FORESTRY COMMISSION OCCASIONAL PAPER 40

Forestry Expansion – a study of technical, economic and ecological factors

Forests as Wildlife Habitat

J. Good, I. Newton, J. Miles, R. Marrs
and J.N. Greatorex-Davies
Institute of Terrestrial Ecology

Forestry Commission, Edinburgh

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INTRODUCTION

The increasing momentum of green politics throughout the EEC is boosting national and international pressure for more effective protection and conservation of the countryside. The change in public opinion which the green movement represents resulted from widespread belief that environmental standards on a wide front are actually or potentially in decline, and that action is required now to prevent further deterioration and to conserve healthy, unpolluted environments for humans and wildlife.

Modern British agriculture, with its high input high output economy, has been strongly criticised by environmentalists, including wildlife conservationists. It is now in a period of contraction because in Europe production exceeds demand. This provides opportunities for the expansion of forestry. Forestry's primary product, wood, is unlikely ever to be over-produced in Europe, and imports of wood and wood-based products costs the EC as a whole, and Britain in particular, a great deal of money (see Whiteman, 1991). Through depletion or destruction of natural forests elsewhere, wood imports also involve global environmental cost.

These factors support an expansion of forestry. However, there is much more to the issue than the economics of wood production, and the land potentially available to forestry as a result of changes in EC agricultural policies will not become afforested as a matter of course. As well as persuading landowners and investors that forestry is an attractive economic alternative to other land use options, the forestry lobby will need to convince decision makers that, among other things, the new forests will be environment-friendly, and hence politically acceptable.

This paper considers the demands for wildlife conservation in new forests, and the constraints which meeting them would place on timber production.

Background

Britain if left to nature would be heavily wooded, but now has only 10% of its land (2.3 million ha) forested. Only 300 000 ha (13%) of this forested area is semi-natural woodland. Nevertheless, our native flora and fauna still contains a high proportion of woodland species (Steele, 1975). Many of these, such as the woodland birds which frequent gardens (e.g. blue tit, blackbird), have been able to adapt to historic deforestation. Others have not, and are now restricted in their range by the availability of suitable woodland habitat (Peterken, 1981). It is the latter group, the woodland specialists, which by definition include many of the rarer species (e.g. oxlip, twinflower, capercaillie), which most need conserving. If their needs can be met in the new forests, most of the other, less discriminating species will also be provided for.

Those who wish to conserve woodland wildlife for its own sake look primarily to the Nature Conservancy Council (NCC), as the agent of Government, to act on their behalf. This the NCC does, mainly using powers derived from the Wildlife and Countryside Act 1981, the Nature Conservancy Council Act 1973, and the National Parks and Access to the Countryside Act 1949. The Government also has a direct role through the Secretary of State under EC Directive 79/409 on the Conservation of Wild Birds.

The NCC has, until recently, seen its main function as being 'the preservation and maintenance of the main types of native communities, with their associated floras and faunas' (Nature Conservancy Council,1977). This task has been achieved partly through the establishment of National Nature Reserves (NNRs) and more extensively through the designation of Sites of Special Scientific Interest (SSSIs). The amount of land afforded statutory protection in Britain ranges between 5.5% in England to 10.6% in Scotland (Table 1). The NNR network includes woods chosen to represent the range of native woodland types. While some NNRs are owned outright by the Crown, most remain in private ownership and are managed by the NCC through leases and agreements with landowners.

Table 1 Amounts of land (ha) with statutory protection in Great Britain

Country	*NNRs*	*SSSIs*	*Total area (ha)*	*Proportion protected (%)*
England	41 272	675 870	13 044 017	5.5
Scotland	112 039	722 973	7 877 226	10.6
Wales	12 172	117 943	2 076 662	6.3
Great Britain	165 483	1 576 786	22 997 906	7.6

SSSIs are not managed by the NCC. However, the Council is able to constrain forestry activities carried out by landowners or their agents in woodland SSSIs which might damage or destroy their conservation values. Management agreements may include payments calculated on the basis of profit foregone, in accordance with financial guidelines issued by the Government (DOE and MAFF Circular 4/83, Welsh Office Circular 6/83).

The costs of compensation in the establishment and maintenance of SSSIs, and the rent or purchase of NNRs, are the most tangible indicators in economic terms of the public demand for wildlife conservation in forests in Britain. Added to these should be the administrative and legal costs, labour costs (e.g. wardens) and material costs (e.g. fencing) associated with conservation of sites (Willis *et al.*, 1983).

While the NCC is the primary government body responsible for promoting nature conservation in forests, the Forestry Commission also has a duty for conservation (see next section). Many other special interest groups are also involved in nature conservation in forests. They include organisations representing individuals professionally concerned with native (including woodland) flora and fauna (notably the British Ecological Society), societies organised by amateurs concerned specifically for woodland conservation (e.g. the Woodland Trust), or with wildlife conservation more generally (e.g. the county naturalist's trusts). There are also the bodies (e.g. the Royal Society for the Protection of Birds) which are involved in the conservation of particular groups of plants or animals which occur in woodland. The existence of these organisations is proof of demand for wildlife conservation among the public, for they can only survive because individuals are willing to pay to support them.

Willingness to pay (WTP) may also be used as a means of assessing more general public demand for wildlife conservation. One approach to this which is widely used in economic analysis to determine the value of recreation experience generally, and which has been employed in wildlife resource evaluation, is the travel cost (TC) method (see Willis *et al.*, 1986 for a full discussion of the problems involved and attempts made to solve them).

Briefly, visits to see, experience or learn about wildlife involve a travel cost which is evidence of willingness to pay. However, it may be difficult to disentangle wildlife benefits from other reasons for visits, and location plays an important part in determining visitor preferences. Therefore a contingent valuation (CV) method may more comprehensively determine WTP. Contingent valuation involves assessing people's expressed preferences, by asking them directly how much they value wildlife rather than by inference as under the TC method. CV, unlike TC methods, can also be used to estimate the value placed on wildlife sites by non-visitors, e.g. by determining intentions of non-visitors to visit and the valuation they place on knowing that the site is there (***existence value***), that they have an ***option*** to visit, and that future generations will retain that opportunity (***bequest value***). The use of this method, and the possible biases involved, are discussed by Willis and Benson (1988).

Deciding how the various demands for wildlife conservation in forests may best be met is perhaps most appropriately done by considering the conservation values of various forestry options in different forest zones of the British Isles. This method is pertinent because regional (particularly east/west) differences in soils and climate determine the forestry options which are, in practice, available, and hence the potential for wildlife habitat creation.

Options considered here relate to woodland types (e.g. native broadleaves, native pine forest, spruce forest), forest design (location, size, structure, species), forest management (planting, weed, pest and disease control, thinning, felling, replanting) and the interrelationships between new forests and adjacent land uses.

Wildlife protection legislation in the UK and the EEC

Current UK wildlife legislation affects owners and managers of woodland as those of any other land cover. One of the most important pieces of legislation is the amendment of the Forestry Act 1967 in Section 4 of the Wildlife and Countryside (Amendment) Act 1985. This places a general duty on the Forestry Commissioners in discharging their functions, 'so far as may be consistent with the proper discharge of those functions', to 'endeavour to achieve a reasonable balance between (a) the development of afforestation, the management of forests and the production and supply of timber, and (b) the conservation and enhancement of natural beauty and the conservation of flora, fauna and geological or physiographic features of interest.' This means that this balance must be sought in any woodland planting approved by the Commissioners for grant aid after the passing of the 1985 Act or done in their own forests.

Apart from this specific duty of the Forestry Commissioners, woodlands are subject to the provisions of the Wildlife and Countryside Act 1981 the same as any other kind of land. This in particular:

1. lists various species of plant which must not be picked, uprooted or destroyed, and of wild animals and birds (*sic*) which must not be taken, injured or killed;

2. allows the Nature Conservancy Council to declare areas (termed sites by the NCC) of special scientific interest (SSSIs);

3. allows the Secretary of State to make provision by order of areas of special protection (SPAs) for birds, which in particular allows the UK to meet its obligations under the EC Council Directive of April 1979 on the Conservation of Wild Birds (Directive 79/409/EC).

SSSIs have only limited legal back-up; they are a restraint on landowners, but not an absolute constraint. Nevertheless, SSSIs are a mainstay of wildlife conservation in the UK and they will remain.

The growing experience of the EC member states with the way in which the European Commission's lawyers interpret the Wild Birds Directive, suggests that an SPA is a much more rigorous restraint to developments which are counter to the interests of wildlife than is an SSSI. A member state can be challenged in the European Court for failure to achieve any of the Directive's aims, though few aspects of the Directive have as yet been reviewed and interpreted by the Court.

In September 1988 the EC Commission published a proposal for a Directive for the Conservation of Natural and Seminatural Habitats and of Wild Flora and Fauna, couched in similar terms to the Wild Birds Directive. It seeks to give all other species of wildlife and their habitats the same protection currently given to birds, with annexes listing species and habitat types to be given special protection. All member states have approved the idea behind the proposal. However, the wording of the text and the content of the annexes are still being discussed, so it is currently unclear what obligations may stem from the eventual directive. It might, however, strengthen the protection given to the SSSI network.

At the wider international level, individual sites or species may be protected under the RAMSAR convention (wetlands), the World Heritage Convention or the Bonn Convention on Migratory Species (Lyster, 1985).

METHODS OF ASSESSING THE WILDLIFE CONSERVATION VALUES OF NEW FORESTS

The NCC's Nature Conservation Review (Ratcliffe, 1977) was based on the use of ten criteria for site evaluation – size, diversity, naturalness, rarity, typicalness, fragility, recorded history, position in an ecological or geographical unit, potential value and intrinsic appeal. These and other criteria have been critically evaluated by various workers (Adams and Rose 1978; Margules and Usher, 1981; Spellerberg, 1981; Goldsmith, 1983). Much of the criticism has been based on the fact that the methods used contain a mixture of criteria which can be more or less precisely defined (e.g. site size, species diversity, relative rarity) and others (e.g. intrinsic appeal) based on social or aesthetic considerations which are more difficult or impossible to measure.

Another major problem concerns the common failure of evaluation systems to define precisely the purpose of the evaluation (Goldsmith, 1983). This difficulty is highlighted when, as in most cases, different users, or potential users, of the same site, have differing requirements. In such cases, different sets of criteria are required to evaluate the site for the various requirements (Adams and Rose, 1978). This has rarely been done when selecting sites primarily, but not exclusively for their nature conservation values.

Bearing these arguments in mind, since certain criteria are commonly used by those officially determining the conservation values of sites, it is necessary to consider them in more detail. The three criteria which seem to be most often used are naturalness, diversity and rarity.

Naturalness

Highest value is generally given to habitats nearest their natural state and least to those most severely disturbed by human activity and dominated by introduced species (NCC 1986). Using this valuation, new forests can be assessed by comparing them with the types which are natural to the area. Have they the same dominant tree species, the same kind of structure, the same range of plants and animals occupying similar niches? If not, to what extent do they resemble the ideal and how do they differ? Can such differences be made good by management or through the passage of time, or are they intrinsic and unalterable?

Changes in the naturalness of an area caused by changes in land use are of great concern to conservationists. It is not only the naturalness of the new forest which is important, but also the loss of naturalness resulting from the replacement of pre-existing habitats. Using this assessment, a new oakwood established on arable farmland of little conservation interest might be judged to have a higher relative conservation value than one planted on unimproved, species-rich grassland (NCC, 1986).

A major problem in assessing naturalness arises from the fact that almost all land in Britain has been affected, directly or indirectly, by man. Thus most vegetation, including much of the upland areas planted with conifers, was maintained in an artificial state by man's activities, notably in the uplands by sheep grazing and fire, before afforestation. It can be argued, therefore, that to replace heather moorland with exotic conifer forests is merely to supplant one unnatural vegetation type with another, and that the loss of naturalness is therefore much less than it seems, particularly since much of the land would have been forested in ancient times.

This argument cannot be sustained, at least in relation to conventional modern forestry, because of the ecological consequences arising from the fact that the new forests bear so little structural relationship to the ancient forests which preceded them, and which still exist in fragments for comparison. The dwarf shrub heath derived from the ground vegetation of the ancient woodland has a similar botanical composition to the ground flora of these ancient forest fragments. It persists, if at all, in only a much impoverished form within the forest blocks after conventional afforestation.

The reduction in naturalness caused by afforestation is even clearer in the case of the wetter upland blanket bogs and raised mires, because they are naturally treeless and bear climatically maintained climax communities.

Diversity Diversity is widely used to assess wildlife conservation value. When considering ecosystems, diversity refers to species. Value is generally considered to be high when the total number and relative abundance of species approaches those which would occur naturally (in the absence of human disturbance). At the larger (landscape, regional, national, international) scales, highest value is given to those areas which contain the greatest diversity of appropriate habitats (NCC, 1986).

The words 'appropriate' and 'naturally' in these definitions are of crucial importance. They indicate that what conservationists favour is <u>natural</u> diversity, not diversity for its own sake. Misunderstandings have repeatedly arisen between foresters and conservationists on this point. Foresters have correctly argued that planting non-native forests in areas dominated by semi-natural ecosystems increases both habitat and species diversity. Some have wrongly concluded from this that it is perverse of conservationists, who claim to favour diversity, not to welcome this increased variety (Garthwaite, 1983). The conservationists' argument is that the increased local species diversity brought about, for example, by largescale Sitka spruce afforestation of moorland, is generally inadequate compensation for the semi-natural habitats that are lost (NCC, 1986). This argument seems to hold for birds, as is shown by the relative paucity of woodland compared with moorland bird species in the annexes of specially protected species in the Wildlife and Countryside Act 1981 and the 1979 EC Wild Birds Directive. However, it does not hold for mammals, and is doubtful for plants.

The NCC are prepared to welcome the creation of artificial habitats only where there is an overall gain in conservation value of an area. This is said to be likely only where the existing habitats are not of special interest, or where the area of new habitat is so small in proportion to the remaining unafforested area as not to influence it appreciably (NCC, 1986).

Rarity Rarity, like diversity, can be considered at different scales – there are rare species, rare communities and rare ecosystems. Generally nature conservationists consider rarity at all these levels as being a relevant measure of value, because rarity is one important aspect of natural diversity (NCC, 1986).

The main concern of the NCC regarding rarity and the effects of afforestation in Britain is that the areas where most tree planting has taken place include habitats which, while relatively common here, are rare internationally. The plant and animal communities of raised and blanket bogs and heather moorland are especially important in this regard. It is argued that even small reductions in these habitats need to be resisted because the small remaining areas in other countries are also threatened and many of the characteristic birds and other animals of these open habitats require large areas.

These arguments would be more easily evaluated if we knew more about the minimum areas of habitats required to support viable communities of the range of dependent plants and animals, and of the effects of afforestation. It is clear, for example, that it is not just the total area of undisturbed habitat which remains after afforestation that is important, but also its spatial distribution in relation to the forested area: e.g. are few large unplanted areas preferable to many smaller areas? Unfortunately, reliable answers to these and similar questions are available for very few species, notably raptorian birds, and even in their case it has proved difficult to be precise in detailing the effects of afforestation (Marquiss *et al.*, 1978). It is sensible in the absence of such information to err on the side of caution, although the degree of prudence required is a matter for debate.

The 'profit and loss' approach

In practice, the best way of assessing potential wildlife conservation values of new forests is to consider the 'profit and loss' that will follow their introduction, so that the net change in conservation value can be determined. Clearly some existing habitats have more conservation value than others; for example, the peat bogs of the Flow country are worth more to conservation than the arable lands of southern England. Therefore the conservation losses will be greater and the gains less in afforesting the Flow country. This being so, on balance it is preferable to afforest arable land in southern England.

In assessing wildlife conservation profit and loss from afforestation it is necessary to take account of the dynamic nature of the forest ecosystem. The different successional stages of forest growth have different conservation value. For example, the period between tree planting and canopy closure on upland sheepwalk is typified by luxuriant ground vegetation, which promotes large populations of voles and their predators (Charles, 1981; Newton and Moss, 1981). Vole densities reach up to 100 times those on the original sheepwalk. These high vole densities lead to a spectacular influx of short-eared and other owls, kestrels and other raptors, which for a time breed at exceptionally high densities. In any one new forest this bonanza lasts about 10 years, ending as the canopy closes and eliminates the grass on which the voles mainly depend.

Comparative assessments of habitat quality is clearly dependent on scale. Account must be taken of:

1. International importance: e.g. peat bogs were once common in Britain but have always been extremely rare on a world scale.

2. National importance: e.g. Calluna moorland is characteristically a British habitat; it is essential for the conservation of the red grouse and important for other species.

3. Regional importance: e.g. a habitat may be common in one part of the country, but rare or much valued in another.

4. Local importance: a given area may have particular scenic or local conservation value in its present form (e.g. many local naturalist trust reserves).

Let us, for the sake of illustration, consider the conservation of heather moorland in more detail. This might be regarded as a plentiful habitat in the Scottish highlands, but as a scarce and much valued resource in Wales. Planting trees on heather moorland might, therefore, be more acceptable in Scotland than in Wales. The situation changes with time, however, and as an increased proportion of the given habitat is planted, then the conservation value of the remainder will increase.

Another factor which is likely to influence the balance of conservation gains and losses, and hence the public acceptance of any future afforestation scheme, is the type of forest created. An area of mixed native broadleaved trees would usually elicit much less hostility than a conifer plantation. We attempt to determine whether this is a fair reaction in the sections which follow.

NEW FORESTS IN RELATION TO NATIVE WOODLAND TYPES

Assessing the demand for forest wildlife habitat requires knowledge of the range of natural woodland types, their extent and distribution, and the wildlife they support. Unfortunately, for several reasons this information is not fully available. First, there is no one widely accepted published classification of British woodlands, so there can be no agreed assessment of the distribution for all types, even if a complete survey had been done, which it has not. Second, woodlands are dynamic entities which change with time, the amount and rate of change being increased by man's activities. Any piece of ground may support more than one kind of woodland, and past and present management often determines the current woodland type rather than natural successional processes. This can be easily confirmed by comparing the relative abundance of different types of woodland now and in the past. As recently as the 1930s a large proportion of British woodlands were classed as oakwoods (Tansley, 1939). This was an appropriate classification at the time, because coppice management with the continuous retention of oak standards was the commonest system over large areas. Since then the coppice system has declined almost to extinction, oak standards have been felled without replacement, and the coppice has grown away. Woods once dominated by oak now consist mainly of ash, maple, lime, birch, sycamore and hazel, and there is no appropriate Tansleyan class into which they can be meaningfully placed (Peterken, 1981).

Modern classifications (e.g. Peterken, 1981; Bunce, 1982; 1989) acknowledge the dynamic nature of woodlands and regard types as being, to varying degrees, indeterminate and interchangeable. They enable the range of possibilities for any particular site to be listed. A synopsis of Peterken's classification is given in Table 2. Distributions for most of the types are mapped in (Peterken, 1981), but these are, to a varying extent, incomplete. They provide only a very general guide to the range and frequency of each type.

Table 2 British woodland types as defined by (Peterken, 1981) and presented by (Rackman, 1986)

Ash – wych-elm woods

1. Limestone and chalk, southern
2. Limestone, northern
3. Wet clay
4. Wet sandy
5. Limestone, N.E. Midland
6. Clay on limestone, with sessile oak
7. Western valley

Ash – maple woods

8. Ash – maple on wet clay
9. Maple on wet clay
10. Ash – maple on wet light soils
11. Ash – maple on dry light soils
12. Ash – maple on dry heavy soils

Hazel – ash woods

13. Heavy acid soils, with pedunculate oak
14. Light acid soils, with pedunculate oak
15. Limestone, southern
16. Limestone, northern
17. Acid soils, with sessile oak

Ash – lime woods

18. Acid soils, with birch
19. With maple, eastern
20. With maple, western
21. With sessile oak

Oak – lime woods

22. With pedunculate oak
23. With sessile oak

Birch – oak woods

24. Highland, with sessile oak
25. Highland, with hazel and sessile oak
26. Highland, with pedunculate oak
27. Highland, with hazel and pedunculate oak
28. Lowland, with sessile oak
29. Lowland, with hazel and sessile oak
30. Kentish sessile oakwood on chalk
31. Lowland, with pedunculate oak
32. Lowland, with hazel and pedunculate oak

Alder woods

33. Acid valleys
34. Calcareous valleys
35. Peaty depressions
36. Calcareous springlines
37. Acid springlines
38. Plateaux
39. Slopes
40. Lowland, with bird-cherry
41. Highland, with bird-cherry

Beech woods

42. Acid with sessile oak
43. Acid with pedunculate oak
44. Dry calcareous, with pedunculate oak, ash lime, wych-elm
45. Damp calcareous, with ash and wych-elm
46. Dry calcareous, with ash and maple
47. Acid, with ash and pedunculate oak
48. Acid, with ash and sessile oak
49. Limestone, with ash and sessile oak

Hornbeam woods

50. With birch, hazel and pedunculate oak
51. With ash and maple
52. Acid, with sessile oak
53. Calcareous, with sessile oak

Suckering elm-woods

54. Invasive
55. Valley

Pine woods

56. Without oak
57. With oak

Birch woods

58. Birch only
59. With hazel

It is not possible in the space available here to consider each type in turn, its relative scarcity, and the effects that scarcity has on wildlife. This would be inappropriate anyway, because scarcity and abundance are not the only, or even necessarily the most worthwhile, indicators of conservation value. The aim should be to conserve representative samples of each native woodland type (Peterken, 1981). This is best achieved by preserving existing woodlands, but it may be possible in some cases to supplement these with new planting.

The significance of predicted climate change in relation to the distribution and abundance of particular vegetation types, including woodlands, has so far received little attention. Britain is probably entering (or already in) a period of more rapid change in climate than any that has occurred before. The various climatic extremes predicted by global climate models have been experienced before, but the warming and cooling have occurred over centuries, or millenia, not decades. There is no doubt that such changes affected the natural ranges of woodland types in the past (Godwin, 1975). What is not known, however, is how quickly trees, with their long generation times, can respond to the much more rapid climate change now apparently in train. Rapid changes in climate would certainly require periodic re-assessments of what vegetation types are 'natural' in particular parts of the country. The uncertainty inherent in such assessments would make the use of naturalness as a criterion for determining conservation value of new forests less reliable than it is at present. It would not, however, affect the fundamental argument that native species are more natural than exotics.

OPTIONS FOR MEETING DEMANDS FOR WILDLIFE HABITAT IN NEW FORESTS

Generally speaking, options for tree species choice increase with site quality. Until recently the only land available for large-scale afforestation was in the uplands and on infertile lowland sites, such as the acid heaths in the east of England and coastal sand dunes systems. Now that most land not protected by statute is potentially available for afforestation, including fertile agricultural land in the lowlands (but see Harvey, 1991 for actual constraints), a new range of options arises. In many situations, native broadleaves can be considered, in various mixtures. Silvicultural options also increase. Uneven-aged forests with continuous canopy cover can be considered as alternatives to even-aged forests which are clearfelled. With the increased range of forestry options come a wider range of opportunities for wildlife habitat creation. Even in the uplands, as afforestation moves 'down the hill' to intermediate altitudes and better soils, the possibilities will increase.

New 'native' woodland, including coppice

The greatest demand from woodland wildlife, and that which is most difficult to meet, is for more 'native' woodland, especially of the types which are now least common. Unfortunately, producing faithful replicas with authentic soils and the full range of component species is probably impossible in most cases on any realistic timescale. The difficulty increases with the complexity of the woodland type, its isolation from other similar woodlands, and the degree to which soils have been altered. Thus it is relatively

easy to extend Scots pine woodland on undisturbed soils in the Highlands because (1) it is a simple woodland type with few tree and shrub species (Peterken, 1981),
(2) expansion and contraction are normal in this woodland type anyway in response to fire and variable grazing pressure (Forster and Morris, 1977). It is much more difficult to extend complex, multi-species lowland woods onto cultivated land, because many of the characteristic field-layer plants have very limited powers of dispersal and colonisation. Creating new woods with high wildlife conservation value in isolation under such circumstances will take even longer periods of time, probably centuries if natural colonisation is relied on alone.

In most instances, the conservation value of new 'native' woodland will not compare with that of old secondary woodland, much less ancient woodland, for centuries, if at all. Having accepted that, it is possible to produce new woods which will eventually contain many of the native species of plants and animals which occur in old 'native' woodland. Maintaining continuous tree cover by reintroducing coppice with standards, or some other form of selection silviculture, is essential if the species particularly associated with mature forests are to be attracted and maintained. There are several reasons for this. First, many species are relatively immobile, and cannot therefore move to other areas when a wood is clearfelled (Peterken, 1981). Second, some (e.g. many of the lichens and invertebrates found only in ancient woods) depend on the continuity of very particular microhabitats and microclimates (Rose, 1974; Elton, 1966). Dead wood is an essential microhabitat in mature native woodland for many different species of lower plants, fungi and invertebrates, which inhabit it at different stages of decay. Dead wood in the crowns of trees has a different range of associated flora and fauna from similar wood lying on the ground. Even the diameter of dead branches affects the species they can support (Elton, 1966). However, 'careful' management of forests has resulted in many of these saprophytic species now having the dubious status of being the rarest in western Europe (personal communication from an EC wildlife adviser).

Introducing appropriate plants and animals may hasten development of native woodland, but much more research needs to be done before the success of this approach can be fully assessed. When introducing woodland plants into woods established on fertile agricultural land, more vigorous woodland species and non-woodland weeds may provide strong competition for decades or longer. The possibility of reducing fertility in such cases before establishing the new woodland needs to be investigated; this would not significantly reduce the growth of the tree crop. Successive cropping with nutrient-demanding agricultural species might be an option, but calculations suggest that on the more fertile sites, many crop rotations would be needed to make the required reduction. Even then, nutrients blown in from the surrounding land in some arable areas might continuously replace any that were stored in the trees or lost from the system.

A major potential advantage for wildlife conservation from planting new 'native' forests in areas where woodlands have become fragmented, is the linking together of formerly isolated woodland communities. Larger woods tend to contain more species than smaller woods of the same kind (Moore and Hooper, 1975; Peterken, 1974, Rackham, 1980). Also, because their populations are generally bigger, species in danger of local extinction are likely to be less vulnerable. Bringing formerly isolated populations together also increases the genetic pools of outbreeding species and hence increases their capacity to respond to environmental change. However, although ecologists feel that woodland fragmentation should lead to loss of species, the evidence suggests that extinctions are minimal for plants (Helliwell, 1976; Whitley and Somerlot 1985), while evidence for the other groups is equivocal (Usher, 1987).

Woodlands predominantly of pine

Three species of pine are commonly planted in British forests, Scots (*Pinus sylvestris*), Corsican (*P. nigra* ssp. *maritima*) and lodgepole (*P. contorta*). Scots pine, the only native species, was the foremost tree in British forestry until the arrival of European larch (*Larix decidua*) and, much later, Sitka spruce (*Picea sitchensis*) and lodgepole pine. It is still quite widely planted in the east of Scotland where the rainfall is suboptimal for Sitka spruce. Unfortunately it is not very fast growing or high yielding, but it tolerates a wide range of site conditions, making it a good general purpose tree. In particular, unlike Sitka spruce it can establish and grow among heather (*Calluna vulgaris*), a useful characteristic if there were a move away from deep ploughing and use of herbicides towards less environmentally-damaging forestry practices.

Scots pine is often considered better for nature conservation in Britain, particularly in areas where it is native, than other conifers (Kirby, 1986). This is generally true for conservation of ground flora because pine casts less shade than most other conifers. Larch, which allows similar amounts of light to reach the floor to pine if similarly thinned, can also support a well developed woodland flora. Other pines can be as good as Scots pine. Thus, heavily thinned stands of Corsican pines growing on coastal dunes at Newborough on the Isle of Anglesey, allowed the development of good growth of ferns and gradual invasion of woodland flowering plants (Hill and Wallace, 1989).

Invertebrates notice the difference between native and introduced pine species. Thus in a study by Kennedy and Southwood (1984), Scots pine was recorded as being host to 82 phytophagous species while lodgepole pine hosted only 18 (Table 3). In this paper Sitka spruce was recorded as hosting only 32 species but in a more recent study (Evans, 1987) 90 phytophagous species were accredited to Sitka spruce while around 160 species were indicated as feeding on Scots pine.

Table 3 Numbers of phytophagous insects associated with each of a range of tree species and genera in the British Isles (after Kennedy and Southwood, 1984)

Tree species	*Numbers of phytophagous insects*
Scots pine	82
Lodgepole pine	18
Sitka spruce	32
Oak (2 species)	423
Birch (2 species)	334
Willow (all native species)	450

No doubt more of the species associated with Scots pine will accept lodgepole and Corsican pines as alternative hosts in time, although the rate of acceptance is difficult to predict. Species feeding on foliage and shoots are thought likely to colonize more readily than those eating woody tissues and associated saprophytic fungi (Welch, 1986). This hypothesis is supported by the fact that most of the ancient forest indicator species fall into the latter group.

In a study of birds in natural Scots pine and Norway spruce (*Picea abies*) forests in Finland (von Haartman, 1971), there was no significant difference in abundance between spruce, pine, or spruce/pine mixtures. This is probably because forest structure is more important than tree species for birds, most of which do not depend for their food on a particular tree species (Newton, 1983). The same is generally true for mammals (Staines, 1986). Thus red squirrels (*Sciurus vulgaris*), which are particularly associated with large pinewoods in Britain, are common in Norway spruce (*Picea abies*) woods in Northern Europe (Pulliainen, 1973).

Lodgepole pine is planted much less in the uplands now than formerly. Pure Sitka spruce is replacing it in most situations. Many sites on particularly wet deep peats in the north and west of Scotland which were planted with lodgepole pine have been dried out sufficiently in the first rotation to enable Sitka spruce to be planted in the second. The widespread damage caused to lodgepole plantations by the pine beauty moth (*panolis flammea*) has also undermined confidence in it. However, self-thinning mixtures of lodgepole pine and Sitka spruce are now the usual first crop on deep peats with the presence of the lodgepole pine greatly increasing nitrogen uptake by Sitka, so removing the need for expensive fertiliser additions.

Corsican pine has proved a very satisfactory species for afforesting sand dunes and other light soils in regions of low rainfall, also for revegetating mine wastes and other disturbed lands. It is preferred to Scots pine because of its more reliable establishment on such soils, and its faster growth. The Newborough study (Hill and Wallace, 1989) showed the forest as a whole to have a very high botanical interest. Not only had all the original species to be found in the adjacent Newborough Warren NNR been retained, but many new species associated with woodland and bare-ground habitats had entered. The retention of the original species was mainly due to the inclusion in the forest of substantial unplanted areas, notably dune slacks, while the road verges provided an important new habitat.

If future climate change results in increased mean annual temperature and summer soil water deficits, Corsican pine, as a Mediterranean species, is likely to become more important. Other pines now little planted in Britain, including radiata pine (*P. radiata*) and ponderosa pine (*P. ponderosa*), may also become economic options. Their wildlife conservation value will be less than that of Scots pine, but will probably differ little from that of Corsican or lodgepole pines.

Woodlands predominantly of spruce

Spruces have a poor reputation with wildlife conservationists, primarily because of the widespread use of Sitka spruce for 'blanket' afforestation in the western uplands (NCC, 1986). In fact, Sitka is probably no better and no worse for wildlife than most other exotics, broadleaved or coniferous. Planting spruce in the uplands dramatically increases species diversity, because many woodland species, especially those belonging to mobile groups such as birds (Moss, 1978), are quickly attracted into the new forest, while most of the open moorland species remain, at least in the early phases of the forest cycle. This imposed increase in diversity may be seen as advantageous, provided the afforestation is not on too large a scale in relation to the ecosystems being displaced, and that care is taken to protect more valuable areas (bogs, upland heaths, herb-rich meadows) (NCC, 1986).

Predicted climate change may adversely affect Sitka spruce in Britain, especially in the more southerly parts of its planted range, because it requires cool wet summers. However, this adverse effect may be offset by increased growth in the north due to increased temperatures. It is not yet clear how the two effects will balance out (Cannell *et al.*, 1989). Species such as Douglas fir (*Pseudotsuga menziesii*), and the true firs, including grand fir (*Abies grandis*) and noble fir (*A. procera*) may be better suited to a warmer drier climate and may thus be more widely planted in future. Their wildlife value is likely to differ little from that of the spruces.

Non-native broadleaved woodland

Broadleaved woodlands bearing only superficial resemblance to native woodland have been established in past centuries all over Britain. They may or may not consist of native species, often being mixtures of both. Where natives were used they were not necessarily natives of the area – beech woodlands are widespread in the north of England and south Scotland, for example, although beech is not now native north of the English midlands.

New broadleaved woods of this kind are easy to produce. Traditionally they would have been planted as conifer/broadleaf mixtures, the conifers (typically Scots pine or larch) being used as nurses for the broadleaves and providing some early revenue as thinnings. This is probably still the most effective means of establishment, providing livestock can be excluded and weeds controlled. But, with the availability of tree shelters, small woods at least are likely to be established with broadleaves alone from the start.

Given similar soils and climate, and assuming comparable availability of seed sources for colonisation, many of the same species of flowering plants, ferns and bryophytes will occur in the ground vegetation in non-native broadleaved woodland as in native broadleaved woodland, if the amount of light reaching the ground is similar. Other things being equal, thinned conifer crops of a range of species also have much the same plant communities as broadleaves (Good *et al.*, 1990). This suggests that the intensity of the shade cast by the trees is of overriding importance in determining ground vegetation, rather than tree species.

The distribution of epiphytic algae, lichens and bryophytes in forests also appears to depend less upon particular species of trees, than upon forest structure and macro- and micro-climatic factors (Barkman, 1958). Rainfall and air humidity have a major effect, a wider range of species and greater density of cover occurring under the high rainfall, high humidity conditions found in forests in the north and west of Britain than in those of drier southern and eastern regions. Some species are only to be found along the Atlantic seaboard (Ratcliffe, 1977).

It is likely that the similar plant communities of non-native and native broadleaved woodland will support many of the same invertebrates. Nevertheless, the non-native broadleaved woodland will normally contain fewer species of invertebrates, because the trees themselves will have a poorer fauna (Kennedy and Southwood, 1984). The extent of the difference will depend on the species of trees being compared. Recently introduced species which are closely related to native trees (e.g. *Nothofagus* spp. and common beech [*Fagus sylvatica*]) are likely to be colonised more quickly by more species than others (e.g. *Eucalyptus* spp.) which have no close relatives native to Britain (R.C. Welch personal communication). They may also overtake others, such as horse chestnut (*Aesculus hippocastanum*), which have been here much longer but which also have no close relatives which are native to Britain.

Numbers of species of invertebrates in non-native broadleaved woodland can probably be increased substantially by planting a range of tree species, each of which will host at least some species not found on the others. However, research is needed to determine the extent of such enhancement of invertebrate diversity.

Because of its preeminent place among non-native trees in broadleaved plantations in Britain, and the antagonism which it traditionally arouses among conservationists, sycamore (*Acer pseudoplatanus*) deserves special consideration. In the Forestry Commission's 1979-1982 census of woodlands (Locke, 1987), 9% of broadleaved woodland in Great Britain was found to be sycamore, compared with 31% oak. Sycamore is also important in clumps, linear formations and as isolated trees, where it contributed 12% of the broadleaves compared with 15% for oak and 20% for ash. Thus sycamore is a significant part of the established broadleaved woodland scene. Furthermore, it is likely to increase in importance, both in plantations and in the countryside generally. It is popular with foresters because of its reliable growth on a range of soils and the steadiness of demand for its timber. The latter is due to the fact that good quality timber can be dependably produced in Britain and that the wood, which is creamy white but readily dyed, is in constant demand for a wide range of uses, including flooring, panelling, and furniture (Harris, 1987).

The major disadvantage of sycamore as a commercial species is its extreme attractiveness to grey squirrels (*Sciurus carolinensis*), which frequently do severe damage to plantations in the south of Britain. Some relief from attack, and insurance against too great a loss should damage occur, may be achieved by planting sycamore in intimate mixtures with other valuable species. Even allowing for its vulnerability to squirrel damage, sycamore is still likely to be a favoured species for farm woodland and agroforestry plantations in the southern lowlands. Further north, and particularly in Scotland, grey squirrel damage is not a threat. Thus sycamore is likely to be planted on an increasing scale there at a low to medium altitudes, particularly if climate warming, which would be favourable to this species, takes place.

In addition to its use in plantations, sycamore is also widely planted as an amenity tree. Additionally, because of its fecundity and tolerance of varied soil and other site conditions, it is an aggressive colonist in many situations.

It is for this reason, and particularly because of its ability to invade existing woods, that sycamore is perceived as an alien threat to native woodland communities. While not yet placed in the absolutely intrusive category of *Rhododendron ponticum*, sycamore is generally regarded as distasteful and is often strenuously discouraged by conservationists.

It is felt by some (Boyd, 1990) that any rapid expansion of broadleaved forestry in lowland Britain could be accompanied by a conservation campaign against the use of exotic species, focused on sycamore. After examining the evidence for and against sycamore in relation to nature conservation, Boyd concludes that while sycamore is an invader of valued semi-natural woodlands in which it has not previously been present, it possesses benefits as well as adverse effects in such habitats. While it has less species of insects associated with it than native oak it has a greater annual biomass of invertebrates, mainly aphids, than most native species of tree and is, in certain circumstances, a preferred food source for insectivorous birds. It also has an associated flora of lichens, mosses and liverworts which is better in quality and quantity than most native species of tree, while its leaf litter is quickly assimilated into soils which are comparatively rich in invertebrates.

Boyd (1990) proposes that, 'a policy on planting and natural occurrence of sycamore should specify management objectives in : (i) ancient, semi-natural woods (ASNW) with native communities including SSSIs; (ii) old, mixed, broadleaved-conifer woods; (iii) conifer plantations; (iv) broadleaved plantations without sycamore crop; (v) mixed conifer-broadleaved plantations without sycamore crop; (vi) sycamore plantations with or without other crop species.' He goes on to suggest that,

> As a rule, where sycamore does not occur in (i) it should not be introduced and where it does occur in (i) it could be eliminated (if in small quantity) or its extent restricted by heavy thinning. In (ii) the old, mixed woodland could include sycamore as a *bona fide* member of the community adjusted in frequency according to disturbance and its proximity to ASNWs – close-to, sycamore might be thinned as within ASNWs. In (iii), (iv) and (v), sycamore could be retained to diversify the woodland edge. In (vi), the rapid growth of sycamore is the primary objective. Where plantations are placed on ancient woodland sites which are not ASNW, but still retain some features of the relict community, sycamore could be retained and managed as in (ii).

Climate change is likely to extend northwards and westwards the geographic range over which broadleaves such as sycamore can be grown commercially in Britain. It is also likely to favour the more extended planting of some exotic broadleaves, e.g. *Eucalyptus* spp. and southern beeches (*Nothofagus* spp.) which are of limited usefulness at present because of their vulnerability to frost damage in severe winters. It is important that more information is obtained on the conservation values of these and other species with commercial potential.

Mixed woodland

Mixed woodlands in which blocks of broadleaves and conifers are mixed through the forest in various patterns and proportions are a feature of some parts of Britain, notably the Scottish Borders, the Lake District and Wales. They are common in hilly areas at low to intermediate altitude, which provide variable site conditions. Conifers are planted on the poorer, more exposed sites, broadleaves in the more sheltered areas with deeper, more fertile soils. More of this sort of afforestation is likely if marginal land of this type becomes increasingly available. These mixed woods can have high wildlife conservation values, especially if the broadleaves are predominantly natives and the favourable management practices described in Chapter 6 of this paper are used. These include developing structural diversity, with small blocks of trees of different ages distributed in an intimate mosaic, retaining some blocks on sheltered sites beyond normal rotation age, and allowing some trees (conifers and broadleaves) to die and decay naturally.

MANAGEMENT AND DESIGN ASPECTS

Forest design and management are of central importance wildlife habitat protection and enhancement in new forests. If they are to be used effectively in particular cases it essential that clear objectives are set.

Design and management for particular objectives

In the past, benefits to wildlife following afforestation have been mainly fortuitous. It is likely, therefore, that positive management for wildlife can achieve great improvements. There is now quite substantial knowledge on some aspects of management for wildlife, e.g. birds (Smart and Andrews, 1985) to support this. Objectives might include the conservation of particular species, the protection of existing important habitats, the creation of new ones, and the more general development of high wildlife conservation value for the whole forest. It is important in any particular case to choose compatible objectives. This may not be difficult in large forests, or smaller but more diverse ones, but may cause problems in very small woods, or larger woods on uniform sites. This is because management designed to favour one species or group of plant or animals is likely to be disadvantageous to others. Keeping rides open and sunny for woodland butterflies, for example, might be harmful for a species such as dormouse (*Muscardinus avellanarius*) which, because it lives in the crowns of understorey trees and shrubs, needs crown contact across rides to be able to move about a wood.

In the larger and/or more diverse woods it is possible, within designs aimed at sustaining the widest possible range of habitats and species, to concentrate on particular, more specific objectives, in different areas. An abandoned hay meadow within the forest might be set aside for preservation and managed accordingly, or a nestbox scheme might be initiated to encourage hole-nesting birds in a young forest where suitable natural holes were few.

In the small (<5 ha) woods that seem likely to be the norm on arable lands in the lowlands it would not be possible to achieve such a wide range of objectives and specialisation might be more appropriate. One wood might be managed primarily for a particular species of insect and its foodplant, another for establishing a range of woodland plants, a third for woodland birds. In some cases it might be possible to manage a group of individual small woods as though they were parts of a larger whole. However, small woods have their problems. For example, predation of birds' eggs and nestlings may be greater in small or fragmented woods (Small and Hunter, 1988).

To avoid inadvertent destruction of important pre-existing habitats and archaeological artefacts, thorough conservation inventories should be done before afforestation. Management requirements of existing representative sites, not necessarily of SSSI status, should be identified and specified.

Habitat creation in the planted areas as compared with unplanted areas in the forest is a largely unexplored topic. The effects of altering forest structure, for example by altering the size of stands of trees; their distribution in relation to unplanted areas; number, location, width and form of rides; management, especially thinning, are probably profound, but largely unknown. It is likely, however, that in general more diverse forest structures with smaller planted areas will favour wildlife more than those of larger scale and greater uniformity. This is because a patchwork of different habitats is created on the smaller scale, with areas of similar habitat separated by smaller distances than is the case where larger blocks of uniform age and structure are planted.

Edge habitat is particularly important as it provides niches for many species which cannot live in the uniform habitat on either side. (Young, 1986) noted that larvae of many woodland Lepidoptera fed on herbs, grasses and fungi in woodland rides and clearings. The amount of edge inside the wood is a function of management. Decreasing the size of felling coupes increases it as does the creation of woodland glades and rides within woods. By managing restocks, glades, rides, riversides and roadsides so as to retain diversity of vegetation structure both locally and throughout the forest, it is possible to create a mosaic of woodland habitats which contain a wide range of plants and animals.

Bibby *et al.*, (1986) investigated the effects of coupe size and adjacent vegetation on song bird diversity and abundance in Wales. They showed that the composition of song bird communities was unrelated to area. They also found that song bird density decreased adjacent to thicket stage conifers but not when adjacent to broadleaves, pre-thicket or pre-felling stage conifers. Small felling coupes may sustain a higher number of some song birds because of their greater edge to area ratio. Thus in Sweden, forest bird density increased near the edge of felled areas (Hansson, 1983). Large felling coupes may be more attractive to some larger and scarcer species, such as curlews (*Numenius arquata*) and short-eared owls (Leslie 1981), while kestrels may breed in old crow nests near the edge of felled areas and exploit the increased abundance of voles which occur in them as compared with the forest blocks (Petty, 1985). Thus for birds, a range of felling coupe sizes will sustain the widest range of species.

More information of this kind is needed for a wider range of plant and animal groups so that possibly conflicting conservation requirements can be identified and resolved. In particular, the quantitative relationships between increasing length of forest edge and increasing diversity and abundance of selected plant and animal communities need to be modelled. Similarly, the importance of isolation on colonization by plants with different dispersal mechanisms, and animals of differing mobility, needs to be investigated. When we have more information on these and many other equally important matters wildlife conservation will be better done.

Probably the best guide now is that used throughout this paper – enhancing natural diversity. Wherever possible, naturalness should be encouraged by maintaining or creating habitats which favour species native to the area. The Forestry Commission broadleaved woodland policy (Forestry Commission, 1984) has helped in this respect, especially in the uplands, by encouraging the planting of a proportion of broadleaves in all plantations. Wherever possible native species appropriate to the area should be used. Better still, natural regeneration should be encouraged, and protected where grazing pressure is a limiting factor to successful establishment.

Establishing a diverse age structure will further enhance wildlife diversity. When starting with new largely even-aged woodlands, the aim should be to develop such diversity over time. It is desirable to advance and/or delay felling some coupes by at least 10 years to give greater age diversity. A long term objective should be the retention of large mature trees distributed through the forest, preferably in clumps or small blocks, for birds such as goshawk (*Accipiter gentilis*) and capercaillie (*Tetrao urogallus*). This can be achieved eventually by giving no felling date to 1-2% of a forest. These oldest trees will develop the most interesting epiphytic floras of liverworts, mosses and lichens, and also the richest invertebrate faunas. Leaving at least some of the trees to die of old age will give the crucial scattering of standing and fallen dead wood. The field layer associated with the oldest trees is also most likely to have acquired the most plant species. It is desirable to maximize the diversity of the flora, as this is a simple way of ensuring that the maximum diversity of phytophagous invertebrates is conserved.

Other positive conservation activities involve enhancing the overall conservation values of afforested sites by improving unplanted areas. Restoring pools, lakes and other wetland habitats in areas where these had previously existed on farmland, but had been removed during agricultural improvements, is a good example. This is easy using heavy earthmoving machinery.

At the policy level, decisions need to be made on whether wildlife conservation inputs should be spread thinly over the whole forest, or be concentrated on limited areas of high existing conservation value or potential. It may be best to optimize timber production on land of low wildlife conservation value, making fewer concessions to wildlife there, if this would permit larger areas of land of high wildlife conservation value to be kept unplanted or be managed sympathetically. Adopting this approach, former arable land in the lowlands is the obvious candidate for intensive timber production.

Habitat diversification

Afforestation almost always increases habitat diversity in Britain, because forest cover is so sparse. It has been seen that this does not necessarily make forestry acceptable to nature conservationists. The deciding factor is whether increased diversity is achieved at the expense of the more highly valued semi-natural communities which are replaced, or complements them (NCC, 1986). Thus an uneven-aged, structurally diverse conifer forest, planted in a large area of common vegetation of low wildlife conservation value, and designed so as to contain large areas of undisturbed land at an appropriate scale, might well be considered beneficial to wildlife. A similar forest, but of uniform age and structure, planted in large blocks with few inclusions of unplanted land, and located in a vegetation type which is of high conservation value would, understandably, be much less acceptable.

Habitat diversification within woodlands is increased by the use of tree-species mixtures, but little is known about the extent to which such mixtures increase wildlife diversity, except of birds. Bird diversity and abundance are greater in conifer forests containing broadleaves than in pure conifer stands (Bibby, 1987). The proportion of broadleaves need only be small and they can be equally effective either scattered through the crop or arranged in groups. Introducing conifers to broadleaved forests also increases bird species diversity and abundance (French *et al.*, 1986), so it seems that mixtures generally are favourable for bird diversity. The reason seems to be partly that some species are restricted to one or the other tree type, partly that mixed forests have greater structural diversity, which means more habitats. Certainly many birds need deciduous trees (e.g. blackcap [*Sylvia atricapilla*], wood warbler [*Phylloscopus sibilatrix*], green woodpecker [*Picus viridis*]), but others conifers (e.g. siskin [*Carduelis spinus*], common crossbill [*Loxia curvirostrata*]).

Choice of areas for planting – location, size and shape

Location of forests has a considerable bearing on their wildlife conservation values. Forests planted where there is little existing woodland generally have higher relative values than those planted in heavily wooded districts, even though their absolute values may be less. Forests planted on land of low wildlife conservation value will have greater relative, though not necessarily greater absolute value than those which displace more important wildlife habitats.

Woods planted near centres of population, and with public access, will be subject to more disturbance than more remote, less accessible woods. Some characteristic animals may be kept out of a wood by repeated disturbance, so conservation value will be less than optimal. If, on the other hand, one aim of new forests is to meet a demand from people who want to use them for recreation, and may hope to experience woodland wildlife as part of the recreation experience, good access is a bonus. It may be possible, in such cases, to limit access to paths, so reducing disturbance, and to provide hides to enable the public to view wildlife without disturbing it unduly.

The size of new forests will fundamentally affect their conservation values. Very small woods, although having large edge to area ratios, are likely to have very restricted ranges of habitats and may attract only woodland edge species. They are also likely to be unstable, losing faunal species in particular by local extinction due to low population numbers. Unfortunately it is not possible to say what the minimum size of woods should be, because such factors as relative isolation and the presence of corridors (hedges, shelterbelts, scrub) will be important, but their influence is only now being investigated.

At the other end of the scale, optimum size for conservation value will never be reached. This is because species diversity continues to increase indefinitely with increasing woodland area. It should be borne in mind, however, that large forests will be displacing equally large areas of other habitat, and may adversely affect even the areas which are not planted (NCC, 1986). In deciding optimal size, as in choosing preferred locations, it is necessary, therefore, to take local factors into account, but also to be aware of the regional, national and possible international significance of these decisions.

Shape is important both on the outside and the inside of new forests. Small woods are probably best with minimum external edge to area ratio, but as the size increases the desirability of increasing the external edge increases. Edges are also created within the forest, along rides, roads, streams and other linear features. Curving edges are not only less visually intrusive than straight ones, especially if they are related to topography, they also increase variability of aspect and reduce wind tunnel effects, both of which are generally advantageous to wildlife.

CONCLUSIONS

The greatest benefits to woodland wildlife conservation from afforestation are likely to be provided by new mixed-species forests containing native trees local to the area. These forests are likely to be of particular local benefit in areas where woodland is scarce. They will gain in wildlife interest more quickly if they can be established near old woodland.

Even under the most favourable circumstances new woods are unlikely to attain the conservation values of ancient woodland on similar sites for many centuries. Assisting natural colonisation by introducing characteristic woodland species may be one way of speeding ecosystem development. However, establishing woodland plants is unlikely to be easy, especially on soils whose structure and fertility have been greatly altered by cultivation since they were last forested. Weed competition, in particular, is likely to pose major and continuing problems, especially where the surrounding land (e.g. arable) provides both a barrier to wildlife immigration and a continuing source of unwanted nutrients.

The larger new forests are, the better in general they are likely to be for woodland wildlife conservation. In practice, however, new woods are likely to be small, in which case attention should be given to reducing their isolation from each other and from existing woods, for example by retaining, or planting, connecting hedges and shelterbelts. If new woodlands can be planned so as to be part of the local, even national and perhaps international woodland wildlife resource, so much the better.

If forests of appropriate native tree species cannot be planted, the next best forest types for wildlife are probably broadleaved/conifer mixtures. These have higher wildlife conservation values than non-native broadleaves or conifers alone, because they are more structurally diverse and hence provide a greater range of niches. They may develop more species diversity than new native woodlands, but may be considered less desirable nevertheless, because they are less natural. Monocultures of non-native species are generally the least desirable for nature conservation, except where they provide habitat for native species of plants or animals which is otherwise unavailable.

Regardless of woodland type, attention should be given at the planning stage to the wildlife conservation potential of individual woods so that informed decisions can be made on management requirements. Examples of positive management are: developing varied age class and structure; introducing characteristic plants and animals; retaining trees to biological maturity and beyond; leaving dead standing and fallen trees and wood.

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'FORESTRY EXPANSION: A STUDY OF TECHNICAL, ECONOMIC AND ECOLOGICAL FACTORS'

This is one of a series of papers which form part of a study to consider the scale, location and nature of forestry expansion in Britain.

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The full list of papers is as follows:

Occasional Paper No	Title	Author
33	Introduction	Professor Ian Cunningham, Macaulay Land Use Research Institute
34	British Forestry in 1990	Hugh Miller, University of Aberdeen
35	International Environmental Impacts: Acid Rain and the Greenhouse Effect	Melvyn Cannell and John Cape, Institute of Terrestrial Ecology
36	The Long Term Global Demand for and Supply of Wood	Mike Arnold, Oxford Forestry Institute
37	UK Demand for and Supply of Wood and Wood Products	Adrian Whiteman, Forestry Commission
38	Development of the British Wood Processing Industries	Iain McNicoll and Peter McGregor, University of Strathclyde and Bill Mutch, Consultan
39	The Demand for Forests for Recreation	John Benson and Ken Willis, University of Newcastle
40	Forests as Wildlife Habitat	John Good, Ian Newton, John Miles, Rob Marrs and John Nicholas Greatorex-Davies, Institute of Terrestrial Ecology
41	Forestry and the Conservation and Enhancement of Landscape	Duncan Campbell and Roddie Fairley, Countryside Commission for Scotland
42	The Impacts on Water Quality and Quantity	Mike Hornung and John Adamson, Institute of Terrestrial Ecology
43	Sporting Recreational Use of Land	James McGilvray and Roger Perman, University of Strathclyde
44	The Agricultural Demand for Land: Its Availability and Cost for Forestry	David Harvey, University of Newcastle
45	Forestry in the Rural Economy	John Strak and Chris Mackel, Consultants
46	New Planting Methods, Costs and Returns	Jim Dewar, Forestry Commission
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