, 1976).

Many birds of prey have benefited from afforestation; partly as a result of increases in both bird and mammal prey, but probably also because they are freed from persecution over very large areas. Upland forest may be one of the few habitats in Britain where raptor populations are regulated more by natural resources than by man, and just as songbird populations change as the forest grows, so do raptor populations (Petty, 1988).

One of the most spectacular additions to these forests is the goshawk. This bird appears to have declined in Britain as a result of deforestation (Chapter 2), and the remaining few were then easily removed by some keepers and egg collectors. They ceased to breed in Britain in the 1880s. Re-introductions since the 1960s by falconers have enabled some populations to become established and to increase dramatically (Marquiss and Newton, 1981; Cooper and Petty, 1988; Petty, 1989). Those that have done best are either in large forests or where persecution is not intense. Unfortunately this is not the case in most of lowland Britain. The future for this bird probably lies in the uplands, where a large amount of apparently suitable but at present unoccupied habitat exists.

Felling and replanting (restocking)

Clear felled areas, although different in many respects from the original moorland or grassland, do provide a temporary treeless habitat within the forest area. At present relatively little of the forest estate has reached this stage (Chapter 6), and few studies have shown how birds react. In this vacuum of information some extravagant claims have been made about the potential value of clear felled areas. These range from the suggestion that they will provide adequate refuges for open country upland species (Garfitt, 1983), to the suggestion that because clear cuts do not contain all stages of the succession and a multi-storeyed structure, they are not desirable on conservation grounds (admittedly this was in broadleaves but the same argument would apply to conifers) (Smart and Andrews, 1985). The truth could be at either extreme, but probably lies somewhere in between, although the real effects remain to be discovered by detailed research.

The potential maximum value of clear felled areas can be surmised by discussing the best of all possible cases; that clear felling provides suitable habitat for all the original upland species. This depends on the length of time that a newly felled area remains open. If we take a value of 10 years, and assume a rotation of 40 years, this would mean that about one quarter of the area would be suitable for moorland species at any one time, assuming an even distribution of age classes. In other words, afforestation would result in around 75 per cent of the land not being available. This seems to put the argument into some perspective. However attractive clear felled areas are to upland species, they will never be able to maintain anything but a small proportion of the original bird communities. It still remains to be seen whether clear felled areas will be attractive to open country birds.

Clear felled areas can be valuable to birds, as illustrated by studies in the Breckland region of East Anglia, where David Lack studied passerine populations on the original plantings. This forest was one of the first planted by the Forestry Commission, and now has large areas of second generation crops. Nightjars and woodlarks now breed on these restocked sites. In fact the majority of Breckland woodlarks are now found on these sites. These facts would probably not have been predicted at the time of planting. However encouraging the events in Breckland are, the species involved and the type of area are very different from those in the uplands.

Leslie (1984) reported the results of 2 years' surveying for nightjars in the North York Moors. In each year nightjars were found on between a third and a half of the new plantations and restocked sites which were surveyed. There was no difference in the occupancy rates between these two habitats. Restocked sites now form the largest habitat for nightjars in Britain (Gribble, 1983).

Two relatively small-scale studies have compared the bird communities found on newly planted and restocked and afforested plots. Leslie (1981) used the mapping technique to record the bird communities on two sites of about 20 ha of 5-year-old Sitka spruce in Harwood Forest, Northumberland. Both the number of species and the densities of birds were higher on the restocked site; however, the plots were on different soil types and this may have influenced the results. Currie and Bamford (1981) made similar comparisons at two sites in Wales. Their sites were both 20 ha with 6 to 9-year-old Sitka spruce, and again a higher density and more species were found on the restocked site. Differences between these two studies were apparent in the species which were recorded on the restocked site. In Wales they were predominantly scrub birds, whereas in Northumberland they were mainly moorland species. These two studies are of sufficient interest to suggest that they should be followed up with much more extensive work.

Bibby *et al.* (1985) carried out an extensive survey using the point-count method, of the birds found on restocked sites in north Wales. These sites had mostly been planted with Sitka spruce between 2 and 11 years earlier. Comparison with newly afforested sites is difficult, partly due to the far greater sample size of Bibby *et al.* (1985) than of any study of afforestation. However, the results certainly make restocked sites seem relatively rich in bird species, which agrees with the findings of Currie and Bamford (1981) and Leslie (1981) who both found the diversity of birds impressive compared with the relatively uniform species found in newly afforested areas (Moss *et al.*, 1979a). Dougall (unpublished) found that only wrens were breeding on a small (5 ha) study plot on a very recently felled site. These studies are the only ones available to assess the bird communities of upland restocked sites.

The species found to be breeding on or using restocked sites include curlew (which was said by Moss *et al.*, 1979b, to be a species which would not be expected to occur on brash covered restocking sites), red grouse, black grouse, siskin, short-eared owl, long-eared owl and hen harrier (Leslie, 1981; Bibby *et al.*, 1985; Petty and Anderson, 1986; Petty, unpublished data); all of which can be regarded as interesting species which enhance the upland bird community. The presence of typical moorland species, the red grouse and curlew, is interesting since it suggests that some moorland species may indeed be able to adapt to restocked areas. However, the small size of the areas sampled and the small number of birds involved means that much more research is needed. In particular, it would be interesting to know the role of the proximity to open moorland in determining the bird communities of restocked areas since it seems probable that restocks would be more likely to be colonised if close to moorland (Chapter 7). Similarly, the occurrence of short-eared owl and hen harrier on restocked sites suggests that their presence in young plantations may not be limited to the first planting with its explosion in vole and bird numbers.

It is sometimes suggested (Newton, 1983) that the explosion in vole numbers, associated with the initial planting may not be repeated at second and subsequent plantings, and so the species which come into young plantations to feed on voles may not be present on restocked sites. Petty (1987b; 1989) working in Kielder Forest, Northumberland, showed that field voles do occur on restocked sites and that their population showed great fluctuation. In one 7-year period there were three peak vole years. Although tawny owls were the most abundant raptor which fed on the voles, short-eared owls nested on restocked sites in the good vole years and appeared to obtain most of their food from these areas (Petty, unpublished data).

Newton (1983) considered that the overall effects of afforestation in this country had so far been beneficial, although he expressed fears about future harmful effects if the level of afforestation dramatically increased. No species of bird has been seriously reduced in numbers through the effects of afforestation though there have been some local declines. Several species have had a large increase in their range as a result of adapting to use forestry plantations.

Threatened moorland birds

The possible harmful effects of afforestation on particular moorland birds have brought conservationists into conflict with forestry interests. It is thought that many moorland species will not adapt to afforestation, and that the increasing areas of forests will reduce both the breeding areas and feeding grounds of some of our rarest birds. Species with a high conservation value, such as the greenshank or merlin, are evocative of the wide expanses of open moorland, rugged scenery, and some of the wildest most beautiful places to be found in Britain. Many interrelated features of forestry shape people's attitudes towards it, but here we consider only the effects of afforestation on moorland birds. This, of course, does not mean that birds are the only considerations, nor necessarily the most important ones.

The breeding species which are most likely to be displaced from an area of afforested moorland are meadow pipit and skylark. These two species are almost certain to be found on any area in the uplands, and many other habitats too (Fuller, 1982). Their local loss cannot be considered to be of any significant conservation importance, particularly since they will be replaced by a greater number of woodland birds. A few other passerine species, including wheatear, ring ousel, wren, whinchat and twite, are likely to be found breeding on open moorlands, and of these only ring ousel and twite can be considered to be almost wholly confined to the uplands. None of these is in danger of disappearing as a British species although some (wheatear, whinchat) have also declined in numbers in southern Britain.

The situation with respect to non-passerines is very different. Here, a considerable number of the species which breed in the uplands are uncommon in Britain, and are unlikely to be found breeding outside the uplands. Examples of these species include golden eagle, red-throated and black-throated divers, raven, greenshank, golden plover, dunlin, merlin, common scoter and wigeon.

Different methods are necessary to study rare or uncommon birds than are used for assessing passerine populations. Ten minute point counts which were used most effectively to monitor populations of passerines in Welsh upland forests would be useless for monitoring raptor and duck populations in Caithness. For these species it is necessary to organise specific studies with which to gauge the effects of afforestation. Some such studies have been completed and will be discussed below.

Raven

We think of the raven as one of the most characteristic birds of the British uplands and we forget that it was once extremely common in lowland Britain. Its demise in the lowlands was related to improvement in hygiene since medieval times, and also to persecution. Its present distribution is closely correlated with the existence of hill ranges and moorland (Sharrock, 1976). In Britain sheep carrion forms a large part of raven diet (Marquiss *et al.,* 1978), so it has been suggested that afforestation might adversely affect raven populations by reducing their food supply. Several studies have investigated the changes in raven population numbers in recent years but their findings are not wholly consistent, and indicate that regional variations in the effects of afforestation might exist.

The first major study of ravens in relation to afforestation was carried out in northern England and southern Scotland (Marquiss *et al.,* 1978). This population had remained stable in numbers from 1946 to the 1960s when it began to decline. In 1974-75 only 55 per cent of the formerly regularly used sites were still occupied. Most of this decline in the raven population was associated both geographically and temporally with afforestation. This coincidence of afforestation and the timing of raven desertion is widely quoted as being excellent evidence for the adverse effects of afforestation on upland birds, so it is well worth examining in some detail.

Marquiss *et al.* (1978) present data from 22 territories where the dates of afforestation and raven desertion were known precisely (Figure 5.3). The years of afforestation and desertion coincide more often than would be expected if desertions occurred randomly through time ($p < 0.001$). Between 1944 and 1953 only one of the 22 raven sites was deserted whereas in the next 10 years five sites were deserted, and in the next 12 years another 16 sites were deserted (including the only three to be deserted in years of non-afforestation). This might suggest that the effect of afforestation became more marked as time went on, so that extra afforestation in a territory was more likely to cause desertion in the study period, than early on. This might be, for example, because as canopy closure occurs, the remaining open moorland areas in a pair's territory become more critical to their continued residence, and so further afforestation would have a bigger effect. Alternatively a third factor, perhaps change in sheep husbandry, might cause the temporal pattern of raven desertions which by chance coincides with the pattern of afforestation.

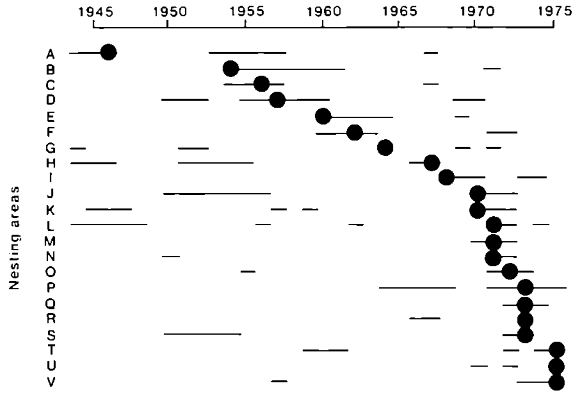


Figure 5.3 The desertion of former nesting areas of raven in relation to afforestation within 3 km of the nest sites. Lines show the years of planting, and filled circles show the last year when the nesting area was still to be occupied (from Marquiss *et al.*, 1978).

Because the data show that there is a decline in raven numbers through time, and a coincident increase in afforestation, this means that the coincidence between desertion and afforestation is less impressive than if there was not a general trend in both sets of data. It is possible that part or all of the link between the two trends is due to spurious correlation but this is unlikely to be true since Marquiss *et al.* (1978) also provide good evidence for a causal link.

Those ravens still nesting showed a decline in productivity with increasing afforestation in their territories, so there seems to be a highly plausible causal connection between raven declines and afforestation. Marquiss *et al.* (1978) suggested that competition with golden eagles, disturbance by climbers and most importantly improvements in sheep management (resulting in less carrion) had also contributed to the decline in the raven numbers. Data from the Lake District, a region where afforestation was practically non-existent, acted as a control area and showed that no comparable decline had taken place. However, these control data were not presented in the paper, so it is not possible to assess the comparability of the two areas and the value of the control.

Mearns (1983) reported that the decline in the area studied by Marquiss *et al.* (1978) had continued but was also mirrored in coastal Galloway where afforestation is much less widespread. The picture is not straightforward. Mearns suggests that a trend towards bringing sheep down to lower ground or into sheds for lambing and overwintering will have been an important contributory factor to the decline in raven numbers. However, the maturation of the forest in already afforested areas may also be important since ravens will feed in the plantations until canopy closure occurs. Also the interpretation of the population trend in the adjacent coastal area of Galloway is not straightforward. This area only acts as a true control if it is known that the two populations are not linked by dispersal of birds from one to the other. If interchange occurs then the decline in one area due to afforestation (perhaps) could be the cause of the decline in the other population.

Newton *et al.* (1982b) studied ravens in Wales and produced results in marked contrast to those found in the Scottish borders. Overall there was no decline in the raven numbers over the period studied. Breeding success of upland ravens was not affected by the amount of forest within 3 km of the nest as they had been in Galloway and Northumberland. This was true whether total forest or the amount of closed canopy forest was used in the analysis, and for a range of territories which had 0–60 per cent of closed forest within either 1 or 3 km of the nest.

In the Welsh study area little afforestation occurred during the course of the study, so it is possible that the ravens had declined during the earlier period of afforestation. This seems unlikely to have been true since the pairs were equally spaced with no apparent gaps between the 'remaining' territories. Raven densities were still very high compared with other upland areas, and there was no apparent effect of amount of forest on the success of the 'remaining' birds, so it is difficult to think of a mechanism by which afforestation should have had an effect, except perhaps by direct disturbance during planting, but this might have been expected to be only temporary.

However, it would be rash to predict the past effects of afforestation without the data from the relevant years. Newton *et al.* (1982b) suggest two reasons why their results on the effects of afforestation on breeding success and occupancy might differ from those of Marquiss *et al.* (1978). First, sheep were more numerous both in (!) and out of the forests in Wales, so that the effects of afforestation may not have been so drastic on the ravens' food supply. Second, that in Wales the trees were younger than in the borders of Scotland and so still allowed some raven foraging.

A factor which was not mentioned, was that the overall amount of afforestation also differed between the two study areas. In Wales the amount of forestry was less than in the Borders. However, allowing for the amount of forestry within a particular territory the occupancy of territories differed between the two areas. For a given amount of forestry within the territory the rate of occupancy was higher in Wales than in the Borders.

These detailed studies of ravens are interesting. We feel that taken together they show that there are occasions when afforestation does not have harmful effects on raven populations, although in other cases it may. It would be instructive to look at these two cases in much more detail to see whether this difference has persisted and to try to understand why afforestation has had such differing effects in the two areas. Newton *et al.* (1982b) suggested some reasons for the differences and it would be interesting to investigate these further. It has to be admitted that if only one of these studies had been carried out then it would have been tempting to assume that the subject was closed, while in retrospect it is possible to see that this would have been incorrect. On the basis of current knowledge it is clear that afforestation can have harmful effects on raven populations but it is not possible to say how often this is the case for the country as a whole. It must also be remembered that ravens are widely distributed in commercially managed Scandinavian conifer forests, so what are the implications for ravens when second generation fellings occur in large even-aged forest areas, such as in Galloway?

Merlin

Many British birds of prey declined dramatically in numbers during the 1960s as a result of poisoning by organochlorines used in agriculture (Newton, 1979a; Chapter 2). The merlin was one of the species affected in this way (Newton, 1973; Newton *et al.,* 1982a). British merlins are in continuing decline and this is certainly not solely due to organochlorines (Bibby, 1986; Newton *et al.,* 1986). Declines have been noted in many areas including Wales (Bibby, 1986), Northumberland (Newton *et al.,* 1986a) and the Peak District (Newton *et al.,* 1981b). Afforestation has been suggested as a possible contributory cause of decline of this species since merlins are usually ground nesters and hunt over open ground (Newton *et al.,* 1978; Watson, 1979).

Bibby (1986) recently investigated site occupancy and breeding success of Welsh merlins in relation to habitat characteristics, including the proximity of coniferous forest. They used discriminant analysis of sites which were either regularly occupied or not recently occupied, and 15 vegetation variables which measured the amounts of five different habitats (heather moorland, bracken, grass moor, conifers and farmland) within three different ranges of distances (0-1, 1-2 and 2-4 km) from merlin nest sites. The analysis confirmed prior expectations that heather moorland was the most important single factor that appears to influence site occupancy. Knowing the amount of heather moorland within 1 km of a nest site allows 79 per cent of sites to be classified correctly as used or not used. Occupied sites contained more heather and bracken and less grass moorland than unoccupied sites. The relationship between the amount of conifer plantation near the nest and occupancy was weak at all distances. Occupied nests tended to have less conifer plantation within 1 km but slightly more between 1 and 4 km than did unoccupied sites. Breeding success was not affected by the amount or proximity of coniferous plantation near the nest.

Bibby (1986) regards the effects of the present level of afforestation in Wales as essentially neutral; in small amounts, forestry may have little and possibly even beneficial effects on merlins. However, extensive areas of afforestation may destroy existing nest sites and feeding areas. It is important to realise that the results of Bibby's analysis refer to the present situation and cannot be extrapolated to predict the effects of future large-scale changes in land use. The study suggested that reclamation of upland areas for farming would be in this area a greater threat to merlins than afforestation.

Newton *et al.* (1978, 1986a) found that merlins will sometimes nest in newly afforested sites but that sites in open land were more regularly used than those in new coniferous plantations. In contrast to Bibby's (1986) study, regularly used sites were not more successful than irregularly used ones. This suggests that merlins would be just as successful nesting in the forests as they are on open moorland, yet for some reason they prefer to nest on the moorland.

Neither of these studies gives much information on the age of the conifer plantations in the study area. This is important since the response of ground-nesting species to large newly afforested sites may be very different from that to the patchwork of stands of trees of different ages which will develop over time. Even if merlins adapt well to newly afforested land, this may merely postpone their disappearance from former moorland areas unless they can also adapt to nesting on or around large clear felled areas, and either hunting over these open areas within the forest or commuting from the forest to the moorland to feed.

Watson (1979) showed that for two pairs of merlins nesting within conifer plantations, most of their prey came from the open moorland, and comprised largely of meadow pipits. Other studies have found that merlins will take goldcrest, coal tit, green woodpecker, young capercaillie, young woodcock, unfledged brambling and redwing (Cramp and Simmons, 1980); all are woodland rather than moorland species. This indicates that merlins can catch prey in forest sites presumably along edges, in very open crops or above the canopy. It is not known how much moorland would be a minimal requirement for a pair of merlins.

The studies of Newton *et al.* (1978; 1986a) and Bibby (1986) differ in their findings regarding the success of tree-nesting pairs. In Northumberland, tree-nesting merlins have high nesting success, perhaps because these nests escape ground predators, whereas in Wales the trend was for heather-nesting pairs to have lower failure rates than those nesting in grass-dominated habitats where most nests are in trees. Tree-nesting merlins use abandoned crow nests and have recently been found to have started using such nests along the edge of coniferous forest in Northumberland (Newton *et al.*, 1986a). This may herald a revival in the fortunes of merlins.

However, in both studies the production of young birds is inadequate to maintain the merlin populations at stable numbers. Newton *et al.* (1978; 1986a) suggest that increased predation rates by ground predators such as foxes, may have been a major cause of the merlin decline in recent years. Change of land use to afforestation may have been one of the factors contributing to the rise in fox numbers in the region by providing cover, reducing the activities of gamekeepers and providing abundant food during the explosion in vole numbers which follows afforestation. However, the role of afforestation is by no means proven and is not suggested by Bibby's data. The decline in merlin numbers may be entirely independent of afforestation in Britain since merlins appear to be declining throughout Britain, even in areas where afforestation is minimal such as The Peak District (Newton *et al.*, 1981b), Orkney (E. Meek, personal communication) and Shetland (P. Ellis, personal communication).

The situation with merlins is rather similar to that of the raven. There is little doubt that afforestation has harmed some merlin nesting areas. However, in many areas there seems to be little evidence for a harmful affect, and in yet other areas merlins are declining for reasons which cannot be connected with afforestation since there are no trees for miles.

Golden eagle

In its British range this species relies to a large extent on the availability of carrion and moorland birds and mammals (Brown and Watson, 1964; Lockie, 1964), so it is likely to suffer from similar reductions in food supply as occur in some raven populations when afforestation takes place. Marquiss *et al.* (1985) describe the fate of a population of up to four pairs of eagles which recolonised south-west Scotland in the early 1940s. In the early 1970s, coincident with large-scale afforestation, two pairs ceased to breed and a third pair rarely produced any fledged young. Studies of diet showed that few prey items were of species associated with forestry. The conclusion drawn is that eagles possibly benefit initially from afforestation due to reduced disturbance from humans but that loss of feeding habitat then affects their breeding success and this can lead to a decline in eagle numbers.

The details given in their paper support their conclusions. The three territories which were occupied during the period from 1945 through to 1963 all had higher nesting success then, despite occasionally being robbed by egg collectors, than they do now. However, the fourth pair colonised its present territory in 1965 when it was already 19 per cent afforested. Despite another 13 per cent of this territory being afforested in the period 1965-73 the success of this pair has been higher in the period 1974-83 than any of the territories which were studied in any of the periods under consideration. Overall, the production of young eagles from the whole study area has been 1.54, 1.47 and 1.20 young per year in the periods 1945-63, 1964-73 and 1974-83 respectively, so there has been a real decline in total productivity.

The different fates of the individual eagle pairs are instructive. Marquiss *et al.* (1985) point out that the worst affected pairs were ones where afforestation had covered very high proportions of the territories (61.4 per cent in territory A, 42.8 per cent in territory B) and that the remaining areas of open country were the tops of the hills, which are probably less productive hunting areas for eagles than the areas which had been afforested within these territories. Supporting this contention is the fact that in territory D (where reproductive success remained high) considerable areas of low elevation, productive ground remained over which the birds could hunt.

Recent research by Watson *et al.* (1987) has attempted to take the eagle story further than was possible with the very small number of pairs studied by Marquiss *et al.* (1985). Watson *et al.* (1987) studied six large areas which were designed to cover the range of available eagle habitats in Scotland. In only one of their study areas (Argyll and Bute) do they consider the effects of afforestation in any detail. Even there little information is given. The Argyll and Bute region is compared with two others which showed similar reductions in sheep numbers since 1965. In these two areas negligible afforestation is said to have occurred and the declines in eagle numbers since 1956-60 have been less (12 and 5 per cent) than in Argyll and Bute (32 per cent decline).

As is unavoidable with species that are rare, the sample sizes are fairly small, particularly when broken down into sub-groups, so that the trend for more eagles to have been lost from Argyll and Bute is not statistically significant. No information is given on the fates of individual eagle pairs in relation to the afforestation of their home ranges as was done by Marquiss *et al.* (1985). Without information of this sort it is quite possible to suggest that some other factor is operating in Argyll and Bute, besides forestry, such as persecution or disturbance which may be the real causes of the decline in eagle numbers. It could also be that the information supplied by amateur observers was either unusually thorough for Argyll and Bute in the late 1950s or unusually incomplete for more recent years.

Watson *et al.* (1987) provide no information on the detailed timing of desertion of eagle territories in the Argyll and Bute area, so it is not even possible to relate the timing of the disappearance of the eagles to the arrival or ageing of the forestry plantations, in the same way that was done for ravens and eagles by Marquiss *et al.* (1978; 1985). Nor do Watson *et al.* (1987) provide detailed information on the individual breeding success of pairs which was demonstrated for ravens and eagles in south-west Scotland, and forms the important causal connection between land use changes and population size. But they do show that the Argyll and Bute area was overall one of the most productive of their nine study areas. Information is not given that would enable a reader to assess whether the interesting suggestions of Marquiss *et al.* (1985), that afforestation affects eagles through depriving them of rich, low hunting areas and forcing them more on to the barren tops, is true throughout Scotland.

A similar decrease in eagle numbers to that observed in mid-Argyll took place in another area (Kincardine and Deeside) but Watson *et al.* (1987) do not suggest that it was caused by afforestation. Rather, it is suggested, this was due to changes in deer management. We find the evidence provided in this report to give inadequate grounds for assessing the effects of afforestation on eagle numbers in Scotland. The report does show that in all areas studied the eagle population had fallen since 1956-60 and so the effects of afforestation are acting in conditions which, at least in recent times, have been unfavourable for eagles. It is possible therefore that afforestation either has, or will, contribute to the decline in eagle numbers by reducing food supplies, but it is not at all possible to be sure of what level of contribution or significance afforestation has had, or will have, to this decline. The results of Watson *et al.* (1987) do little to take the argument over afforestation and eagles beyond the point reached by Marquiss *et al.* (1985) which dealt with just four pairs.

Waders

The British uplands hold important numbers of breeding wading birds. Species such as golden plover are very closely associated with moorland during the breeding season (D. A. Ratcliffe, 1976). Other species of wader which breed in the uplands include dunlin, common sandpiper, redshank, snipe, curlew, woodcock, greenshank and lapwing. Of these only golden plover, dunlin and greenshank might be seriously threatened by very extensive afforestation. However, for the other species, the uplands could become more and more important to their British status, were land drainage in the lowlands to continue. A few scarce or very scarce waders also breed in the uplands, these are dotterel, whimbrel, purple sandpiper, Temminck's stint, wood sandpiper and ruff. Of these dotterel, purple sandpiper and whimbrel occur mostly at altitudes and latitudes beyond tree planting limits.

Until recently there were no published studies that examined the question as to whether afforestation has caused the loss of important numbers of breeding waders. Langslow (1983) states that, "many moorland areas within Caithness and Sutherland which formerly supported greenshank, dunlin and golden plover have also been afforested", suggesting that there is scope for such studies. Langslow states also that, "Preliminary evidence for three sites surveyed in Caithness in 1980 (before planting) and in 1983 (after planting) shows the complete loss of breeding waders for the afforested ground. Planting of sites led to a loss of species richness and abundance in adjacent areas; for example, curlew and golden plover declined at all three sites". There is no indication of the numbers of these species found on control areas.

Despite the lack of published data on the actual effects of afforestation on wading birds it is not unreasonable to suppose that they exist. Of the relatively common British species of breeding wader, only the woodcock is commonly associated with mature woodlands or plantations; the other species are all birds of open country. However, it would be surprising if planting immediately made areas unsuitable for waders, so the interesting, and important, question becomes, "for how long can wading birds persist on planted land?"

How reasonable is the assumption that species such as golden plover, dunlin and greenshank will be seriously harmed by afforestation? We take these species in turn and examine the small amount of evidence that relates to these questions.

The dunlin is a breeding species which numbers about 6000 pairs (Sharrock, 1976) in Britain and Ireland. The birds favour small pools and dubh lochans where they may breed semicolonially. Their apparent dependence on pools and wet areas means that they are likely to be affected by land drainage prior to afforestation. We know of no examples of dunlins breeding in plantations or natural forests, so this does seem to be a species that avoids afforested areas.

Golden plovers are typical of upland moorlands in Britain (D. A. Ratcliffe, 1976). The British breeding population is probably about 30 000 birds, of which two thirds occur in Scotland (Sharrock, 1976). In Britain this species is a bird of open country but in Scandinavia it can be found nesting in willow scrub as well as on the open moors (Avery, personal observation). Ahlen (1975, 1976) in Sweden regards this as a species which suffers because of the afforestation of open land, but importantly, also one which benefits from the creation of clear-cut areas. The interaction between forestry and bird communities in Scandinavia is very different from in Britain, mainly because much of the area is naturally forested rather than open land. Afforestation in that situation leads to the felling of plantations, with a consequent loss of some forest birds of high conservation value such as capercaillie, waxwing and black woodpecker (Jarvinen *et al.*, 1977). Golden plover was one of the species added in an area during a period of drainage, felling and afforestation in Finland (Vaisanen and Rauhala, 1983). Ulmanen and Valste (1965) also found golden plovers nesting on clear felled areas in Finland. These observations suggest that golden plovers may be one of the species which may move on to clear felled areas in Britain when these become available, but this may only partially compensate for the loss of original moorland.

The greenshank is said to disappear from afforested land in Britain (Langslow, 1983). However, it is not at all clear whether this will be the general trend. Nethersole-Thompson and Nethersole-Thompson (1979) state that greenshanks sometimes nest in the young conifer plantations in Sutherland and used to nest in forest bogs in Speyside. In their review of the nesting haunts of greenshanks abroad, they show that in many parts of its range this species is found in forest clearings and associated with forest pools. It would be interesting to know the extent to which greenshanks use forestry plantations of different ages in this country and whether particular features are favoured. Greenshanks are found on clear felled areas in Finland (Helle, 1985a; 1985b).

Of the rarer waders none is likely to gain greatly from afforestation but two species, wood sandpiper and whimbrel, have bred in clear felled areas in Scandinavia (Helle, 1985a; 1985b).

Regional differences

General analysis

The species to benefit and to lose from the effects of afforestation will differ from place to place. This obvious fact sometimes appears to be overlooked in popular discussions of the pros and cons of forestry practices, where one would sometimes gain the impression that every spruce tree to be planted will be on top of a greenshank's nest! The general trend in Britain (Chapter 4) is for species richness of woodlands to decrease from south to north, yet the trend for species richness in moorlands is in the opposite direction.

Fuller (1982) lists species breeding on moorland (Table 5.1) and many of these such as golden eagle, twite, red-throated diver, arctic skua, are found in Scotland but not in southern England or Wales. There are no primarily moorland species which show an opposite pattern of being more likely to be found in southern Britain than northern Britain. This means that afforestation in Scotland, particularly the very north of Scotland, is likely to have much greater detrimental effects on upland species of high conservation value than would afforestation in southern British uplands. To some extent, this loss will be offset by colonisation of plantations by woodland and scrub birds. Since Scotland has a relatively impoverished woodland avifauna the gains there in woodland birds may be greater than in southern England. How does the balance sheet of potential gains and losses through afforestation look for different parts of the country?

Assessing the gains and losses which may

Table 5.1 Birds used in the regional afforestation analysis which were classified by Fuller (1982) as being either woodland or upland species

Woodland species used in the regional analysis
Sparrowhawk, kestrel, capercaillie, pheasant, woodpigeon, stock dove, wood pigeon, turtle dove, tawny owl, long-eared owl, nightjar, green woodpecker, great spotted woodpecker, carrion crow, jackdaw, magpie, jay, great tit, blue tit, coal tit, crested tit, willow tit, long-tailed tit, nuthatch, treecreeper, wren, mistle thrush, song thrush, redwing, blackbird, redstart, robin, grasshopper warbler, blackcap, garden warbler, whitethroat, lesser whitethroat, willow warbler, chiffchaff, wood warbler, goldcrest, spotted flycatcher, pied flycatcher, dunnock, tree pipit, starling, greenfinch, siskin, linnet, redpoll, bullfinch, crossbill, chaffinch, yellowhammer, reed bunting.
Woodland species not used in the regional analysis
Grey heron, buzzard, red kite, osprey, hen harrier, hobby, little owl, barn owl, black grouse, golden pheasant, Lady Amherst's pheasant, lesser-spotted woodpecker, collared dove, rook, marsh tit, nightingale, tree sparrow, goldfinch, hawfinch.
Upland species used in the regional analysis
Red-throated diver, black-throated diver, wigeon, common scoter, greylag goose, golden eagle, merlin, peregrine, red grouse, ptarmigan, golden plover, curlew, snipe, redshank, greenshank, dunlin, arctic skua, common gull, blackheaded gull, raven, dipper, ring ousel, wheatear, twite.
Upland species not used in regional analysis
Manx shearwater, grey heron, teal, mallard, eider, hen harrier, buzzard, kestrel, oystercatcher, ringed plover, lapwing, wood sandpiper, great skua, lesser black-backed gull, herring gull, greater black-backed gull, common term, arctic tern, cuckoo, snowy owl, stock dove, barn owl, short-eared owl, nightjar, house martin, skylark, shorelark, meadow pipit, rock pipit, pied wagtail, yellow wagtail, wren, stonechat, blackbird, mistle thrush, jackdaw, carrion/hooded crow, starling, reed bunting, snow bunting, lapland bunting.

result from tree planting and its contingent activities is a very difficult exercise. However, it is one which must be attempted if an informed approach is to be maintained. Here we describe an attempt to identify those species which are likely to lose and gain from upland afforestation. The approach used is kept as objective as possible but must, by its nature, be regarded as highly provisional. We defined two categories of birds, upland and forest species, and then used the published data on their British distributions to assess the effects of afforestation in seven different localities in the British uplands.

For the purpose of this analysis upland bird species are defined as those that occur on Fuller's list and are primarily restricted to this habitat in their breeding distribution in Britain. Species which do breed on moorland but are found widely on farmland (for instance skylark, lapwing, yellow wagtail, reed bunting), or are primarily coastal (Manx shearwater, eider, arctic tern, common tern), or can be regarded as primarily woodland species (wren, stock dove, mistle thrush, blackbird), are excluded from our analysis of upland species. Other species whose responses to afforestation might be said to be ambivalent such as hen harrier, cuckoo, short-eared owl are also excluded. This process is to some extent subjective but inspection of the data suggests that it will not have greatly influenced the results.

The definition of woodland bird species also relied on Fuller's classification. Birds on this list were included in the analysis if there were published records of their occurrence in commercial upland coniferous forests (for source see Tables 4.1 and 4.2) or for a very few species if their exclusion seemed foolish to us even if we could find no records of their occurrence in such forests. This again entails some subjectivity. Seven localities were chosen along a latitudinal gradient. In each locality one single 10 × 10 km square was chosen from the BTO Atlas (Sharrock, 1976) to fulfil the following criteria: the square must contain some land of over 300 m in altitude but none greater than 820 m; it should be adjoined on all four sides by squares which also contained some land of at least 330 m; and the square should contain some moorland habitat. The information was derived from the acetate overlays available with the Atlas and which were compiled by the Biological Records Centre.

These constraints were constructed in order to pick squares that were similar in habitat characteristics, and were of the type of land which is suitable for afforestation. We have no personal knowledge of any of these squares and they can be regarded as an essentially unbiased selection of upland squares. Squares surrounded on all sides by other upland squares were chosen to maximise the chance that the squares contained appreciable amounts of upland habitats in them. The chosen squares should not be regarded as comprising wholly upland, but they can be expected to be typical of the uplands in their own regions. The seven areas where such squares were located were Dartmoor, Exmoor, south Wales, north Wales, south Scotland, mid Scotland and north Scotland. These areas are conveniently on a latitudinal gradient but at very similar longitudes. The lists of woodland and upland bird species were compiled before the atlas squares were chosen and were not altered afterwards.

After the species and atlas squares were chosen the presence or absence of each of the woodland and upland bird species was recorded for each of the squares. In addition, for the woodland birds the presence or absence of the species in the adjacent eight squares (sideways and diagonally in each direction) was recorded. Woodland birds which were not recorded in the focal square, but were present in an adjacent square were regarded as potential colonists for the square if afforestation were to occur.

Our definitions of the two groups of species, upland and woodland, differ in their stringency. Upland bird species were only so classified if they relied on this habitat to a large extent for nesting, whereas a looser definition of woodland bird species was used; the species only had to be known to have probably bred in upland conifer forests to qualify, no reliance on the habitat was required. The differences in the stringencies of the definitions can be seen to be sensible because the groups were used as ones which were potentially threatened (upland bird species) or potentially assisted (woodland bird species) by afforestation. Thus we needed upland species which would be likely to be present only because the square contained upland moor, and therefore would certainly be threatened by afforestation. The woodland species were regarded as potential colonists to the square but would only be so counted if they were not already present there. Thus we needed these to be species which could live in conifer forests if they were available but these did not have to be species only found in that habitat.

The results of this analysis (Tables 5.2 and 5.3) show some interesting effects. For upland bird species there is a clear gradient from north to south of decreasing species richness. The northern-most square in the analysis contained approximately two-and-a-half times more upland species than the southern-most one. If only those species which were not found in all seven of these upland squares are considered then the disparity is even greater. The northern British uplands contain a much larger proportion of our upland bird species than can be found in southern Britain. Another important part of this pattern is that none of the upland species that are found in southern Britain is restricted to that part of the country; the lower species richness of the southern uplands is due to the absence of some northern species which are not replaced by any uniquely southern species.

The pattern for the woodland bird species is to some extent that expected from Fuller's analysis; the northern-most squares are notably species-poor (although the overall latitudinal pattern is not extreme) in breeding species. The more interesting comparison though is that of the number of potential colonising species which is highest in the north too. This result was not expected at the start of this analysis. It might be thought that southern squares would have a much greater species-pool of potential colonisers than northern ones simply because some species are restricted to the south of the country. Our

Table 5.2 Results of the afforestation analysis for upland species

Species present in all sites: curlew, snipe, raven, dipper, ring ousel, wheatear.

Species absent from all sites: common scoter, golden eagle, ptarmigan, arctic skua.

Species present in some sites:

Species	*D*	*E*	*SW*	*NW*	*SS*	*MS*	*NS*
Red-throated diver	–	–	–	–	–	–	*
Black-throated diver	–	–	–	–	–	–	*
Wigeon	–	–	–	–	–	*	*
Greylag goose	–	–	–	–	*	*	*
Peregrine	?	?	–	–	*	*	–
Merlin	*	*	*	*	*	–	*
Red grouse	*	–	–	*	*	*	*
Golden plover	–	–	–	*	*	*	*
Redshank	–	–	–	–	*	–	*
Greenshank	–	–	–	–	–	–	*
Dunlin	–	–	–	–	*	–	*
Common gull	–	–	–	–	*	*	*
Black-headed gull	–	–	–	*	*	*	*
Twite	–	–	–	–	–	–	*
All species	8	7	7	10	15	13	19

D = Dartmoor; *E* = Exmoor; *SW* = south Wales; *NW* = north Wales; *SS* = south Scotland; *MS* = mid Scotland; *NS* = north Scotland; * present; – absent; ? unknown; † present in neighbouring square.

anlysis shows that this is not true; the species pool is very constant across the country except for the extreme north, where it is reduced. This may be in part because many of the woodland bird species which are restricted to the south of the country have not been reported in commercial conifer forests (marsh tit, nightingale, hawfinch, lesser spotted woodpecker) and so were not considered here by us. However, their number is sufficiently low that their inclusion in the analysis would not greatly affect it.

Thus we can compare the species that are likely to gain from afforestation with those likely to lose from it for different parts of the country. Some generalisations can be made. First, in all sites the number of potential colonists is smaller than the number of species that would come under threat from afforestation. This is perhaps surprising since the pool of woodland bird species is so much greater than that of upland bird species and potential colonists were drawn from eight neighbouring squares. An examination of the species lists which we have used and the relevant Atlas maps suggests that this result would be robust to small alterations in methods and assumptions from the ones used here, so we believe the result fairly reflects reality.

Species richness is not the sole criterion for assessing the conservation repercussions of changes in land use. The conservation value of the species concerned should also be taken into account. It is difficult to arrive at an objective method of doing this. Fuller (1982) suggests that three attributes of bird communities should be used in evaluating ornithological sites; population level, species richness and rarity. Population sizes in the different areas are not considered here; that information is not easily available. The national rarity of species can be assessed from the Atlas. We used the number of squares in which breeding may have taken place (confirmed, probable or possible) to rank all British breeding species, and then compared the ranks of the threatened upland species with those of the potential woodland colonists for each of the seven sites. This shows that the potential losers from afforestation are generally species of

Table 5.3 Results of the afforestation analysis for woodland species. See footnote to Table 5.2 for definition of area abbreviations

Species found in all sites: kestrel, woodpigeon, cuckoo, carrion crow, coal tit, wren, mistle thrush, song thrush, blackbird, redstart, robin, stonechat, willow warbler, goldcrest, spotted flycatcher, dunnock, tree pipit, starling, redpoll, chaffinch, yellowhammer, reed bunting.

Species found in no sites: crested tit.

Species found in some sites:

Species	*D*	*E*	*SW*	*NW*	*SS*	*MS*	*NS*
Sparrowhawk	*	*	*	†	*	*	*
Hen harrier	–	–	–	?	*	*	†
Black grouse	–	*	†	*	*	*	†
Capercaillie	–	–	–	–	–	*	–
Pheasant	*	*	*	*	*	*	–
Woodcock	–	†	*	†	*	*	*
Stock dove	*	*	*	*	*	*	†
Turtle dove	†	†	†	–	†	–	–
Tawny owl	*	*	*	*	*	*	†
Long-eared owl	–	*	†	†	*	†	†
Short-eared owl	–	–	†	*	*	*	*
Nightjar	†	†	†	†	*	–	–
Green woodpecker	*	*	*	*	*	*	–
Great spotted woodpecker	*	*	*	*	*	*	†
Jackdaw	*	*	*	*	*	*	†
Magpie	*	*	*	*	*	–	–
Jay	*	*	*	*	*	–	–
Great tit	*	*	*	*	*	*	†
Blue tit	*	*	*	*	*	*	†
Willow tit	†	*	*	†	*	–	–
Long-tailed tit	*	*	*	*	*	*	†
Nuthatch	*	*	*	*	–	–	–
Treecreeper	*	*	*	*	*	*	†
Redwing	–	–	–	–	–	†	*
Stonechat	*	*	†	†	*	*	*
Grasshopper warbler	*	*	*	†	*	*	–
Blackcap	*	*	*	*	*	†	–
Garden warbler	*	*	*	*	*	*	–
Whitethroat	*	*	*	*	*	*	–
Lesser whitethroat	–	*	†	–	–	–	–
Chiffchaff	*	*	*	*	*	*	–
Wood warbler	*	*	*	*	*	*	–
Pied flycatcher	*	*	*	*	*	*	–
Greenfinch	*	*	*	*	*	*	†
Siskin	*	–	†	*	*	*	*
Linnet	*	*	*	*	*	*	†
Bullfinch	*	*	*	*	*	*	–
Crossbill	†	–	–	–	*	†	–
All species present	48	51	48	47	55	49	28
Potential colonists	4	3	8	7	1	4	13

more restricted range than the ones which are likely to gain from afforestation.

This analysis shows two things. First, in all of the areas examined, unless some of the species listed in Table 5.2 can be accommodated in the type of new forests now being approved, the expected losses of bird species due to afforestation will exceed the expected gains. In general more species, and rarer species, could be adversely rather than favourably affected by afforestation. The second major finding is that population size of the key moorland species varies between different parts of the British Isles and generally follows a latitudinal trend of increasing loss towards the north of the country.

Caithness and Sutherland: a detailed example

The foregoing analysis identified northern Scotland as the area where afforestation would potentially do most harm to moorland birds of high conservation value. This is an area where afforestation has taken place and where some is still planned for the future. This has resulted in a considerable number of censuses and surveys of the birds of this region being undertaken although rather few of these have yet been published (Reed *et al.,* 1983; Stroud *et al.,* 1987). Because of the high conservation value of the area and the consequently high level of concern over its future, we consider that some analysis of its avifauna is required here. For this we have identified the areas where afforestation is proceeding or is planned and again used Sharrock (1976) as our source for information on bird use of the areas.

The boundaries of the flow country of Sutherland and Caithness are rather loosely defined and mean different things to different people. The RSPB delimited an area which they consider to be the core of the flow country (Bainbridge *et al.,* 1987), but recently NCC has widened the argument to include the whole of the peatlands of Caithness and Sutherland (Stroud *et al.,* 1987).

Stroud *et al.* (1987) attempt to assess the total breeding populations of three species of wader for the Caithness and Sutherland districts in order to assess the effects of the recent afforestation in the area. The total land area (764 094 ha) was classified by land use. Those areas unsuitable for moorland breeding waders: long-established woodland (12 204 ha); forestry plantations (73 046 ha); agricultural land (104 090 ha); fresh water (25 170 ha); and land too steep or too high for waders (279 484 ha), were excluded from analysis. The densities of dunlin, golden plover and greenshank on each of three land use categories (A, B and C; corresponding to good, medium and poor habitats for waders) were assessed from a sample of half of the sites which NCC had surveyed in the area. No information is presented as to whether these three categories differed significantly in their wader densities although the chances are that at least some of the differences are significant since they are often large, but the sample sizes are often small.

A test was performed to see whether a second person classifying the other half of the sites would produce similar answers. Comparing the observed densities of each species of wader between land use categories recognised by different observers produced no significant differences in wader densities. The power of this test can be questioned on two main grounds. First, unless the first observer's classification is shown to split the sites into three groups which have significantly different wader densities, then the lack of significant differences between observers is meaningless. Second, after splitting the plots, which had been surveyed, in half and then into three land use categories the sample sizes are very small (land use category A, has 25 versus 18; category B, 5 versus 12; and category C, 9 versus 4) so that the power to detect differences between the two classifications is probably low. No information is given on the actual densities of birds in each of the land use categories for each of the observers, so there is no way that it is possible to see how closely matched they were.

The estimates of densities of waders for the different landform categories are then multiplied up by the total representation of those areas for the whole of Caithness and Sutherland, to estimate surviving wader populations. A similar calculation is done for the areas which have been planted. The estimates of percentages of wader numbers which have been lost to afforestation are as follows; 19 per cent golden plover, 17 per cent dunlin and 17 per cent greenshank.

Several points are worth making about these estimates. First, no confidence intervals are given of the estimate. Second, the numbers of sites surveyed by NCC in each landform category differed significantly from the representation of the three landform categories in the whole of Caithness and Sutherland (landform categories A:B:C; in NCC survey sites, 43:17:13; in whole of Caithness and Sutherland, 82 900 ha:99 300 ha:88 000 ha), so that the estimates would have been made more accurate by a different apportionment of survey effort. Little information is given on the relative variability of the wader numbers in the different landform categories, but what there is suggests that a more even division of survey effort should have been followed to produce the best estimates of wader numbers. Third, although the methods used to produce these estimates of the numbers of waders lost to afforestation are probably not ideal, they can be shown to give sensible results by a simple method.

Assuming that the afforestation which has occurred, has all been on land suitable for waders, then the percentage of land lost to forestry can be calculated as 73 046 ÷ (73 046 + 270 200); or 21 per cent. Because some land unsuitable for waders has been planted (Stroud *et al.*, 1987, Figure 1.4) this simple calculation indicates that the NCC estimates are likely to be of the right order of size.

It is possible, using the NCC's data to look at this estimate in more detail. Since the densities of each species of wader on each of the landform categories is known, and the total number of waders lost to afforestation is known, and the amount of each landform category which was lost must be the same for each wader species, it is possible to calculate (by simultaneous equations) the amounts of each landform category lost to afforestation. The figures are 16 450 ha, 24 300 ha and 26 040 ha respectively for A, B and C. These represent 16.6, 19.7 and 22.8 per cent of the initial areas of these landform categories respectively. So although it is true to say that most afforestation (91.6 per cent in fact) has occurred on land suitable for moorland waders, it is not true that within that category there has been a tendency to plant the areas which are best for waders; in fact there has been a tendency for the opposite to occur. We believe that this difference occurs for good silvicultural reasons; the best areas for waders are the worst areas for growing trees.

Stroud *et al.* (1987) describe the bird communities of this area and their international importance. This area has the highest species richness of any moorland bird community in Britain and contains practically all Britain's upland breeding species. Only a few species which breed at higher altitudes such as dotterel and ptarmigan, or at lower altitude such as nightjar and barn owl, are missing from Fuller's list of upland bird species. The area includes significant proportions of the British breeding populations of internationally rare species, although the estimates of the exact proportions of the British populations present in different parts of the area have been subject to change.

For example, it was originally suggested on the basis of NCC surveys that the area regarded as being the flow country by RSPB (about one quarter of the total area of Caithness and Sutherland), may contain 69 per cent of British greenshank (Bainbridge *et al.*, 1987) but Stroud *et al.* (1987) state that the whole of Caithness and Sutherland contain 66 per cent of the British populations. The RSPB report suggests that the area also contains up to 9 per cent of the British population of merlin whereas NCC regard the whole of Caithness and Sutherland as holding 5 per cent of Britain's merlins. Other nationally scarce species such as red-throated and black-throated divers, greylag geese, common scoter, golden eagle, whimbrel, wood sandpiper and raven are also present.

Afforestation

Conservationists often express fears about the effects on moorland birds of 'blanket afforestation' without defining what is meant by the term 'blanket'. Extensive afforestation is here assumed to refer to planting of conifers which gives rise initially to large even-aged stands of trees. Criticisms of this type of afforestation tend to confuse two issues which are better kept separate; the total amount of afforestation and the size of individual areas of forestry.

Any bird species which is adversely affected by afforestation is likely to be worse affected the greater the area of newly afforested land, although we will demonstrate that this is not always the case once a threshold level has been passed. However, this is very different from assuming that if a species is adversely affected by forestry then it will survive better in land where the afforestation is distributed in small forest blocks rather than if the same area of afforestation is found in one single plantation. The effects of total amount of afforestation and patch size of forestry are quite separate issues. Different information is needed to investigate these two effects. On the whole it is likely that it will be easier to investigate the effects of total amount of afforestation than to assess the influence of patch size.

An example of a study which demonstrates the adverse effects of afforestation on an upland species is the study of the raven in southern Scotland and northern England which we dealt with earlier (Marquiss *et al.,* 1978). There the evidence that afforestation reduced the productivity of ravens was convincing. The last sentence of that paper is, "Almost certainly the species will decline further if blanket afforestation continues to expand over former upland sheepwalk". 'Blanket' presumably means forest which forms a continuous covering rather than one which is patchy or discontinuous. This sentence certainly summarises the main result of the paper that extensive afforestation seriously decreases the productivity of raven populations but it could also be taken to mean that continuous afforestation has worse effects on ravens than would patchy afforestation.

This possibility is examined in some detail below. Newton (1983) states, "presumably, with increasing afforestation, more and more areas could come under threat from blanket forestry . . ., any naturalist would accept that a varied landscape containing a mixture of different habitats, will support more varied widlife than a monotonous one. It is again a matter of scale". Moss (1979) suggests that, "in conclusion, the best management of upland forestry for birds appears to be a policy favouring small, even-aged plantations forming a patchwork of blocks of differing ages within any one forest area, rather than large-scale cover by forests of uniform age".

Ravens and afforestation

Marquiss *et al.* (1978) showed that raven populations decline in productivity due to afforestation; the greater the amount of plantations close to the nest site, the lower is the chance that ravens will breed and will produce fledged young. How will the pattern of afforestation affect the productivity of a raven population? To examine this question we can construct a very simple graphical model (Figure 5.4) with which to examine the possibilities. The graph plots imaginary relationships between productivity and the percentage afforestation of an individual territory. In all three cases the relationship is a negative one because we assume that afforestation decreases raven productivity.

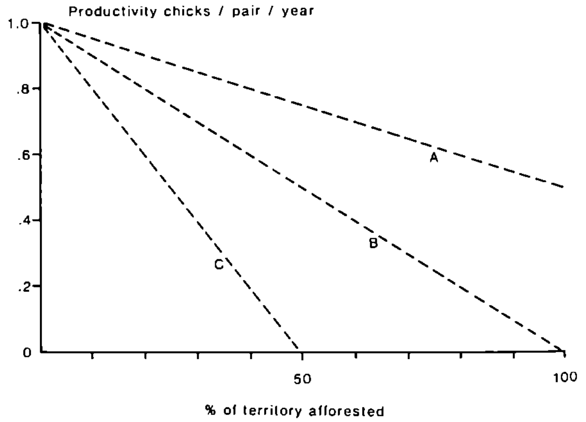


Figure 5.4 Model of theoretical relationships between productivity in ravens and the percentage of individual territories afforested (see text).

Now we can examine the effect of different patterns of afforesting the same overall area by reading off values from the graphs, but first we have to assume some overall level of afforestation. To begin with we assume 50 per cent afforestation. If the situation shown by line A leads to 50 per cent afforestation in all territories, then the productivity of the population will be 0.75 chicks/territory/year. But this is also the productivity if 50 per cent of territories are totally afforested (and produce 0.5 chicks/year) and the other 50 per cent of territories are completely unafforested (and produce 1 chick/year). In this case the pattern of afforestation has no effect on the raven productivity. This is also true for the situation shown by line B, although here the productivity is lower, 0.5 chicks/territory/year.

In case C, however, the pattern changes. If all territories are 50 per cent afforested then none of them produce any chicks. As we move the planting regime to continuous afforestation, the territories with less afforestation become productive and those with more afforestation do not become less productive (they can't since they are already producing no chicks). The net effect is to increase productivity. Maximum productivity of 0.5 chicks/territory/year is reached when half the territories are fully afforested (producing no chicks) and half are unafforested (producing 1 chick each). This effect is easy to understand intuitively; when the effect of afforestation on productivity is very severe (the line relating the two factors has a steep negative slope) then it is better to plant completely areas which are partly afforested, assuming that this frees other areas from being planted.

The example used here is imaginary but it indicates the types of data and analysis that are necessary to gauge the effects of the pattern of afforestation in an area. In particular it suggests that those bird species which are most adversely affected by afforestation will be better conserved, for a given area of afforestation, if some of their range is totally afforested and other parts left totally unafforested, than if afforestation proceeds piecemeal and produces a patchy mosaic of afforestation. This finding is counter-intuitive and is contrary to what most conservationists would have supposed. We should stress that this discussion has been centred around the assumption that the only decision to be made is on the form of forestry, continuous or mosaic, and not on the absolute amount of forestry to occur.

From the data provided by Marquiss *et al.* (1978), it appears that raven productivity has a relationship with afforestation which is similar to that shown by line C, suggesting that this species would be better conserved if planting were concentrated in particular areas, rather than the same amount planted over a wide area.

In any real example the quality of the land to be afforested will be an important consideration in determining whether continuous or mosaic afforestation will be least injurious to particular bird species or the moorland bird community in general. Continuous afforestation of the very best widlife areas and non-planting of less good sites often proves to be a worse option than mosaic planting. However, it will still often be true that continuous afforestation of the worse areas and non-planting in the best areas may be better than mosaic afforestation.

The simple analysis presented above will hold true for homogeneous environments. Its value is that it does show that continuous afforestation may in some cases be the preferred planting strategy for preserving upland bird species that are adversely affected by afforestation.

Unplanted islands

In most commercial forests, areas of land are left unplanted which can be regarded as islands of original habitat within a sea of forestry. This may occur through patterns of land ownership preventing total afforestation, or sometimes because the unplanted areas are not economically plantable. It has been suggested that in the uplands these moorland remnants might act as havens for the original moorland bird community and thus mitigate any deleterious effects of afforestation. If this is the case then these unplanted areas would be important conservation features of forests and their existence should be planned and encouraged in future afforestation. However, it may be that moorland species will not nest in small enclosed moorland fragments or that they require remnants of at least a minimum size. Only one study has investigated such problems so we will discuss it in detail here.

Rankin and Taylor (no date) investigated the birds to be found in areas of unplanted ground within afforested areas in northern England and southern Scotland. Twenty-nine sites ranging in size from 8 ha to 850 ha were censused. The number of breeding species on these sites ranged from one to 20, of which between one and 17 species were regarded as being typical upland species. The variation in breeding numbers was significantly correlated with the area of the plot; big plots contained more breeding species than small plots. Using a multiple regression analysis Rankin and Taylor concluded that both area and habitat diversity of the plot affected species richness. The species lists for the different plots indicated that some species were present on only the biggest plots, whereas others were present on nearly all plots. From this, Rankin and Taylor conclude that their results show that some species (for example merlin, ring ousel) will not breed on small sites. They go further to suggest that a representative upland avifauna would not be maintained on a large number of small sites. This conclusion is unwarranted for the following reasons.

In showing that small sites contain few breeding species, it is not possible to demonstrate that the absent species will not use such small sites, without a very large sample of such small sites. Because they are rare, one needs to examine a very large number of small sites before one can be sure that merlins will not use small sites (otherwise all that is shown is that they do not use all, or most, small sites which is hardly surprising since they are so rare). In fact the incidence of occurrence of species on the moorland islands was highly correlated ($r = 0.688$, $p < 0.05$) with their abundance on two large open moorland sites in the area. Thus the species/area relationship is highly likely to be at least partly due to the effects of taking different sized samples from a common species pool, and cannot convincingly be used to recommend the minimum of necessary moorland areas which should be left in afforested areas. The only convincing recommendation that could be made from these data would be that there is no evidence that small patches of moorland are much better than large ones in supporting moorland birds, so there is no need, as a result of this work, to plan to include such patches within plantations as a way of significantly assisting upland bird conservation.

Rankin and Taylor's smallest site was one hundredth the size of their biggest one. One would therefore need a huge number of them to show that their combined bird community was less rich than the larger area. For example, the largest site was 850 ha and contained nine pairs of curlew; it can be calculated that 45, 8 ha sites, all devoid of curlew, would be necessary to show a significant preference for large sites. In fact even this, strictly speaking, would not be sufficient since the expected value for curlews on the island area would not reach the threshold that would allow a χ^2 test to be used; a much larger number of large sites would also be necessary. Curlew is the third commonest species on the large site, so an even larger sample would be needed to demonstrate avoidance of the small sites by the rarer species such as golden plover. Even then, the result would be very dependent on that one large site so it would be safer to census several large sites to make sure that a representative sample had been selected.

Further, the result that habitat diversity affects species number independently of area would also contribute to the species/area relationship. If particular species need particular habitats (curlew needs bogs) then rare habitats, and their characteristic species, will be often absent from small sites. But this does not mean that those species would be absent from small sites if their preferred habitat was present.

An alternative way to use the data is to examine sites of the same size which are moorland islands in forestry blocks and those which form parts of large open moors. Rankin and Taylor present data collected by NCC teams on large moors which were not afforested. These sites are similar in size to the largest moorland patches in Rankin and Taylor's sample, so it is possible to compare their bird lists although this must be done with caution since different personnel were involved in collecting the data, so they may not be directly comparable. However, the two true moorland sites contained eight and 10 upland species respectively. Taking just the richer of these two sites, the figure of 10 upland species is exceeded or equalled by 11 of the 'island' sites, all but three of which are smaller than the true moorland plots. This aspect of their data is not discussed by Rankin and Taylor, but at face value suggests that equal areas of moorland island and open moorland are at least equally valuable in the number of species they support, and that the remnants may actually be richer than the open moorland. Examination of the data does not suggest that the species found on the moorland islands are of lower conservation value than those found on the open moorland.

The comparability of large and small sites, and moorland remnant and open moorland sites, can also be questioned. It is not necessarily true that small unplanted areas are left unplanted for the same reasons as are large unplanted sites. For example, small sites may be left within forests because they are not economically plantable, whereas larger areas may be left unplanted because the land was not available for purchase. Such factors may be correlated with bird species richness and could bias the results. It is not clear in which direction such biases would act. Separate from the issue of whether comparing large and small areas is fairly comparing like with like, is the question of whether the unplanted areas in this study are representative of the unplanted areas that are left in other forests. Assuming that similar factors govern which areas of which size are left unplanted in different parts of the country then these results will be applicable to those areas.

Clearly, the conclusion that "a representative avifauna would not be maintained on a large number of small sites . . ." is of sufficiently doubtful validity that it would be rash to base any management proposals on it. If such a statement could be supported by firm evidence then it would be a reason for advocating continuous afforestation rather than mosaic afforestation, particularly in areas where the proportion of land coverage by afforestation was to be large, since this would maximise the sizes of individual areas of unplanted land.

Edge effects

Forests may have effects on birds which extend beyond the forest boundaries. Langslow (1983) suggests that tree planting usually leads to the cessation of burning of the adjacent moorland (because of the risk of forest fires). This will affect both the height and the species composition of the vegetation and may reduce its suitability for species such as waders. Drainage of an area in preparation for planting may also reduce the water table of surrounding areas and affect species of waders and waterfowl. Coniferous trees may also have effects on the water runoff from upland areas. Any enhanced acidification of streams may lead to decreases in the streams' productivity of invertebrates with consequent effects on bird species (Smith, 1980; Harriman and Morrison, 1980; Ormerod *et al.*, 1985).

Stroud and Reed (1986) suggested that although the vegetation differed at different distances from forestry plantations in Sutherland and Caithness, it was still true that taking this effect into account the numbers of waders were smaller close to the forest than far away. It has been pointed out that their analysis contains both arithmetic errors and an analytical procedure which tends to bias the results in the direction of the difference which they claimed their results show. These data are being reanalysed.

As a result of the findings of Stroud and Reed (1986), the Forestry Commission commissioned research from RSPB to investigate this subject further. In that work 42 sites, each adjacent to 4 km of plantation in Sutherland and Caithness were surveyed on eight transect lines at different distances from the forest edge (100 m, 300 m, 500 m out to 1500 m) on each of 2 days. These data suggest that although there are fewer golden plover and dunlin in the area immediately adjacent to the edges of the oldest forests, this is solely due to differences in vegetation; there is no effect of proximity to the forest which acts over and above this. It remains an open question whether or not part or all of these vegetation changes have been caused by the presence of trees or whether they existed before the forests were planted.

Other edge effects may include increased predation on species nesting near to the forest edge, by predators which find nest sites or shelter within the forest. Likely predators of this type are foxes and carrion crows. A large scale experiment has been undertaken in Sutherland to attempt to investigate this subject. Grids of chicken eggs were place next to forestry plantations and the rate of disappearance of the eggs was monitored. Although more eggs disappeared near to the forest edge than far from it, this difference was explicable in terms of vegetation differences rather than as a real effect of distance.

Those edge effects so far mentioned have been ones where harmful effects of the forest may extend beyond its boundaries. There may also be favourable edge effects for birds. These do not appear to have been studied but might be expected to consist of slightly higher densities of some species at the forest edge where the forest provides nest sites for species which feed in the open. Such species might include thrushes and some of the warblers and flycatchers.

Concluding remarks

Both foresters and conservationists are used to thinking in the long term; trees take a long time to grow and no bird community is static. It is therefore appropriate to consider the long-term effects of afforestation in the British uplands.

General

A common view put forward by conservationists, is that afforestation leads to the replacement of scarce moorland birds, with many widely distributed and common woodland birds. While this does happen, it can be considered an unbalanced comparison. For example, it would be equally naive of foresters to argue that goshawk, capercaillie and crested tit replace common moorland birds such as meadow pipit and skylark. A better approach is to compare either the common birds in forests or moorland habitat or the scarce birds. Forestry does lead to a dramatic change in the bird populations, but it also brings in many rare and scarce species. In the end it may come down to a subjective judgement of the value of a goshawk against golden eagle, black grouse against red grouse or meadow pipit against chaffinch.

Are there any long-term trends in British bird numbers which might suggest that the losses of moorland species would occur even without the added pressures of afforestation, or any suggestions that these losses might be to some extent offset by the acquisition of new breeding species? If so, then the long-term effects of forestry will only be accelerating a long-term trend. If such a trend did exist it would not necessarily mean that the objections to upland afforestation would disappear but they should be considered against the long term picture.

Pressures on the uplands are likely to increase, of which forestry is only one. Bibby (1986), for example regarded the change of managed grouse moor to improved pasture to be a greater present threat to the merlin in Wales than afforestation. There is evidence that raven numbers in Galloway are continuing to decline even in areas which are not afforested (Mearns, 1983) and it is not clear that all of the decline in afforested areas can be allotted to the effects of forestry. At the very least the effects of afforestation might be lessened if other pressures were removed. It is not clear – and never is – to what extent current agricultural practices will remain constant in the next say 40 years, the period of time after which trees planted today will reach felling age. Forty years ago no one would have foreseen the drastic effects of pesticides which have come and almost gone in recent years, nor would they have predicted the effects of the Common Agricultural Policy on British agriculture.

Moorland birds face threats from factors other than afforestation although afforestation is perceived widely as the main threat. The loss of heather moorland to agricultural improvement or to overgrazing has reduced the areas of good habitat for birds very significantly in some areas. Some of this is due to overgrazing by sheep. However, dead sheep form important food reserves for many moorland birds. The consequences for carrion feeders such as eagles of a change in agricultural support schemes in the uplands might be very serious; even if other species of birds benefited from such changes.

Chapter 6

The current forest resource and its management

Introduction

Chapter 2 described the decline of native forests up to the formation of the Forestry Commission in 1919. This Chapter looks at the growth and management of forestry since 1919, provides a background to working practices in upland forestry, and serves as an introduction to the next Chapter where we describe how forests may be improved for birds.

The forest resource

A difficulty which we have generally ignored in this review has been defining what is upland and what is lowland. The most recent forest inventory data available are the Forestry Commission's Census of Woodland and Trees 1979-82, which provides a comprehensive account up to 1980. The data are presented separately for each of the Forestry Commission Conservancies (old style Conservancies prior to a reorganisation of the Forestry Commission in 1984), for each country (England, Scotland and Wales), and for Great Britain as a whole. These inventory data cannot be readily split into lowland and upland areas so we have compared different conservancies or countries to highlight differences between predominantly lowland or predominantly upland areas.

Distribution and area of forest

The distribution of woodland in Great Britain is shown in Figure 6.1. In the lowlands, woodland areas are predominantly small and widely dispersed, apart from a few exceptions such as the Forest of Dean, New Forest and Thetford Forest, while in the uplands of northern England, Wales and Scotland, forests are larger. The most extensive forests occur in the England/Scotland border area, in Galloway in south-west Scotland,

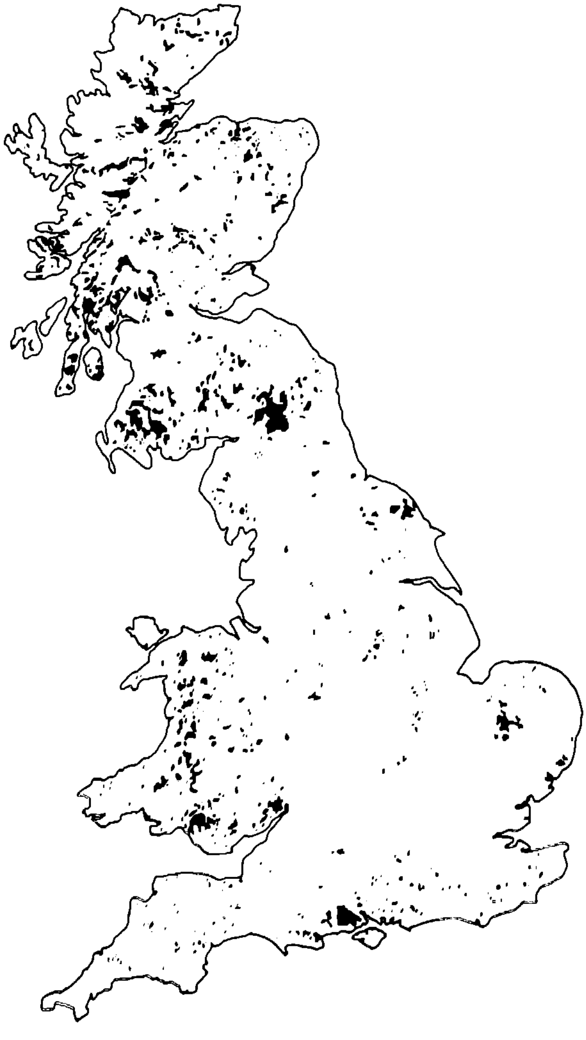


Figure 6.1 The distribution of forest in the British Isles.

and in parts of west and north Scotland.

Woodlands of over 0.25 ha cover 2.1 million ha and comprise 9.4 per cent of the surface area of Great Britain. There are differences between the three countries. Forests in Scotland cover 12.6 per cent (0.9 million ha) of its area, Wales 11.6 per cent (0.2 million ha) and England 7.3 per cent (0.9 million ha). There are also marked differences in the proportion of afforested land within the conservancies in each country, with South Scotland, North Wales and North-East England being the most heavily forested (Figure 6.2). In 1980, high forest (see definition in Table 6.1) amounted to 1.9 million ha or 89 per cent of the total forest area of Great Britain. The remaining 11 per cent comprised of coppice, scrub and cleared ground, which are not considered further in the sections below.

Tree species and age distribution

Comparing England, Wales and Scotland, there are differences in the composition of high forest (Table 6.1). There is a decrease in the amount of broadleaves from the south to the north (Figure 6.3). The ratio (ha) of broadleaves to conifers is 1:0.9 in England, 1:2.8 in Wales and 1:10.0 in Scotland. In England a wide range of high forest species are used, with oak being most widespread (Table 6.3). In comparison, Wales and Scotland have a large proportion of the forest area planted with Sitka spruce. Wales has more Douglas fir and larch in contrast to Scotland where Scots and lodgepole pine have been more widely used. These differences become more obvious when the five most important trees in each country are ranked according to the area they cover (Table 6.2).

Table 6.1 Area of high forest* by tree species and country (data from Forestry Commission, 1983a, b and c and 1984)

Species	*England*	*Wales*	*Scotland*	*Great Britain*
Scots pine	91 074	5 592	144 371	241 037
Corsican pine	40 212	3 693	3 346	47 251
Lodgepole pine	15 249	7 895	103 924	127 068
Sitka spruce	75 599	85 701	364 601	525 901
Norway spruce	43 499	18 641	54 707	116 847
European larch	21 862	2 595	15 957	40 414
Japanese/hybrid larch	35 742	23 461	52 146	113 349
Douglas fir	25 063	10 708	11 628	47 399
Other conifers	17 442	7 906	6 312	31 660
Mixed conifers	21 665	1 744	8 641	32 050
Total conifers	387 407	167 936	765 633	1 320 976
Oak	129 352	26 087	16 551	171 990
Beech	57 828	5 612	10 496	73 936
Sycamore	36 204	3 833	9 389	49 426
Ash	56 092	9 387	4 102	69 581
Birch	45 901	5 583	16 647	68 131
Poplar	12 757	504	329	13 590
Sweet chestnut	9 451	412	8	9 871
Elm	5 545	358	3 611	9 514
Other broadleaves	20 294	3 718	5 115	29 127
Mixed broadleaves	50 914	3 851	10 319	65 084
Total broadleaves	424 338	59 345	76 567	560 250
Total	811 745	227 281	842 200	1 881 226

*High forest = stands of trees having a canopy density of 20 per cent or more, or, in the case of young stands that have not closed canopy, occupying 20 per cent or more of the ground at normal spacing. More than half of the crop should eventually be capable of producing 3 m timber lengths of good form and be of merchantable species (Forestry Commission, 1984).

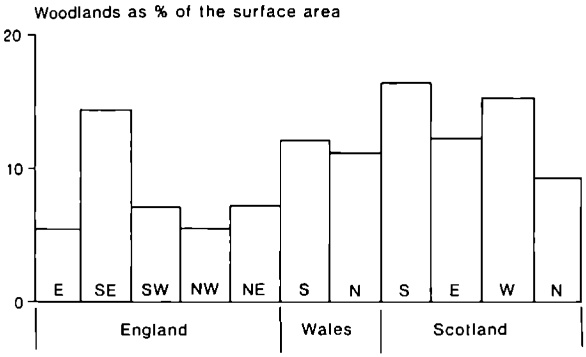


Figure 6.2 The proportion of the surface area of each Forestry Commission Conservancy (old style) which is covered by woodland (adapted from Forestry Commission, 1983a, b and c).

Table 6.2 Five principal high forest species ranked according to the area they occupy in each country

Rank	*England*	*Wales*	*Scotland*
1	Oak	Sitka spruce	Sitka spruce
2	Scots pine	Oak	Scots pine
3	Sitka spruce	Jap./hybrid larch	Lodgepole pine
4	Beech	Norway spruce	Norway spruce
5	Birch	Douglas fir	Jap./hybrid larch

Table 6.3 Area of high forest by planting year classes and country (data from Forestry Commission, 1983a, b and c; 1984)

Planting year classes	*England*	*Wales*	*Scotland*	*Great Britain*
1971-80	69 415	28 093	250 369	347 877
1961-70	133 888	49 769	217 608	401 265
1951-60	151 457	58 771	166 438	376 666
1941-50	98 088	32 720	61 378	192 186
1931-40	74 054	19 458	43 490	137 002
1921-30	69 586	9 160	31 568	110 314
1911-20	31 622	7 767	9 246	48 634
1901-10	37 315	6 063	13 302	56 680
1861-1900	99 542	10 665	31 147	141 381
Pre-1861	46 778	4 815	17 627	69 220
Total	811 745	227 281	842 200	1 881 226

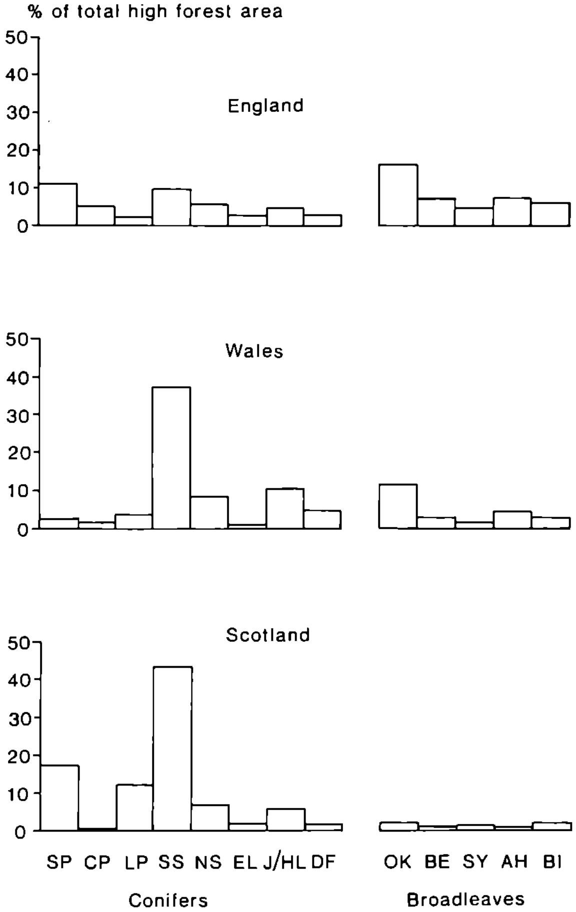


Figure 6.3 The main high forest species in England, Wales and Scotland (data from Forestry Commission, 1983a, b and c).

The age class distribution of high forest in Great Britain reflects the increasing area afforested in most decades since the formation of the Forestry Commission in 1919 (Table 6.3). In England and Wales planting peaked in the 1950s and then declined. In contrast, planting in Scotland has continued to increase in every decade up to 1980 (Figure 6.4).

Another trend has been the change in the importance of different species through time demonstrated by ranking the 13 most important species in Scotland by planting-year classes (Table 6.4). The use of Sitka spruce, lodgepole pine and Japanese/hybrid larch has increased considerably, with a corresponding decrease in the use of Scots pine and many broadleaved species.

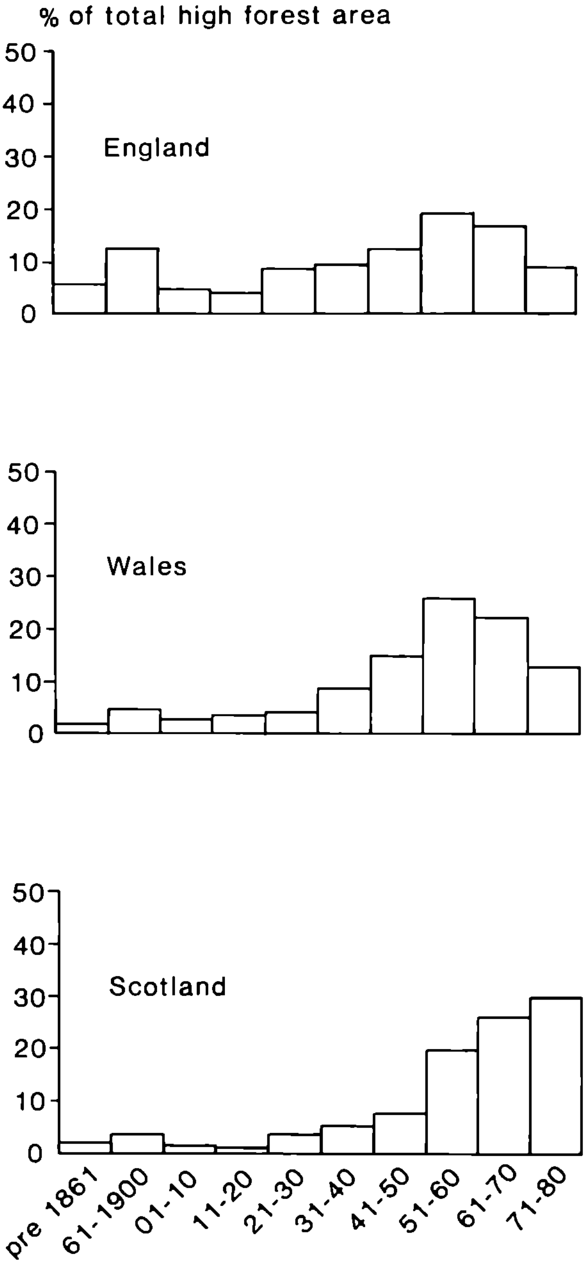


Figure 6.4 The age class distribution of high forest in England, Wales and Scotland (data from Forestry Commission, 1983a, b and c).

Ownership

Ownership of high forest is split evenly between the Forestry Commission (46.5 per cent) and private ownership (53.5 per cent) and there are differences in the distribution of high forest types between owners and countries (Table 6.5). In England, the Forestry Commission owns around 30 per cent of all high forest, whereas in Scotland and Wales it owns just under 60 per cent. Most of this is coniferous, with little broadleaved high forest – the proportion of which declines between England, Wales and Scotland. In contrast, the proportion of high forest in private ownership is mainly broadleaved in England (68 per cent) and Wales (58 per cent) but mainly coniferous in Scotland (80 per cent).

Forest management

We have approached this by looking at the various operations and decisions that have to be taken throughout the life of a commercial forest. Hibberd (1986) and Savill and Evans (1986) provide comprehensive reviews of forestry practice, much of which is relevant to the uplands and Low (1987) provides a good account of Sitka spruce silviculture in Scotland. The management of broadleaved woodland is covered by Evans (1984), but this relates mainly to lowland conditions, whereas Low (1986) looks specifically at the selection and establishment of broadleaves in upland forests.

Site classification and species choice

Site classification is essential to enable the correct species to be planted and the right cultural techniques to be employed. The most important factors influencing tree performance are soil type and exposure. Soils in upland forests fall into three categories; mineral soils with well aerated subsoil such as brown earths, podzols and ironpans; mineral soils with poorly aerated subsoil such as peaty gleys, surface-water and ground-water gleys; and peatlands which can be divided into flushed and unflushed peats (Pyatt, 1970; 1982). Busby (1974) and Busby and Grayson (1981), used these soil types together with altitudinal zones to determine cultural prescriptions, species choice and anticipated yield of forest sites in upland Britain. Hibberd (1986) also used Pyatt's classification to

Table 6.4 Principal high forest species in Scotland ranked according to the area planted in each year class (data from Forestry Commission, 1983a, b and c; 1984)

Species	*Pre-1861*	*1861-1900*	*1901-1910*	*1911-1920*	*1921-1930*	*1931-1940*	*1941-1950*	*1951-1960*	*1961-1970*	*1971-1980*
Scots pine	2	1	1	1	1	2	2	2	3	4
Corsican pine	12	12	12	13	10	11	10	9	10	12
Lodgepole pine	12	13	13	12	13	8	8	4	2	2
Sitka spruce	8	10	11	9	2	1	1	1	1	1
Norway spruce	9	9	7	4	3	3	3	5	4	5
European larch	6	5	3	3	6	4	6	7	7	7
Japanese/hybrid larch	11	11	10	7	5	5	5	3	5	3
Douglas fir	7	8	9	8	7	10	9	6	6	6
Oak	1	2	2	6	11	6	13	10	13	13
Beech	3	3	6	10	12	12	11	12	12	10
Sycamore	4	4	4	5	9	9	7	11	9	8
Ash	5	6	8	11	8	13	12	13	11	11
Birch	10	7	5	2	4	7	4	8	8	9

Table 6.5 High forest type and ownership by area in thousands of hectares (data Forestry Commission, 1983a, b and c; 1984)

Ownership	*Forest type*	*England*		*Wales*		*Scotland*		*Great Britain*	
		Area	*%*	*Area*	*%*	*Area*	*%*	*Area*	*%*
Forestry Commission	Coniferous high forest	204.4	25.2	129.9	57.1	485.0	57.6	819.3	43.6
	Broadleaved high forest	44.0	5.4	6.1	2.7	4.0	0.5	54.1	2.9
Private Woodlands	Coniferous high forest	178.1	21.9	38.1	16.8	281.4	33.4	497.5	26.4
	Broadleaved high forest	385.2	47.5	53.2	23.4	71.8	8.5	510.3	27.1
Total	All high forest	811.7	100.0	227.3	100.0	842.2	100.0	1881.2	100.0

produce a simplified guide to soils and species choice (Table 6.6).

The general trend in upland forestry has been to plant fewer rather than more species. In particular, Sitka spruce has been favoured because, it will grow on a wide range of soil types, it is relatively easy to establish, and it yields a high volume of usable timber much sought after by industry. Other conifer species are used for landscaping or conservation reasons; on the better soils such as brown earths a higher yield of timber may be achieved with species like Douglas fir, or on the less fertile peats lodgepole pine can be grown pure or in mixture with Sitka spruce, while on some of the drier east coast sites Scots pine is used. Low (1985) compares the advantages and disadvantages of alternative conifer species.

Broadleaves are not a commercial alternative to conifers on most upland sites, mainly because they yield far less timber and are more expensive and difficult to establish. Broadleaves are planted in the uplands largely for conservation and landscaping purposes. The introduction of the new Broadleaved Woodland Policy in 1985 (Forestry Commission 1985a; 1985b; Low 1986) has ensured the expansion of broadleaved planting and it is expected they will ultimately form no less than 5 per cent of the total woodland area in upland Britain, and substantially more in the lowlands with the incentives now being offered to establish farm woodlands (Hibberd, 1988; Insley, 1988).

Fencing

In the uplands deer, rabbits and domestic stock, such as sheep, cattle and goats can damage trees.

Table 6.6 Soil types and choice of commercial species in upland forests (adapted from Hibberd, 1986)

Soil	*Soil codes (Pyatt, 1982)*	*Conifer species*		*Broadleaved species (sheltered site soils)*	*Remarks*
		Sheltered	*Exposed*		
Brown earths	1	Douglas fir Larches	Sitka spruce	Beech	Most tree species will grow well on this site type
Podzols	3	Scots pine Larches Sitka spruce	Sitka spruce Lodgepole pine	Birch	Sitka spruce may suffer heather check unless planted in mixture with pines and larches
Ironpan soils	4	Sitka spruce	Sitka spruce	Birch	Mixtures of spruce and pine or larch may be used
Gleys and peaty soils	5, 6 & 7	Sitka spruce Norway spruce (where non-peaty)	Sitka spruce Norway spruce (where non-peaty)	Birch Oak Alder (where non-peaty)	
Peatlands, flushed basin and blanket bogs	8 & 9	Sitka spruce	Sitka spruce	Birch	Birch mainly for amenity and conservation
Peatlands, unflushed bogs	10 & 11	Sitka spruce Lodgepole pine	Sitka spruce Lodgepole pine		Mixtures of Sitka spruce with pines or larches may be used

When areas are planted for the first time, fencing will usually be required to prevent excessive damage to newly planted trees from grazing mammals (domestic and/or wild) when these are present on the adjacent land.

Fencing is expensive, so the specification should be the minimum required to protect the crop. In many areas all that is required is a stock fence to exclude sheep and cattle. However, in northern Scotland, where high numbers of red deer are present on the open hill, a more expensive type of fence may need to be erected. Pepper and Tee (1986) give details of a range of fence designs using spring steel wire. At present these designs are considered to be the most cost effective for forest fencing.

In northern England and Scotland, despite fencing, deer (roe, red and/or sika depending on the area) invariably become established in the forest during the first rotation. Deer may cause severe damage at the time of restocking. In some upland forests, restocked sites are not fenced to exclude deer, and here the choice of tree species is limited to those that will tolerate browsing and bark stripping. Sitka spruce is by far the most resilient of the commercially grown conifers and birch and alder among the broadleaves. In parts of Northumberland and north-east Scotland, restocked sites are fenced against deer. This allows a wider range of species to be planted and it also enables broadleaved herbs and trees to regenerate successfully.

Ground preparation

Trees grow poorly, if at all, when they are planted directly into many upland soils. This is a result of adverse competition from the ground vegetation and waterlogged soils or a combination of both. To overcome these obstacles many of the older forests in the uplands were established by planting trees on upturned turves, which had been cut and spread by hand, from a network of drains which removed the surface water. Turves provided both a vegetation free and well drained site, which resulted in the quick, successful establishment of conifers.

This practice was superseded by ploughing which became the main type of ground cultivation and drainage for a wide range of upland sites. Thompson (1978 and 1984) describes the types of plough available and the techniques for ploughing the different soil types are discussed by Pyatt (1982). Complete ploughing is rarely practised for forestry, instead furrows are produced at the desired spacing between tree rows. Furrows provide a well-drained, weed-free site in which to establish trees. Depending on the soil type, ploughing can also assist in soil mixing and if a tine is used, impeded layers such as ironpans can be broken to increase rooting depth. Ploughing is a cultivation technique which then needs a drainage system to remove excess water. The design of such a system is important to prevent excessive runoff and erosion (Thompson, 1979).

Ploughing and draining are the most important operations which determine the layout of the forest for the foreseeable future. It is therefore essential to ensure that sites to be ploughed, and those to be left unploughed, are clearly marked before the plough arrives on the site. Many of the improvements suggested in Chapter 7 depend on detailed planning being implemented on the site at this stage.

Planting

The trees used for planting are usually 2-4 years of age. They are raised in nurseries. For the first 1-2 years they are grown in a seedbed, after which they will be transplanted into rows for a further 1-2 years. The plants usually arrive on the site to be planted as bare-rooted stock, and these are planted on the side or top of the upturned furrow. Over the years there has been a tendency to plant trees further apart. However, this now seems to have stabilised at around 2 m × 2 m (2500 trees/ha) spacing for spruces and 2.2 m × 2.2 m (2000 trees/ha) for pines, larches and Douglas fir. Wider spacing may deleteriously affect timber quality by producing heavy persistent branching which causes knots in sawn timber.

Restocking produces different problems from those at afforestation (Low, 1985). Recently felled sites contain a lot of branches and tops from the previous crop (brash) and little ground vegetation. Some sites may be waterlogged. The presence of stumps and brash make ploughing both expensive and difficult and therefore it is seldom done except on ironpan soils which were not adequately cultivated at the time of afforestation. The machines that extract timber from a felling site usually spread and break up the brash enough to allow trees to be planted. Sturdier transplants are usually preferred for restocked compared to newly planted sites. In the absence of ploughing, trees are usually planted shallowly and against stumps on wet sites, and more deeply on dry sites. On both afforested and restocked sites the replacement of dead/missing trees (beating up) may be necessary in the years following planting to ensure that an adequate stocking density is achieved.

Fertilising

Peats and podzols are usually deficient in phosphorus (Binns *et al.*, 1980) and young crops on these soils are fertilised with rock phosphate (McIntosh, 1984) using a helicopter, fixed-wing aircraft or by hand. Applications may be made before or after planting and more than one application may be necessary on some sites. By the time canopy closure occurs most crops are capable of re-cycling nutrients and further fertiliser treatments are not usually required (Miller and Miller, 1987). On the poorer nitrogen deficient upland soils, mixing pine or larch with spruce has a similar effect to fertilising pure spruce crops (Miller and Miller, 1987). Occasionally some of these poorer upland sites may also be deficient in potassium. Crops growing on richer brown earths and gleyed soils rarely need fertiliser treatment.

Weeding and cleaning

Vegetation which is competing with the growth or survival of the crop may require removal by hand, mechanical or chemical methods. Weeding is the control of weeds in the first few years after planting. Cleaning occurs at a later stage, usually before canopy closure, and often entails the removal of a proportion of broadleaved trees such as birch and willows. Both operations are more often required on richer lowland rather than upland sites.

On upland sites, where trees are planted on plough furrows, the trees usually outgrow competing weeds well before the furrows are colonised by vegetation. Only on some of the richer mineral soil sites will weeding be necessary. There is a wide range of techniques and materials available (Sale *et al.*, 1986). Herbicide weeding is more often required on restocked sites, particularly when there is a time lag between felling and planting, allowed vegetation to become established. When weeding is in the uplands, it is often the control of grasses and bracken rather than broadleaved plants. On some heather sites, the growth of spruce may virtually cease. This may be overcome by fertilising the crop, or by planting spruce/pine mixture on sites where this problem can be anticipated.

The most frequent form of cleaning practised in upland forests, is where natural regeneration of birch and other broadleaves has occurred. Sitka spruce together with most of the important north-west American conifers often produce good seed crops which can result in prolific, dense regeneration. When this occurs it is necessary to reduce the stocking to the normal density for the species. This is an expensive operation and should be done as early as possible.

Brashing and pruning

The removal of the lower branches (brashing) up to about 1.8 m, to provide access into thicket stage or older crops is seldom done. The operation is expensive and unnecessary, particularly in crops grown on no-thinning regimes (see later). Brashing is only likely to be undertaken in crops at a lower elevation where there are environmental considerations, or in areas to be selectively thinned.

Pruning is the progressive removal of branches up to 5-6 m high carried out in the early life of a crop with the aim of producing knot free timber. It appears only to be justified when dealing with high quality broadleaves and conifers, and is rarely practised in the uplands.

Timber production

The production of timber involves forecasting how much timber will be produced and harvesting the crop.

Forecasting

It is important to be able to predict future timber yields, both for the development of timber-using industries and to plan for the optimum labour and machinery requirements within forests. We shall briefly describe the system used within the Forestry Commission.

Forecasting timber yield is based upon a knowledge of the area occupied by different species of trees and their rates of growth. This is based on 1:10 000 scale stock maps. The important unit of land is the sub-compartment which is an undefined area containing trees of a single species of the same age and growth rate: sub-compartments are grouped into compartments for administrative regions. Stock maps show these compartment and sub-compartment boundaries and the species and planting year within each sub-division. These maps are regularly updated to account for felling, replanting, changes in growth rate and windblow, and provide the source for a computerised inventory database.

To forecast future production it is necessary to predict timber yield from this growing stock. This is achieved through the yield class system (Edwards and Christie, 1981). The cumulative volume of a stand divided by its age is called the Mean Annual Increment (MAI). During the growth of a stand, MAI increases to a maximum and then declines. The maximum MAI indicates the yield class of the stand, so that a crop with a maximum MAI of 12 m^3/ha is equivalent to yield class 12 (Figure 6.5). The top height and age of a stand can be used to predict its general yield class. Forecasts of timber yield can be made by using yield models which are available for most commercial species (Edwards and Christie, 1981). These models incorporate planting distance, yield class and thinning/no-thinning options.

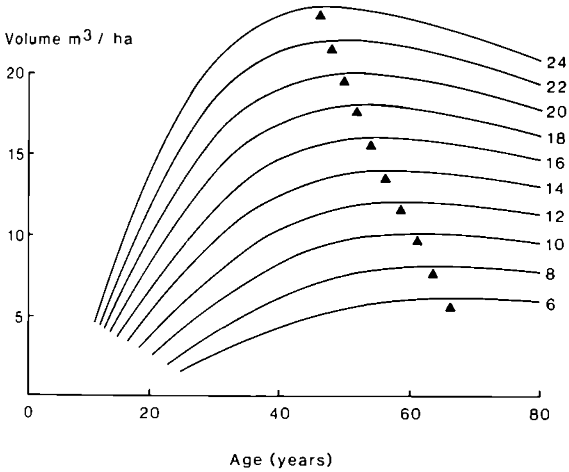


Figure 6.5 Mean annual volume increment curves for Sitka spruce (from Edwards and Christie, 1981)

Harvesting

Trees can be harvested either part-way through a rotation as thinnings, or at the end of a rotation by clear felling.

Thinning

Thinning removes a proportion of the standing trees and gives the remainder more space. Thinning does not increase the total timber yield from a stand but it does increase the proportion of the more valuable larger diameter trees. In upland areas, thinning is only practical on the more sheltered, stable and lower sites. The majority of high elevation crops will be grown as unthinned stands as thinning can rarely be done at a profit and it makes the standing crop more liable to windthrow (see later).

When thinning is practised, various options are available and these are described in detail by Busby and Grayson (1981) and Rollinson (1985). There are two main types of thinning, these are selective and systematic. The various methods of selective thinning are: low thinning which removes only the sub-dominant trees from the crop; crown thinning which removes trees from the crown so that the remaining dominant trees are better spaced; or combinations of both of these methods. Racks (removal of one row of trees) are provided so that the felled trees can be removed from the thinned crop. Racks are usually as straight as possible and their width and siting depends on the type of extraction equipment that is to be used.

Systematic thinning involves removing timber in a pre-determined pattern, such as line thinning, where for instance every third or fourth row may be removed (Hamilton, 1980). This can sometimes be combined with a selective thinning in between the removed rows. The advantages of line thinning is that it allows good access into the crop to remove the felled timber, and it is usually more cost effective than are selective thinning methods. The main disadvantage is that in many upland areas it can often make the crops more susceptible to windthrow.

Where thinning occurs it is usually undertaken on a 4-6 year cycle. However, this period may increase as the crop ages. Determining the interval between thinnings, is usually a compromise between profitability which favours larger intervals and more timber removed at each, and crop stability which favours more regular thinnings which remove less timber.

Clear felling

In most upland forests a rotation is terminated by clear felling. Felling systems that have a less dramatic ecological effect are usually unsuitable for most upland sites, as they greatly increase the chance of windthrow and are more costly to operate. However, in sheltered and windfirm areas, methods of selection felling may be used where it is desirable to retain a permanent forest cover (Helliwell, 1982). Such areas may include extended rotation areas and woodlands of recreation or conservation value such as native pinewoods. Selection fellings remove small groups or even single trees. Trees are then planted or natural regeneration is encouraged in the gaps provided. Where deer are present, the young may need protecting by fences or guards.

Before clear felling it is essential to plan how the harvesting operation is to be undertaken. The way in which felled timber will be extracted from a site governs how the crop is felled. Within the Forestry Commission a terrain classification system determines the type of extraction machinery to be used (Rowan, 1977). This method uses three factors: ground conditions based on soil type which reflect the load bearing capacity of the ground; ground roughness indicates the presence of obstacles which may limit movement; and slope. Terrain classification is also used to determine suitable forest road density, which relates to the type of extraction equipment.

There are three main types of timber extraction machinery used in forests. These are skidders, forwarders and cable cranes. **Skidders** range from agricultural type tractors to purpose built machines. They pull loads of felled trees from the felling site to the nearest road, either by the tree tip or butt with the other end dragging along the ground. A **forwarder** is a tractor with a trailer and loading crane; they can range from converted agricultural trailers to highly efficient purpose built Scandinavian machines. The crane is used to load timber lengths into the trailer at the felling site and to unload it at the roadside. Forwarders are being used more frequently because they are better suited to many of the working conditions present in upland forests, such as a low density of roads and small tree size, they are also the cheapest method of extracting timber. They can work on most of the terrain classes where skidders have been used in the past. **Cable cranes** extract timber by cables, using a tractor-powered winch which is positioned on a forest road. This is the most expensive type of timber extraction but it is the only method suitable for steep terrain.

All these extraction methods are dependent on the crop being felled in a suitable manner (Hibberd, 1986). Nowadays, most trees are felled and snedded (branches removed) with a chain saw. The felled trees can be extracted to roadside where they are crosscut into the final product, this method is better suited for skidder extraction. Trees can also be crosscut where they are felled and then extracted and stacked at roadside, which is better suited for skidder and cable crane extraction. Recently even more mechanised systems have been introduced; these use processors which sned and crosscut trees that have been felled by hand, or harvesters which both fell, sned and crosscut trees. The use of both types of machine is likely to increase in the future.

Hazards to forests in the uplands

There are a number of natural hazards which may occasionally produce forest health problems or influence the type of silviculture that is practised in upland forests. These are mainly concerned with wind, insect damage, disease and animals (see under fencing). Such factors are not just characteristic of man-made forest but can also cause catastrophic damage to natural boreal forests (Ford, 1982; Peterken, 1987).

Windthrow

Windthrow is a serious problem in upland forests and occurs when trees are blown over by strong winds due to the root plate becoming detached from the soil. Two types of windblow are recognised, these are catastrophic and endemic windblow. The former can occur on virtually any site after particularly severe gales, as occurred in west and central Scotland in January 1968 (Holtam, 1971) and in south-east England in October 1987. It is not possible to predict when and where catastrophic windblow will occur. Endemic windthrow occurs after canopy closure and typically starts with a few trees on a wet exposure site during normal winter gales. These initial windthrow holes may then extend progressively until they affect large areas. Endemic windthrow is influenced by site conditions and silvicultural practices and it is possible to predict and manage for its occurrence.

Miller (1985) describes a method of windthrow hazard classification that scores four factors – regional wind zones, elevation, topography and soil types – to produce critical top heights for stands at the onset of windblow (Table 6.7). Thinning results in windthrow commencing at an early stage compared with unthinned stands. In the least stable hazard classes, 5 and 6, it is recommended that crops are not thinned. Crops in classes 3 and 4 may be thinned with care, while stands in classes 1 and 2 would be routinely thinned.

Insect pests

A large proportion of upland forests are young and have been created on previously unafforested sites. These two factors have enabled most afforestation to occur with few problems from insect pests. As our forests grow larger and older, with second and subsequent rotations being established, then the number of potential insect pests is likely to increase (Hibberd, 1986; Stoakley, 1986).

Upland forests have relatively few insect pests, and the correct choice of tree species and attention to details of forest hygiene go a long way towards preventing outbreaks of some secondary insect pests. However, these measures will not prevent the occasional problem from primary insect pests which are likely to attack apparently healthy trees, and may cause considerable damage. In many instances, insect damage will reach a peak and the population will then collapse naturally. Only rarely is it necessary to use an overall insecticide treatment, both because it may be difficult to predict when outbreaks will occur, and when they do, they rarely cause tree death. The only insecticides used are those which have been fully cleared through the Pesticides Safety Precautions Scheme (PSPS).

Stoakley (1986) reviews the management strategy for the insects of real forest importance in the uplands. Here we aim to highlight just some of the entomological problems that can occur at different stages in the growth of a commercial forest.

A widespread problem with newly planted conifers on restocked but not afforested sites, is root and bark damage from two beetles, the large pine weevil (*Hylobius abietis*) and the black pine beetle (*Hylastes ater*), both of which breed in the stumps of a former conifer crop (Scott and King, 1974). The large adults of the former species can remove the bark from just above the root collar which often results in girdling and death of the young tree, while the smaller adults of the latter species burrow beneath the bark, below the root collar. It is routine practice in most areas to dip conifers in gamma HCH prior to planting on restocked sites, and this usually affords them protection from these beetles in the vulnerable period following planting.

Later on in the life of a conifer crop a number of insects may occasionally cause serious defoliation by feeding on the sap of needles or the needles themselves. The main problems occur with aphids and the caterpillars of moths and sawflies. One of the most frequent defoliators of Sitka spruce is the green spruce aphid (*Elatobium abietinum*) (Carter, 1972). Although severe attacks may cause considerable loss of increment, insecticide treatment has never been used because it is difficult to predict outbreaks and treatment would be unlikely to be cost effective, even if it was ecologically acceptable.

Young pines may occasionally be defoliated by pine sawflies. For one species, *Neodiprion sertifer,* a naturally occurring nuclear polyhedrosis virus has been prepared and marketed as an effective control measure.

Recently, a widely reported case of defoliation and tree death has been by the pine beauty moth (*Panolis flammea*) on lodgepole pine, primarily in northern Scotland. This moth occurs naturally on Scots pine, and it was not until 1976 and 1977 that serious attacks were first noted on lodgepole pine in Sutherland (Stoakley, 1979). A satisfactory control of this species in the worst affected areas has been achieved by aerial applications of the organophosphorus insecticide fenitrothion. This insecticide has a low toxicity

Table 6.7 Critical heights for different windthrow hazard classes (from Miller, 1985)

Windthrow hazard class	*Critical height (m)*		
	Unthinned stands	*Selectively thinned stands*	*Systematically thinned stands*
1	28.0	25.0	25.0
2	25.0	22.0	22.0
3	22.0	19.0	18.0
4	19.0	16.0	14.5
5	16.0	13.0	11.5
6	13.0	10.0	9.0

to vertebrates (Crick, 1986).

Serious problems can occur occasionally from numerous bark and wood feeding beetles, of which we shall mention just two. The pine shoot beetle (*Tomicus piniperda*) causes damage to pine trees as the young adults feed by boring up the centre of shoots. These shoots may then break off in windy conditions. Quite often this leads to the loss of the terminal shoot which results in a distortion of the main stem (Bevan, 1962). The beetle lays its eggs between the bark and sapwood of recently dead pine trees, where the larvae feed. The main way of reducing damage from this beetle is to ensure that pine trees are quickly removed from the site after felling. This prevents the beetles completing their life cycle.

The great spruce bark beetle (*Dendroctonus micans*) is a major forest pest throughout Europe and was discovered in Wales in 1982 where it is now well established. It can attack and kill all species of spruce by producing extensive tunnels under the bark of mature and apparently healthy spruce trees. Recently a biological control programme has been initiated using a predatory beetle *Rhizophagus grandis*. At present the great spruce bark beetle has not spread into the extensive spruce forests of the English/Scottish border region or Scotland.

Diseases

Diseases of trees can be caused by non-living agents, such as chemicals, climate or adverse site conditions, and by pathogens such as fungi, bacteria or viruses. Quite often two or more factors combine to produce the disease symptoms (Peace, 1962; Phillips and Burdekin, 1982).

Much concern recently has centred around whether British forests are affected by low levels of air pollution, to the same extent that has been reported in Europe. It has been accepted for many years that trees may suffer and can die in the vicinity of severe industrial pollution. A comprehensive survey undertaken annually by the Forestry Commission since 1984 has shown no signs of the type of damage occurring in West Germany (Binns *et al.*, 1985 and 1986). This type of survey provides a valuable baseline against which any future change can be measured. The British Isles have a highly variable climate, and this combined with a high proportion of exotic conifers in our forests can sometimes lead to widespread climatic damage occurring; particularly after unusual weather conditions, such as summer droughts, prolonged east winds and low temperatures in the winter or unseasonal effects like late spring frosts.

Amongst the living agents that can cause diseases, fungi are the most important in upland conifer forests. However, the incidence of fungal disease is low when compared with the total area of forest. Many fungal problems can be overcome or reduced by good silvicultural practices. We shall briefly mention one of the most important diseases.

Root and butt rot caused by *Heterobasidion annosum* can affect most conifer species (Forestry Commission, 1970). Afforested sites are usually free of the disease but it may become established as soon as felling commences. Windblown spores settle on cut stumps and then spread via root contact to adjacent trees. It is standard practice to treat cut stumps with urea which prevents the spores becoming established. In pine areas, a spore suspension of a competing fungus *Peniophora gigantea* is used to treat stumps. This species does not attack living trees.

Summary

1. This chapter provides a background to forestry in the British uplands.
2. The forest resource which covers 2.1 million ha is described. Particular attention is given to the 1.7 million ha of high forest and regional differences in tree species, age class and ownership are highlighted.
3. The management of plantations looks first at how sites are classified to enable the correct choice of species to be made. The range of operations involved in the establishment and maintenance of plantations are then described together with the forecasting and operation of harvesting.
4. Finally some of the main hazards of commercial forests are discussed. These include windthrow, insect pests and tree diseases.

Chapter 7

Management of forest bird communities in the uplands

Introduction

The knowledge required to improve forests for birds is often incomplete, but it is necessary to make recommendations based on our present understanding which can be modified in the light of future research. Because of this most of the recommendations in this chapter are preliminary in nature. It will be important not only to try new management practices but also carefully to monitor and report their effects. The following should therefore be regarded as reasoned suggestions but based as yet on what we consider to be insufficient data.

In this chapter various methods are suggested by which the conservation value of a forest plantation can be enhanced for birds. These range from actions to be taken within areas, that are specifically set aside to be managed for conservation benefits, to other measures that can be applied throughout the forest.

Management objectives for particular habitats must be decided after considering the relative conservation value of different options. It is often difficult to make objective comparisons between the value of different species or communities, particularly when a management proposal would have favourable effects on one species but unfavourable effects on another. An example of such a conflict would be if a species or species group were favoured by mosaic planting of trees whereas other species might benefit by amalgamating forest areas to leave the maximum amount of uninterrupted moorland. Wherever such conflicts exist the optimal solution will depend on local factors, for example the density, composition and rarity of the existing community.

In the absence of generally agreed management aims, we suggest the following criteria as a basis for management decisions. In the afforestation phase the main aim should be to minimise harmful effects on the pre-existing moorland bird community. Once a forest is established we suggest the aim should be to produce as diverse a forest bird community as possible. Here we are only concerned with the effects of upland forest management practices on bird communities but, of course, management policies will take account of other issues including other wildlife, amenity value, landscape and economics.

Most opportunities for conserving birds in upland forests occur at two distinct periods; before planting, and before felling, when extensive first generation forests can be restructured. These operations usually deal with large areas at any one time and careful planning beforehand can greatly affect the way in which a forest and its avifauna develop. Decisions taken at these times will affect the character of the forest throughout a rotation, a period which is unlikely to be less than 40 years. During the growth of a forest little can be done to alter forest structure on a large scale, although minor but important modifications can be made. We will discuss the various management options under three main headings: afforestation, restructuring, and standing crop improvements.

Afforestation

Some people argue that any afforestation is harmful to upland birds and should be avoided (Thompson *et al.*, 1988). This ignores two facts. First, that many species use upland forests and some are of high conservation value; second, that many of the British uplands are of low conservation value for birds. The priority, therefore, is to make positive recommendations that will (a) help to improve new upland forests for birds and (b) safeguard the richer moorlands and their associated bird communities.

Land acquisition

The acquisition of land for tree planting can be discussed at various levels from the national to the very local. In Chapter 5 we showed that species richness in moorland bird communities follows a latitudinal gradient, with the northern uplands having many more rare breeding species (see also Table 7.1). In some areas afforestation leads to a net bird conservation loss. This effect would be more pronounced in the Caithness and Sutherland flows or on the best heather moorland, than it is in parts of England or on some of the grass moors of west Scotland and the southern uplands of Scotland.

Studies of raven, red kite and buzzard in Wales (Newton *et al.*, 1981a; Newton *et al.*, 1982b) suggest that the density and productivity of these species might be little affected by afforestation if it is distributed in smaller areas separated by sizeable areas of open land. A similar principle may also apply to merlins in Wales (Bibby, 1986). These examples suggest

Table 7.1 Birds listed on Schedule 1 of the Wildlife and Countryside Act 1981

Part I: Protected by special penalties at all times

Avocet
Bee-eater
Bittern
Bittern, little
Bluethroat
Brambling
Bunting, cirl
Bunting, Lapland
Bunting, snow
Buzzard, honey
Chough
Corncrake
Crake, spotted
Crossbills (all species)
Curlew, stone
Diver, black-throated
Diver, great northern
Diver, red-throated
Dotterel
Duck, long-tailed
Eagle, golden
Eagle, white-tailed
Falcon, gyr
Fieldfare
Firecrest
Garganey
Godwit, black-tailed
Goshawk
Grebe, black-necked
Greenshank
Gull, little
Gull, Mediterranean
Harrier (all species)
Heron, purple
Hobby
Hoopoe
Kingfisher
Kite, red
Merlin
Oriole, golden
Osprey
Owl, barn
Owl, snowy
Peregrine
Petrel, Leach's
Phalarope, red-necked
Plover, Kentish
Plover, little ringed
Quail, common
Redstart, black
Rosefinch, scarlet
Ruff
Sandpiper, green
Sandpiper, purple
Sandpiper, wood
Scaup
Scoter, common
Scoter, velvet
Serin
Shorelark
Shrike, red-backed
Spoonbill
Stilt, black-winged
Stint, Temminck's
Swan, Bewick's
Swan, whooper
Tern, black
Tern, little
Tern, roseate
Tit, bearded
Tit, crested
Treecreeper, short-toed
Warbler, Cetti's
Warbler, Dartford
Warbler, marsh
Warbler, Savi's
Whimbrel
Woodlark
Wryneck

Part II: Protected by special penalties during the close season

Goldeneye
Goose, Greylag (in Outer Hebrides, Caithness, Sutherland and Wester Ross only)
Pintail

that afforestation will increase species diversity, by retaining most moorland species (albeit at reduced densities) and adding a wide range of woodland birds. This is most likely to be true if forestry is concentrated on the lower slopes, with the higher ground left unplanted, although Marquiss *et al.* (1985) points out the value of leaving some unplanted land below 300 m.

This can be better illustrated using a simple model (Figure 7.1) which looks at three birds: the golden eagle which is dependent on moorland; the tawny owl on forest; and the goshawk which nests in the forest but feeds mainly outside. Assuming a sigmoid function for both increasing and decreasing populations, it suggests that the greatest species richness and diversity is likely to occur when around half of any given area is planted.

This hypothetical example neatly exposes two problems in suggesting management advice based on the probable effects of different management options on birds. First, the detailed information is usually not available. Although Figure 7.1 captures the essence of the changes that may occur in some parts of Britain the actual slope of the functions is unknown and may well vary between different parts of the country. Second, even if the suggested relationships were precisely known, there is no accepted way at present, of assessing the conservation values of these changes on species or communities.

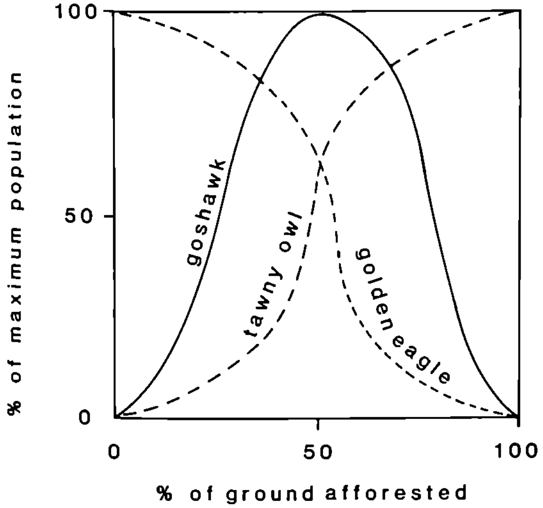


Figure 7.1 Model demonstrating the possible population changes with differing levels of afforestation, in a raptor of moorland (golden eagle), woodland (tawny owl) and one which uses both open and forested habitats (goshawk) (from Petty, 1988).

In this case we believe the arguing could go as follows. The tawny owl is common throughout Europe, including Britain, so we would not seek to maximise their numbers. Goshawks are very rare breeders in the UK, much rarer than golden eagles (Dennis *et al.*, 1984; Spencer *et al.*, 1986) so we would wish to promote their numbers. However, goshawks are relatively common in Europe (Gensbol, 1984; Cramp, 1985) compared with golden eagles, but both species include several rare races which are widespread in the Holarctic region (Figure 1.1) (Brown and Amadon, 1968). How do we strike the right balance? This is not a question that foresters can easily answer, but it is one which needs to be answered in order to provide soundly-based conservation advice. We consider this problem to be an important area which as yet has received only cursory attention and deserves more research (see Chapter 8).

Designing a new forest

Once the decision has been taken to plant an area, then it is essential to achieve the best design for wildlife and landscape. Pre-planting surveys are required, to assess the silvicultural potential of an area and *inter alia* its ecological value for birds. Help can be sought from bodies such as the NCC, the RSPB, the Institute of Terrestrial Ecology (ITE), the Scottish Wildlife Trust (SWT), local naturalist organisations or other experts. Afforestation is a dramatic change in land use so, in addition to the extensive consultation procedures, environmental impact assessments are sometimes needed to decide whether and how an area should be planted (Forestry Commission, 1988a). Prescriptions for an area should aim to enrich the main body of productive forest by sensibly incorporating areas of conservation value or potential. In addition there may be special requirements for individual species or communities.

Conservation areas

All forests should have conservation as a management objective and will often have specific areas where it is the primary one. The importance of conservation areas is that most are permanent, unlike the rest of the productive forest which changes dramatically, particularly at the end of each rotation. Such features provide permanent bird habitat, retain representative sections of the original vegetation (Hill, 1983), and provide a refuge for sedentary animals such as amphibians, reptiles, and small mammals during the closed-canopy stage in the conifer matrix. This pool of species may then be able to colonise adjacent areas when they are felled. There are many other advantages too; forest nesting species, such as tawny owl, or forest edge birds such as black grouse, will use these areas for feeding and they have an important role in the control of deer (Ratcliffe, 1985).

In upland forests, watercourses and roads provide an important interlinking network connecting these permanent features. These can be used to divide the forest into blocks that can be treated independently at the time of felling. We will describe the different types of conservation areas in turn.

Watercourses

Our term 'watercourses' refers to the whole water system, commencing high on the hills in the bogs and pools and descending in streams through the forests, then into rivers, lakes or sea. Watercourses should be thought of as the arterial system of a forest. Lack of regard for these features will lead to a much poorer forest ecosystem (Forestry Commission, 1988b).

Watercourses provide habitat for a unique range of birds, many of whch are solely restricted to these areas. Pools and bogs provide nesting and feeding sites for many species which in the best Scottish areas would include red-throated diver, dunlin, teal, scoter, wigeon, greenshank, snipe, golden plover, redshank, curlew, greylag. In many parts of upland Britain streams and rivers hold common sandpiper, dipper, grey wagtail and heron, while in the north, goldeneye, goosander, red-breasted merganser also occur. In Scotland, larger lochs, particularly those with islands, can hold such scarce species as black-throated diver and osprey.

Planting conifers right up to, or directing drains straight into watercourses, may have damaging effects (Miles, 1986), including reduction in the amount of light reaching the water surface and banks resulting in the loss or reduction of bank and aquatic vegetation, deposition of conifer needle litter directly into the water and excessive silting. These changes can lead to a reduction in the productivity of a stream and as a result fewer fish (Mills, 1980). This will reduce the carrying capacity for some of the birds which are dependent on invertebrates or fish for food (see Ormerod *et al.*, 1985 for dippers).

Watercourses require careful, sensitive and individual management. Broad guidelines should be followed (Mills, 1980; Forestry Commission, 1988b). The aim should be to retain areas of natural vegetation along all waterside areas. Along streams and lake sides it is difficult at the time of planting to envisage what the final stand will look like in 50 years time. Very often unplanted strips that are left along burn sides are adequate in the early growth stages of a conifer crop, but are rapidly shaded out as the trees grow. To avoid this, the distance from the water's edge to the tree crop should be at least equivalent to the anticipated top height of the conifer crop. For example, yield class 12 Sitka spruce, felled at the age of maximum mean annual increment, will range from 23 to 25 m top height (Hamilton and Christie, 1971) (Figure 7.2).

Opportunites should be taken to create areas that include a blend of open spaces intermixed with widely spaced broadleaves (Figure 7.3). In many instances topographical features that are uneconomic to plant, such as steep ground, gorges and boggy areas, can be incorporated into the riparian management areas. Many of these are valuable because of their relatively undisturbed natural vegetation. Ideally, forest drains

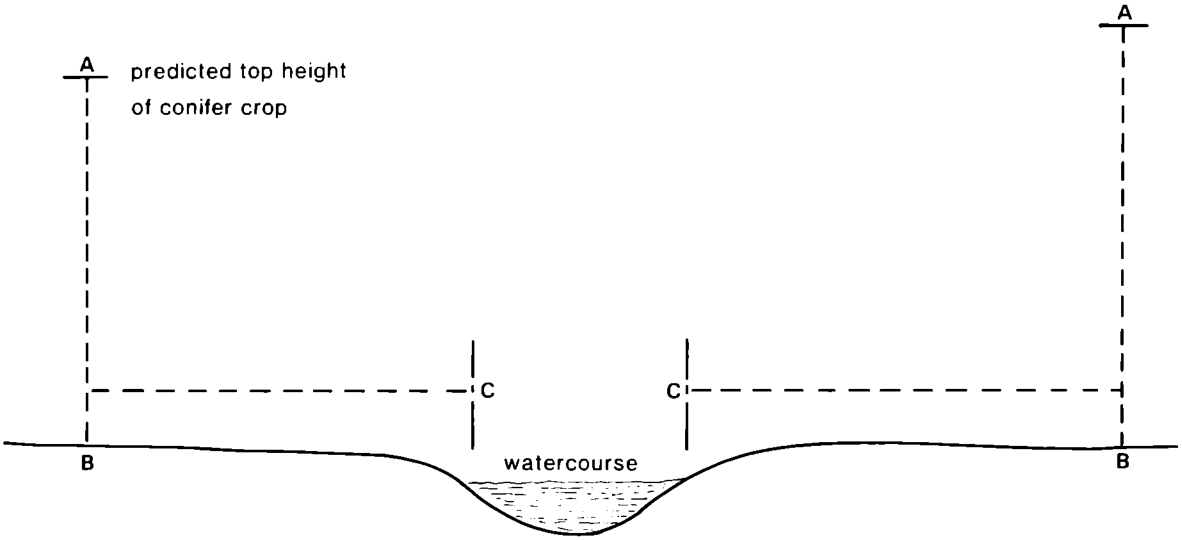


Figure 7.2 Conifer crops should be kept well back from the water's edge. The distance BC should be at least that of AB and may need to be more on steep terrain or north-facing slopes.

should not extend into these areas and if they do they should stop well before the watercourse. Alongside lochans and bogs it may be essential to have more extensive unplanted areas, for example, to allow flightways for divers. Once land is acquired for planting, lochans and small water bodies which are used by divers and waterfowl may experience increased disturbance once land is acquired for planting, from road development, planting operations and increased recreation use. Where such sites are identified as being important for these birds, then management should aim to limit disturbance particularly during the nesting season, either by preventing or discouraging access to the smaller lochans or by zoning access to only part of the larger lakes.

Divers are specially protected during the breeding season and it is an offence to cause disturbance in the vicinity of an occupied nest. Disturbance resulting from fishing is likely to cause the worst conflict. Once divers are put off their nests they are reluctant to return with people in the vicinity and their eggs are then vulnerable to crow and gull predation. In certain instances the provision of well designed artificial islands may overcome or reduce disturbance and predation problems (Merrie, 1979).

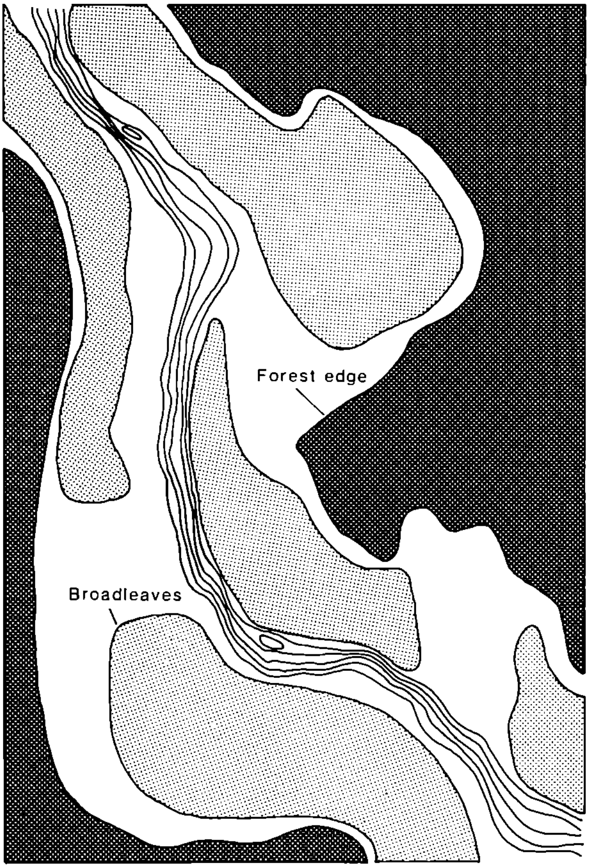


Figure 7.3 Design for streamside treatment (per D. Campbell).

Forest roads, rides and wayleaves

Forest roads are usually sited in ecologically less interesting parts of the forest, particularly when compared with the varied habitats that can occur around watercourses. However, they may still retain remnants of the original vegetation, and in base-poor areas roadside verges may be enriched by the addition of base-rich stone used in the construction of forest roads. Therefore, the trees should be kept well back from roadside verges (Figure 7.4). This is also important for civil engineering reasons so that roads are kept open and dry (Granfield *et al.*, 1980). Road side margins, incorporating areas of unplanted ground and groups of broadleaves, can retain birds associated with earlier growth stages and edges, such as willow warbler and tree pipit, as the adjacent tree crop passes through the less interesting thicket stage. The developing ground vegetation attracts small mammals and their avian predators such as tawny owl, long-eared owl, barn owl, kestrel and buzzard. Where good quality heather and bilberry occur, this provides food and nesting cover for black grouse, and in north-east Scotland for capercaillie. When populations of either species of woodland grouse occur, it will be beneficial to thin heavily young conifers in areas with the best ericaceous vegetation well before canopy closure, with the aim of perpetuating patches of heather and bilberry interspersed with widely spaced conifers.

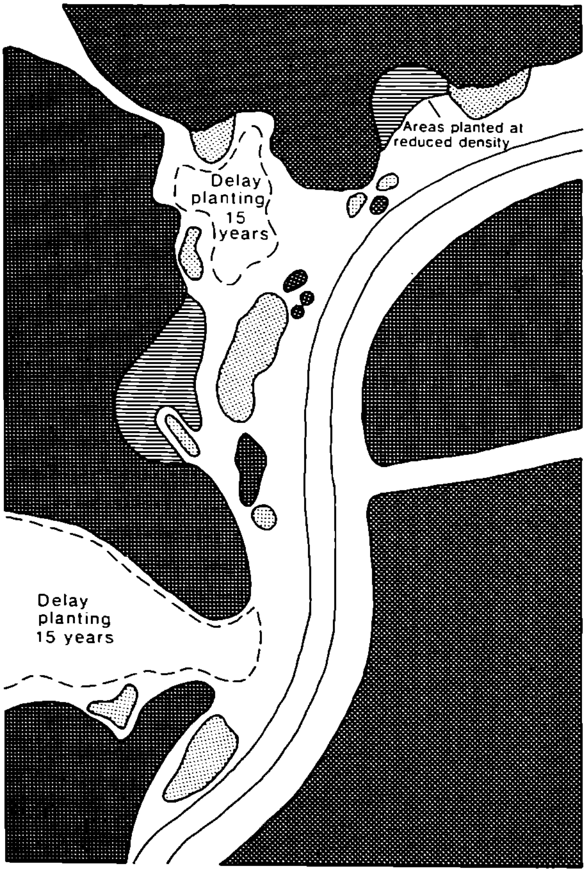


Figure 7.4 Design for roadside treatment (per D. Campbell).

One of the most obvious uses of open spaces within upland forests, has been for **rides,** which have been used to demarcate compartments. There appears to be less reason for the continuing use of rides in upland forests, unlike their lowland counterparts (Carter and Anderson, 1987). Many upland rides have little ecological value apart from being able to retain small linear remnants of former moorland plant communities (Hill, 1983). These often suffer from excessive shading and are unable to retain their own bird communities as the adjacent crops grow. We consider the use of rides should be rationalised. Occasional contour rides are needed as future road or extraction lines, and these can be treated in a similar way to roads. However, we consider that the use of other rides, particularly those running at sharp angles to the contours should be discontinued. Many in this last group have no practical purpose, they add an even greater degree of artificiality to the landscape, and the area devoted to them could be more profitably incorporated into other conservation areas, such as watercourses or roads, which can then be used as management boundaries.

Wayleaves are wide tracts of treeless ground which are left either side of powerlines. Conifer and tall scrub growth are regularly removed by herbicide or mechanical means. This results in many wayleaves developing a grassy, ericaceous or low scrub vegetation. The value of wayleaves for birds has not been quantified in Britain, but because they are usually substantially wider than rides, they should have the potential to attract edge species, small mammals and their avian predators where a grassy vegetation exists, and woodland grouse where the vegetation comprises mainly heather and bilberry.

Existing woodland

When land in the uplands is acquired for planting, it may contain some existing woodland. These may range from ancient woodlands of native trees, to created sites such as shelterbelts which may comprise exotic and native species. Ancient woodlands have the highest conservation value, but all retained woodlands will add to the diversity of birds in an area and usually comprise a small proportion of the area to be planted.

Ancient woodlands in the uplands are usually relics of their former glory. Most have survived, either because they grow on the damper and steeper ground, where it was difficult to clear woodland and to keep the conditions open enough for sheep by repeated burning, or they have in the past been managed for timber production, such as oakwoods for charcoal and pinewoods for timber. In these latter cases, more extensive areas have sometimes survived.

Upland broadleaved woodlands have a characteristic bird communities that are particularly noted for their pied flycatcher, wood warbler, redstart, buzzard and red kite (Wales).

Native pinewoods in northern Scotland, along Speyside and Deeside, provide habitat for a unique group of species including crested tit, capercaillie and Scottish crossbill. The former two birds having spread extensively, albeit at a low density, into plantations particularly of Scots pine adjacent to their original range (Sharrock, 1976; Jones, 1982a, b and c; Cook, 1982). Golden eagles also nest in very open native pinewood sites in north-eastern Scotland (Nethersole-Thompson and Watson, 1974), a habitat that was undoubtedly more frequent when these native woodlands were more extensive.

Small groups of broadleaves or pines, and glades within larger areas, are also important for such attractive and widespread species as redstart, spotted flycatcher and tree pipit, while dead trees are important as a nest site for great spotted woodpecker and tree creeper, and tree holes for blue tit, great tit, pied flycatcher and other hole-nesting birds.

Planted woodland of either native or exotic species, or mixtures, are also important for birds; examples of these are the policy woodlands created around lodges and houses on private estates, and shelterbelts. The larger woodlands are likely to hold a similar range of birds to those present in native woodland of a similar structure in the same locality. Old shelterbelts, in small blocks and strips, are often sited in otherwise tree-less landscapes which are usually extensively grazed by sheep. In these situations they can provide breeding sites for a wide range of tree nesting species such as mistle thrush, heron, rook, crow and some birds of prey.

The crow is widespread throughout the uplands and has the habit of building a new nest most years. Old crow nests are used by owls and falcons, two families of raptors which do not build their own nests. Afforested moorland surrounding old shelterbelts, provide particularly rich feeding areas in the establishment stage for predators of field voles, such as kestrel and long-eared owls, which frequently breed in old crow nests (Village, 1980; 1981; 1983). Merlins may also use crow nests in shelterbelts situated in newly afforested areas or surrounded by sheep-walk (Newton *et al.*, 1978; 1986a). Tawny owls are often present in the larger areas of woodland. Depending on the structure and location of these shelterbelts, they may also be used by other raptors capable of building their own nests such as sparrowhawk, buzzard and very occasionally golden eagle and osprey. Many of these species will continue to nest in shelterbelts long after the surrounding area has been planted. We know of sites in Northumberland where long-eared owls still breed, which are now surrounded by 20-year-old plantations.

Forest design before planting should ensure that, whenever possible, all areas of existing woodland are retained (or enlarged in area – see next section) as permanent woodland sites. These should be managed and regenerated in ways which avoid wholesale clear felling (Helliwell, 1982), to create a diverse structure containing a variety of tree species, tree ages, canopy heights and glade sizes which are so important for enriching song-bird communities (French *et al.*, 1986). Prior to being incorporated into a new planting scheme, many upland woods are badly neglected. The cessation of grazing by domestic stock, coupled with active management including natural regeneration and planting, often safeguards their future.

Planting of the timber crop should be kept well back to prevent shading and death of trees in the retained woodland. The same principle should apply here, as with waterways, that the minimum distance between the planted and retained crop should be equivalent to the anticipated total height of the crop. Wherever possible it would be desirable to connect these retained crops to other permanent features such as waterways or roadsides.

Establishing areas of broadleaved woodland

This section deals mainly with the establishment of permanent areas of broadleaved woodland at the afforestation stage. We envisage these being planted in three types of areas, all are important for birds, these are:

1. planting groups of broadleaves along permanent linear features such as watercourses and roadsides;
2. extending areas of existing woodland; and
3. establishing new areas of broadleaves.

Few studies have directly compared the birds to be found in deciduous and coniferous woodlands in the same area (Chapter 4). However, those that have all give similar results. Broadleaved woods usually contain more species and more individuals than do coniferous woods, and conifer/broadleaf mixtures are even better (Yapp, 1962; Williamson, 1969; Simms, 1971; Adams and Edington, 1973; Adams, 1974; Moss, 1979; French *et al.,* 1986; Peck, 1989). Native species generally appear to have a richer invertebrate fauna than exotic trees (Southwood, 1961; Kennedy and Southwood, 1984). They may also fruit more frequently, although some exotic species also produce very good seed crops. When trees are being planted for conservation reasons, those species which are native to the locality are preferred, even though these may no longer occur naturally in the area. Insects and seeds are important food for birds, and birds are probably more efficient at feeding in tree species with which they have had a long association. While some exotic broadleaves such as *Nothofagus* and *Eucalyptus* can produce good seed crops, birds may not be able to extract their seed efficiently.

The objective for broadleaf planting should be clearly established; in most upland forests timber production is not the main objective, so species should be selected for environmental purposes. Low (1985) describes a range of broadleaved species, including exotics, which may be used in the uplands, together with their site requirements and establishment techniques. From his list the following are likely to be of particular value for birds because of their rich insect fauna and/or their edible fruit or seed and/or for the nest-holes and shelter they provide; sessile oak, alder, downy and silver birch, aspen, ash, wych elm, rowan, holly, bird cherry, gean, hawthorn, hazel and a number of native willows.

In addition to these species, sycamore which is an exotic in Britain, and beech a native of southern England, have both been planted widely in the uplands where they are of considerable value to birds. The large aphid community which is invariably present on sycamore, is a major attraction for *Phylloscopus* warblers (leaf warblers including wood warbler, willow warbler and chiffchaff), while in good mast years beech provides a seed bonanza that is eaten by a wide range of birds, such as woodpigeon, jay, pheasant, chaffinch, brambling and great tit.

Bibby *et al.* (1988) in Wales, counted birds in patches of broadleaves within conifer forests. Twelve species, including eight of conservation interest, were associated with the presence of broadleaves, particularly oak. This relationship indicated, that a one hectare patch of broadleaves, would support about six times more of these species if it was distributed in a 100 small patches (effectively single mature trees) rather than concentrated in one area. The ability of small areas of broadleaves to attract birds that are normally absent from conifer forests, may be peculiar to the southern uplands of Britain and lowland areas, where these discriminating birds are abundant in habitats adjacent to conifer forests. In the northern uplands some of these birds are absent, scarce or patchily distributed. Therefore, the response of birds in the north may be quite different, and broadleaved areas may need to be larger to have a similar effect (French *et al.*, 1986).

Crags

Crags with good ledges provide nest sites for raven and a wide range of birds of prey including golden eagle, peregrine, buzzard and kestrel, while steep heather banks are also used by merlin (Cramp and Simmons, 1980). In some areas the cavities present in rock faces or block scree at the base of crags may be used as nest sites by barn owl and occasionally tawny owl, quite often with peregrine, buzzard or raven nesting on the crag above. All these species tend to use traditional nest sites, many of which have a long history of usage, such as the golden eagle and peregrine nest sites that were known by egg collectors last century and are still in use today.

In any area the use of crags depends on both the availability of alternative sites and the species present. When crags are abundant, such as in parts of the Scottish Highlands, then each species may have numerous alternatives within its home range or territory, leading to a low proportion of crags being occupied in any one year. Where crags are scarce such as in Northumberland or parts of mid-Wales, then they are used repeatedly. There may also be competition for the better sites with the larger, earlier nesting birds of prey displacing the smaller, later nesting raptors. For instance, golden eagle will displace peregrine, and peregrine will displace kestrel or merlin.

These traditional crag nest sites can continue to be successfully used after planting, providing a few simple precautions are taken. It is important not to restrict flight paths by planting trees too close. It is easy to misjudge this at planting, just as with waterways when it may be difficult to visualise what the area will look like when a mature crop develops. Therefore, crags should be left unplanted on the downhill side, so that at the end of the rotation they are still open. The proximity of planting to crags may vary depending on the adjacent land form, with small crags needing relatively more unplanted ground compared to large ones. Quite often, crags are located close to watercourses where they can be easily incorporated into the unplanted ground in these areas, sometimes with little or no extension.

Unplanted areas

Forests incorporate large areas which are considered unplantable for numerous reasons; this ground plays an important part in diversifying bird communities in all upland forests. Statistics for Forestry Commission land in 1985 show that 246 140 ha or over 20 per cent of the total land holding is in this category (Table 7.2), with the proportion increasing towards the north of Britain (Figure 7.5). To place this in perspective, the total area of National Nature Reserves (NNRs) administered by the NCC amounts to 150 470 ha (Nature Conservancy Council, 1985). While the Forestry Commission's non-forest area will generally be of lower ecological value than NNRs, it may have significant interest as it includes many SSSIs, of which some 350 are on Forestry Commission ground (Forestry Commission, 1985c). Most of this area falls into two distinct categories; agricultural land, grazings or forest worker holdings (42 per cent); and unplantable and miscellaneous (58 per cent). We shall discuss the value to birds of the lower agricultural land within forests, and the higher grazed or ungrazed areas usually above the tree line.

The low agricultural land within, or adjacent to forests, provides not only habitat for open country birds but also for birds which nest in the forest but forage outside. Much of the agricultural land owned by the Forestry Commission in the uplands consists of unimproved moorland, pastures and hay meadows with frequent wet *Juncus* patches persisting on the poorly drained parts. These areas are used by a number of waders such as curlew, lapwing, snipe and redshank, and by passerines such as skylark, meadow pipit, wheatear and occasionally yellow wagtail. Post-breeding flocks of starling, and migrating redwing and fieldfare, also feed on these pastures, and black grouse leks are often

Table 7.2 Forestry Commission land use in ha (% of total area) at March 1985 (adapted from Forestry Commission, 1985c)

Land category	*England*	*Wales*	*Scotland*	*Great Britain*
Forest land (including plantations, other woodland and land to be planted)	247 011 (85.8)	137 337 (91.0)	550 486 (74.2)	934 834 (79.2)
Agricultural and grazing	10 571 (3.7)	6 981 (4.6)	82 962 (11.2)	100 514 (8.5)
Forest worker holdings	417 (0.1)	233 (0.2)	2 609 (0.3)	3 259 (0.3)
Unplantable and miscellaneous	30 081 (10.4)	6 350 (10.4)	105 936 (14.3)	142 367 (12.0)
Total other land	41 069 (14.2)	13 564 (9.0)	191 507 (25.8)	246 140 (20.8)
Total area	288 080	150 901	741 993	1 180 974

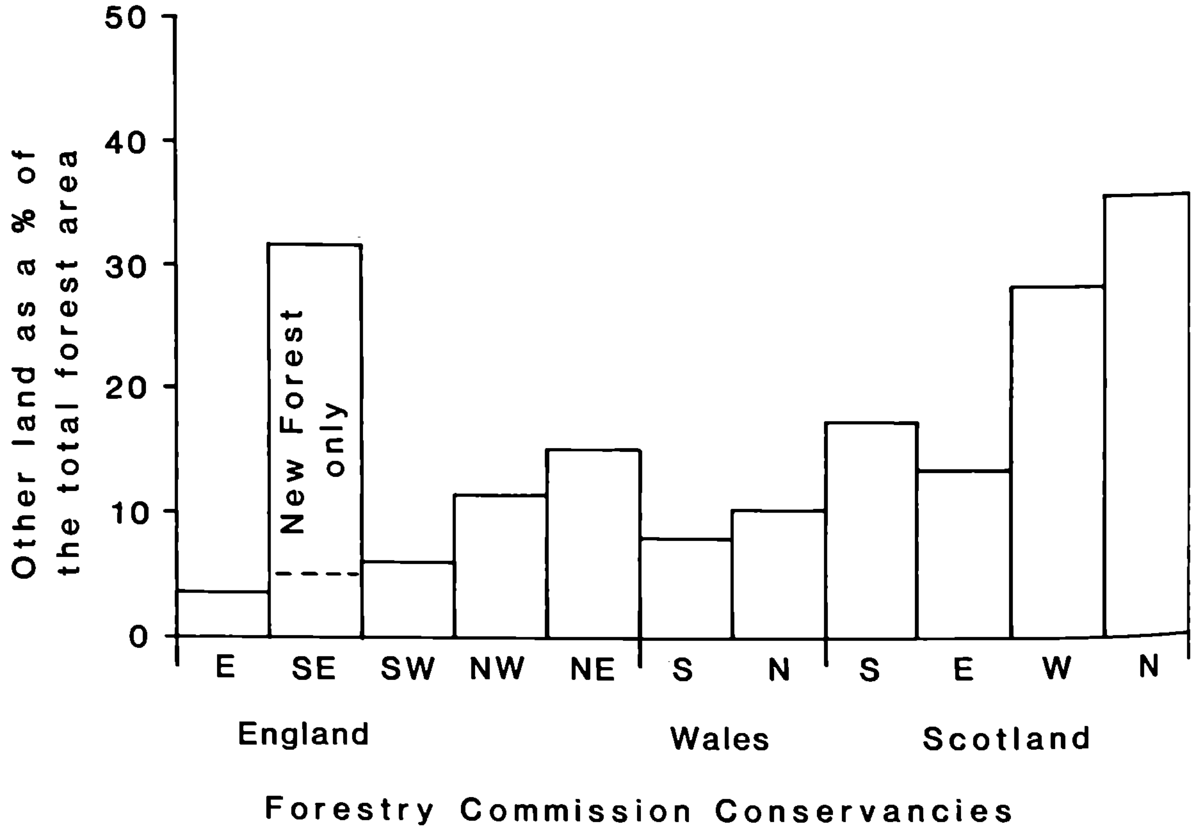


Figure 7.5 The proportion of non-forest land in Forestry Commission Conservancies (adapted from data in Forestry Commission, 1985c).

sited here too. Little-used or derelict buildings provide nest sites for swallow, house martin, kestrel and barn owl.

Forest nesting birds such as mistle thrush may forage in fields regularly or just when food is abundant, as occurs when buzzard feed on carrion or earthworms and sparrowhawk commence hunting the abundant post-breeding song bird flocks in June and July, just at the time when they have large nestlings to feed (Newton, 1986b). Hirons and Bickford-Smith (1983) using a radio tracking technique in the lowlands, showed that woodcock, a forest nesting bird, often foraged out on the adjacent pasture for earthworms at night. Woodcocks regularly occur in upland conifer forests but it is not known how dependant they are on agricultural land. No study has quantified the value for birds of these agricultural areas within forests, however the wide range of species that use them suggest that these areas do help to diversify the avifauna present in a forest.

Upland forests usually have areas above the commercial tree line. If we look at three upland areas, Northumberland, Galloway and Argyll, select the largest Forest District in each, namely Kielder, Newton Stewart and Cowal, we find that 19.7, 27.6 and 31.2 per cent respectively of the total forest area is in the non-forest land category (Forestry Commission, 1985c). These proportions probably reflect the increasing ruggedness of the terrain between the three areas. The value of these areas to moorland birds is very difficult to assess (Chapter 5). However, the fact that many important moorland birds either nest (golden plover and dunlin) or forage (golden eagle and raven) at higher rather than lower elevations, suggests that these areas may be of greater value than has been hitherto thought. They would be less valuable if adverse edge effects from the forest occurred, by for instance increasing predator pressure, but this has yet to be demonstrated (Chapter 5).

Little is known about how grazing regimes by domestic stock affect birds in these areas. Generally only on the largest moorlands is it cost effective to fence for grazing. Where grazing ceases, it is reasonable to anticipate that birds will respond in a similar way as during the early stages of afforestation, when there is an increase in both songbird and small mammal density and their respective avian predators. On the negative side, increased growth of ground vegetation may inhibit some waders (golden plover?), which breed in short vegetation, and the loss of sheep carrion may affect raven, golden eagle and buzzard unless this loss is offset by an increase in natural food sources (voles, deer).

The right mixture of conservation areas

We have discussed the various types of conservation areas, but how should they be mixed? We believe that the answer to this question depends on the location of the forest. In general terms, the conservation value of the woodland bird community will be greatest in the south of Britain, whereas the upland bird community is likely to be highest in the north. So we suggest that the emphasis between encouraging broadleaves and retaining unplanted areas should vary latitudinally. In the south of the country the emphasis should be on broadleaves whereas in the north a higher priority should be given to leaving areas unplanted. In more specific terms it is important to know what the original bird community was before planting. With this knowledge it will be possible to leave unplanted those areas known to be best for birds.

The forest crop

In a new forest, we consider it essential first to design the network of conservation areas which interlink whenever possible, and complement other design factors such as landscaping and recreation. Then, it is relatively easy to plant the remaining areas, where the main object of management is timber production. These areas will consist mainly of conifers. Below we discuss various factors which can be used to improve new planting for birds.

Planting sequence

When land is acquired for afforestation, it is usually all planted within a few years. This frequently leads to large even-aged plantations. Within the present system of land acquisition and forest management there are few opportunities for making any substantial change.

The abundance of any forest bird changes in relation to the growth stage of its habitat. Therefore, extensive even-aged new planting with conifers tends to support a bird community comprising a few species. This is similar to the situation that occurs on grazed moorland, but forests, because they are structurally more complex, tend to have more species associated with them (Moss, 1978a). One way of increasing the range of birds to be found in a forest is to stagger new planting so that a mosaic of different ages is produced. There are financial penalties associated with this tactic whenever it is employed.

Choice of conifer species

While the use of conservation areas can help to diversify some parts of the forest, for good reasons new planting in the uplands will continue to be predominantly with conifers. Sitka spruce is the principal species planted in the uplands and is likely to remain so owing to its high yield of valuable timber. In Great Britain, the proportion of the total forest land planted with Sitka spruce has risen in every decade since 1920. This is most pronounced in Scotland and Wales, where in the decade 1971-80, nearly 70 per cent of the land planted was with Sitka spruce (Figure 7.6). This increased use of Sitka spruce has largely been at the expense of Scots pine and Norway spruce.

What effects will reliance on a single species have on bird communities? We shall look at its effect on birds that are habitat generalists and specialists, because the choice of conifer species is likely to have quite different effects on these two groups.

Habitat generalists. Many birds that occur in upland conifer forests are not dependent on the trees themselves but on the habitat structure provided by the trees. They are habitat generalists and therefore occupy a wide range of woodland habitats irrespective of tree species. For these birds, the conifer planted is likely to affect them less than the inherent characteristics of the site, such as soil type, elevation, and the chosen structure of the forest.

The value to birds of North American exotics, such as Sitka spruce, compared with European conifers, has been little investigated. Newton and Moss (1981) found a similar range of species in Norway and Sitka spruce. Sitka spruce appears to sustain a considerable insect fauna (Evans, 1987) which is undoubtedly exploited by insectivorous song birds. Moss (1979; and 1978b) and Newton and Moss (1981) indicated that greater numbers of songbirds are found in spruce rather than pine with larch somewhere in between. Many of the birds recorded are insectivorous during the breeding season. Spruce carry about twice the weight of needles compared with pine (Miller and Miller, 1987), which may contain more invertebrate food for canopy feeding birds, such as the goldcrest, which is usually the most abundant species in this type of forest.

Von Haartman (1971) summarising the results from Finnish forests, showed similar differences between spruce and pine avifaunas. He also showed that for a given tree species, bird density increased on more productive sites. This suggests that an assessment of forest growth potential using the yield class system (Hamilton and Christie, 1971) for example, may also in time be used to predict bird density. Von Haartman (1971) also showed that spruce/birch mixtures held a greater density of birds for a given site type than did either pure conifers or conifer mixtures, and that these differences were greatest on the most productive sites. Bibby *et al.* (1989) demonstrated that broadleaves interspersed through a conifer crop resulted in the presence of some rarer songbirds, which were associated with broadleaves and were absent in pure conifer crops.

Habitat specialists. Birds that can be classed as conifer specialists are those that are dependent on conifer seed for food, together with the unique group of species that are found in native pinewoods and adjacent habitats. All are of considerable conservation interest.

The most characteristic birds that eat conifer seed are the crossbill and siskin (Newton, 1972a). Both species have dramatically extended their range in Britain as a result of afforestation (Sharrock, 1976). When conifer seed is particularly abundant other species will also feed on it,

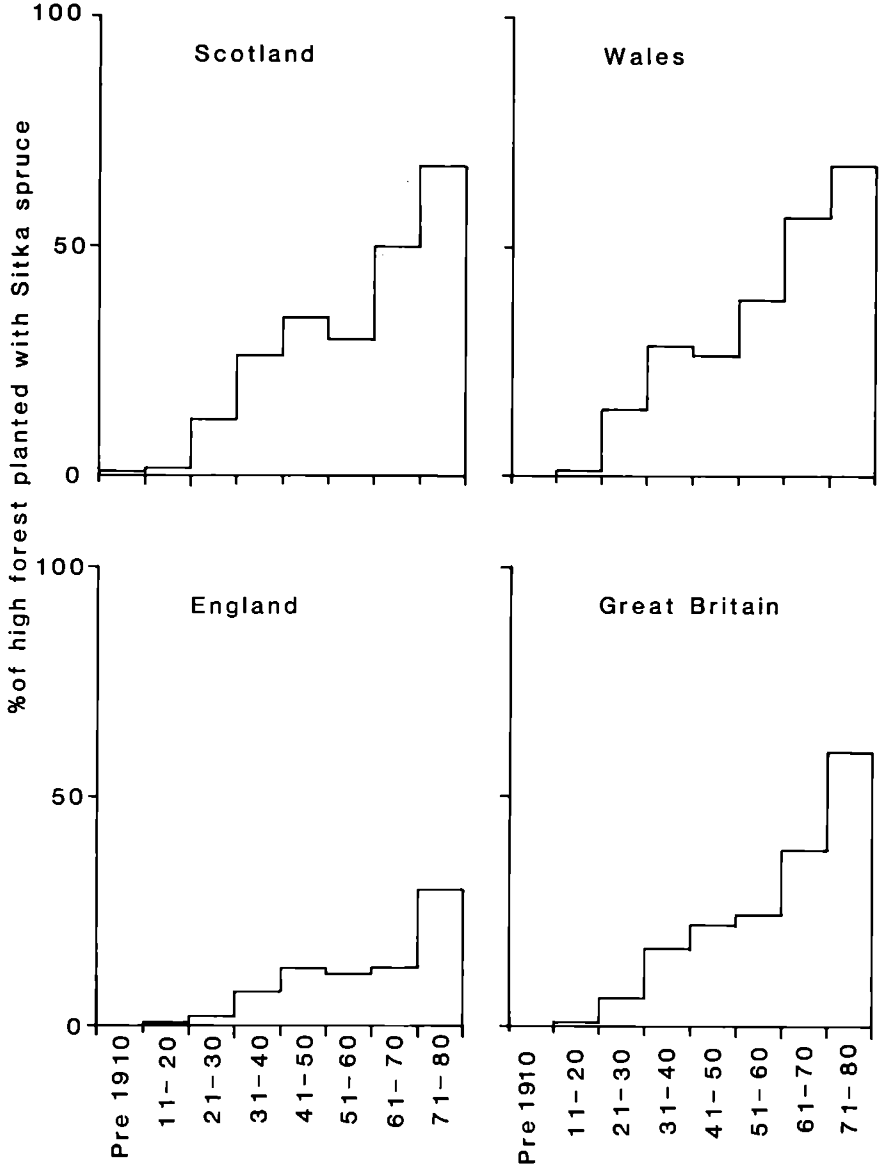


Figure 7.6 The increasing use of Sitka spruce in British forestry.

these include great spotted woodpecker (Cramp, 1985) and coal tit, marsh tit, willow tit and crested tit (Haftorn, 1959). Many of these species have evolved in north European forests of predominantly Scots pine and Norway spruce. Both trees have relatively large cones and seeds compared to the lighter cones and very much smaller seed of Sitka spruce (Staines *et al.*, 1987).

In Kielder Forest, Northumberland, which comprises largely Sitka and Norway spruce, a cone index on a scale from 0-10 (no cones – bumper crop) was kept for the years 1974-86. During this 12-year period there were good cone crops (scale 8-10) on Sitka spruce in 4 years and on Norway spruce in 2 years, one year was common to both tree species. Very high common crossbill and siskin populations were associated with both of the good Norway spruce cone crops but lower populations occurred during the three good Sitka spruce cone crops (Petty, unpublished data). This may be related to both the earlier shedding of seed and to the smaller seed size of Sitka spruce, which may therefore require birds to expend more energy to extract a similar weight of seed than in Norway spruce.

In northern Europe, high crossbill populations are also associated with good Norway spruce cone crops (Newton, 1972a). The availability of other trees, particularly Scots pine, probably helps to maintain crossbill populations in those years when Norway spruce fails to produce cones. Similar environmental conditions appear to be responsible for heavy flowering in both Norway spruce and Scots pine with the former cones maturing in the first year after flowering, whereas with the latter species cones take a year longer. This often results in a good spruce cone crop being followed a year later by a good pine cone crop. Common crossbills are not as efficient at extracting seed from the much stronger Scots pine cones. In northern Europe a larger crossbill with a heavy beak, the parrot crossbill, has evolved to cope with pine cones but this species has not established a permanent British population.

In the native pinewoods of Scotland yet another crossbill occurs (Nethersole-Thompson, 1975). This is thought to be a form of the common crossbill, which through time has evolved a larger beak to cope with pine cones. Some authorities now consider this to be a separate species, the Scottish crossbill (Knox, 1975; Voous, 1978). There is no indication that the Scottish crossbill has extended its range into Sitka spruce forests in response to the expansion of forestry, where because of the smaller cones and seed size it would presumably be less efficient at extracting seed than the common crossbill.

In the pine forests of north-east Scotland two other characteristic species occur, the crested tit and the capercaillie. Both are present in a wide range of forest habitats in this area, including conifer forest. However, the native pinewoods appear to form the core area holding the greatest density of these birds (Sharrock, 1976). Within Europe neither species is restricted to Scots pine and both occur widely in spruce and mixed forests. However, both species appear to require a particular forest structure comprising of open forest with rich ericaceous field layer. The latter is usually absent in spruce plantations and in high rainfall areas, which may account for the easterly distribution of both capercaillie and crested tit in Scotland. In addition to those requirements, crested tits also need dead wood or nestboxes in which to breed, and capercaillie need display sites (leks) which should be retained and not felled and where disturbance should be kept to a minimum. It should be possible to improve the quality of conifer forest for them, but this is dependent on a better understanding of their requirements.

We consider that the following guidelines on the choice of conifer species will be beneficial to birds.

1. European conifers, particularly Norway spruce and Scots pine, and also larch, should comprise around 5-10 per cent of any new planting proposals wherever the site conditions are suitable. This will help to diversify the cone production and benefit those birds that are dependent on conifer seed for food.
2. The choice of conifer species is likely to have less effect on songbird density than is site quality. One method of increasing the density and range of songbirds present in conifer crops would be to incorporate a scatter of broadleaves throughout the area in small groups rather than large areas. The effect of this practice is likely to be more pronounced on the most productive sites, particularly on well drained valley bottom sites, and in the south rather than the north of Britain, where there is a greater pool of bird species which select for broadleaves.
3. Safeguard the present area of native pinewoods and wherever possible extend their area.

Restructuring

When forests reach the end of the first rotation, the opportunity occurs to alter dramatically what may have been till then a large even-aged planting. Prior to the commencement of clear felling and restocking, the opportunity should be taken to plan the restructuring of the forest. This will eventually result in a more varied habitat, containing a patchwork of different growth stages. The aim should be to improve the distribution of age classes by exploiting any differences in rates of development in the present stands at different elevations, thus varying the rotation length. The start of restocking provides the chance to produce a far more varied forest ecosystem for birds.

Rotation length

To create a mosaic of different ages out of an even-aged plantation requires a commitment both to shorten and extend the rotation length in a proportion of the forest. In many upland forests the differences in growth potential vary widely due to factors such as altitude, site conditions and species. Therefore, optimum economic rotation lengths even in the same species, can show considerable differences over a relatively small area. High elevation stands will usually have a lower yield class and be less windfirm and be felled earlier than crops growing at lower elevations. There may be some financial loss incurred as a result of trying to spread out age classes, but this may be offset to some degree if the differences in site potential are fully used. Stands grown on both shortened and extended rotation tend to be richer for birds than normal forestry rotations, because they maximise the interesting pre-thicket and mature forest stages (Chapter 5).

Shortened rotations

The establishment and pre-thicket stages are particularly rich for birds (Chapter 5). Shortened rotations lead to a greater proportion of a forest area being in these stages (Figure 7.7). On exposed sites the possibility of improving the vertical structure within a stand is limited. In these areas, plantations are usually unthinned and need to be clear felled before a critical top height is reached, otherwise windthrow will cause serious problems (Chapter 6). In these stands we see shortened rotations as being the main method of increasing the attractiveness of these higher elevation plantations for birds.

Extended rotations

Given the prevailing discount factors, current economic rotations are short in relation to biological maturity. Stands have developed few of the structural characteristics which are important for a range of birds which are more abundant in old growth forests. Currie and Bamford (1982a) in Wales, showed that 100 year old conifer crops of three species had around twice the bird density and species richness compared with pole-stage stands of the same tree species. The breeding birds found in the older plots included species not usually associated with conifer stands; redstart, wood warbler, pied flycatcher and spotted flycatcher. Studies from America suggest that the value of such old trees is very dependent on age and is greatest for trees of over 15 m in height and 100 years old (Mannan *et al.*, 1980).

There are four major components of old natural forests which rarely occur in managed forest. First, the trees are much bigger and their larger crowns tend to produce more seed (see Bergsten, 1983 for Scots pine) and provide better nesting opportunities particularly for some raptors which build large nests. Second, the spacing of the dominant trees becomes wider and more irregular through time, as trees are felled or die. This leads to the development of sheltered glades which provide foraging sites for such birds as redstarts and flycatchers. Third, as more light reaches the forest floor, the ground vegetation slowly develops and may provide nest sites for some birds including ground-nesting warblers. The structure of the ground layer, including the regeneration of broadleaves and conifers which may later form a secondary canopy, depends on the level of grazing. Where grazing levels are high, from sheep and/or deer, the ground vegetation may become dominated by grasses and bryophytes with little or no development of a secondary tree or scrub canopy. Fourth, the quantity of large-diameter dead wood is likely to increase as crops age either as a result of wind

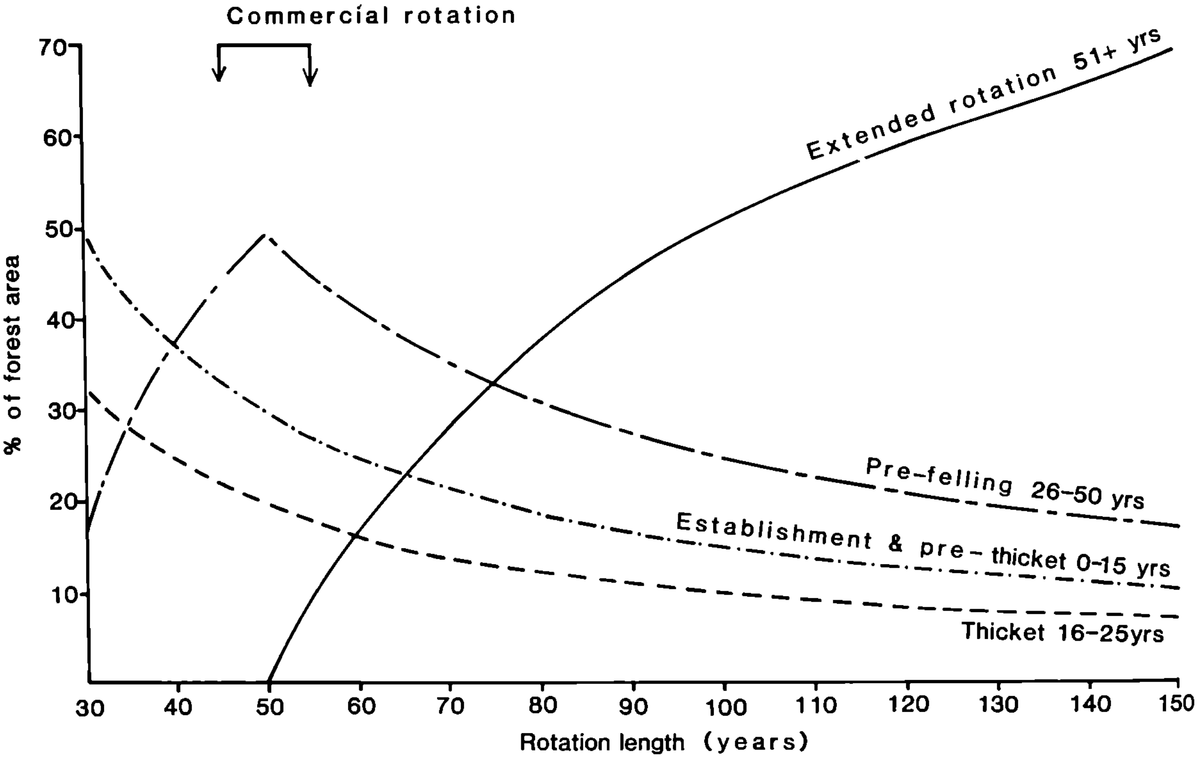


Figure 7.7 The influence of rotation length on the proportion of different forest growth stages, based on yield class 12 Sitka spruce (from Ratcliffe and Petty, 1986).

damage or fungal decay. Dead conifers and broadleaves provides both foraging and nesting opportunities for species such as woodpeckers, tits and tree creepers. Larger cavities within these dead trees provide nest sites for a range of species such as tawny owl, stock dove, jackdaw and goosander.

It is expensive to extend the rotation length beyond the age of maximum discounted revenue (Chapter 6), although for the first ten years or so, little is lost. However, we consider that a proportion of old growth is a highly desirable ingredient and more research into the management and effectiveness of extended rotation crops is required.

There are two distinct roles for extended rotations and both will benefit birds; the first is to break up even-aged stands, while the second is to provide rich old growth habitat. Rotation lengths should vary accordingly, with up to 1.5 times the normal rotation for the former, and greater than 1.5 for the latter including a small proportion going on to biological maturity. Most extended rotations should be sited on well drained, sheltered valley bottom sites to minimise the risk of windblow. Within a forest, a scatter of sites, ranging from a few trees to a maximum of 10-15 ha, would be of more value to birds, than concentrating the area into a few or a single large block.

Felling coupes

In commercial forests in the uplands, clear felling is the preferred silvicultural system. This is the most efficient way of harvesting timber due to site conditions, silvicultural techniques and commercial considerations. Given these constraints, there are nevertheless ways in which the clear felling system can be used to improve the quality of a forest for birds, mainly through planning the spatial arrangement and size of felling crops. We will look at both of these aspects in turn.

Spatial arrangement

In Chapters 4 and 5 we have shown how bird populations change in relation to forest succession. Within any one forest growth stage, the range of song-birds is relatively small. Restructuring a forest to provide an intimate patchwork of different growth stages will result in a more diverse song-bird community, although individual species will show abundance peaks in the growth stage which is most favourable to them. This patchwork will also favour some scarcer birds which require a mix of different habitats within their home range, these birds are generally of higher conservation value.

Examples are birds of prey and woodpeckers which generally exhibit a regular spacing between pairs when nesting habitat is not limited. Newton *et al.* (1979) showed that the nearest neighbour distances between sparrowhawks nesting territories in 12 study areas ranged between 0.46 km and 2.06 km. This distance between pairs was correlated with altitude and land productivity with the densest populations on the lowest most productive sites. Petty (1987a) showed that tawny owl territories were regularly spaced at a mean distance of 0.81 and 1.01 km along watercourses in two upland study areas. Both these species nest in pre-felling or older conifers, and tawny owls in particular obtain a lot of their food from restocked sites. Therefore, the provision of a patchwork of different aged growth stages throughout a forest can provide both nesting and feeding areas for some of these scarcer species. If the forest is felled in the same sequence in which it was planted, then there will be large periods when there are no nesting areas available for these more interesting species.

Size

Advice on the optimum size of felling coupes depends on how birds are distributed within them. If all species are distributed randomly then coupe size is irrelevant. This will lead to large areas having more species on them, simply because there is a greater chance of scarce species occurring in a large area independent of whether this is in a single 100 ha coupe or ten × 10 ha coupes.

Bibby *et al.* (1985) working in Wales showed this effect for a wide range of the most abundant birds. Only one species, the redpoll, showed a preference for larger coupes. However, out of 31 species recorded in 3258 contacts, only five species accounting for 1.1 per cent of the contacts could be considered moorland species. The bulk were birds associated with the various growth stages of forest succession, and these may have been less likely to have shown an area preference.

In contrast, Leslie (1981) in northern England, showed that moorland birds predominated on restocked sites. This may be more typical of the northern uplands, where many of the species recorded by Bibby *et al.* (1985) are much scarcer, and where poorer soil types and high deer numbers prevent the development of luxuriant scrub and broadleaved vegetation as occurred in Wales. It is hoped (Chapter 5) that clear felled areas may provide, for a short period, suitable habitat for the return of at least some of the original moorland birds.

A similar situation has already occurred in the lowlands, where some typical and scarce birds of heaths, such as nightjar and woodlark, now have substantial populations breeding on restocked sites (Gribble, 1983; Sitters, 1986). With this in mind, we recommend that clear felled areas should be made as large as possible in those areas where such a return is most likely, particularly where nearby the local moorland bird community is rich.

If there are species that are area sensitive, then it is likely to result from a preference for, or aversion to, the edge between the standing crop and felled area. Hansson (1983) in Sweden showed that there were fewer open land birds, in the 100 m adjacent to the standing crop, compared with further out on felled areas. If it were considered desirable to manage clear felling to the benefit of these birds, then the percentage of a clear fell within this 100 m strip, falls off relatively slowly on areas of 50 ha and over (Figure 7.8), suggesting that felling coupes should be over 50 ha and square shaped rather than long and narrow. Hansson also showed that overall bird density was greatest in the 50 m of forest closest to the edge. This was caused by forest bird density increasing in this area, and the addition of edge species such as tree pipit and great spotted woodpecker. To produce the maximum edge effect, felled areas should be below 20 ha and long and narrow (Figure 7.8).

Hibberd (1985) provides an account of restructuring in Kielder Forest, Northumberland. Here the forest has been grouped into three windblow zones based on elevation; low, medium and high risk. Roads and watercourses have been used as the main felling coupe divisions which are closer together at lower elevations. This has resulted in suggested felling coupe size for the low risk areas of 5-25 ha, the medium risk area of 25-50 ha and the high risk area of 50-100 ha. Until more evidence is available this type of approach provides a range of coupe sizes of the right order in the best situations. Small coupes at low elevation will maximise the edge to area ratio, while larger coupes at a higher altitude may provide habitat for a range of moorland species, some of which are likely to prefer larger areas.

Standing crop improvements

Once an area has been planted, either at afforestation or restocking, there is relatively little that can be done to improve the habitat for birds. Major changes in the composition of the bird community will be governed by the structure of the developing crop. However, certain silvicultural operations and other techniques are generally considered to be beneficial to birds.

Retaining non-crop vegetation

Operations such as weeding and cleaning, remove vegetation that is competing with the desired trees. Often it is the presence of this vegetation, particularly herbaceous plants and broadleaved regeneration, that accounts for the high density and diversity of birds to be found on these sites. Therefore, weeding or cleaning should be restricted to the immediate vicinity of

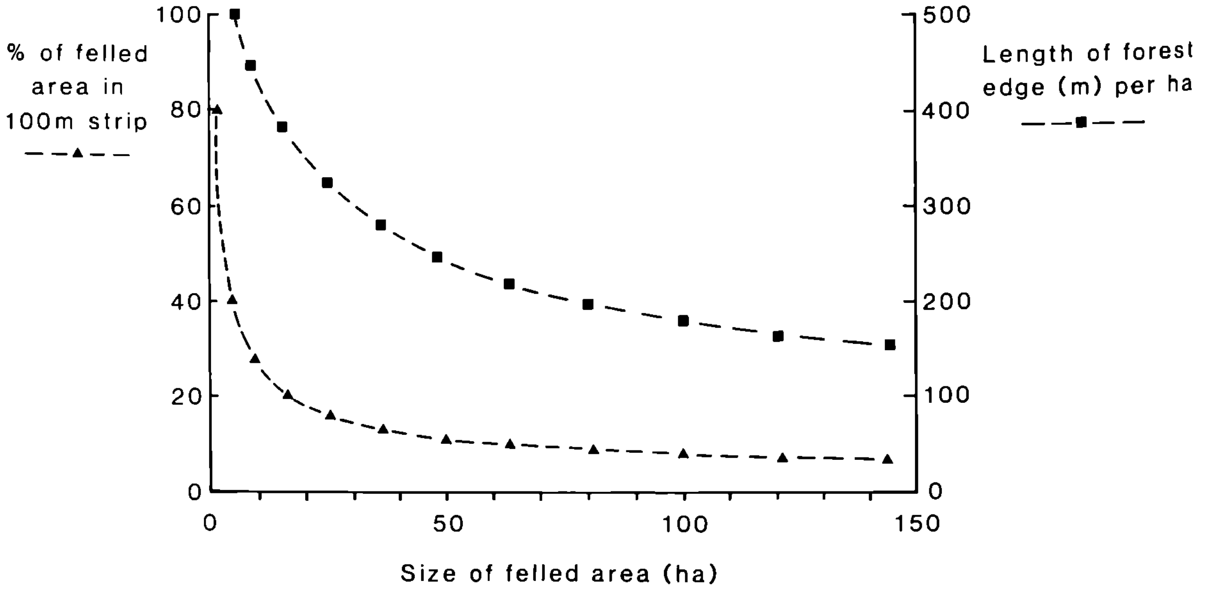


Figure 7.8 The relationship between forest edge (length of edge and area in a 100 m strip) and the size of a felled area. The model assumes a squared shaped felling coupe.

the desired trees only. In many upland forests, weeding and cleaning are only required on the richer sites or on some heather sites.

Thinning

Thinning is often considered to be beneficial to birds but there has been no research to test this assumption. Thinning is only likely to be carried out on the lower, faster growing and more stable sites. Many higher elevation crops will be grown on non-thinning regimes (Busby, 1974).

Thinning advances the ecological age of the stand, and these structural changes affect birds in two ways. First, it opens up the stand and allows below canopy access for a range of larger birds which are usually absent or scarce in unthinned stands. Sparrowhawks often commence nesting in stands for the first time almost immediately after thinning. Tawny owls and woodcock also become established at this time. Second, it encourages the growth of ground vegetation as more light reaches the forest floor, and the branches and tops from the felled trees are usually left on the ground, thus providing nest sites and feeding areas, particularly for wrens.

Artificial nest sites

In this section we look at the value of artificial nest sites in upland forests. We consider that these are suitable only for widespread use if they result in an increased density of a species, particularly if it is of conservation value. This may occur when a lack of suitable natural nest sites limits a population to below carrying capacity. However, some species with an adequate supply of natural nest sites may switch when provided with artificial sites. For instance, this is known to occur when nest boxes are provided for tawny owls (Southern, 1970; Petty, 1987a). Although useful for research purposes, they appear to have little effect on tawny owl density and productivity (Petty, unpublished data). Nestboxes are also widely advocated for kestrels, but in upland areas there is often an adequate supply of natural sites such as crags or crow nests, and nestboxes may be little used (Petty, 1985). Details of artificial nest sites can be found in De Feu (1985), Saurola (1978) and Petty (1987a; 1988).

Hole-nesting birds are conspicuously absent from the early stages of new forests, sometimes because of the absence of nest holes, but often because the habitat structure is not suitable for them. In managed forests few trees are allowed to get to an age when natural cavities are common. This means that nestboxes can occasionally result in spectacular increases to the density of birds in forests. The most notable example of this effect was obtained with pied flycatchers in Scandinavia, where densities of up to 2000 pairs/km^2 were observed, a higher density than any other bird community without nestboxes (von Haartman, 1971).

In Britain one study has monitored the effect of the introduction of nestboxes into study plots of oak, beech, or larch in Wales (Currie and Bamford, 1982b). In each case the number of hole-nesting species (great tit, blue tit, pied flycatcher) increased in the year after the provision of the boxes. In the second year after the boxes were provided the numbers rose even higher. Small numbers of willow tit, tree creeper, coal tit and nuthatch also used the boxes for nesting.

The effect of nestbox provision on pied flycatchers was impressive. In the year before nestboxes were provided, only one female pied flycatcher was seen on any of the plots. Two years later over 70 pairs were present. This species obtains much of its food from the canopy of oak trees, but here it was able to feed and rear young successfully in the larch plantation, although much foraging was undertaken in adjacent broadleaved areas. No control plots were surveyed and a few non-hole-nesting species (robin, wood warbler) also showed increases during the study period which suggested that perhaps some of the increase in hole-nesters may have occurred anyway. However, the high rate of uptake and use of nestboxes certainly suggests that nestbox provision will be a useful management technique in well thinned, lower elevation conifer stands, particularly those that are near to broadleaved areas and particularly where pied flycatcher are present in adjacent habitats.

In contrast, Currie and Bamford (1982b) also placed nestboxes in higher elevation spruce plantations and had very little success.

Nestboxes need to be cleaned out after they have been used. If not, they can become blocked with old nests which contain many parasites. Local natural history organisations can often be encouraged to monitor and maintain nestbox studies.

Larger nestboxes have also been used successfully to establish a breeding population of goldeneye in Scotland (Dennis and Dow, 1984). This duck may further extend its range in Britain if the use of nestboxes becomes more widespread. Another hole-nesting duck, the goosander, can also be encouraged to use even bigger nestboxes, if they are sited along rivers and lochs (Petty, unpublished data). A number of raptors will also use artificial nests. Village (1981 and 1983) showed that artificial crow nests, placed in shelterbelts in a newly planted area, led to an increase in the density of kestrel and long-eared owl when natural crow nests were lacking. Artificial platforms can also be successful in encouraging species such as goshawk, golden eagle and osprey to stay and breed in areas where they have been seen prospecting (Saurola, 1978; Dennis, 1987).

Summary

1. Upland forests should have areas devoted to permanent, conservation/landscape features. These should form an interlinking network, based on roads as the main horizontal divisions and watercourses as the vertical divisions. Whenever possible, other areas such as crags, broadleaved woodland, agricultural land and moorland should be linked into this system. These areas will provide semi-permanent bird habitat containing a pool of species from the former flora and fauna which can colonise the adjacent forest particularly after clear felling.
2. First generation upland conifer forests typically consist of large areas of even-aged stands, mainly of Sitka spruce. The addition of between 5 and 10 per cent of Norway spruce, Scots pine and larch may greatly enhance these forests for a number of species that are dependent on conifer seed, all of which are of high conservation value. The density and diversity of the more widely distributed insectivorous song birds is likely to be affected more by land productivity and habitat structure than the species of conifer. The use of broadleaf/conifer mixtures may further enhance the density and diversity of this last group of birds, particularly on better soils and near to broadleaved woodland.
3. At the end of the first rotation, the opportunity should be taken to restructure the forest to produce a patchwork of felling coupes of different ages, segmented by conservation areas. There should be a range of felling coupe sizes, with the smallest areas in the valleys and the larger areas at a higher elevation. Restructuring may involve some shortening and extending of the normal planned rotations. We believe that both are important for birds, particularly the latter.
4. During the rotation, little can be done to alter a forest for birds. Changes in the bird community are governed primarily by structural changes associated with the growth of the stand. Thinning may provide some larger birds with access into the crop at an earlier stage than in unthinned crops. In selected areas, the use of artificial nest sites can have important benefits to some species, either through increasing density or adding new birds to the forest avifauna.

Chapter 8

Future bird research in upland forests

Introduction

Most ecological subjects would benefit from more research. There is always a tendency to extrapolate from findings in one site or species or time to other sites, species and times, which may or may not be justified; only replicated work will decide this. Despite the need for more work on all subjects it is possible to identify topics that are of the greatest interest and value. Here we suggest topics for future research effort, and also in some cases our recommendations for the way in which it should be carried out.

Comments on methods

It is difficult to give advice on research into forest birds that will be relevant to all studies and we do not intend to do so here. However, the following failings of previous research should be borne in mind.

Too many studies of bird communities have used the time-intensive mapping method, when there was no intention of using the great amount of information which is collected (Chapter 3). The mapping method is invaluable when information is needed on the spatial location of individual territories or accurate estimates of bird density. A further justification for mapping territories would be when the detectabilities of birds varied in different study sites and their densities were to be compared. An example of such a situation would be a study of forest succession where it is likely that detectability of birds might be expected to vary with the growth stages of the crop.

Studies using the mapping technique have investigated only a small number of sites which differ in a large number of ways. This greatly reduces the value of the results because of the probable existence of confounding variables (Chapter 3). Future studies should preferably sample a large number of sites within a confined area (Bibby *et al.*, 1985) since such studies can lead to safer generalisations. With limited resources, increasing the number of sites visited can only be done at the expense of making fewer or shorter visits to each site. This clearly indicates point-counts as the method that should play an increasingly important role.

Some workers, particularly amateurs, seem to dislike this method on the grounds that it is a quick technique and therefore, in their mind, it cannot be as good as a more intensive method. This is not necessarily so. It would be better to know that a species occurred in most of a large sample of one type of wood and in a few of a large sample of another type of wood, than to know the exact number of breeding pairs in a small sample of both types of wood. This would be true even if the occurrence of the species was underestimated in the large samples for both types of wood.

When designing a study, ensure that the point-count locations are representative of the chosen area. This is best done by gridding the area and then using a random number table to pick the coordinates of the point-counts. When there are distinct habitat differences in the area to be studied, it may be necessary to stratify the area by habitat, before randomly determining a similar or minimum number of point-count locations in each strata. In previous work there has been a tendency to survey areas of high quality woodland. This is understandable if the mapping technique is being used, as few plots can be visited and much time is spent in each site; who wants to spend their time in birdless areas? Many of the woodland surveys that have been published have been of nature reserves and therefore probably of unusually high quality as regards their bird fauna. Such studies are useful for monitoring changes of bird species and densities in optimal habitats but cannot be used to generalise about the average woodland.

The census techniques that are used to assess the community structure of song-birds are usually unsuitable for gaining information on the scarcer song-birds and non-passerines, because in the conventional census the sample sizes of these birds are too small to be of use. For low-density birds, it is sometimes necessary to adapt existing techniques. The Nature Conservancy Council use a modified territory-mapping method for their Upland Bird Survey (Stroud *et al.*, 1987). Often studies of individual species are required, and methods and analytical techniques have to be developed for the bird concerned. Some good examples of recent studies of bird/habitat associations are given in Marquiss *et al.* (1978); Newton *et al.* (1981a); Newton *et al.* (1982b) and Bibby *et al.* (1986).

Research recommendations

Research is required to improve our understanding of how birds adapt to upland forests and how these forests can be improved for birds. We have divided the subject into three areas where we consider that further research is required, these are; bird community (synecological) studies, studies of individual species (autecological), and the effects of forest management practices.

Synecological studies

In Chapters 4 and 5 we have shown that much of the information on forest song-bird communities is based on small samples from restricted areas, and often from sites that are considered to be good for birds. To improve on our present knowledge requires a large sampling effort to describe the song-bird communities associated with a wide range of upland forest types and the growth stages of each, and taking into account both site and forest stand variables (Table 8.1). While both first and second generation stands require study, it is worth emphasising that afforestation only occurs once on a site, whereas restocking occurs repeatedly at the end of each rotation. Most bird research has so far investigated first generation stands, future research should concentrate on second generation stands, with appropriate comparisons with afforestation and the subsequent growth stages. Long-term monitoring plots may be required as the populations of many passerines in the uplands show considerable fluctuations in response to seasons, climate and variations in food supply.

The rationale behind developing such a data-

Table 8.1 Variables which need to be considered in future studies of forest song-bird communities

Forest crops	*Site*	*Song-bird communities*
1. Tree species (the main upland conifer (C) and broadleaf (B) species, plus C B, C C and B B mixtures)	1. Altitude	1. Density
2. Growth stage (including the range from planting to felling in both first and second generation crops)	2. Latitude (including both S>N and E>W gradients)	2. Species richness
	3. Land productivity (investigate relationships between the yield class system of predicting crop production and bird densities)	3. Breeding bird community
	4. Climate	4. Winter bird community
	5. Soil	5. Annual fluctuations
	6. Non-crop vegetation	6. Food supply
		7. Competition

base would be to provide a predictive 'forest bird population model', which managers could use in conjunction with the forest inventory database to forecast bird population trends (locally, regionally or nationally) in relation to different management options.

Autecological studies

Many of the scarcer upland birds are of conservation importance, and the success of different management options may often be judged by how these rarer birds are affected (either beneficially or detrimentally). Community studies seldom give enough information about these rarer birds. It is therefore necessary to initiate research into the requirements of individual species or closely related groups of birds, to provide managers with the advice about how to manage habitats for these species. Such advice should always be based on sound fundamental research which will often compare the effects of habitat variability on basic demographic factors such as productivity, survival and density.

Species benefiting from forestry

We have indicated that some species gain from forestry in the uplands. To some extent these species can be left alone since they appear to be favoured by the status quo. However, research could indicate which aspects of the forest are most attractive and therefore suggest future management to encourage their presence. Again research should concentrate on those species that are at present unusual or rare but could be helped to increase by sensitive management (Table 8.2).

The potential importance of clear fells to both moorland and woodland birds and the paucity of data to assess it has already been indicated in Chapter 5. The proportion of restocked land in British forests will increase markedly over the next 20 years. There is therefore scope to study the birds of restocked areas. The following questions need answers.

1. *Which birds occur on restocked sites?*
   This information is necessary to complete the picture of the total effects of afforestation on an area. Simple censuses would rapidly give information on the types of species which might be expected to use such areas for nesting. Such surveys should ideally cover large areas and concentrate on species of key conservation interests such as merlin, short-eared owl, hen harrier, red and black grouse, golden plover and greenshank.
2. *What is the optimum size for clear felled areas?*
   If birds of high conservation value will nest on clear felled areas, then these could be managed to make them more attractive. Research is needed to suggest management guidelines. An important aspect of clear felled sites may be their area. Species which generally nest in open habitats may avoid the wooded edges of clear felled areas so that if all such areas are too small they may never attract moorland species (see Chapter 4; Hansson, 1983). It has already been pointed out that to show avoidance of small plots, it is necessary to have large sample sizes. Such studies should concentrate on non-passerine

Table 8.2 Scarce species which have benefited from upland afforestation and where research is required to provide advice on habitat management

Species	*Reasons for research*
Red kite Hen harrier Barn owl Long-eared owl Short-eared owl Black grouse Buzzard Nightjar	Benefit from the early stages of afforestation, but no data are available on how they use second generation forests (establishment stage) and whether this habitat can be improved.
Capercaillie Woodcock Great spotted woodpecker Goshawk Crested tit Crossbill Siskin	Substantial range expansion as a result of afforestation but little data available on how to improve forest habitats.
Scottish crossbill	Determine how if separated from common crossbill, genetically, behaviourally, by habitat and by feeding ecology. Habitat management advice needed.

species of high conservation value, since there is now some baseline information on the passerine communities that can exist on restocked plots.

3. *Does the survival and productivity of species differ between restocked and newly planted sites?*
   Just as it is important to know how afforestation decreases (or increases) the output of wide-ranging species such as raven, so it is important to know whether species whose territories are wholly or partly contained within restocked sites are more or less productive when they nest on open moorland or on newly planted areas. Information of this type would allow assessment of the conservation value of clear fells, and eventually would allow a calculation of the equivalent area of moorland which is 'represented' by an area of clear fell.
4. *How do restocked sites benefit forest nesting birds?*
   Many larger birds require a combination of different habitats within their territory/home range. Large areas of even-aged forest, typical of the first generation, often lack or have low densities of these birds. Restocking may provide the habitat diversity that is required by species such as buzzard, kestrel, long-eared and tawny owl, sparrowhawk and great spotted woodpecker.

Species adversely affected by afforestation

These species have already been reviewed at some length in Chapter 5. Species regarded as being of high conservation value are not easily defined in objective terms but there is little disagreement among conservationists about which species these are, although the relative order of importance would probably vary widely between different individuals. The birds on which afforestation will have most effect are those that appear unable to live in managed forests and whose breeding range is largely within areas that are liable to be afforested. Some of Britain's less common wader species, carrion-dependent species and raptors come into this category. The breeding ranges of greenshank, dunlin, golden plover, raven, golden eagle, merlin, and also red-throated and black-throated divers, arctic skua, and ring ouzel are all restricted to upland areas, some of which may be afforested. Research on the effects of forestry on raven and merlin is sufficient to show that it will not be wise to generalise from studies of one species to the next, nor from studies in one area to another. For such species the following information is needed.

1. *What is the effect of afforestation on their residence, survival and productivity?*
   For some of the species thought to be at threat from afforestation there is generally a lack of published studies which show that such an effect exists. Although this does not mean that afforestation will not have harmful effects on these species, the case for the harmful effects of afforestation must be demonstrated not surmised. To show an effect of afforestation on residence, survival and productivity, it will be most convincing to show a before/after effect within sites which are afforested (providing suitable controls show no such effect of course) rather than comparing these measures for different sites which have and have not been afforested. This is because the areas afforested are highly likely to differ in important respects (e.g. altitude, isolation, soil quality) from those which have not been afforested.
2. *Which habitats and areas are favoured by the species at the moment?*
   Information on the present areas and habitats preferred by key species will be of value in identifying areas of prime conservation importance. This information could be used to establish Sites of Special Scientific Interest or in negotiating the areas to be left unplanted. Even in areas where, from the conservationist's point of view, there should be no afforestation at all, it would still be preferable to know which are the less vulnerable areas.
3. *What is the minimum area of moorland left within a forest area which can support these key species?*
   For species such as the greenshank and merlin which may nest within afforested

land, at least during the establishment phase, it may be possible to set aside particular areas within forests to allow these species to breed. This would require research to identify the habitat types required by these species and also the minimum size of area necessary to support a breeding pair.

Assuming that afforestation continues in the uplands in the near future, research should be concerned not only with demonstrating what the harmful effects of such afforestation may be on key species but also on determining the form in which afforestation would be most beneficial, therefore the following questions are posed.

4. *What is the effect of the spatial pattern of a given amount of afforestation on the avifauna of an area?*
The size of the area planted may affect the amount that productivity is reduced in a species. We demonstrated this with the imaginary example based on the raven in Chapter 5. More theoretical work on this issue would demonstrate the possibilities and probably suggest the best form in which data should be collected. Another complementary approach would be to investigate the effects of amount of forestry, dispersion of forest, and orientation relative to nest site on productivity of key species. This would be merely to extend the approach that has already been used to assess the effects of afforestation on species such as the raven. We also consider that it is important to continue monitoring at 5 or 10-year intervals, some of the studies which have already been established for merlin, red kite, raven, buzzard. The aim should be to show how these species are affected as the forests grow and are felled and replanted.
5. *What are the effects of afforestation on birds on the adjacent moorlands?*
Planting an area with trees may affect a much larger area than simply that which is planted; drainage of the plantable area may conceivably alter the water table of adjacent areas; increased human disturbance from forest workers (or decreased persecution by gamekeepers) may affect particular shy species; predatory birds or mammals may find nest sites or shelter within the forest and then prey on species outside the forest; management techniques may alter on land adjacent to afforested areas. If any of these suggestions are true then the adverse or favourable effects of afforestation may be felt over a much larger area than what is actually planted. The magnitude of these possible effects is an important subject for research. If significant harmful edge effects exist then this would suggest that afforestation would be less harmful if carried out in large areas than if carried out in small patches, assuming of course that there were also some large tracts left unplanted; this is opposite to much conventional conservation advice to foresters to create a mosaic of afforested and unafforested land.

Forest management practices

While we have outlined our ideas about future research on both bird communities and individual species we feel that there is a great scope for monitoring the effect of different forest management practices and options on birds. Many of our recommendations on improving upland forests for birds (Chapter 7) are based on what we intuitively think are best. Much of this advice needs validating and in many cases this can be done by comparing different treatments with untreated controls. Suggested areas of research are as follows.

1. The effect of different watercourse treatments (bankside planting) on waterway birds (grey wagtail, dipper, common sandpiper, goosander and heron).
2. The effects of different roadside treatments on birds.
3. The value of unplanted moorland (size and spatial arrangement) within forests to both moorland and forest birds.
4. The value of farmland, particularly pastures within forests (size and spatial distribution), to forest and open ground birds.
5. The value of power-line wayleaves for birds and ways of improving them.

6. The value of developing a permanent zone of widely spaced trees (conifers and broadleaves) at forest/moorland edge for species such as black grouse, merlin, long-eared owl and golden eagle.
7. The effects of broadleaves (species, size of area and dispersal) within conifer stands on birds.
8. The importance of conifer mixtures as opposed to monocultures for birds (and mammals) which are dependent on conifer seed for food.
9. The values of extended rotation (tree species, length of extention, size of area, spatial arrangement) areas for birds.
10. The value of deadwood (diameter, species, spatial distribution) to birds.
11. The effects of thinned compared with unthinned crops on birds.
12. The effects of felling-coupe size and spatial distribution on birds (including scarce species).
13. The effect of edges within the forest on birds (on either side of the edge), particularly the edges created at clear felling, and ways of improving edges.
14. The identification and management of areas rich in small mammals, which form a large part of the diet of numerous raptors and mammalian predators.

REFERENCES

ABLE, K. P. and NOON, B. R. (1976). Avian community structure along elevational gradients in the northeastern United States. *Oecologia* **26,** 275-294.

ADAMS, M. W. (1974). Effects of forest management practices on song-bird populations. *Report on Forest Research 1974,* 69-72. HMSO, London.

ADAMS, M. W. and EDINGTON, J. M. (1973). Comparisons of song-bird populations in mature coniferous and broadleaved woods. *Forestry* **46,** 191-202.

AHLÉN, I. (1975). Forestry and bird fauna in Sweden. *Ornis Fennica* **52,** 39-44.

AHLÉN, I. (1976). Forestry and vertebrate fauna. In, *Man and the boreal forest,* ed. C. O. Tamm, Ecological Bulletins **21,** 59-62. Swedish Natural Science Research Council, Stockholm.

AMBUEL, B. and TEMPLE, S. A. (1983). Area-dependent changes in the bird communities and vegetation of southern Wisconsin forests. *Ecology* **64,** 1057-1068.

ANDERSON, B. W., OHMART, R. D. and RICE, J. (1983). Avian and vegetation community structure and their seasonal relationships in the lower Colorado river valley. *Condor* **85,** 392-405.

ANDERSON, S. H. (1979). *Habitat structure, succession and bird communities.* USDA Forest Service, General Technical Report NC-51, 9-21.

ANDERSON, S. H., MANN, K. and SHUGART, H. H. (1977). The effect of transmission line corridors on bird populations. *American Midland Naturalist* **97,** 216-221.

ANDERSON, S. H. and ROBBINS, C. S. (1981). Habitat size and bird community management. *Transactions of the North American Wildlife and Natural Resources Conference* **46,** 511-520.

BAINBRIDGE, I. P., HOUSDEN, S. D., MINNS, D. W. and LANCE, A. N. (1987). *Forestry in the flows of Caithness and Sutherland.* RSPB, Sandy.

BALDA, R. P. (1975). *Vegetation structure and breeding bird diversity.* USDA Forest Service, General Technical Report WO-1, 59-80.

BATTEN, L. A. and POMEROY, D. E. (1969). Effects of reafforestation on the birds of Rhum, Scotland. *Bird Study* **16,** 13-16.

BEEDY, E. C. (1981). Bird communities and forest structure in the Sierra Nevada of California. *Condor* **83,** 97-105.

BERGSTEN, U. (1983). Cone and seed properties in a young and old stand of Pinus sylvestris. *Studia Forestalia Suecica* **168,** 1-13.

BEVAN, D. (1962). *Pine shoot beetles.* Forestry Commission Leaflet 3. HMSO, London.

BIBBY, C. J. (1986). Merlins in Wales; site occupancy and breeding in relation to vegetation. *Journal of applied Ecology* **23,** 1-12.

BIBBY, C. J. (1987). Foods of breeding merlins *Falco columbarius* from Wales. *Bird Study* **34,** 64-70.

BIBBY, C. J., ASTON, N. and BELLAMY, P. E. (1989). Effects of broadleaf trees on birds of upland conifer plantations in North Wales. *Biological Conservation* **49,** 17-30.

BIBBY, C. J., PHILLIPS, B. N. and SEDDON, A. J. E. (1985). Birds of restocked conifer plantations in Wales. *Journal of applied Ecology* **22,** 619-633.

BIJLEVELD, M. (1974). *Birds of prey in Europe.* Macmillan Press, London.

BINNS, W. O., MAYHEAD, G. J. and MACKENZIE, J. M. (1980). *Nutrient deficiencies of conifers in British forests.* Forestry Commission Leaflet 76. HMSO, London.

BINNS, W. O., REDFERN, D. B., RENNOLLS, K. and BETTS, A. J. A. (1985). *Forest health and air pollution 1984 survey.* Forestry Commission Research and Development Paper 142. Forestry Commission, Edinburgh.

BINNS, W. O., REDFERN, D. B., BOSWELL, R. and BETTS, A. J. A. (1986). *Forest health and air pollution 1985 survey.* Forestry Commission Research and Development Paper 147. Forestry Commission, Edinburgh.

BIRKS, H. J. B., DEACON, J. and PEGLAR, S.

(1975). Pollen maps for the British Isles 5000 years ago. *Proceedings of the Royal Society of London B* **189,** 87-105.

BLAKE, J. G. and KARR, J. R. (1984). Species composition of bird communities and the conservation benefit of large versus small forests. *Biological Conservation* **30,** 173-187.

BLONDEL, J. and CUVILLIER, R. (1977). Une méthode simple et rapide pour décrire les habitats d'oiseaux: le stratiscope. *Oikos* **29,** 316-331.

BLONDEL, J., FERRY, C. and FROCHOT, B. (1973). Avifaune et végétation essai d'analyse de la diversité. *Alauda* **41,** 63-84.

BÖRSET, E. and KRAFFT, A. (1973). Black grouse *Lyrurus tetrix* and capercaillie *Tetrao urogallus* brood habitats in a Norwegian spruce forest. *Oikos* **24,** 1-7.

BRAMWELL, D. (1960). Some research into bird distribution in Britain during the late-glacial and post-glacial periods. *Bird Report of the Merseyside Naturalists' Association 1959-60,* 51-58.

BROWN, L. H. and AMADON, D. (1968). *Eagles, hawks and falcons of the world.* Country Life Books, Feltham.

BROWN, L. H. and WATSON, A. (1964). The golden eagle in relation to its food supply. *Ibis* **106,** 78-100.

BURGESS, R. L. and SHARPE, D. M. (eds) (1981). *Forest island dynamics in man-dominated landscapes.* Springer-Verlag, New York.

BUSBY, R. J. N. (1974). *Forest site yield guide to upland Britain.* Forestry Commission Forest Record 97. HMSO, London.

BUSBY, R. J. N. and GRAYSON, A. J. (1981). *Investment appraisal in forestry.* Forestry Commission Booklet 47. HMSO, London.

CADBURY, J. (1980). *Silent death: the destruction of birds and mammals through the deliberate misuse of poisons in Britain.* RSPB, Sandy.

CADMAN, W. A. (1949). Distribution of black grouse in north Wales forests. *British Birds* **42,** 365-367.

CARTER, C. I. (1972). *Winter temperature and survival of the green spruce aphid.* Forestry Commission Forest Record 84. HMSO, London.

CARTER, C. I. and ANDERSON, M. A. (1987). *Enhancement of lowland forest ridesides and roadsides to benefit wild plants and butterflies.* Forestry Commission Research Information Note 126. Forestry Commission, Edinburgh.

CENTRE FOR AGRICULTURAL STRATEGY (1980). *Strategy for the UK forest industry.* CAS report number 6. Reading.

CHARLES, W. N. (1956). The effect of a vole plague in the Carron Valley, Stirlingshire. *Scottish Forestry* **10,** 201-204.

CHARLES, W. N. (1981). Abundance of the field vole (*Microtus agrestris*) in conifer plantations. In, *Forest and woodland ecology,* eds F. T. Last and A. S. Gardiner, 135-137. Institute of Terrestrial Ecology, Cambridge.

CHITTY, D. (1952). Mortality amongst voles (*Microtus agrestris*) at Lake Vyrnwy, Montgomeryshire in 1936-39. *Philosophical Transactions of the Royal Society of London B* **236,** 505-552.

COOK, M. J. H. (1982). Breeding status of the crested tit. *Scottish Birds* **12,** 97-106.

COOKE, A. S., BELL, A. A. and HAAS, M. B. (1982). *Predatory birds, pesticides and pollution.* Institute of Terrestrial Ecology, Huntingdon.

COOPER, J. E. and PETTY, S. J. (1988). Trichomoniasis in free-living goshawks *Accipiter gentilis gentilis* from Great Britain. *Journal of Wildlife Diseases* **21,** 80-87.

CORBET, G. B. and SOUTHERN, H. N. (1977). *The handbook of British mammals.* Blackwell, Oxford.

CRAMP, S. (ed.) (1985). *The birds of the western Palearctic, Volume IV.* Oxford University Press, Oxford.

CRAMP, S. and SIMMONS, K. E. L. (eds) (1980). *The birds of the western Palearctic, Volume II.* Oxford University Press, Oxford.

CRICK, H. Q. P. (1986). The effects of fenitrothion applications to control pine beauty moth on non-target animals in Scotland. In, *Trees and wildlife in the Scottish uplands,* ed. D. Jenkins, 101-107. Institute of Terrestrial Ecology, Huntingdon.

CURRIE, F. A. and BAMFORD, R. (1981). Bird populations of sample pre-thicket forest plantations. *Quarterly Journal of Forestry* **75,** 75-82.

CURRIE, F. A. and BAMFORD, R. (1982a). The value to wildlife of retaining small conifer stands beyond normal felling age within forests. *Quarterly Journal of Forestry* **76,** 153-160.

CURRIE, F. A. and BAMFORD, R. (1982b). Songbird nestbox studies in forests in North Wales. *Quarterly Journal of Forestry* **76,** 250-255.

DENNIS, R. H. (1987). Osprey recolonisation. *RSPB Conservation Review* **1,** 88-90.

DENNIS, R. H. and DOW, H. (1984). The establishment of a population of goldeneyes *Bucephala clangula* breeding in Scotland. *Bird Study* **31,** 217-227

DENNIS, R. H., ELLIS, P. M., BROAD, R. A. and LANGSLOW, D. R. (1984). The status of the golden eagle in Britain. *British Birds* **77,** 592-607.

DIAMOND, J. M. (1975). Assembly of species communities. In, *Ecology and evolution of communities,* eds M. L. Cody and J. M. Diamond, 342-444. Harvard University Press.

DICKSON, J. G., CONNER, R. N. and WILLIAMSON, J. H. (1984). Bird community changes in a young pine plantation in east Texas. *Southern Journal of applied Forestry* **8,** 47-51.

DIMBLEBY, G. W. (1984). Anthropogenic changes from Neolithic through Medieval times. *New Phytologist* **98,** 57-72.

DU FEU, C. (1985). *Nestboxes.* British Trust for Ornithology Field Guide Number 20. BTO, Tring.

EDWARDS, P. H. and CHRISTIE, J. M. (1981). *Yield models for forest management.* Forestry Commission Booklet 48. Forestry Commission, Edinburgh.

ENEMAR, A. (1959). On the determination of the size and composition of a passerine bird population during the breeding season. *Var Fagelvarld* **18,** supplement 2, 1-114.

ENGSTROM, R. T., CRAWFORD, R. L. and BAKER, W. W. (1984). Breeding bird populations in relation to changing forest structure following fire exclusion: a 15-year study. *Wilson Bulletin* **96,** 437-450.

ERDELEN, M. (1984). Bird communities and vegetation structure: I. Correlations and comparisons of simple diversity indices. *Oecologia (Berlin)* **61,** 277-284.

EVANS, H. F. (1987). Sitka spruce insects: past, present and future. *Proceedings of the Royal Society of Edinburgh* **93B,** 157-167.

EVANS, J. (1984). *Silviculture of broadleaved woodland.* Forestry Commission Bulletin 62. HMSO, London.

FAEGRI, K. and IVERSEN, J. (1964). *Textbook of pollen analysis.* Blackwell, Oxford.

FERRY, C. and FROCHOT, B. (1970). L'avifauna nidificatrice d'une forêt de chênes pédoncules en Bourgogne: étude de deux successions écologiques. *Terre et Vie* **117,** 153-250.

FLEGG, J. J. M. and BENNETT, T. J. (1974). The birds of oak woodlands. In, *The British oak: its history and natural history,* eds M. G. Morris and F. H. Perring, 324-340. Published for the Botanical Society of the British Isles by E. W. Classey, Oxford.

FLOWER, R. J., BATTARBEE, R. W. and APPLEBY, R. G. (1987). The recent palaeolimnology of acid lakes in Galloway, south-west Scotland: diatom analysis, pH trends, and the role of afforestation. *Journal of Ecology* **75,** 797-824.

FORD, E. D. (1982). Catastrophe and disruption in forest ecosystems and their implications for plantation forestry. *Scottish Forestry* **36,** 9-24.

FORD, E. D., MALCOLM, D. C. and ATTERSON, J. (eds) (1979). *The ecology of even-aged forest plantations.* Institute of Terrestrial Ecology, Cambridge.

FORESTRY COMMISSION (1970). *Fomes annosus: a fungus causing butt rot, root rot* *and death in conifers*. Forestry Commission Leaflet 5. HMSO, London.

FORESTRY COMMISSION (1977). *The wood production outlook in Britain: a review*. Forestry Commission, Edinburgh.

FORESTRY COMMISSION (1983a). *Census of woodland and trees, England*. Forestry Commission, Edinburgh.

FORESTRY COMMISSION (1983b). *Census of woodland and trees, Wales*. Forestry Commission, Edinburgh.

FORESTRY COMMISSION (1983c). *Census of woodland and trees, Scotland*. Forestry Commission, Edinburgh.

FORESTRY COMMISSION (1984). *Census of woodland and trees, Great Britain*. Forestry Commission, Edinburgh.

FORESTRY COMMISSION (1985a). *Broadleaved woodland grant scheme*. Forestry Commission, Edinburgh.

FORESTRY COMMISSION (1985b). *The policy for broadleaved woodlands*. Policy Paper 5. Forestry Commission, Edinburgh.

FORESTRY COMMISSION (1985c). *Forestry Commission 65th Annual Report and Accounts* 1984-85. HMSO, London.

FORESTRY COMMISSION (1988a). *Environmental assessment of afforestation projects*. Forestry Commission, Edinburgh.

FORESTRY COMMISSION (1988b). *Forests and water guidelines*. Forestry Commission, Edinburgh.

FORESTRY COMMISSION. (1989). *Native pinewoods – grants and guidelines*. Forestry Commission, Edinburgh.

FORMAN, R. T. T., GALLI, A. E. and LECK, C. F. (1976). Forest size and avian diversity in New Jersey woodlots with some land use implications. *Oecologia (Berlin)* **26,** 1-8.

FRENCH, D. D., JENKINS, D. and CONROY, J. W. H. (1986). Guidelines for managing woods in Aberdeenshire for songbirds. In, *Trees and wildlife in the Scottish uplands,* ed. D. Jenkins, 129-143. Institute of Terrestrial Ecology, Huntingdon.

FULLER, R. J. (1982). *Bird habitats in Britain.* Poyser, Calton.

FULLER, R. J. and MORETON, B. D. (1987). Breeding bird populations of Kentish sweet chestnut *Castanea sativa* coppice in relation to age and structure of the coppice. *Journal of applied Ecology* **24,** 13-28.

FULLER, R. J. and TAYLOR, K. (1984). Breeding birds of some Lincolnshire limewoods in 1982. *Transactions of the Lincolnshire Naturalists' Union* **21,** 47-54.

FULLER, R. J. and WHITTINGTON, P. A. (1987). Breeding bird distribution within Lincolnshire ash-lime woodlands: the influence of rides and the woodland edge. *Acta Oecologia Oecologia Generalis* **8,** 259-268.

GALLI, A. E., LECK, C. F. and FORMAN, R. T. T. (1976). Avian distribution patterns in forest islands of different sizes in central New Jersey. *Auk* **93,** 356-364.

GARFITT, J. E. (1983). Afforestation and upland birds. *Quarterly Journal of Forestry* **77,** 253-255.

GÉNSBØL, B. (1984). *Collins guide to the birds of prey of Britain and Europe, North Africa and the Middle East.* Collins, London.

GIBBS, J. A. (1960). Populations of tits and goldcrests and their food supply in pine plantations. *Ibis* **102,** 163-208.

GŁOWACINSKI, Z. (1972). Secondary succession of birds in an oak-hornbeam forest. *Bulletin, Polish Academy of Sciences, Biology* **20,** 705-710.

GŁOWACINSKI, Z. (1975a). Succession of bird communities in the Niepolomice Forest (southern Poland). *Ekologia Polska* **23,** 231-263.

GŁOWACINSKI, Z. (1975b). Birds of the Niepolomice Forest (a faunistic-ecological study). *Acta Zoologica Cracoviensia* **20,** 1-87.

GŁOWACINSKI, Z. (1979). Some ecological parameters of avian communities in the successional stages of a cultivated pine forest. *Bulletin, Polish Academy of Sciences, Biology* **27,** 169-177.

GŁOWACINSKI, Z. (1981a). Secondary succession of birds in a maturing forest ecosystem (syntheses). *Studia Naturae (Seria A)* **26,** 1-64.

GŁOWACINSKI, Z. (1981b). Stability in bird communities during the secondary succession of a forest ecosystem. *Ekologia Polska* **29,** 73-95.

GŁOWACINSKI, Z. and JÄRVINEN, O. (1975). Rate of secondary succession in forest bird communities. *Ornis Scandinavica* **6,** 33-40.

GŁOWACINSKI, Z. and WEINER, J. (1975). A bird community of a mature deciduous forest: its organisation, standing crop and energy balance (IBP 'Ispina project'). *Bulletin, Polish Academy of Sciences, Biology* **23,** 691-697.

GODDARD, T. R. (1935). A census of the short-eared owl (*Asio f. flammeus*) at Newcastleton, Roxburghshire, 1934. *Journal of Animal Ecology* **4,** 113-118 and 289-290.

GODWIN, H. (1975). *The history of the British flora.* Cambridge University Press, Cambridge.

GRANFIELD, E. F., MACMAHON, C. D. and MITHEN, D. A. (1980). *Developments in forest road planning.* Forestry Commission Research and Develoment Paper 127. Forestry Commission, Edinburgh.

GRIBBLE, F. C. (1983). Nightjars in Britain and Ireland 1981. *Bird Study* **30,** 165-177.

GROMADSKI, M. (1970). Breeding communities of birds in mid-field afforested areas. *Ekologia Polska* **18,** 307-350.

GROVE, S. J., HOPE JONES, P., MALKINSON, A. R., THOMAS, D. H. and WILLIAMS, I. (1988). Black grouse in Wales, spring 1986. *British Birds* **81,** 2-9.

HAAPANEN, A. (1965). Bird fauna of Finnish forests in relation to forest succession. *Annales Zoologici Fennici* **2,** 153-196.

HAAPANEN, A. (1966). Bird fauna of Finnish forests in relation to forest succession. *Annales Zoologici Fennici* **3,** 176-200.

HAFTORN, S. (1959). The proportion of spruce seed removed by tits in a Norwegian spruce forest in 1954-55. *Det Kongelige Norske Videnskabers Selskabs Forhandlinger* **32,** 121-125.

HAILA, Y., JÄRVINEN, O. and VÄISÄNEN, R. A. (1980). Habitat distribution and species associations of land bird populations on the Aland Island, SW Finland. *Annales Zoologici Fennici* **17,** 87-106.

HAMILTON, G. J. (1980). *Line thinning.* Forestry Commission Leaflet 77. HMSO, London.

HAMILTON, G. J. and CHRISTIE, J. M. (1971). *Forest management tables (metric).* Forestry Commission Booklet 34. HMSO, London.

HAMILTON, R. B. and NOBLE, R. E. (1975). *Plant succession and interactions with fauna.* USDA Forest Service, General Technical Report WO 1, 96-114.

HANSSON, L. (1983). Bird numbers across edges between mature conifer forest and clearcuts in central Sweden. *Ornis Scandinavica* **14,** 97-103.

HARRIMAN, R. and MORRISON, B. R. S. (1982). Ecology of streams draining forested and non-forested catchments in an area of central Scotland subject to acid precipitation. *Hydrobiologica* **88,** 251-263.

HARTLEY, P. H. T. (1953). An ecological study of the feeding habits of the English titmice. *Journal of Animal Ecology* **22,** 261-288.

HARVEY-BROWN, J. A. (1879). *The capercaillie in Scotland.* D. Douglas, Edinburgh.

HELLE, P. (1984). Effects of habitat area on breeding bird communities in northeastern Finland. *Annales Zoologici Fennici* **21,** 421-425.

HELLE, P. (1985a). Habitat selection of breeding birds in relation to forest succession in northeastern Finland. *Ornis Fennica* **62,** 113-123.

HELLE, P. (1985b). Effects of forest regeneration on the structure of bird communities in northern Finland. *Holarctic Ecology* **8,** 120-132.

HELLE, E. and HELLE, P. (1982). Edge effect on forest bird densities on offshore islands in the northern Gulf of Bothnia. *Annales Zoologici Fennici* **19,** 165-169.

HELLIWELL, D. R. (1982). *Options in forestry.* Packard Publishing, Chichester.

HIBBERD, B. G. (1985). Restructuring of plantations in Kielder Forest District. *Forestry* **58,** 119-129.

HIBBERD, B. G. (ed.) (1986). *Forestry practice.* Forestry Commission Bulletin 14. HMSO, London.

HIBBERD, B. G. (ed.) (1988). *Farm woodland practice.* Forestry Commission Handbook 3. HMSO, London.

HILL, M. O. (1983). Plants in woodlands. In, *Forestry and conservation,* ed. E. H. M. Harris, 56-65. Royal Forestry Society, Tring.

HINO, T. (1985). Relationships between bird community and habitat structure in shelterbelts of Hokkaido, Japan. *Oecologia (Berlin)* **65,** 442-448.

HIRONS, G. J. M., and BICKFORD-SMITH, P. (1983). The diet and behaviour of Eurasian woodcock wintering in Cornwall. In, *Second European woodcock and snipe workshop,* ed. H. Kalchreuter, 11-17. International Waterfowl Research Bureau, Slimbridge.

HOGSTAD, O. (1967). The edge effect on species and population density of some passerine birds. *Nytt Magasin for Zoologi* **15,** 40-43.

HOLMES, G. D. (1975). History of forestry and forest management. *Philosophical Transactions of the Royal Society of London B* **271,** 69-80.

HOLMES, R. T. and STURGES, F. W. (1975). Bird community dynamics and energetics in a northern hardwood ecosystem. *Journal of Animal Ecology* **44,** 175-200.

HOLTAM, B. W. (1971). *Windblow of Scottish forests in January 1968.* Forestry Commission Bulletin 45. HMSO, London.

INSLEY, H. (ed.) (1988). *Farm woodland planning.* Forestry Commission Bulletin 80. HMSO, London.

IRVINE, J. (1977). Breeding birds in New Forest broadleaved woodland. *Bird Study* **24,** 105-111.

JALKANIN, P. (1960). Palkaneen linnustota. (Zusammenfassung: Uber die Vogelfauna von Palkane). *Palkaneseuran julk* **2,** 1-122.

JAMES, F. C. (1971). Ordinations of habitat relationships among breeding birds. *Wilson Bulletin* **83,** 215-229, 235-236.

JAMES, F. C. and RATHBUN, S. (1981). Rarefaction, relative abundance, and diversity of avian communities. *Auk* **98,** 785-800.

JAMES, F. C. and WAMER, N. O. (1982). Relationships between temperate forest bird communities and vegetation structure. *Ecology* **63,** 159-171.

JÄRVINEN, O., KUUSELA, K. and VÄISÄNEN, R. A. (1977). [Effects of modern forestry on the numbers of breeding birds in Finland 1945-1975]. *Silva Fennica* **11,** 284-294.

JESSOP, R. M. (1982). The impact of afforestation on the avifauna of a Scottish moor. *Arboricultural Journal* **6,** 107-119.

JOHNSON, V. R. (1947). Breeding birds of the forest edge in Illinois. *Condor* **70,** 348-357.

JOHNSON, A. S. and LANDERS, J. L. (1982). Habitat relationships of summer resident birds in slash pine flatwoods. *Journal of Wildlife Management* **46,** 416-428.

JOHNSON, D. W. and ODUM, E. P. (1956). Breeding bird populations in relation to plant succession on the Piedmont of Georgia. *Ecology* **37,** 50-62.

JOHNSTONE, G. W. (1967). Blackgame and capercaillie in relation to forestry in Britain. *Forestry (supplement 1967),* 68-77.

JONES, A. M. (1982a). Capercaillie in Scotland – towards a conservation strategy. In, *Grouse,* ed. T. W. I. Lovel, 60-73. World Pheasant Association, Suffolk.

JONES, A. M. (1982b). Winter habitat relationships of capercaillie in Scottish plantations. *Journal of the World Pheasant Association* **7,** 9-27.

JONES, A. M. (1982c). *Aspects of the ecology and behaviour of capercaillie Tetrao urogallus in two Scottish plantations.* Unpublished PhD Thesis, University of Aberdeen.

JONES, P. H. (1972). Succession in breeding bird populations of sample Welsh oakwoods. *British Birds* **65,** 291-299.

KARR, J. R. (1971). Structure of avian communities in selected Panama and Illinois habitats. *Ecological Monographs* **41,** 207-233.

KENNEDY, C. E. J. and SOUTHWOOD, T. R. E. (1984). The number of species of insect associated with British trees: a re-analysis. *Journal of Animal Ecology* **53,** 455-478.

KNOX, A. G. (1975). Crossbill taxonomy. In, *Pine crossbills,* by D. Nethersole-Thompson, 191-201. Poyser, Berkhamsted.

KROODSMA, R. L. (1982a). Edge effect on breeding forest birds along a power-line corridor. *Journal of applied Ecology* **19,** 361-370.

KROODSMA, R. L. (1982b). Bird community ecology on power-line corridors in East Tennessee. *Biological Conservation* **23,** 79-94.

KROODSMA, R. L. (1984). Effect of edge on breeding forest bird species. *Wilson Bulletin* **96,** 426-436.

LACK, D. (1933). Habitat selection in birds, with special reference to the effects of afforestation on the Breckland avifauna. *Journal of Animal Ecology* **2,** 239-262.

LACK, D. (1939). Further changes in the Breckland avifauna caused by afforestation. *Journal of Animal Ecology* **8,** 277-285.

LACK, D. (1966). *Population studies of birds.* Clarendon Press, Oxford.

LACK, D. and LACK, E. (1951). Further changes in bird-life caused by afforestation. *Journal of Animal Ecology* **20,** 173-179.

LANGSLOW, D. R. (1983). The impacts of afforestation on breeding shorebirds and raptors in the uplands of Britain. In, *Shorebirds and large waterbirds conservation,* eds P. R. Evans, H. Hafner and P. Hermit, 17-25. Commission of the European Communities, Brussels.

LAST, F. T., JEFFERS, J. N. R., BUNCE, R. G. H., CLARIDGE, C. J., BALDWIN, M. B. and CAMERON, R. J. (1986). Whither forestry? The scene in AD 2025. In, *Trees and wildlife in the Scottish uplands,* ed. D. Jenkins, 20-32. Institute of Terrestrial Ecology, Edinburgh.

LAY, D. W. (1938). How valuable are woodland clearings to birdlife. *Wilson Bulletin* **45,** 254-256.

LEBRETON, P. and BROYER, J. (1981). Contributions to the study of avifauna-altitude relationships. I. In the Rhone-Alpes region, France. *Oiseau Revue Française d'Ornithologie* **51,** 265-285.

LESLIE, R. (1981). Birds of the north-east England forests. *Quarterly Journal of Forestry* **75,** 153-158.

LESLIE, R. (1984). The population and distribution of nightjars *(Caprimulgus europaeus)* on the North York moors. *The Naturalist* **110,** 23-28.

LEVER, C. L. (1977). *The naturalised animals of the British Isles.* Hutchinson, London.

LINDÉN, H. (1984). The rate of energy and resin contents in the selective feeding of pine needles by the capercaillie. *Annales Zoologica Fennici* **21,** 435-439.

LOCKIE, J. (1955). The breeding habits of short-eared owls after a vole plague. *Bird Study* **2,** 53-69.

LOCKIE, J. D. (1964). The breeding density of the golden eagle and fox in relation to food supply in Wester Ross, Scotland. *Scottish Naturalist* **71,** 67-77.

LOVE, J. A. (1983). *The return of the sea eagle.* Cambridge University Press, Cambridge.

LOVE, J. A. (1987). The reintroduction of the sea eagle to Scotland. In *Breeding and management in birds of prey,* ed. D. J. Hill, 133-144. University of Bristol, Bristol.

LOW, A. J. (1985). *Guide to upland restocking practice.* Forestry Commission Leaflet 84. HMSO, London.

LOW, A. J. (1986). *Use of broadleaved species in upland forests.* Forestry Commission Leaflet 88. HMSO, London.

LOW, A. J. (1987). Sitka spruce silviculture in Scottish forests. *Proceedings of the Royal Society of Edinburgh* **93 B,** 93-106.

LYNCH, J. F. and WHIGHAM, D. F. (1984). Effects of forest fragmentation on breeding bird communities in Maryland, USA. *Biological Conservation* **28,** 287-324.

MACARTHUR, R. (1974). Patterns of terrestrial bird communities. In, *Avian biology,* eds D. S. Farner and J. R. King, 189-221. Academic Press, New York.

MACARTHUR, R. H. and MACARTHUR, J. W. (1961). On bird species diversity. *Ecology* **42,** 594-598.

MACARTHUR, R. H., MACARTHUR, J. W. and PREER, J. (1962). On bird species diversity: II. Prediction of bird census from habitat measurements. *American Naturalist* **96,** 167-174.

MACARTHUR, R. H., RECHER, H. and CODY, M. (1966). On the relation between habitat selection and species diversity. *American Naturalist* **100,** 319-332.

MACCLINTOCK, L., WHITCOMB, R. F. and WHITCOMB, B. L. (1977). Island biogeography and 'habitat islands' of eastern forests. 2. Evidence for the value of corridors and minimisation of isolation in preservation of biotic diversity. *American Birds* **31,** 6-16.

MCELVEEN, J. D. (1979). The edge effect on a forest bird community in north Florida. *Proceedings of the Annual Conference of the South East Association of Fish and Wildlife Agencies* **31,** 212-215.

MCINTOSH, R. (1984). *Phosphate fertilisers in upland forestry – types, application rates and placement methods.* Research Information Note 89/84/SILN. Forestry Commission, Edinburgh.

MAGURRAN, A. E. (1988). *Ecological diversity and its measurement.* Croom Helm, London.

MANNAN, R. W., MESLOW, E. C. and WIGHT, H. M. (1980). Use of snags by birds in Douglas fir forest, Western Oregon. *Journal of Wildlife Management* **44,** 787-797.

MARGALEF, R. (1968). *Perspectives in ecological theory.* University of Chicago Press, Chicago.

MARQUISS, M. and NEWTON, I. (1981). The goshawk in Britain. *British Birds* **75,** 243-260.

MARQUISS, M., NEWTON, I. and RATCLIFFE, D. A. (1978). The decline of the raven *(Corvus corax)* in relation to afforestation in southern Scotland and northern England. *Journal of applied Ecology* **15,** 129-144.

MARQUISS, M., RATCLIFFE, D. A. and ROXBURGH, R. (1985). The numbers, breeding success and diet of golden eagles in southern Scotland in relation to changes in land use. *Biological Conservation* **33,** 1-17.

MASSEY, M. E. (1974). The effect of woodland structure on breeding bird communities in sample woods in south central Wales. *Nature in Wales* **14,** 95-105.

MEARNS, R. (1983). The status of the raven in southern Scotland and Northumbria. *Scottish Birds* **12,** 211-218.

MERRIE, T. D. H. (1979). Success of artificial island nest sites for divers. *British Birds* **72,** 32-33.

MESLOW, E. C. and WIGHT, H. M. (1975). *Avifauna and succession in Douglas fir forests of the Pacific Northwest.* USDA Forestry Service, General Technical Report WO-1, 266-271.

MILES, J. (1986). What are the effects of trees on soil? In, *Trees and wildlife in the Scottish uplands,* ed. D. Jenkins, 55-62. Institute of Terrestrial Ecology, Huntingdon.

MILLER, H. G. and MILLER, J. D. (1987). Nutritional requirements of Sitka spruce. *Proceedings of the Royal Society of Edinburgh* **93B,** 75-83.

MILLER, K. F. (1985). *Windthrow hazard classification.* Forestry Commission Leaflet 85. HMSO, London.

MILLS, D. H. (1980). *The management of forest streams.* Forestry Commission Leaflet 78. HMSO, London.

MOORE, N. W. (1957). The past and present status of the buzzard in the British Isles. *British Birds* **50,** 173-197.

MOORE, N. W. and HOOPER, M. D. (1975). On the number of bird species in British woods. *Biological Conservation* **8,** 239-250.

MORRISON, M. L. and MESLOW, E. C. (1983). *Avifauna associated with early growth vegeta-*

tion on clearcuts in the Oregon Coast Ranges. USDA Forest Service Research Paper PNW-305.

MOSS, D. (1978a). Diversity of woodland songbird populations. *Journal of Animal Ecology* **47,** 521-527.

MOSS, D. (1978b). Song-bird populations in forestry plantations. *Quarterly Journal of Forestry* **72,** 5-14.

MOSS, D. (1979). Even-aged plantations as a habitat for birds. In, *The ecology of even-aged forest plantations,* eds E. D. Ford, D. C. Malcolm and J. Atterson, 413-427. Institute of Terrestrial Ecology, Cambridge.

MOSS, D., TAYLOR, P. N. and EASTERBEE, N. (1979a). The effects on song-bird populations of upland afforestation with spruce. *Forestry* **52,** 129-147.

MOSS, R., WEIR, D. and JONES, A. (1979b). Capercaillie management in Scotland. In, *Woodland grouse 1978,* ed. Lovel, T.W.I., 140-155. World Pheasant Association, Suffolk.

NATURE CONSERVANCY COUNCIL (1985). *Eleventh annual report, covering the period 1 April 1984-31 March 1985.* Nature Conservancy Council, Peterborough.

NATURE CONSERVANCY COUNCIL (1986). *Nature conservation and afforestation in Britain.* Nature Conservancy Council, Peterborough.

NETHERSOLE-THOMPSON, D. (1975). *Pine crossbills.* Poyser, Berkhamsted.

NETHERSOLE-THOMPSON, D. and NETHERSOLE-THOMPSON, M. (1979). *Greenshanks.* Poyser, Berkhamsted.

NETHERSOLE-THOMPSON, D. and WATSON, A. (1974). *The Cairngorms, their natural history and scenery.* Collins, London.

NEWTON, I. (1972a). *Finches.* Collins, London.

NEWTON, I. (1972b). Birds of prey in Scotland: Some conservation problems. *Scottish Birds* **7,** 5-23.

NEWTON, I. (1973). Egg breakages and breeding failures in British merlins. *Bird Study* **20,** 241-244.

NEWTON, I. (1979a). *Population ecology of raptors.* Poyser, Berkhamsted.

NEWTON, I. (1979b). Effects of human persecution on European raptors. *Raptor Research* **13,** 65-78.

NEWTON, I. (1983). Birds and forestry. In, *Forestry and conservation,* ed. E. H. M. Harris, 21-30. Royal Forestry Society, Tring.

NEWTON, I. (1986a). Principles underlying bird numbers in Scottish woodlands. In, *Trees and wildlife in the Scottish uplands,* ed. D. Jenkins, 121-128. Institute of Terrestrial Ecology, Huntingdon.

NEWTON, I. (1986b). *The sparrowhawk.* Poyser, Berkhamsted.

NEWTON, I. and MOSS, D. (1981). Factors affecting the breeding of sparrowhawks and the occurrence of their song-bird prey in woodlands. In, *Forest and woodland ecology,* eds F. T. Last and A. F. Gardiner, 125-131. Institute of Terrestrial Ecology, Cambridge.

NEWTON, I., MEEK, E. and LITTLE, B. (1978). Breeding ecology of the merlin in Northumberland. *British Birds* **71,** 376-398.

NEWTON, I., MARQUISS, M., WEIR, D. N. and MOSS, D. (1979). Spacing of sparrowhawk nesting territories. *Journal of Animal Ecology* **46,** 425-441.

NEWTON, I., DAVIS, P. E. and MOSS, D. (1981a). Distribution and breeding of red kites in relation to land use in Wales. *Journal of applied Ecology* **18,** 173-186.

NEWTON, I., ROBSON, J. E. and YALDEN, D. W. (1981b). Decline of the merlin in the Peak District. *Bird Study* **28,** 225-234.

NEWTON, I., BOGAN, J., MEEK, E. R. and LITTLE, B. (1982a). Organochlorine compounds and shell thinning in British merlins. *Falco columbarius. Ibis* **124,** 328-355.

NEWTON, I., DAVIS, P. and DAVIS J. (1982b). Ravens, buzzards and land-use in Wales. *Journal of applied Ecology* **19,** 681-706.

NEWTON, I., MEEK, E. and LITTLE, B. (1986a). Population and breeding of Northumbrian merlins. *British Birds* **79,** 155-170.

NEWTON, I., WYLLIE, I. and MEARNS, R.

(1986b). Spacing of sparrowhawks in relation to food supply. *Journal of Animal ecology* **55,** 361-370.

NIEMI, G. J. and HANOWSKI, J. M. (1984). Effects of a transmission line on bird populations in the Red Lake peatland, northern Minnesota. *Auk* **101,** 487-498.

NILES, J. (1971). Forestry and conservation. 2. Strictly for the birds. *Journal of the Devon Trust for Nature Conservation* **3,** 80-85.

NORTHCOTE, E. M. (1980). Some Cambridgeshire Neolithic to Bronze Age birds and their presence or absence in England in the late-glacial and early Flandrian. *Journal of Archaeological Science* **7,** 379-383.

ODUM, E. P. (1950). Bird populations of the Highlands (North Carolina) Plateau in relation to plant succession and avian invasion. *Ecology* **31,** 587-605.

ODUM, E. P. (1971). *Fundamentals of ecology.* W. B. Saunders Co., Philadelphia.

OELKE, M. (1966). 35 years of breeding bird census work in Europe. *Audubon Field Notes* **20,** 635-642.

OPDAM, P., RIJSDIJK, G. and HUSTINGS, F. (1985). Bird communities in small woods in an agricultural landscape: Effects of area and isolation. *Biological Conservation* **34,** 333-352.

ORMEROD, S. J., TYLER, S. J. and LEWIS, J. M. S. (1985). Is the breeding distribution of dippers influenced by stream acidity? *Bird Study* **32,** 33-39.

O'SULLIVAN, P. E. (1977). Vegetation history of the native pinewoods. In, *Native pinewoods of Scotland,* eds R. G. H. Bunce and J. N. R. Jeffers, 60-69. Institute of Terrestrial Ecology, Cambridge.

PALMER, C. E. (1976). *The capercaillie.* Forestry Commission Forest Record 109. HMSO, London.

PARSLOW, J. (1973). *Breeding birds of Britain and Ireland.* Poyser, Berkhamsted.

PARSONS, A. J. (1976). Birds of a Somerset wood. *Bird Study* **23,** 287-293.

PEACE, T. R. (1962). *Pathology of trees and shrubs.* Clarendon Press, Oxford.

PEARS, N. V. (1969). Post-glacial tree-lines of the Cairngorm Mountains, Scotland: some modifications based on radiocarbon dating. *Transaction of the Botanical Society of Edinburgh* **40,** 536-544.

PEARS, N. V. (1975). Tree stumps in Scottish hill peats. *Scottish Forestry* **29,** 255-259.

PEARSON, D. L. (1975). The relation of foliage complexity to ecological diversity of three Amazonian bird communities. *Condor* **77,** 453-466.

PECK, K. M. (1989). Tree species preferences shown by foraging birds in a forest plantation in northern England. *Biological Conservation* **48,** 41-57.

PEET, R. K. (1974). The measurement of species diversity. *Annual Review of Ecology and Systematics* **5,** 285-307.

PEMBERTON, J. E. (1981). *Birdwatcher's yearbook 1982.* Buckingham Press, Buckingham.

PENNIE, I. D. (1950-51). The history and distribution of capercaillie in Scotland. *Scottish Naturalist* **62,** 65-87; **62,** 157-178; **63,** 4-17.

PEPPER, H. W. and TEE, L. A. (1986). *Forest fencing.* Forestry Commission Leaflet 87. HMSO, London.

PERRINS, C. M. (1979). *British tits.* Collins, London.

PETERKEN, G. F. (1987). Natural features in the management of upland conifer forests. *Proceedings of the Royal Society of Edinburgh* **93B,** 223-234.

PETTY, S. J. (1985). A negative response of kestrels *Falco tinnunculus* to nestboxes in upland forests. *Bird Study* **32,** 194-195.

PETTY, S. J. (1987a). The design and use of a nestbox for tawny owls *Strix aluco* in upland forests. *Quarterly Journal of Forestry* **81,** 103-109.

PETTY, S. J. (1987b). Breeding of tawny owls (*Strix aluco*) in relation to their food supply in an upland forest. In, *Breeding and management in birds of prey,* ed. D. J. Hill, 167-179. University of Bristol, Bristol.

PETTY, S. J. (1988). The management of raptors in upland forests. In, *Wildlife management in forests*, ed. D. C. Jardine, 7-23. Institute of Chartered Foresters, Edinburgh.

PETTY, S. J. (1989). *Goshawks: their status, requirements and management.* Forestry Commission Bulletin 81. HMSO, London.

PETTY, S. J. (1989). Productivity and density of tawny owls (*Strix aluco*) in relation to the structure of a British spruce forest. *Annales Zoologici Fennici* **26,** 227-233.

PETTY, S. J. and ANDERSON, D. (1986). Breeding by hen harriers *Circus cyaneus* on restocked sites in upland forests. *Bird Study* **33,** 177-178.

PHILLIPS, D. H. and BURDEKIN, D. A. (1982). *Diseases of forest and ornamental trees.* Macmillan Press Ltd., Basingstoke.

PIANKA, E. R. (1974). *Evolutionary ecology.* Harper and Row, New York.

PIELOU, E. C. (1966). Species diversity and pattern diversity in the study of ecological succession. *Journal of Theoretical Biology* **10,** 370-383.

PYATT, D. G. (1970). *Soil groups of upland forests.* Forestry Commission Forest Record 71. HMSO, London.

PYATT, D. G. (1982). *Soil classification.* Research Information Note 68/82/SSN. Forestry Commission, Edinburgh.

RALPH, C. J. and SCOTT, J. M. (1981). *Estimating numbers of terrestrial birds.* Cooper Ornithological Society, Allen Press, Kansas.

RANKIN, G. D. and TAYLOR, I. R. (no date). *Changes within afforested but unplanted ground: birds.* NCC Contract number HF3/03/175.

RANNEY, J. W., BRUNER, M. C. and LEVINSON, J. B. (1981). The importance of edge in the structure and dynamics of forest islands. In, *Forest island dynamics in man-dominated landscapes,* eds R. L. Burgess and D. M. Sharpe, 67-96. Springer-Verlag, New York.

RATCLIFFE, D. (1980). *The peregrine falcon.* Poyser, Calton.

RATCLIFFE, D. A. (1976). Observations on the breeding of the golden plover in Great Britain. *Bird Study* **23,** 63-116.

RATCLIFFE, D. A. (1984). Post-medieval and recent changes in the British vegetation: The culmination of human influences. *New Phytologist* **98,** 73-100.

RATCLIFFE, P. R. (1985). *Glades for deer control in upland forests.* Forestry Commission Leaflet 86. HMSO, London.

RATCLIFFE, P. R. and PETTY, S. J. (1986). The management of commercial forests for wildlife. In, *Trees and wildlife in the Scottish uplands,* ed. D. Jenkins, 177-187. Institute of Terrestrial Ecology, Huntingdon.

REED, T. M., LANGSLOW, D. R. and SYMONDS, F. L. (1983). The breeding waders of the Caithness flows. *Scottish Birds* **12,** 180-186.

RITCHIE, J. (1920). *The influence of man on animal life in Scotland.* Cambridge University Press, Cambridge.

ROBBINS, C. S. (1979). *Effect of forest fragmentation on bird populations.* USDA Forestry Service, General Technical Report NC-51, 198-212.

ROBBINS, C. S. (1980). Effects of forest fragmentation on breeding bird populations in the Piedmonts of the mid-Atlantic region. *Atlantic Naturalist* **33,** 31-36.

ROLLINSON, T. J. D. (1985). *Thinning control.* Forestry Commission Booklet 54. HMSO, London.

ROTH, R. R. (1976). Spatial heterogeneity and bird species diversity. *Ecology* **57,** 773-782.

RØV, N. (1975). Breeding bird community structure and species diversity along an ecological gradient in deciduous forest in western Norway. *Ornis Scandinavica* **6,** 1-14.

ROWAN, A. A. (1979). *Terrain classification.* Forestry Commission Forest Record 114. HMSO, London.

SALE, J. S. P., TABBUSH, P. M. and LANE, P. B. (1986). *The use of herbicides in the forest.* Forestry Commission Booklet 51. Forestry Commission, Edinburgh.

SAMMALISKO, L. (1957). The effect of

woodland–open peatland edge on some peatland birds in South Finland. *Ornis Fennica* **34,** 81-89.

SAUROLA, P. (1978). Artificial nest construction in Europe. In, *Birds of prey management techniques,* ed. T. A. Geer, 72-81. British Falconers Club, Oxford.

SAVILL, P. S. and EVANS, J. (1986). *Plantation silviculture in temperate regions.* Clarendon Press, Oxford.

SCOTT, T. M. and KING, C. J. (1974). *The large pine weevil and black pine beetles.* Forestry Commission Leaflet 58. HMSO, London.

SHACKLETON, N. J. and TURNER, C. (1967). Correlations between marine and terrestrial Pleistocene successions. *Nature, London* **216,** 1079.

SHANNON, C. E. and WEAVER, W. (1949). *The mathematical theory of communication.* University of Illinois Press, Urbana.

SHARROCK, J. T. R. (1976). *The atlas of breeding birds in Britain and Ireland.* Poyser, Berkhamsted.

SHAW, G. (1976). The breeding bird community of the hillside oakwoods of Loch Lomondside. *Western Naturalist* **5,** 41-51.

SHAWYER, C. R. (1987). *The barn owl in the British Isles: its past, present and future.* Hawk Trust, London.

SHUGART, H. H., ANDERSON, S. H. and STRAND, R. H. (1975). *Dominant patterns in bird populations of the eastern deciduous forest biome.* USDA Forestry Service, General Technical Report WO-1, 90-95.

SHUGART, H. H. and JAMES, D. (1973). Ecological succession of breeding bird populations in northwestern Arkansas. *Auk* **90,** 62-67.

SIMMS, E. (1971). *Woodland birds.* Collins, London.

SITTERS, H. P. (1986). Woodlarks in Britain. *British Birds* **79,** 105-116.

SMART, N. and ANDREWS, J. (1985). *Birds and broadleaves handbook: a guide to further the conservation of birds in broadleaved woodland.* RSPB, Sandy.

SMITH, A. G. and PILCHER, J. R. (1973). Radiocarbon dates and vegetational history of the British Isles. *New Phytologist* **72,** 903-914.

SMITH, B. D. (1980). The effects of afforestation on the trout of a small stream in southern Scotland. *Fish Management* **11,** 39-57.

SOUTHERN, H. N. (1970). The natural control of a population of tawny owls (*Strix aluco*). *Journal of Zoology (London)* **162,** 197-285.

SOUTHWOOD, T. R. E. (1961). The number of species of insect associated with various trees. *Journal of Animal Ecology* **30,** 1-8.

SOUTHWOOD, T. R. E. (1978). *Ecological methods.* Chapman and Hall, London.

SPENCER, R. and THE RARE BIRDS BREEDING PANEL (1986). Rare breeding birds in the UK in 1984. *British Birds* **79,** 470-495.

STAINES, B. W., PETTY, S. J. and RATCLIFFE, P. R. (1987). Sitka spruce as a habitat for birds and mammals. *Proceedings of the Royal Society of Edinburgh* **93B,** 169-181.

STOAKLEY, J. T. (1979). *Pine beauty moth.* Forestry Commission Forest Record 120. HMSO, London.

STOAKLEY, J. T. (1986). Protecting the timber resource. In, *Trees and wildlife in the Scottish uplands,* ed. D. Jenkins, 108-111. Institute of Terrestrial Ecology, Huntingdon.

STORAAS, T. and WEGGE, P. (1987). Nesting habitats and nest predation in sympatric populations of capercaillie and black grouse. *Journal of Wildlife Management* **51,** 167-172.

STRELKE, W. K. and DICKSON, J. G. (1980). Effect of forest clear cut edge on breeding birds in east Texas. *Journal of Wildlife Management* **44,** 559-568.

STROUD, D. A. and REED, T. M. (1986). The effects of plantation proximity on moorland breeding waders. *Wader Study Group Bulletin* **46,** 25-28.

STROUD, D. A., REED, T. M., PIENKOWSKI, M. W. and LINDSEY, R. A. (1987). *Birds, bogs and forestry: the peatlands of Caithness and Sutherland.* Nature Conservancy Council, Peterborough.

SUESS, H. (1967). Bristle-cone pine calibration of the radiocarbon time-scale from 4100 BC to 1500 BC. In, *Radiocarbon dating and methods of low-level counting*. International Atomic Energy Agency, Vienna.

SUESS, H. (1970). Bristle-cone pine calibration of the radiocarbon time-scale from 5200 BC to the present. *Nobel Symposium* **12,** 303.

SYKES, J. M., LOWE, V. P. W. and BRIGGS, D. R. (1985). Changes in plants and animals in the first 10 years after upland afforestation. *NERC/ITE Annual Report 1984,* 16-21.

SZARO, R. S. and BALDA, R. P. (1979). *Effects of harvesting ponderosa pine on non-game bird populations*. USDA Research Paper RM-212.

TALLIS, J. H. and SWITSUR, V. R. (1983). Forest and moorland in the mid-Flandrian period. I. Macrofossil evidence of the former forest cover. *Journal of Ecology* **71,** 585-600.

THOMPSON, D. A. (1978). *Forest ploughs*. Forestry Commission Leaflet 70. HMSO, London.

THOMPSON, D. A. (1979). *Forest drainage schemes*. Forestry Commission Leaflet 72. HMSO, London.

THOMPSON, D. A. (1984). *Ploughing of forest soils*. Forestry Commission Leaflet 71. HMSO, London.

THOMPSON, D. B. A., STROUD, D. A. and PIENKOWSKI, M. W. (1988). Afforestation and upland birds: consequences for population ecology. In, *Ecological change in the uplands,* eds M. B. Usher and D. B. A. Thompson, 237-259. Blackwell Scientific Publications, Oxford.

TICEHURST, N. F. (1920). On the former abundance of the kite, buzzard and raven in Kent. *British Birds* **14,** 34-37.

TURČEK, F. J. (1957). The bird succession in the conifer plantations on mat-grass land in Slovakia (SSR). *Ibis* **99,** 587-593.

TURNER, C. and WEST, R. G. (1968). The subdivision and zonation of interglacial periods. *Eiszeitalter und Gegenwart* **19,** 93-101.

TURNER, J. and HODGSON, J. (1979). Studies of the vegetational history of the northern Pennines. I. Variation in the composition of the early Fladrian forests. *Journal of Ecology* **62,** 629-646.

TURNER, J. and HODGSON, J. (1983). Studies of the vegetational history of the northern Pennines. III. Variations in the composition of the mid-Flandrian forests. *Journal of Ecology* **71,** 95-118.

ULFSTRAND, S. (1975). Bird flocks in relation to vegetation diversification in a south Swedish coniferous plantation during winter. *Oikos* **26,** 65-73.

ULMAMEN, I. and VALSTE, J. (1965). Sodankylan Kiotelaiskairan linnustosta 1962-64. *Lintumies* **1,** 51-56.

USHER, M. V. (1983). Species diversity: a comment on a paper by W. B. Yapp. *Field Studies* **5,** 825-832.

VÄISÄNEN, R. A. and RAUHALA, P. (1983). Succession of land bird communities on large areas of peatland drained for forestry. *Annales Zoologica Fennici* **20,** 115-127.

VILLAGE, A. (1980). *The ecology of the kestrel (Falco tinnunculus) in relation to vole abundance of Eskdalemuir, south Scotland*. PhD Thesis, Edinburgh University.

VILLAGE, A. (1981). The diet and breeding of long-eared owls in relation to vole numbers. *Bird Study* **28,** 215-224.

VILLAGE, A. (1983). The role of nest-site availability and territorial behaviour in limiting the breeding density of kestrels. *Journal of Animal Ecology* **52,** 635-647.

VON HAARTMAN, L. (1971). Population dynamics. In, *Avian biology, Volume 1,* eds D. S. Farner and J. R. King, 391-459. Acdemic Press, London.

VOOUS, K. H. (1978). The Scottish crossbill: *Loxia scotica*. *British Birds* **71,** 3-10.

WATSON, A. (1969). Preliminary counts of birds in central highland pine woods. *Bird Study* **16,** 158-163.

WATSON, D. (1977). *The hen harrier*. Poyser, Berkhamsted.

WATSON, J. (1979). Food of merlins nesting in young conifer forest. *Bird Study* **26,** 253-258.

WATSON, J., LANGSLOW, D. R. and RAE, S. R. (1987). *The impact of land use changes on golden eagles (Aquila chrysaetos) in the Scottish Highlands.* Nature Conservancy Council, Peterborough.

WEST, R. G. (1968). *Pleistocene geology and biology.* Longmans and Green, London.

WEST, R. G. (1970). Pollen zones in the Pleistocene of Great Britain and their correlation. *New Phytologist* **69,** 1179-1183.

WHITCOMB, R. F. (1977). Island biogeography and 'habitat islands' of eastern forest. I. Introduction. *American Birds* **31,** 3-5.

WHITCOMB, R. F., ROBBINS, C. S., LYNCH, J. F., WHITCOMB, B. L., KLIMKIEWICZ, M. K. and BYSTRAK, D. (1981). Effect of forest fragmentation on avifauna of the eastern deciduous forest. In, *Forest island dynamics in man-dominated landscapes,* eds R. Burgess and D. M. Sharp, 125-205. Springer-Verlag, New York.

WILLIAMSON, K. (1964). Bird census work in woodland. *Bird Study* **11,** 1-22.

WILLIAMSON, K. (1968). Bird communities in the Malham Tarn region of the Pennines. *Field Studies* **2,** 651-668.

WILLIAMSON, K. (1969). Bird communities in woodland habitats in Wester Ross, Scotland. *Quarterly Journal of Forestry* **63,** 305-328.

WILLIAMSON, K. (1971). The breeding birds of a century-old grove of coast and sierra redwoods in Wales. *Quarterly Journal of Forestry* **65,** 109-121.

WILLIAMSON, K. (1972). Breeding birds of Ariundle oakwood forest nature reserve. *Quarterly Journal of Forestry* **66,** 243-255.

WILLIAMSON, K. (1974). Oak wood breeding bird communities in the Loch Lomond National Nature Reserve. *Quarterly Journal of Forestry* **68,** 9-29.

WILLIAMSON, K. (1975). Bird colonisation of new plantations on the moorland of Rhum, Inner Hebrides. *Quarterly Journal of Forestry* **69,** 157-165.

WILLIAMSON, K. (1976). Bird-life in the wood of Cree, Galloway. *Quarterly Journal of Forestry* **70,** 206-215.

WILLIAMSON, R. and WILLIAMSON, K. (1973). The bird community of yew woodland at Kingley Vale, Sussex. *British Birds* **66,** 12-23.

WILLSON, M. F. (1974). Avian community organisation and habitat structure. *Ecology* **55,** 1017-1029.

WOOLHOUSE, M. E. J. (1983). The theory and practice of the species-area effect applied to breeding birds of British woods. *Biological Conservation* **27,** 315-332.

YAPP, W. B. (1955a). The birds of Welsh high level oakwoods. *Nature in Wales* **1,** 161-166.

YAPP, W. B. (1955b). The succession of birds in developing Quercetum petraea. *North Western Naturalist* **26,** 58-67.

YAPP, W. B. (1962). *Birds and woods.* Oxford University Press, London.

YAPP, W. B. (1974). Birds of the northwest Highland birchwoods. *Scottish Birds* **8,** 16-31.

ZWICKEL, F. C. (1966). Winter food habitats of capercaillie in north-east Scotland. *British Birds* **59,** 325-336.

Printed in the United Kingdom for HMSO
Dd 8998078 C15 10/90