



Forestry Commission

Managing Forests for Biodiversity

Edited by Richard Ferris-Kaan



Technical Paper 8



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Managing Forests for Biodiversity

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*Wildlife and Conservation Research Branch
The Forestry Authority*

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EDITOR'S NOTE

Managing Forests for Biodiversity was the title of a one-day symposium, organised by the British Ecological Society's Forest Ecology Group, and held at the Maybury Hotel, Edinburgh on 2 September 1992. The Group wishes to acknowledge the financial support of the British Ecological Society, without which neither the symposium nor this Technical Paper would have been possible.

This Technical Paper is a record of papers presented at the symposium plus additional information, preceded by a brief introductory review. The symposium was seen as a first step towards examining the very complex issues of biodiversity and forest management. The formation of an interdisciplinary Biodiversity Project Team within the Research Division of the Forestry Authority is intended to progress this work.

KEYWORDS: Biodiversity, Ecosystems, Forest management, Forestry, Nature conservation

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Introduction

Forests and biodiversity – a brief review of the issues

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Biodiversity has become a popular word, its use no longer restricted to the scientific world. It is now widely used by the media, the general public and politicians. Although interpreted in a variety of ways, there is general agreement that biodiversity is worthy of conservation, and that its loss will have significant economic, social and ecological consequences (Society of American Foresters, 1992).

Defining what is meant by biodiversity is problematic, since it is an umbrella term which describes the variety of nature (Fenger, 1990). Depending on context and scale, biodiversity can refer to alleles or genotypes within a population, to species or life forms within a biotic community, and to species or ecosystems across a landscape or even the planet (Burton *et al.*, 1992). The ecological structures, functions and processes at all of these levels are also important components of biodiversity.

Forests are increasingly viewed as valuable reserves in which large tracts of land can be 'managed' to protect biodiversity. However, this does not consist solely of preserving virgin forests. Protected areas cover less than three percent of the earth's land area (Wilcove, 1989), and it seems clear that strict preserves will not be sufficient to protect the full range of species and populations on earth. Furthermore, it needs to be realised that some natural systems are biologically diverse and some are depauperate; some of man's economic activities enhance diversity and some of them threaten it (Burton *et al.*, 1992).

The challenge for foresters is to balance economic development and biodiversity, and to achieve this there must be a commitment to strategic management of all resources, with goals attached to each (Bonar, 1989). No matter how well stated, technical definitions and objectives are rarely of

operational use, and biodiversity will not receive the attention it deserves as long as it remains an abstract concept in the minds of managers and policymakers (Wilcove, 1989). It is necessary to develop clearly defined, measurable targets, and to avoid viewing biodiversity in terms of local species richness. Management to maximise species richness at the local level often favours generalists at the expense of habitat specialists (Probst and Crow, 1991).

It is clear that there are no simple solutions, although the application of some general recommendations can help to maintain and enhance biodiversity in multiple-use forests.

- **Make an inventory of the diversity of certain taxonomic groups.** Managing for biodiversity requires better inventories of all biological resources.
- **Identify appropriate management units.** Plan and manage over large areas rather than using a stand-by-stand approach, and use a regional perspective when considering biodiversity.
- **Consider linkages between habitats over a landscape scale.** Maintain or create spatial patterns that enhance conditions for target or problem species, and avoid fragmentation of habitats.
- **Promote tree crop diversity.** Emphasise multi-species and ecosystem management instead of single-species and tree management.
- **Explore the use of alternative silvicultural systems.** These include management for a variety of canopy structures, the retention of leave trees during harvesting operations, and extended rotation lengths to provide large diameter, over-mature trees.

- **Set-up and monitor a network of ecological benchmarks** (standards against which comparisons can be made). ‘Natural’ stands can serve as ecological benchmarks against which to monitor the effects of management; a long-term commitment to monitoring is needed.

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Symposium Papers

Ecological diversity in managed forests

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Summary

The 1991 re-statement of forest policy in Great Britain and an emphasis on multi-objectivity have led to changes in forest management objectives. In recent years the landscape, recreation and wildlife benefits of forestry have been reinforced.

The low wildlife value of managed forests in Great Britain perceived by many people is, I suggest, a result of the management regime rather than the tree species simply not being British. Sensitive management, with nature conservation as an objective, can produce forests which are ecologically diverse with high wildlife value.

Biodiversity must be a product of rather *low* levels of intervention, as it is neither ecologically sound nor economically sensible to pursue biodiversity to a high level without consideration of the site potential. However, *non*-intervention can result in 'undesirable' changes occurring, such as invasion by *Rhododendron ponticum* or over-grazing by deer.

But, what should we aim for in terms of biodiversity? In this paper I suggest that we attempt to mimic natural forest ecosystems, paying special attention to ecological processes.

Introduction

Increases in biodiversity might seek to maximise the number of species and genotypes living in each of a maximum number of ecosystems which can be supported by the physical-chemical environment. This necessarily infers a structural dimension in terms of the number of vertical layers, and a spatial dimension in terms of the number and the distribution of patches. Maximising biodiversity must

be managed within the constraints imposed by the physical-chemical environment, thereby avoiding high external inputs to preserve the *status quo*. Attempts to increase biodiversity may jeopardise some species, and conflicting objectives must be reconciled first.

First, we must consider what sorts of ecosystems we are keen to develop. Clearly historic land use in Britain has left a legacy in terms of what is currently available. We have forests derived from broadleaved and coniferous plantations, and the remnants of semi-natural woodlands remain. However, in the pursuit of maximising biodiversity, what are the possibilities and prospects in forests dominated by exotic tree species?

Natural ecosystems

A reasonable approach might be to attempt to mimic some aspects of 'similar' natural systems, such as the native coniferous forests of Scandinavia and north western America, and the broadleaved forests of Poland and Belarus. These forests are renowned for their high biodiversity and are highly valued representatives of temperate and boreal rain forests. It is widely accepted that managed plantation forests can never replace or substitute for semi-natural native forests, and it is vital that semi-natural remnants are safeguarded.

The existing remnants of natural forest are very different from the primeval wild wood, and the species abundance and composition have almost certainly changed dramatically. For example, small-leaved lime and elm are no longer present at the density and distribution that the fossil and pollen evidence suggest were the case (Godwin, 1956); large predators such as wolf and bear have

become extinct (Langley and Yalden, 1977; Rackham, 1986); and exotic species have been introduced, for example grey squirrel (Gurnell, 1991) and Japanese sika deer (Ratcliffe, 1987). Therefore, in spite of the desirability of recreating the primeval wild wood, it is hardly possible. It seems that a realistic objective is to retain those historical components which still exist, but beyond this should an increase in biodiversity be pursued? Non- or minimal-intervention policies will almost certainly allow the increase of 'undesirable' species, such as grey squirrels, sycamore and *Rhododendron ponticum*, and result in losing some of the valued components. Instead, it might be more realistic to allow some management in order to achieve specific objectives relevant to some future natural state.

Having stated the importance of naturalness, or at least degrees of it, how can forests established from plantations, often of exotic species, support nature conservation objectives?

The 'naturalness' philosophy implies that only ecosystems which have always been present, albeit in an altered form, are valuable. Of course these are very valuable, but artificial ecosystems can provide valuable wildlife assemblages and they

certainly function by natural processes. Indeed, it is frequently necessary to halt or delay natural successional changes to maintain valued habitats such as rides and other ecotones, chalk grasslands and coppice woodlands. This increases the degree of artificiality but increases structural and spatial diversity.

It seems that management systems which have been practised over long periods are at least as important as the long-lived ecosystems which they may produce. For example, established ride systems and grasslands which have received similar treatment over long periods tend toward high biodiversity. The process of restructuring coniferous plantation, such as that currently underway at Kielder Forest (Hibberd, 1985), is an application in forestry where rides and stream ecotones, which surround mosaics of different age classes of spruce forest, will be managed in perpetuity.

Thus, the nature conservation classification of present ecosystems as good or bad, whereby natural is good and artificial is bad, is an oversimplification, and the dynamic temporal elements which are imposed upon these ecosystems drive a change in their relative classification (see Figure 1)

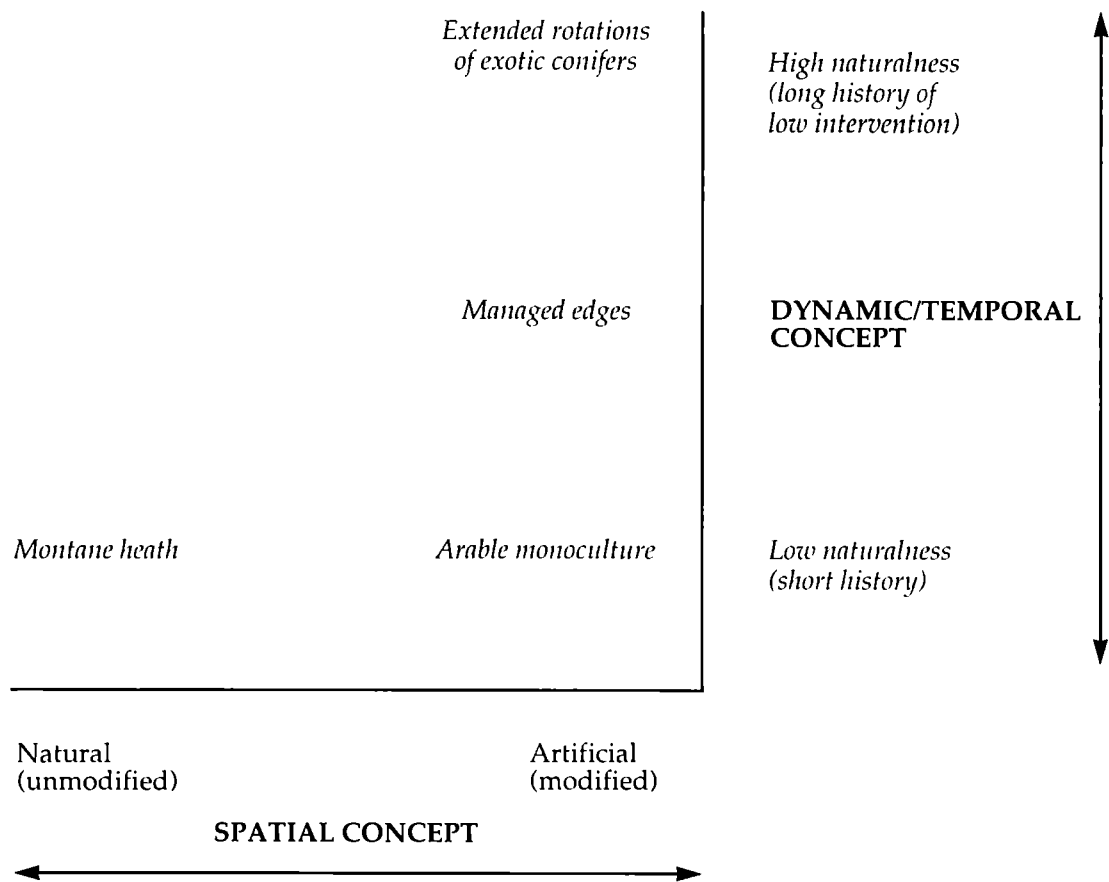


Figure 1 The influence of time and intervention on increasing forest biodiversity.

(Ratcliffe, 1991). In this way, artificial ecosystems all have relative value and long-established, managed edges and extended rotations of exotic conifers, for example, can be correctly seen as interesting, valued and diverse ecosystems. This philosophy avoids any conceptual problems arising from reconciling high management input and artificiality with nature conservation value.

Structural diversity

There are inherent dangers in maximising biodiversity *per se* and it is important to recognise the regional ecology, which is dependent on site conditions. Many natural ecosystems do not appear to have high biodiversity and, more importantly, their most valued components often rely on rather low diversity, for instance, goshawks and red squirrels in mature coniferous forests. Therefore, management should only maximise biodiversity within the physical-chemical limitations of the site (e.g. the inherent conditions of nutrition, soils and climate) and in terms of the prevailing structure of the forest (e.g. establishment, thicket, pole stage, or over-mature, old growth or extended rotation). For example, maximising biodiversity in a 70 to 100-year-old upland spruce forest would entail attempts to fill the available niches in that ecosystem at its current stage of development (such as encouraging dead and decaying wood communities including fungi, lichens, plants, invertebrates and vertebrate animals) and the development of a shrub layer. It would not entail fragmenting the forest by creating more rides and introducing broadleaved trees, unless these had been carefully considered and decided upon as separate and desirable objectives.

Forty to sixty per cent of birds and 65 to 75 per cent of terrestrial mammals which breed in Europe and North America breed in forests (Bunnell, 1990). In North America, 20 per cent of birds and 50 per cent of terrestrial mammals depend on old growth, i.e. forests retained beyond normal felling age, with high structural diversity including a developed shrub layer and a high deadwood component. Wildlife species generally associate with particular seral stages of forests, mainly due to their differing feeding and breeding requirements (Ratcliffe and Petty, 1986).

Unfortunately, extended rotation forests incorporating a high deadwood component are not common, and more effort should be made to increase these, even though the risks of windthrow may be high. Indeed, windthrow is a natural process resulting in deadwood and providing structural diversity and regeneration opportuni-

ties. Bunnell (1990) has suggested that the maintenance of species dependent on downed wood and snags (lying and standing deadwood) is incompatible with a high-yielding forest. This presents a challenge to foresters, and multi-objective forestry must cater for these species by providing increased amounts of extended rotation management (Peterken, 1992).

Spatial diversity

Wildlife species will be maximised if a range of structural types (tree age classes) are present at one point in time. Again, the restructuring of forests is attempting to provide this.

A further important consideration is the scale of spatial biodiversity; this will depend upon the plants, but particularly on the mobile animals which are present. For example, large, wide ranging species such as European bison and wolves need very large scale mosaics, whereas woodland butterflies are maximised in small scale patches. Clearly, the scale of mosaic must be related to the mobility of important species. Many of them are indicators of permanent ancient woodland and include fungi, lichens, bryophytes, ferns, vascular plants, molluscs, insects and arachnids. Because of a limited ability to spread, woodland fragmentation can cause the eventual loss of these species. Recently established forests are unlikely to be naturally re-colonised by many of these species unless the forests are in close proximity to existing ancient woodlands. However, this view is based on rather limited evidence, and the capacity for spread in many organisms may be greater than so far thought. In any case, the avoidance of fragmentation, the establishment of permanent management coupes and ecotones, and careful management of sedentary species are necessary in the management of spatial diversity.

Open spaces

Natural forests encompass variable amounts of open space without trees, and clearly this adds markedly to spatial diversity. Man-made forests sometimes lack this important component and every effort should be made through forest design to maintain and recreate open space habitats. Following the general principles discussed earlier, the most valuable sites are likely to be those that represent long standing open habitats such as unimproved pastures, heaths, moorlands, mires and riparian areas adjacent to rivers, streams and lakes. The same principles of management apply to open space as to woodland areas, and large

areas are necessary to support large, wide ranging species; afforestation should avoid fragmentation of important large expanses of open space. However, if open space is scarce it can be created; opportunities arise, especially at the end of rotations, and tree lines can be modified, bogs and mires expanded, and ponds and riparian areas created.

Management of ecological processes

We have established that perhaps the most important nature conservation objectives are centred on the management of processes such as natural succession, colonisation and dispersal. For example, management intervention can dictate the speed and direction of successional change, and it can even halt it completely. We can also manage the processes of natural regeneration and natural thinning to alter tree species composition and density. This is an important management principle applicable to all ecosystems and the particular form of process management will be driven by the objectives. For example, in a semi-natural woodland the process of colonisation by sycamore might be prevented, and on a heathland patch natural regeneration might be removed to prevent successional changes to woodland.

Management of resources

In all ecosystems, important resources which support large numbers of species can often be identified. Water is perhaps the most obvious resource of this sort and the provision and management of water can clearly add substantially to the range of species present. Similarly, tree seeds are important food for a range of animals and this resource requires careful management to provide a range of tree species which will in turn provide seed continuously. At the single species level, the field vole, *Microtus agrestis*, is the major food for a wide variety of avian and mammalian predators in upland ecosystems and the dynamics of its cyclical changes must be understood.

Habitat creation and introduction

It will soon become clear that many potential habitats, or even entire ecosystems, are absent. For example woodland glades, riparian zones, broadleaved and open patches within conifers and, in some unafforested landscapes, entire

woodlands may be appropriate. Thus habitat creation is a logical step in creating maximum spatial diversity, and suitable sites should be chosen and opportunities taken to create new habitats. As we have seen earlier, these permanent features will attract more species and become inherently more valuable through time.

Perhaps a logical extension of this theme is to introduce new species, and clearly many of our woodland habitats already include exotic species such as Sitka spruce, sycamore, rabbits, and fallow, sika and muntjac deer. However, although this may seem logical, the history of introduced species throughout the world gives cause for concern. Much of New Zealand's indigenous wildlife has been irretrievably damaged by introduced species; in Britain, grey squirrels and rabbits modify plant succession and cause considerable financial losses to forestry and agriculture, while the introduction of Japanese sika deer threatens the genetic conservation of indigenous red deer by hybridisation (Ratcliffe, 1987).

Clearly, very careful consideration is needed before new species are introduced, but in principle it is a method of maximising biodiversity by filling vacant niches, and few would question recent translocations of sea eagles and otters. The Joint Committee for the Conservation of British Insects (JCCBI) has published a code of conduct for the re-establishment of insects for reasons of pest control, research and wildlife conservation (JCCBI, 1986). Many sedentary species of plants and animals are unlikely to colonise new habitats without assistance (Ratcliffe and Petty, 1986).

These initiatives all focus on the re-establishment of indigenous species. More care may be needed if any consideration is given to the introduction of non-native species, particularly if they have invasive or opportunistic tendencies. In any case, the detailed ecological requirements of the species must be known, the likely impact on other species understood and the introduction carefully monitored (Anon, 1979). This applies equally to extinct indigenous species when land use and available habitats have been drastically modified since their extinction.

In the future Geographic Information Systems (GIS) and hierarchical landscape models, incorporating nature conservation and landscape objectives will almost certainly become the tools for managing structural and spatial biodiversity, but these need to be linked with autecological data.

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Biodiversity measurement

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Summary

Species are by definition different from each other. New methods show how the degree of difference between species can be included in an index. The functional aspect of species diversity measurement is strengthened by incorporating other differences between species, such as body size, predator or parasite, as a component of diversity. The choice between certain existing diversity indices is discussed; the size and density of the largest predator may also be an indicator of the overall state of the biological system. Finally it may be useful to make an independent measure of human impact to compare with biodiversity measurement, as a basis of long-term policy assessment.

Introduction

Maximising biodiversity is not necessarily achieved by increasing the number of species on a species list. Recent developments in both conservation biology (Vane-Wright *et al.*, 1991; Faith, 1992) and in functional ecology (Cousins, 1980; Harvey and Godfray, 1987) have pointed away from treating all species as equal units on a species list. *Which* species are on the list becomes important rather than, or as well as, *how many* species are on the list. These developments have potential application in forestry practice as guides to maximising biodiversity.

This paper briefly reviews these new approaches, and offers some suggested improvements to the traditional measurement of species diversity in which species are treated as being of equal importance. How the new methods relate in detail to each other and to ecological theory in general is dealt with elsewhere (Cousins, 1994).

Improvements to existing diversity measures

Species diversity measurement is based around the species number curve. Each time an individual organism is added to a count, N , of organisms in an observed sample, it can be of the same species as organisms already present or it can be a new species in which case S , the number of species in the observed sample, increases by one also. As N increases, S can also increase although it is progressively less likely that, as the number of individuals in the sample increases, the next individual encountered will be of a species not previously met. Functionally, N is proportional to area, such that if the area is doubled within a similar kind of habitat the number of individuals encountered is likely to double also although, for the reasons given above, the number of species increases by a lesser amount.

In summary:

$$N = k \cdot \text{Area} \quad (1)$$

$$S = f(N) \quad (2)$$

$$S = f(\text{Area}) \quad (3)$$

This simple set of relationships has had two basic effects on the measurement of species diversity. While the number of individuals is proportional to the area sampled, and measurement is achieved by number per unit area, i.e. a number density with units of $N \text{ m}^{-2}$, the approach to measuring species diversity has been twofold. Species diversity is described either as a measure of the number of species per unit area, giving a species density measure with units of $S \text{ km}^{-2}$, as typified by the species atlases (Dony, 1976; Sharrock, 1976), or by making measures which

are assumed to be sample size independent and are described by the function (f), as typified by Williams (1964). The frequently used Shannon index is assumed (Pielou, 1975) to fall into the class of sample size independent indices although, as will be shown below, this is not the case for samples of the size normally encountered in the ecological literature.

In a study of breeding birds on farms in the United Kingdom (Cousins, 1977), using Common Bird Census data from the British Trust for Ornithology,

the Shannon indices H' and J were analysed for sample size effects. From the simple relationship which defines H' and J ,

$$H' = J \log^2 S \tag{4}$$

We know that S increases with sample size and that, therefore, from (4), H' and J cannot both be constants, and either H' or J or both are sample size dependent.

Figure 2 shows that for the 85 farm plots (average area 40 hectares), as S increases so does H' .

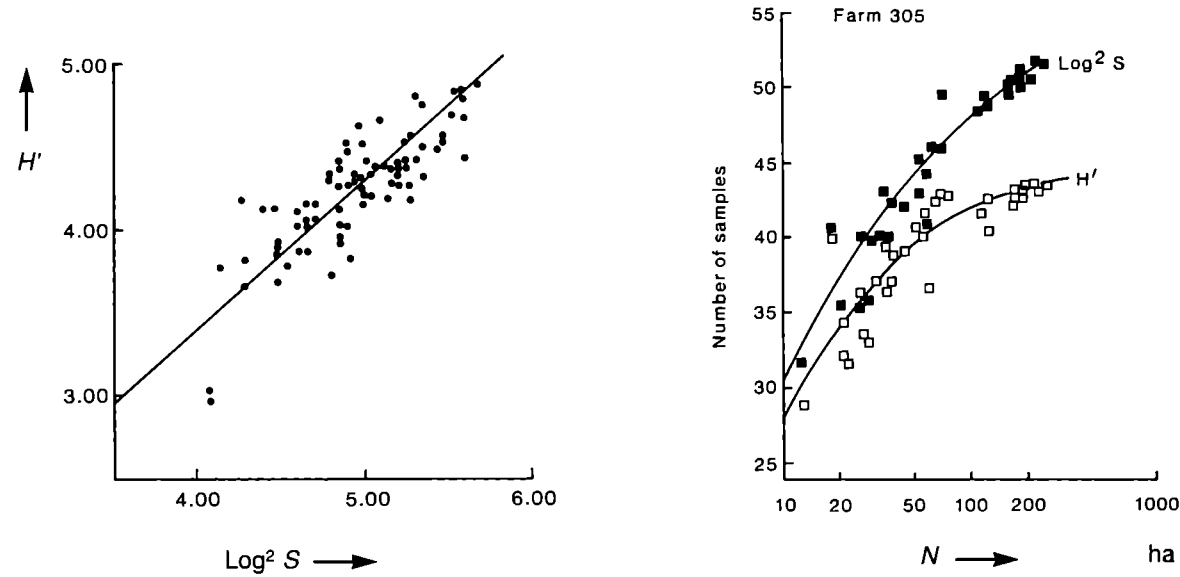


Figure 2 The relationship between H' and S , the number of species in each BTO farm census plot in 1973 (after Cousins, 1977).

Figure 3 The effect of area samples on H' and $\log^2 S$ for a single farm determined by random aggregation of sub-samples (after Cousins, 1977).

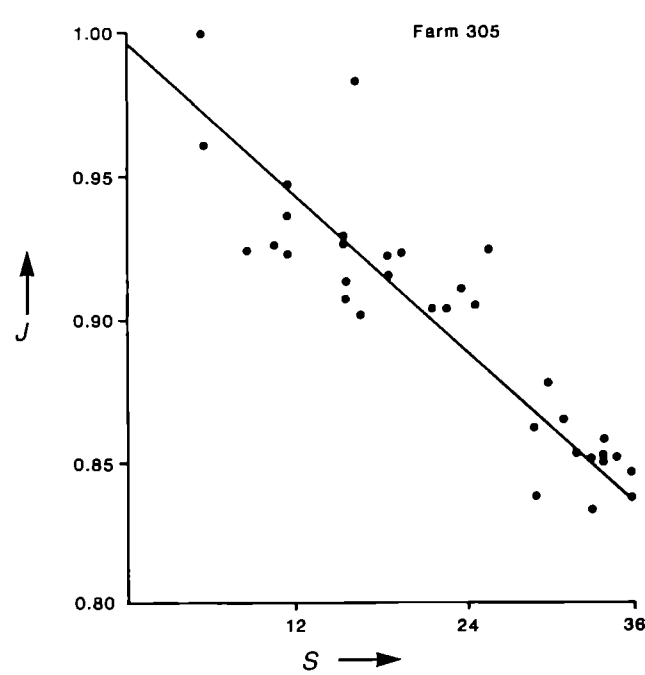


Figure 4 The relationship between J and the number of species present on a single farm determined by random aggregation of sub-samples (after Cousins, 1989).

Table 1 Comparison of representations of bird diversity on three UK farms, using diversity indices H' and J (after Cousins, 1977)

	Farm code		
	315	209	072
N	133	209	250
S	23	41	34
H'	3.90	4.28	4.22
J	0.86	0.80	0.83
$S/10$ ha	9	9	15
$S/20$ ha	14	15	22
$S/40$	21	22	30

S = number of species.
 N = number of individuals.

When individual farms are analysed for sample size effect through the random aggregation of subsamples, the sample size dependence is clearly shown in Figure 3 for both S and H' . J is inversely related to the number of species in the sample and is therefore again sample size related (Figure 4). Therefore, as a conservation evaluation technique, the Shannon derived indices are very difficult to interpret. At a minimum the area of the sample should be specified. The comparison between samples remains difficult if the sampling is not standardised, as is shown in Table 1 where three farms are compared.

The highest diversity measured by H' is 4.28 for farm 209. This is a particularly large farm with the highest total number of individuals (N) and species (S) identified. However, farm 209 has the lowest evenness value (J). We need to know if this is an artefact of sample size, as would be expected from the relationship of J with S shown in Figure 4, or whether it is really different to the value of J for the other two farms. Clearly the most even distribution ($J = 0.86$) is produced by farm 315, but this similarly can be a result of the low number of species (23) found in that plot. In fact when we

compare the species densities at 10 hectares to 40 hectares, farms 133 and 209 are in effect identical in their species densities. Although J values were not calculated for sub-samples corresponding to these species densities, the J value for farm 209 would be expected to rise if that farm were sampled at the same scale as farm 315. Finally, it is farm 072 which turns out to have a much greater species density than the other two. The lack of sample size independence of H' , $\log^2 S$ and J requires that each of these parameters be plotted against sample area to establish comparability between sites (as occurs in Figure 3), although the ratio of H' to $\log^2 S$ would also require plotting to give J .

Given these sample size problems for H' and J , it is preferable to use clearly understood measures, such as species density and the density of individuals, to describe the variety and quantity of biota present in an area. A measure of the relative abundance is provided by Williams (1964),

$$S = \alpha \log e (1+N/\alpha) \quad (5)$$

which can be used as a sample size independent measure and, more importantly, can be used to calculate species densities at a common scale by substituting N' , the number of individuals per unit area, in equation (5) where S and N are known for the whole sample. In some habitats strong edge effects of the sample plot need to be removed from the data before calculating the density, $N' \text{ m}^{-2}$ (Cousins, 1977).

Ordinal measures

Each of the above methods is a cardinal measure of species diversity in which each of the species is treated as equal and additive, forming a count or index. A second group of methods is possible using ordinal measures of species diversity (Cousins, 1991), where species are ranked in an order and not added together. This difference is important for functional biotic measurement. Earlier it was stated that if species diversity is measured as a species density then this represents in some way the variety of energy paths present in a given area. Members of species of different sizes or of different trophic habit also have different quantitative effects on those energy flows, and these can be represented in an ordinal index.

Within the cardinal indices a sparrow counts the same as an eagle in the count of number of species present. In an ordinal representation, such as the number of species in weight classes, different levels of importance can be attached to species

which individually have very different impacts on energy flow. Figure 5 shows the geographical distribution of average body size for species of UK breeding land birds (Cousins, 1989).

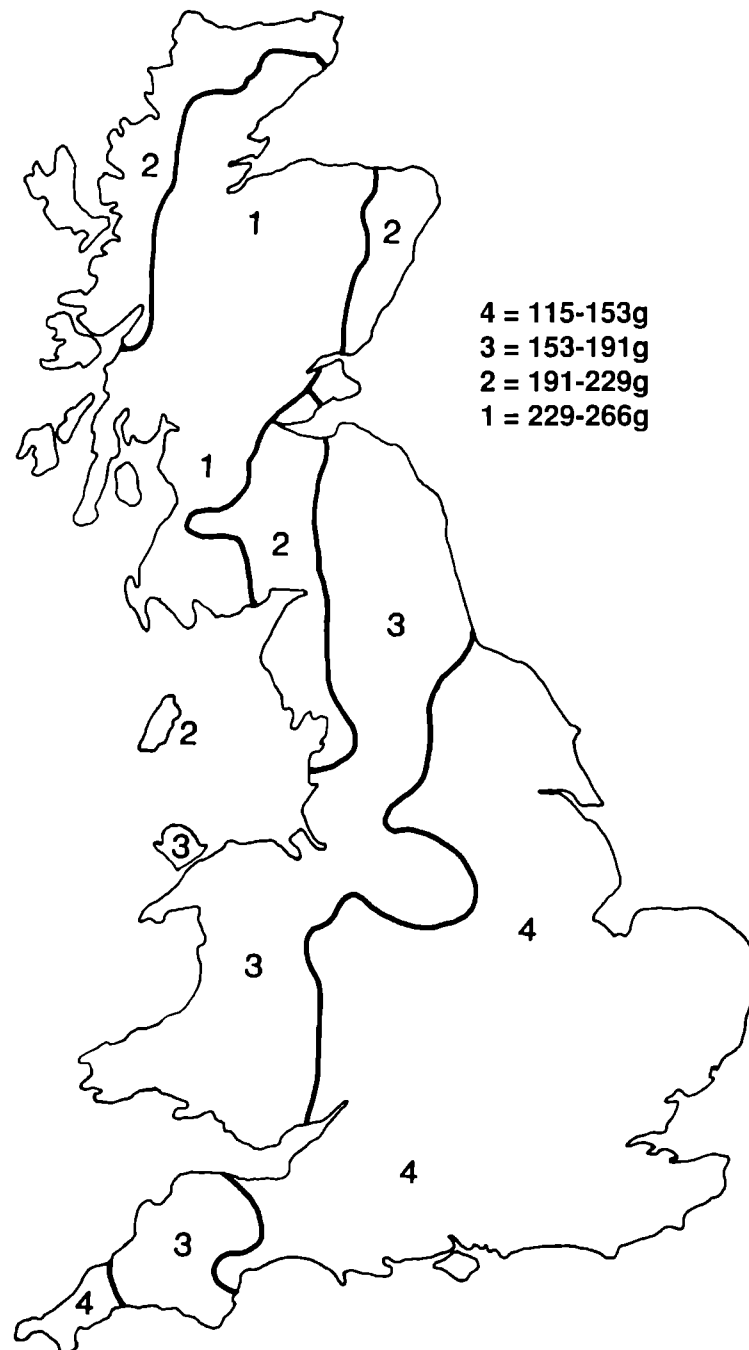


Figure 5 Average species weight (g) of land breeding birds (after Cousins, 1989).

Finally, evidence from a survey of the feeding habits of British insects (Price, 1977) shows that species of parasite (72.1% of the fauna) are much more numerous than species of predator (6.3% of the fauna, including non-parasitic herbivores). In this case parasites are about 10 times as numerous as predators; the remaining species are saprophytes.

This brief review of ‘functional’ diversity indices has suggested simple alternatives to the more familiar Shannon index. Species density, species size and species as predators, saprophytes or parasites provide a set of functional descriptions of ecosystems. These categories may also be applied to the relations in a taxonomic hierarchy of Vane-Wright *et al.* (1991) and used to refine conservation priorities. For example, a closely related but very differently sized organism may change its ranking in a species priority list.

Taxonomic uniqueness measures

Species differ from each other within the range at which sibling species are very similar. Species are increasingly different if they belong to dissimilar, progressively higher taxa. These higher taxa reflect progressively greater differences in anatomy or body plan. Conventional species diversity indices, such as Shannon’s H' (Pielou, 1975), are typically applied to clearly defined taxonomic groups. The precise taxon level differs for the group studied. Thus, for birds the indices are at the level of Class, while for butterflies or moths the taxonomic level is the Order. Plant diversity measurement is often made at the Kingdom level, without reference to the plant divisions, but with observations limited to particular size categories, e.g. diversity of trees or of field layer plants. Thus, existing

indices treat each species as being equal within apparently arbitrary limits of taxonomy and treat species as being different if they lie outside these limits.

In contrast to this, a structured approach to the uniqueness of biological form has been developed for use in conservation biology. Atkinson (1989) states ‘Given two threatened taxa, one a species not closely related to other living species and the other [related to a] widespread and common species, it seems reasonable to give priority to the taxonomically distinct form’. Vane-Wright *et al.* (1991) have explored the implications of measures of taxonomic distinctiveness. They use the hierarchical taxonomic classification to calculate an information content for species dependent on the branch points of the classification tree. They are able to show the value of their technique by a study of the worldwide distribution of bumble-bees in the *Bombus sibiricus* group. If a simple species count is used to locate the grid square of maximal diversity then the Ecuador square is selected with 10 species (23% of world total). However, when taxonomic distinctiveness is allowed for, Gansu in China is selected, with 23% of the world total as against Ecuador’s 15%.

This method can be applied at any spatial scale and indeed it has been argued (Cousins, 1994) that the method is particularly suited to the local ecosystem scale and thus would be useful in forestry applications.

The actual taxonomic distinctiveness technique is based upon the branching hierarchies of the taxonomic relationships between species. There are a number of ways of approaching this but a typical one is as follows (see Figure 6).

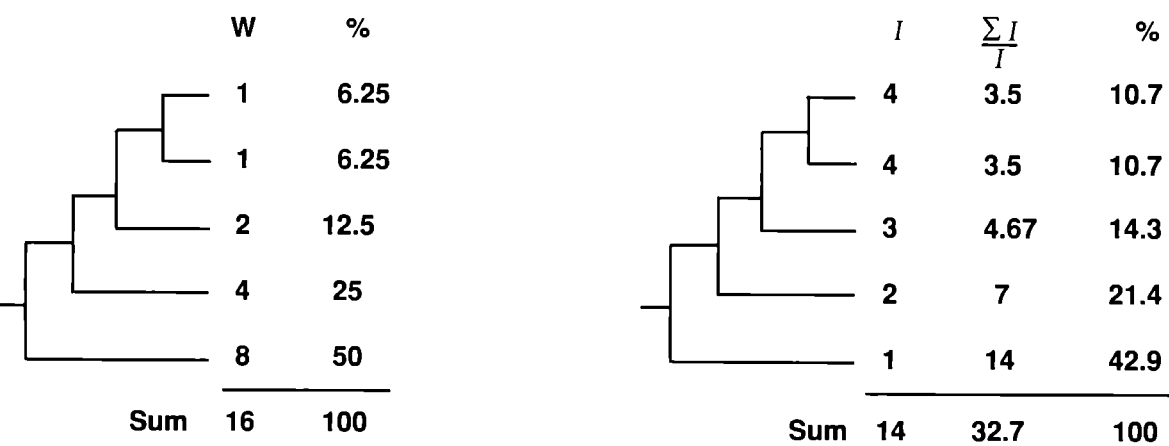


Figure 6 Methods of calculating taxonomic information indices (after Vane-Wright *et al.*, 1991).

The species which has the most branches between the stem and the tip is set equal to 1, then the sister group to this is given a score equal to the sum of the existing branch values. This is repeated until all species have been included. The weightings can be expressed as a percentage. However, this appears to overweight the value of the taxonomically distinct species, since the most distinct will always be equal in value to the sum of all the other species. To amend this approach, Vane-Wright *et al.* (1991) have proposed an 'information' index

based on the number of branchings in the tree that include the species whose characteristics are being measured. They then divide the sum of the branches affecting each species by the value for the individual species itself. Finally, this is expressed as the percentage contribution each terminal taxon makes to the total diversity, as measured by *I*. An example of the use of their scheme is shown in Figure 7 for parrots in the Amazon Basin forests.

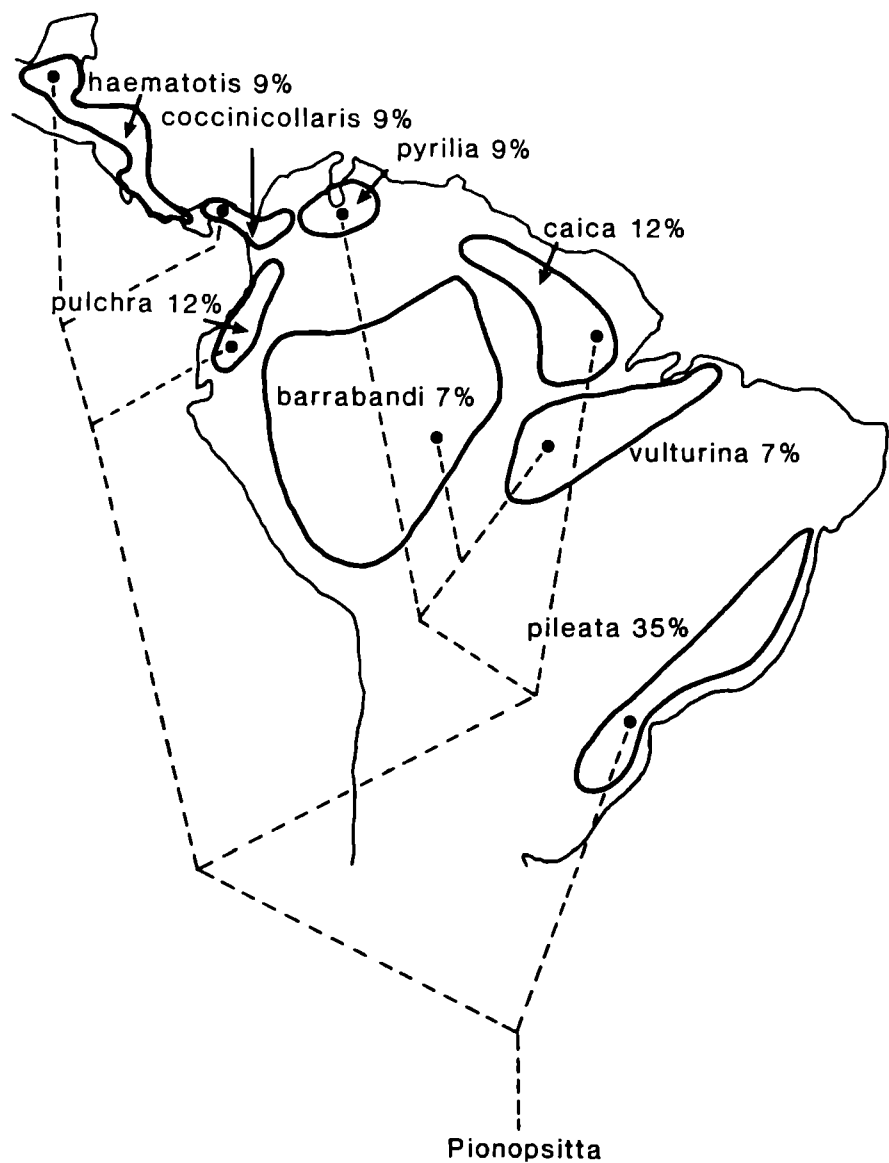


Figure 7 Phylogeny and distribution of parrots of the genus *Pionopsitta* in South and Central America. The relative amount of conservation effort that ought to be allocated to each species is indicated as a percentage following the taxonomic weighting scheme of Vane-Wright *et al.* (1991) (after Barrowclough, 1992).

Ecosystem concept and ecosystem object

The Lindeman (1942) definition of an ecosystem, 'the system of physical-chemical-biological processes active within a space-time unit of any magnitude' is problematic for conservation evaluation since, whatever change occurs, the ecosystem always remains. The ecosystem object (hereafter referred to as the Ecosystem Trophic Module (ETM); Cousins, 1990), unlike Lindeman's ecosystem concept, is countable and has a characteristic spatial scale, because the ETM is identified as the territory occupied by a single social group of the top predator. These predators can be counted and it is then possible to evaluate particular effects on ecosystems, including the loss of an ETM or its downgrading to a smaller scale, by the loss of the primitively largest species of predator and its replacement by smaller predators.

In Britain until recent times, the top predators were brown bears (*Ursus arctos*) and wolves (*Canis lupus*), although their abundance prior to extensive human modification of the landscape is not known. Harting (1894) reports that it was probable that bears were extinct in Britain by the tenth century, whereas the wolf survived until about 1500 in England and Wales, 1740 in Scotland and 1770 in Ireland. In contemporary Britain the fox is the largest predator.

The loss, from the United Kingdom, of the primitive top predators and their replacement by smaller ones raises interesting research questions. The primitive top predators are ground based since they are too large to fly or burrow, and so the system can be viewed as two dimensional. Smaller replacement predators must be considered as inhabiting a three dimensional space, which is of particular relevance to forestry. This points to special priority for avian and smaller mammalian predators.

Top predators tend to have extensive species distributions. For example, apart from its absence from some mountainous regions, the fox is ubiquitous in the UK. This does not mean, however, that there is only one *type* of ETM in the UK. The ETM, although defined by the area occupied by the social group of the top predator, includes, as its parts, all individuals of whatever species are present in that area at any given time. Traditionally, plant functional types have defined biomes

and, at a smaller scale, plant taxonomic categories have defined UK ecosystem types, for example grassland, oak woodland, heather moorland, and so on. Similarly, it is convenient to define types of ETM by plant types and there may be links here to the biogeoclimatic classification of ecosystem types presented by Pyatt (pages 28–31) and Nixon (pages 32–34).

Human impact assessment

The measurement of biodiversity has commonly been used as an index of human impact on the environment. However, for the purposes of conservation evaluation, human impact needs to be seen as an independent variable which is measured and capable of being equated against the results of different conservation strategies. There is no consensus on how this measurement of human impact should be achieved, but a number of lines of inquiry are opening up and these need to be explored fully. The ETM provides one framework to examine these issues.

Hunting, appropriation of the products of photosynthesis and pollution, in total, reduce the possible density of primitive top predators and, ultimately, the size of top predator species that can be sustained in an area. Counts of top predators thus also provide an integrative measure of the status of natural ecosystems (Cousins, 1990), suitable as a conservation evaluation tool. Significantly, it is not the number of types, but the quantity of top predators that provides the evaluation tool.

Conclusion

The familiar and well-established methods of species diversity measurement (Pielou, 1975) are being extended by new developments in taxonomic and functional measurement of biodiversity, in which species are ascribed different relative weightings. The new taxonomic methods may be applied at any spatial scale. In this paper, the ecosystem scale, defined by the size of the ecotrophic module, has been stressed as being of particular importance. It is suggested that functional methods of biodiversity measurement, including species density, species body size and trophic categories of predator, parasite or saprophyte, ensure a sound functional basis for choices concerning biodiversity. The case is made for improving local conservation evaluation by using species density measures and Williams' α rather than the Shannon H' and J indices.

New aspects of biodiversity measurement arise from monitoring populations of top predators. It is suggested that methods of monitoring human impact are needed to assess independently complimentary levels of biodiversity. These developments, when taken together, hold out the possibility of setting standards for levels of biodiversity for particular land classes and geographical locations.

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Genetical aspects of small population size in relation to conservation

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Summary

Both random genetic drift and inbreeding increase as population size decreases. The resulting erosion of genetic variability, fixation of deleterious alleles and inbreeding depression increase the probability that a small isolated population will become extinct. For reasons to be discussed, natural populations usually behave genetically as if they are smaller than in fact they are. This is termed the genetically effective population size.

In the 1980s there was considerable enthusiasm for the concept of a minimum viable population (MVP) – the size that would ensure a certain probability of persistence for a given length of time. However, it is becoming increasingly recognised that MVPs based on genetic considerations alone may ignore crucial demographic consequences of small population size. Both ecology and genetics, and the way in which they interact, are important.

Introduction

Following Shaffer (1987), the risk that a population will become extinct is influenced mainly by three types of factor: genetic, demographic and environmental. Chance plays a role in all three, and they all become more important in their possible consequences as population size decreases. Their effects interact with one another, and may do so synergistically so that the slope to extinction is steeper than consideration of each in isolation would suggest.

In the early 1980s biological conservation embraced genetical principles with great enthusiasm, and the relative neglect of other factors had some undesirable consequences (Lande, 1988). The processes described below are equally applicable to plants and animals, and although some examples refer to inhabitants of woodlands this

need not be considered an important criterion. Interestingly, in the genetical chapter of *The fragmented forest* (Harris, 1984) examples such as the northern elephant seal and the passenger pigeon are used.

The Hardy-Weinberg Law states that, given a number of assumptions including infinite population size, allele and genotype frequencies remain constant from generation to generation. In real, finite populations, two interrelated processes occur: random genetic drift and inbreeding. A particularly clear account of these finite population effects can be found in Falconer (1989).

Random genetic drift

In a population of constant size with N individuals, $2N$ gametes are picked at random to form the zygotes at the beginning of each new generation. If the population size is small the frequency of an allele among the successful gametes may differ for purely statistical reasons from its frequency in the parent population and the gametes they produce. The result is that allele frequencies fluctuate at random from generation to generation. This is random genetic drift, and its main consequences are as follows:

1. genetic differentiation occurs between isolated populations; and
2. genetic variation within populations is reduced through chance loss of alleles.

The loss of genetic variability leads to an increase in homozygosity and reduced potential for evolutionary adaptation to environmental changes. Rare alleles are lost more rapidly and these may include potentially useful ones – for example, those that might confer resistance against some future infection. Cheetahs are notable for their lack of genetic diversity; more than half the cheetahs in American

zoos succumbed to a feline virus which rarely kills domestic cats (O'Brien *et al.*, 1985).

The rate of loss of genetic variability is about $1/(2N)$ per generation, where N is the population size. (Note that in this context there is a very important qualification attached to the definition of N : this is dealt with below). If N remains constant at ten individuals, after ten generations 40 per cent of the original genetic variation will be lost, whereas if N is 100, only 5 per cent of genetic variation will be lost after ten generations. In two species of plants currently declining in The Netherlands, *Salvia pratensis* and *Scabiosa columbaria*, significant positive correlations have been found between population size and levels within populations of allozyme variation and variation in quantitative morphological characters (van Treuren *et al.*, 1991; Ouborg *et al.*, 1991).

The degree to which drift affects allele frequencies depends, in practice, not only upon population size but also upon whether the locus in question is influenced by natural selection or whether gene flow occurs between populations. Even quite low levels of gene flow, say the exchange of one successful migrant per generation, can significantly slow down the rate of genetic divergence between populations. It should be noted, however, that isolation is often associated with rarity or small population sizes.

Inbreeding

Even if the individuals in a small population mate strictly at random, some of the matings will by chance be between close or more distant relatives. This leads to inbreeding, the results of which are reduced heterozygosity and inbreeding depression, resulting from recessive or partially recessive deleterious alleles becoming homozygous. Inbreeding depression may have many effects, but particularly important are reduced viability and fecundity.

It is usually stated that the amount of depression resulting from a given level of inbreeding depends on how rapidly the inbreeding occurs and on the natural breeding system of the species. Under conditions of slow inbreeding, natural selection can purge deleterious recessive alleles from the gene pool, as they become homozygous, so that little permanent inbreeding depression results. Furthermore, those species which regularly inbreed, as do many plants, will have been subject already to this purging process, so that the imposition of more severe inbreeding should result in

little inbreeding depression. However, even in natural inbreeders, outcrosses usually result in improved vigour and, in general, when the effects of inbreeding have been properly sought they have usually been found (Charlesworth and Charlesworth, 1987).

The reduction in heterozygosity in an inbred population compared with the level expected in an infinite randomly mating population is measured by F , the inbreeding coefficient. The increment in F per generation in a finite randomly mating population is approximately $1/(2N)$, i.e. equivalent to the rate of loss of genetic variability through random genetic drift. The same qualification is attached to the definition of N as in the case of drift and will now be considered.

Genetically effective population size

The per generation rates of $1/(2N)$ only apply to a population of N *breeding* individuals for which the following conditions hold:

1. The number of breeding individuals must remain constant at N over each successive generation.
2. All individuals are hermaphrodite and self-fertilisation occurs at its random frequency of $1/N$.
3. All breeding individuals have an equal *chance* of transmitting genes to the next generation.
4. Generations are non-developing.

A population that conforms to these conditions is called an *idealised* population. Natural populations do not, and the effects of deviations from the ideal condition cause them to behave genetically as if they are smaller than they actually are.

Wright (1931, 1938; summarised in 1969) introduced the very useful idea of the *genetically effective population size* (N_e), the reduced size of an idealised population that would show the same amount of drift of inbreeding as the real population under consideration. He provided formulae by which N can be converted to N_e , given that the deviations of the real population from the ideal condition can be quantified. A clear and helpful discussion of the application of these formulae in the context of conservation is provided by Lande and Barrowclough (1987).

Many plants and most animals are not hermaphrodite but have separate male and female individuals. The absence of self-fertilisation is of little consequence, but if the sex ratio amongst breeders deviates from 1:1 the effective population size is reduced. For example, if there are 70 breeding individuals, 20 males and 50 females, the population will behave genetically as if it were really a population with a size of only 57: in $1/(2N)$, $N_e=57$ must be substituted for $N=70$ to predict the amount of drift or inbreeding.

Anyone who has participated in a long-term species monitoring programme will have been impressed by the way numbers of plants or animals in a population fluctuate. The aggregate genetic drift or inbreeding over a number of generations is equivalent to that of a population with a constant size N_e , equal to the harmonic mean of the population sizes in each generation. Periods when the population is small in size have a highly weighted and lingering effect. For example, if an effective size $N_e=100$ is required but, say, in 1 in $t=10$ generations on average we can expect the population to crash to about $N^*=20$ individuals, then in the remaining generations the population size must be at least $N=180$. Remarkably, if $N^*/N_e \leq 1/t$ (i.e. in this case the crash is to 10 individuals on average 1-in-10 generations or to 20 individuals 1-in-5 generations) then the population size in the remaining generations must be infinite.

In many cases, particularly with plants, the most important reason why effective sizes will be less than actual sizes is because individuals do not have equal chances of transmitting genes to the next generation. If they do, offspring number should follow a Poisson distribution with each individual having on average two offspring (assuming constant population size), and the variance in offspring number should be equal to the mean. In practice, fecundity distributions are often L-shaped with a small number of individuals being super-fecund and many individuals being relatively poor as parents; the variance:mean ratio is then greater than one, the value appropriate for a Poisson distribution. A striking example concerns the poppy, *Papaver dubium* (Mackay, 1980; discussed by Crawford, 1984). In a population of 2316 plants, 50 per cent of all seed were produced by only 2 per cent of the population and 4.6 per cent of seed were produced by the single most fecund plant. The variance:mean ratio suggested that the effective size of the population was only 7 per cent of the actual size for this reason alone.

Each of the above examples has dealt with one deviation from an idealised population, in isolation from other reasons for deviation that may also be relevant; Lande and Barrowclough (1987) show how the different effects may be superimposed to provide a combined estimate of effective population size. Begon (1977) studied five successive generations of *Drosophila subobscura* in a woodland near Leeds. Population size estimates increased from 923 to 15 787, with a harmonic mean of only 3507. Under experimental conditions the variance:mean ratio of family sizes was 14.8, so that over the five generations the population would behave as if it were constant at 56 individuals as far as drift is concerned.

The critical question is what sort of values of the N_e will usually be of the order of one-tenth, or less, of N (see Dobson *et al.*, 1992).

Minimum viable population size

The early 1980s saw the emergence of the concept of minimum viable population size (MVP), the genetically effective population size that has to be maintained in order to ensure that a population can have a certain probability (say 0.95) of surviving for a given length of time (say 100 generations), in spite of genetic erosion from random genetic drift and inbreeding (Franklin, 1980; Soulé, 1980; Frankel and Soulé, 1981). For an historical appraisal see Simberloff (1988).

Taking drift first, the loss of genetic variability could be balanced by new variation from mutation if the effective population size were sufficiently large. It has been suggested from experimental work on *Drosophila*, mice and maize that the frequency of mutations affecting quantitative traits is of the order of 10^{-3} per character per generation. Equating $1/(2N_e)$ to 10^{-3} implies that an effective size of 500 individuals is required to nullify the effects of drift.

Turning to inbreeding depression, the general experience of animal breeders has been that an increase in the inbreeding coefficient of one per cent per generation can be sustained. Equating $1/(2N_e)$ to 0.01 suggests that an effective size of 50 individuals is required to reduce inbreeding depression to acceptable levels.

It is important to remember that these effective population sizes of 50 or 500 imply larger actual population sizes because real populations deviate from the idealised condition as discussed above.

These conditions were seized upon with great enthusiasm and soon became known as 'the basic rule of conservation biology' or 'the 50/500 rule'. To call them a rule totally ignores a number of problems associated with their derivation.

1. The estimates of mutation rates were obtained from laboratory or cultivated stocks and concerned characters such as bristle number in *Drosophila*. Different values might be more appropriate for characters of conservation relevance that are also subject to natural selection, for example viability, fecundity, competitive ability or life-history characters.
2. The relevant mutation rates for characters controlled by single gene loci are lower, 10^{-5} to 10^{-6} per locus per generation, implying MVPs of 50 000 to 500 000.
3. The equations used to calculate effective population sizes when populations deviate from the idealised condition are approximations. In particular, they assume that generations are non-overlapping and that populations have a stable age-structure.

Nevertheless, the initial uncritical application of the MVP concept based on genetic criteria has generated some healthy scepticism; the relative importance of genetic and demographic factors in preventing population extinction is a matter of current debate (e.g. Simberloff, 1986, 1988; Soulé, 1987; Lande, 1988).

Demographic factors

Two examples of demographic factors are described here, mainly as a contrast to the genetic factors already covered.

In the early 19th century the passenger pigeon (*Ectopistes migratorius*) numbered billions in eastern North America and probably accounted for more than half of all the terrestrial birds. They migrated in flocks that were so large they darkened the sky. They became subject to heavy hunting pressure, and from about 1879 the population abruptly fell in size. The last individual died on 1st September 1914 in the Cincinnati Zoo. It is thought that the huge flock sizes were necessary to stimulate reproductive behaviour and that breeding ceased below a critical threshold flock size.

This type of non-genetic effect on breeding behaviour as a result of decreased population size is known as an Allele effect. It is not restricted to animals. Small populations of insect-pollinated plants may fail to attract their pollinators if their

floral display becomes too reduced. This will result in reduced seed production if the plant is self-incompatible (Widén, in press).

In small populations random fluctuations in demographic variables such as birth rates, death rates and sex ratios may become important. For example, in very small populations it may happen by chance that all the individuals in reproductive condition at the same time happen to be either all male or all female. What is becoming increasingly recognised is that demographic factors and genetic factors interact with one another. Inbreeding depression can affect one sex more than the other and cause the sex ratio to deviate from 1:1. This, in turn, causes effective population size to be reduced with the result of further inbreeding depression. The vicious circle could then bring the population to the point where only one sex is available for reproduction and demographic extinction occurs.

Many plants without a self-incompatibility system show a mixture of outcrossing and selfing. In low density populations a greater proportion of ovules tends to be self-fertilised, or outcrossed with pollen from a single source. This is a feature of both wind-pollinated and insect-pollinated plants (Murawski and Hamrick, 1991) and will tend to reduce effective population size and increased inbreeding.

Byers and Meagher (1992) have shown how random genetic drift may lead to reduced genetic diversity at the self-incompatibility S-locus. Because a plant can only be fertilised by pollen from another plant that has at least one different S-allele, the number of available mates will be limited and seed-set reduced. Furthermore, the variance in the number of mates available to different plants will tend to increase, leading to greater differences between plants in their levels of seed-set. This will cause further reductions in the effective population size.

Complex interactions between genetic and non-genetic factors are usually the case. The problems of small population sizes in conservation biology require solutions that integrate genetics and ecology.

A final example

Lande (1988) discussed the case of the North American red-cockaded woodpecker (*Picoides borealis*), an inhabitant of old pine forests and officially classified as endangered. They prefer forests with openings caused by fires that prevent the invasion of hardwoods. The official Recovery Plan was based on genetic criteria and aimed for local

populations of 500 breeding individuals. However, the study showed that the bird was continuing to decline rapidly, mainly because of fire prevention measures.

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Species richness and the application of island biogeography theory to farm woodlands

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Summary

The aim of the Farm Woodland Scheme was to increase timber production as well as to produce wildlife benefits. By using sets of woodlands planted during the first half of the 20th century, this paper aims to determine what factors affect these wildlife benefits.

The island nature of a farm woodland is a reflection of the mobility of different groups of species. Thus, for soil mites, the surrounding fields were almost uninhabitable, while for small mammals the woodlands were not like islands at all. Larger woodlands are more species rich than smaller woodlands. A minimum size should be 1.0–1.5 hectares (ha), though woodlands of 5 ha and more should be encouraged.

In the British lowland agricultural environment, isolation is not seen to be important in determining species richness, but it might be more important if the genetical structure of woodland populations were to be investigated.

For the characteristic woodland species, as opposed to the total number of species, more compact woodlands are shown to be more important than shelter belts or other narrow woodlands. The central core zone has a more truly woodland environment.

Introduction

The Farm Woodland Scheme, introduced in 1988, aimed to bring some land currently in agricultural production under woodland (Insley, 1988). As well as attempting to increase timber production, the scheme was seen as having environmental and recreational benefits, and as contributing to farm income through sporting interest and the like. The studies discussed here

were carried out in farm woodlands in the Vale of York and aimed to investigate aspects of their species richness.

The term 'biodiversity' contains two essential components. First, there is the species richness component. Conservationists have traditionally valued species-rich communities (see, for example, Ratcliffe, 1977). Second, there is the genetic diversity component. Crawford (pages 16–21) has looked at aspects of population genetics, but in terms of farm woodlands the important aspects are whether there will be small, isolated populations that will suffer from inbreeding, etc., or whether in an archipelago of farm woodlands a species will act like a metapopulation, with gene flow between sub-populations.

With a new scheme intended to create farm woodlands, it would obviously be useful to track the change in species complement and abundance, from those that occur in arable land to those that occur in mature woodland. However, such studies would inevitably have to be extremely long term – probably lasting more than a century – while the whole colonisation, extinction and successional process was examined. In order to short-circuit the need for such long-term research, it was decided to survey farm woodlands in the Vale of York that were known or inferred to have been established this century; the woodlands were all thought to be between 40 and 90 years of age. The aim of this paper is to examine the species richness of many groups of plants and animals in these woods.

Woodland margin

When considering the island biogeography of farm woodlands, the first question that needs to be addressed is just how distinct the woodland margin is. In other words, how distinct are the

communities of plants and animals on each side of the margin?

Working in a single farm woodland, Sgardelis and Usher (1994) sampled the moss mites (Acari: *Cryptostigmata*) along transects between the farm woodland and the surrounding arable field. The communities 1 m, 4 m, 9 m and 18 m into the woodland were all very similar in their species composition, having between 29 and 33 species. On the margin between the woodland and the field, the number of species was only 24, while 1 m into the field this had dropped to 11. From 4 m onwards into the field, only five species were recorded, four of them very infrequently. For example, *Tectocephus velatus*, which had densities of more than 100 individuals in a soil core in the woodland, had densities of less than one individual per core in the field. This study indicates that, for the soil inhabiting mites, the woodland margin was 'hard' and, therefore, that the woodlands are likely to be more like true islands.

A study of ground beetles and spiders across woodland margins (Bedford and Usher, in press) has indicated that the margin is more likely to be 'fuzzy'. Some species showed a strong preference for woodlands, though they were also found in pitfall traps in the surrounding arable fields. Thus, for example, *Callathus piceus* had a probability of 0.94 of occurring in a pitfall trap within the woodland, 0.82 of occurring in a trap on the wood/field margin, and a probability of 0.48 of occurring in a trap in the field. Other species of ground beetle, such as *Nebria brevicollis*, showed a distinct preference for the field environment, with probabilities for occurring in woodland, edge and field traps of 0.27, 0.53 and 0.63, respectively. One species of ground beetle occurred only within the woodland, and three species only within the field. Similar results were obtained for the spiders, though there were rather more confined to the woodland and fewer confined to the fields.

The studies of the small mammals by Zhang and Usher (1991) have indicated that the woodland margin is not recognised at all by these species. Although there is some evidence that hedgerows may be used by wood mice (*Apodemus sylvaticus*) for movement between woods, there was plenty of evidence to suggest that these species moved across the surrounding fields, especially when there was some vegetation cover.

There is, therefore, no clear answer to the question of whether a farm woodland surrounded by arable land is analogous to an island. For some groups of organisms – those that are less mobile or confined to the soil environment – it is probable that farm woodlands are exact analogues of islands, a suitable habitat surrounded by a sea of unsuitable habitat. For highly mobile species – the small mammals and birds, and perhaps some of the more strongly flying insects – there is essentially no analogy to islands. The species are perfectly able to use the intervening habitats, and move from wood to wood. However, between these two extremes, there is a large number of species for which the margins of the farm woodland are 'fuzzy'. These species may tend to rely on the farm woodland habitat, but they also occur to some extent in the surrounding habitats.

Woodland size

If woodlands are behaving like islands, one would expect more species in larger woodlands. This leads immediately to a dilemma; how does one determine the number of species in any woodland? Obviously, one wants to get as complete a list of species as possible, though this is often more or less impossible. Disney (1986) has indicated how difficult it is to get towards completeness of a species list for a large group of insects. Hence, the species in the habitat have to be sampled and the species richness estimated from these samples. There are two ways to sample: either a standard sample for each island, or a sample that is proportional to the island area. For the former, one might be taking a constant number of quadrats, putting out a constant number of traps, or searching for a constant amount of time. For the latter, the number of quadrats, the number of traps, or the length of time for the search would be proportional to the area, so that one woodland twice the size of another woodland would have twice the sampling effort. Whichever method is used, either attempting to get a complete list or a list based on one of the sampling strategies, the results of subsequent analyses are likely to vary.

For the plants in farm woodlands, Usher *et al.* (1992) attempted to obtain a complete list for each woodland. There was a reasonably close relationship between the number of higher plants and the area of the woodland (Figure 8). Interestingly, the type of woodland, be it deciduous, coniferous or mixed, did not seem to affect the species

richness. For the moths, there are some interesting differences (Keiller and Usher, unpublished). Thus, the Noctuidae, which are strong fliers, showed no significant relationship between the number of species and the area of woodland. However, the

Geometridae, which are much weaker fliers, showed a species-area relationship. These differing results probably reflect the degree of 'islandness' of the woodlands in relation to the abilities of the moths to move from wood to wood.

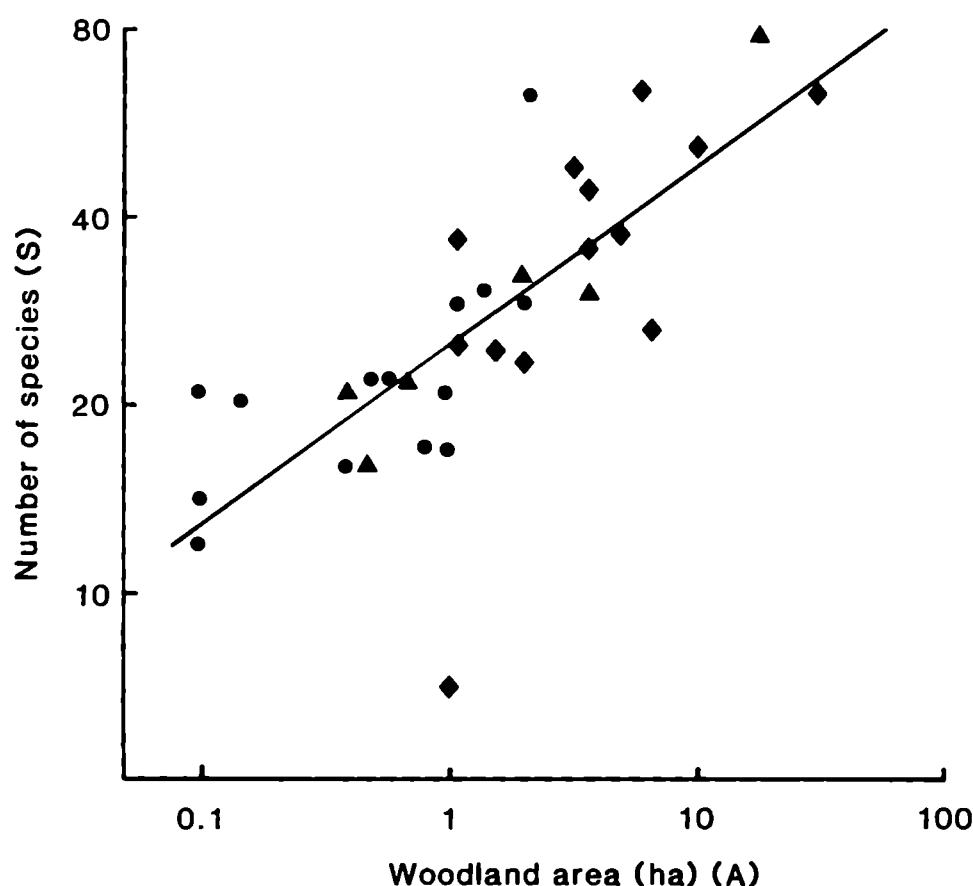


Figure 8 The species–area relationship for herbaceous plants in 33 farm woodlands in the Vale of York. The regression line, $S = 1.81 A^{0.284}$, is shown. The woodland canopies are deciduous (circles), coniferous (triangles) or mixed (diamonds) (from Usher *et al.*, 1992).

Using a standard set of samples, eight pitfall traps for ground beetles and spiders, there was again some evidence for the effect of size of woodland. Thus, although this effect was not noticed for all ground beetles, there was a strong effect of area on the community of ground beetle species that occur predominantly in woodlands; the larger a farm woodland the greater the probability that woodland species would occur. A simple analysis for spiders indicated that there was a weak species-area relationship. Thus, for these two groups of invertebrates, for which the boundary

was shown to be 'fuzzy', there is not a strong species–area relationship.

This range of studies in the Vale of York tends to indicate that the hardness or otherwise of the woodland boundary to any particular group of species determines the likelihood of finding a species-area relationship. The more distinct the boundary appears to a group of species, then the greater the likelihood that there will be a close species-area relationship, with the number of species increasing as a function of the area.

Woodland isolation

Isolation is a relative factor. In island biogeography, we are probably thinking of islands or archipelagos that are separated from the mainland by hundreds or even thousands of kilometres. For the farm woodlands in the Vale of York, the maximum possible separation is about 2 km. It also has to be borne in mind that there are likely to be many small pockets of woody vegetation, such as hedgerows, which could act as stepping stones, thus reducing any possible isolation factor even further.

Given these caveats, it is probably not surprising that isolation tends not to determine the species richness of farm woodlands. Usher *et al.* (1992) found no effect of isolation on the plant species richness. Zhang and Usher (1991), in documenting the movements of wood mice and bank voles between farm woodlands, concluded that isolation was not a factor for the colonisation of farm woodlands by these small mammals. Keiller and Usher (unpublished) also found no effect of wood isolation either on the total moth species richness, or on the species richness of either of the main families of moths. The only group studied which showed an effect of isolation is the spiders. Usher *et al.* (in press) have shown that the number of species of spider in a farm woodland can be predicted from the equation:

$$S = 6.86 \ln N - 1.59 I - 7.6$$

where S is the number of species, N is the number of individuals and I is an index of isolation (ranging from 0.02 for the least isolated to 4.40 for the most isolated, for the 28 woodlands included in the study).

Isolation from other farm woodlands, at least in a reasonably well-wooded landscape, therefore seems to have little relevance to the species richness of those farm woodlands. It may, however, have a greater effect on the genetic diversity of the organisms, and on the flow of genes from population to population. However, there are no data available either to support or reject this hypothesis.

Woodland shape

What effect might one expect shape to have on the species richness of farm woodlands? At one extreme, it could be said that a wood that is as compact as possible, i.e. approximately circular, is more likely to have a large number of species. This argument would essentially say that a

compact woodland has a smaller margin, consequently a smaller marginal zone around it, and more true woodland habitat in the interior; thereby, the woodland species would be able to survive. At the other extreme, it could be hypothesised that a long, narrow woodland would intercept species that were dispersing in the landscape. The woodland would act, therefore, rather like a butterfly net, catching a large number of species, and the woodland would therefore be very species rich.

The farm woodlands in the Vale of York had a variety of shapes. Although none were circular, some approximated to squares, while others were long and narrow. Analysis of the plant data indicated that the number of species was independent of the woodland shape (Usher *et al.*, 1992). For the ground beetles, shape seems to be important, with longer and thinner woods having more species (Usher *et al.*, in press). For the moths, there are mixed results. Family Noctuidae showed no relationship with shape, whereas family Geometridae showed that compact woods were more species rich than long, thin woods (Keiller and Usher, unpublished). Similarly, looking at the diversity of all woodland moth species (i.e. excluding from the dataset all those moth species whose larvae do not feed on woodland plants), then once again there is a negative relationship between woodland shape and moth species richness.

The studies in the Vale of York do not, therefore, give an indication of whether the design of woodlands should be compact or linear. On balance, however, it appears that shape has little effect on species richness, though compact woods may be preferable for the conservation of characteristic woodland species.

Discussion

The first question which should be addressed is whether we are interested in biodiversity *per se*, or whether we are interested in some aspects of the diversity of wildlife in the countryside. Already, the concept of biodiversity has been narrowed to species richness, because there are insufficient studies of the genetic variation within the metapopulations found in areas with farm woodlands. Should species richness be considered in its totality or in some way restricted? Our contention would be to focus purely and simply on the richness of woodland species, i.e. those species that rely for their habitat or food on woodlands. If we are looking at farm woodlands, we should not be concerned with species which are occurring accidentally within them; these species

could not undertake the whole of their life-cycle within the woodlands. Thus, the subset of woodland species are those which rely exclusively on woodland environments either for the whole of their life-cycle, or for an integral part of their life-cycle. These species would become extinct locally if there was no woodland habitat. We should, therefore, in looking at species richness, be excluding those species which do not rely exclusively on a woodland habitat, at least for part of their life-cycle.

In making these restrictions, the results of biogeographical studies appear rather differently. For woodland plants, there is a strong species-area relationship. For woodland moths, there is a weak species-area relationship, and also an indication that more compact woods, for a given area, are more species rich than elongated woods. Similarly, for the woodland ground beetles, there is a weak species-area relationship. Insufficient is known about the ecology of individual spider species to be able to determine which subset of the total species list could be allocated to the woodland category. Therefore, there has been no attempt to recognise a woodland subset, though again there are some indications that woodland species are more frequent in larger woodlands.

Farm woodlands can only contribute to the total diversity of a geographical area if they contain woodland species. Hence, biogeographical studies on the total species list are likely to produce abnormal results, as found for the southern English heathlands by Hopkins and Webb (1984). If one can recognise the subset of woodland species, then these are clearly the ones that the design of farm woodlands should be encouraging. From the studies in the Vale of York, there appear to be two factors which are important in increasing the richness of the woodland species. The first, a factor that comes from most studies, relates to woodland area: the larger the farm woodland the greater the likelihood that it will contain a larger number of woodland species. The second factor relates to woodland shape: the more compact the woodland, the more of it that can be considered as real woodland habitat, as opposed to wood margin habitat, and again there is an increase in the number of woodland species.

What sort of guidelines might there be, then, for the design of farm woodlands? Using four woodland indicator plant species (derived from the lists produced by R. Gulliver, personal communication), only one, the bluebell (*Hyacinthoides non-scripta*), occurred in woodlands of under 0.5 ha. This is joined by the wood speedwell

(*Veronica montana*) in woodlands up to 1.5 ha, as well as by wood sorrel (*Oxalis acetosella*) in woodland of up to 5 ha. The fourth of the woodland indicator species, primrose (*Primula vulgaris*), only occurred in woodlands of over 5 ha. Thus, for the plants, there is a strong indication that farm woodlands should be more than 5 ha in extent (Usher *et al.*, 1992). Surprisingly similar results were obtained for the moths (Keiller and Usher, unpublished); the species lists were generally very poor for woodlands under 1 ha in extent. Interestingly, with compact woods, there seem to be few benefits in increasing the size above 5 ha. Thus, based on the plants and one group of herbivores, there seems to be a reasonable consensus that farm woodlands should be 5 ha or more in extent. A reasonable diversity of woodland species can be expected in woodlands that are between 1.5 and 5 ha in extent. However, smaller than that, the species richness of true woodland species is almost negligible. Hence, a woodland of less than 1–1.5 ha is essentially a landscape feature. Woodlands of greater than this size contribute both to landscape and wildlife conservation.

Acknowledgements

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A biogeoclimatic ecosystem classification for Britain

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Summary

The Forestry Authority Research Division has recently decided to develop a biogeoclimatic ecosystem classification (BEC) as the basis for the application of site-related and ecosystem-related research results in forest management. The system being developed will also have application for conservation and other forms of land use generally, that is on all land up to the mountain tops. It is a system primarily for use at the stand level, for guiding decisions about management operations. It will, for example, integrate with and refine the existing Forestry Commission site classification based on lithologies, soil types and phases.

BEC aims to be a classification of ecosystems or potential ecosystems. The definition of an ecosystem includes the physical environment of a piece of land and its biota; in practice we select climate, soil moisture, soil nutrients and vegetation for primary study.

The use of BEC in Canadian forestry

BEC is the name given in British Columbia (BC), Canada to a system of classification of vegetation, mainly forest types, and their associated climate, soils and other biota (Pojar *et al.*, 1987). The system was developed at the University of BC, Vancouver and has been in use by the Provincial Forest Service for 15 years, during which time many refinements have been made as the system has been extended and mapped throughout most of the country, an area four times the size of Britain (Meidinger and Pojar, 1991). The main exponent of the system at present is Karel Klinka of the University of BC, Faculty of Forestry.

Chris Nixon and I (Forestry Authority Research Division) spent 3 weeks in BC in autumn 1991, part of the time with Karel Klinka, and were impressed by the system and how it was applied in forest management. It seemed to us to offer a basis for classification of ecosystems as well as sustainable production from multi-purpose forests with enhanced biodiversity. The similarity of climate between the temperate rainforests of the Coast Mountains region of BC and western Britain suggested to us that of the various national or regional systems of land classification which we might use as a guide, this was likely to be the most appropriate (Klinka *et al.*, 1991).

Adaptation of BEC for use in Britain

The primary factor responsible for variations in ecosystems is considered to be climate. As a first approximation, the bioclimatic classification of Birse and Dry (1970) and Birse (1971) for Scotland, and Bendelow and Hartnup (1980) for England and Wales, will form the basis for a BEC in Britain. Five bioclimatic *zones* are shown in Figure 9. These will later be subdivided into a number of *sub-zones*. Within each zone or sub-zone, site classification will involve the recognition of nine classes of soil moisture regime (SMR) and six classes of soil nutrient regime (SNR) combined in an *edatopic* grid (Figure 10). Portions of the edatopic grid, each equal to two to five cells of the grid, will constitute *site types*, and the combination of climatic zone or sub-zone, site type and associated biota will

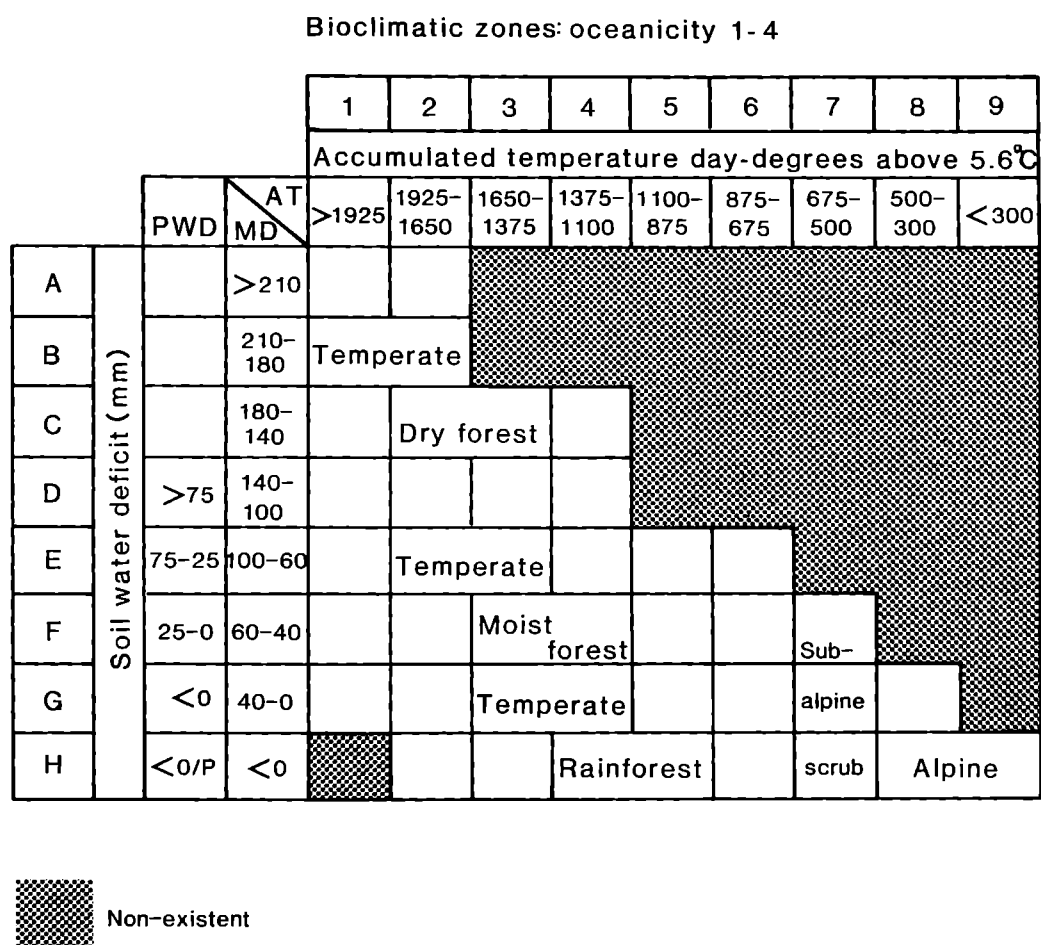


Figure 9 Suggested bioclimatic zones for Britain based on accumulated temperature (AT) and potential soil water deficit (PWD) or moisture deficit.

comprise the *ecosystem*. This is probably equivalent to the sub-community level of the National Vegetation Classification (NVC) (Rodwell, 1991). It is intended that BEC will be integrated with the NVC as far as possible (e.g. Figure 11).

Classes of SMR differ in the availability of moisture for transpiration during the growing season and, in the four wettest classes, in the depth and duration of the water-table restriction, soil aeration and rooting depth. Classes 1–5 of SNR differ mainly in the availability of nitrogen, although Class 6 comprises calcareous soils, and Class 5 is likely to contain base-rich soils of pH above 5.5.

The composition of the ground vegetation (species and cover fraction) will be used to aid recognition

of SMR and SNR, via *ecological indicator values* (e.g. Ellenberg, 1988). Additionally, a classification of forest humus types (Green *et al.*, in press) will help to characterise site quality, especially when ground vegetation is lacking.

Much research needs to be done to test whether methods developed in BC will work as well in Britain. It seems likely that the variation in geology and soil types is greater in Britain than in BC, and that our traditional emphasis on soil parent material as a major factor in site classification should not be forsaken lightly.

BEC has relevance not only in forest management. If it succeeds in classifying ecosystems well it will be equally useful to conservation and other land-uses.

EDATOPIC GRID

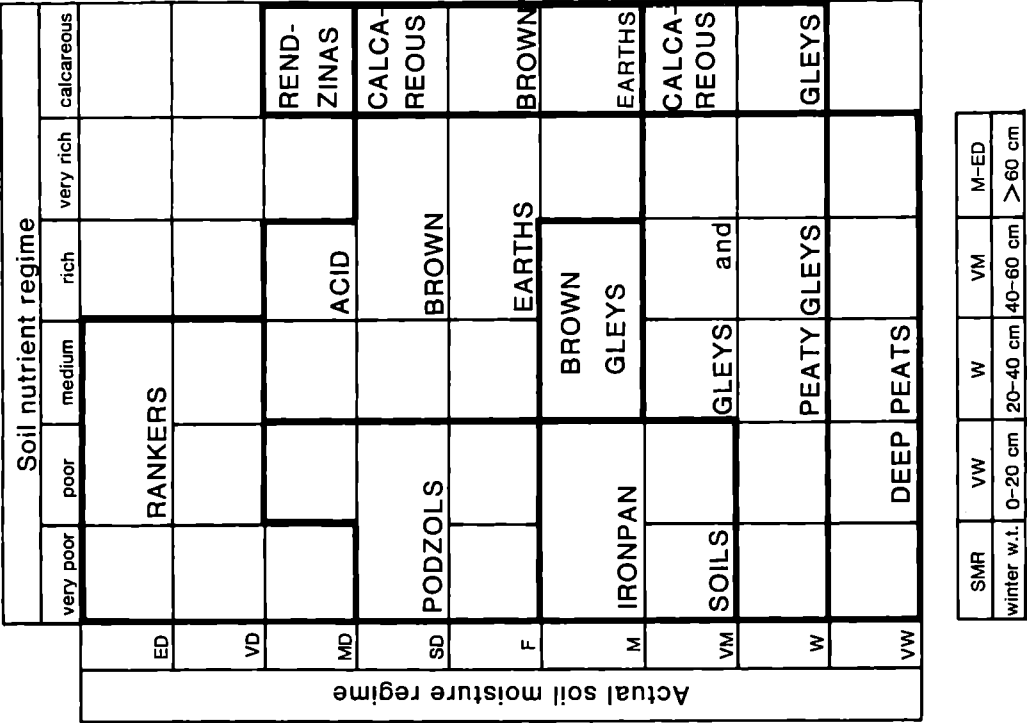


Figure 10 Edatopic grid based on classes of soil moisture and soil nutrient regime showing approximate positions of major soil groups or types. (In practice there is considerable overlap between the groups.) ED = extremely dry, VD = very dry, MD = moderately dry, SD = slightly dry, F = fresh, M = moist, VM = very moist, W = wet, VW = very wet. Winter w.t. = mean depth to water table in winter, SMR = soil moisture regime.

EDATOPIC GRID

ZONE: TEMPERATE RAINFOREST

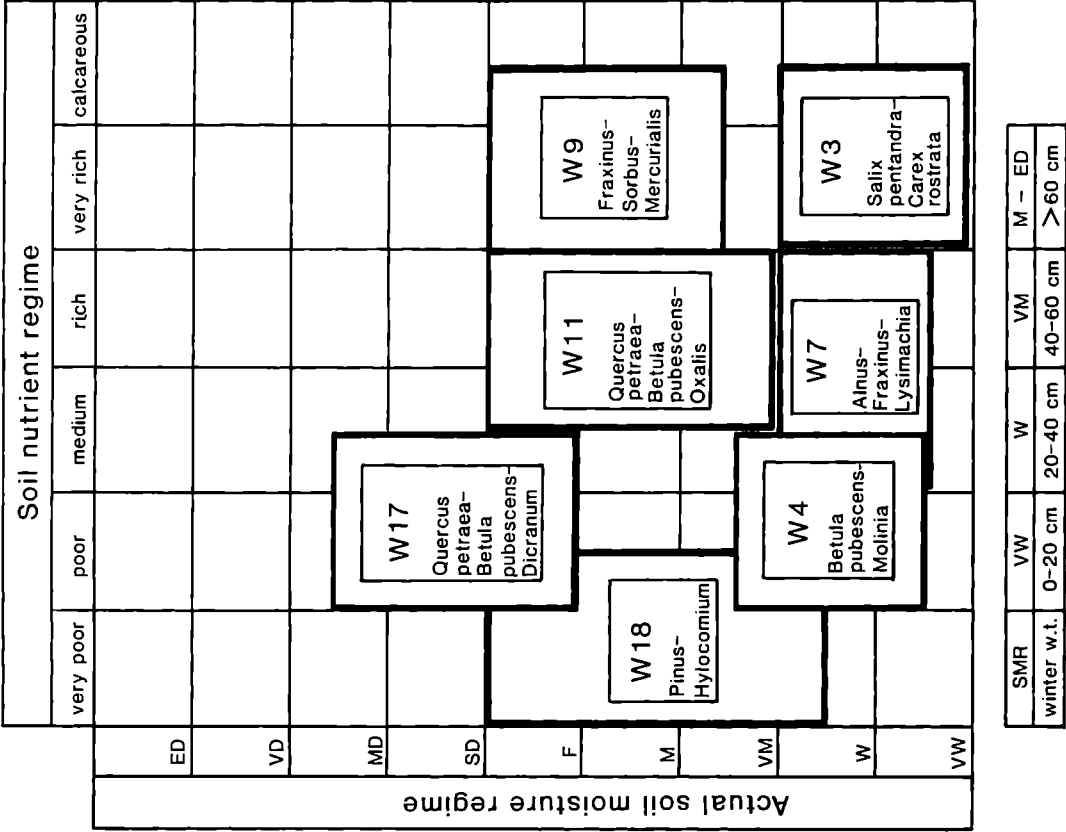


Figure 11 Tentative assignment of NVC woodland types to parts of the edatopic grid within the Temperate Rainforest Zone (based on Rodwell, 1991).

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Biogeoclimatic ecosystem classification and forest management for biodiversity

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Summary

This paper aims to illustrate how the interpretation of a biogeoclimatic ecosystem classification (BEC) system can aid forest management for biodiversity. Several practical examples are described.

Within the BEC system, site types, or units, are portrayed in an edatopic grid in relation to the soil moisture regime and soil nutrient regime. The classification can be made much more flexible by attaching additional descriptive terms to each site unit which relate to other characteristics of the ecosystem or potential ecosystem. For example, the basic site units can be grouped into closely related 'site series' which can support stands of the same species composition and structure, have similar productivity potential, and can be managed by the same silvicultural system.

Site classification

In Britain it is widely recognised that diversification, particularly of our conifer forests, is a desirable goal. What is less clear is the extent to which this can or should be achieved. Despite this, predetermined and deliberate forest management must play a key role in conserving and enhancing biological diversity.

To help achieve an appropriate balance, some form of site classification is essential to provide a framework for site-related management. BEC is a system of ecological classification widely used in British Columbia, Canada where the nature of the climate and topography results in a complex variety of forest ecosystems with a daunting level of diversity. The development of the BEC system was undertaken primarily to describe the variation present and to assist the management of forests on an ecosystem-specific basis (MacKinnon *et al.*, 1992).

BEC provides a hierarchical system which organises ecosystems at three levels – local (site type), regional (zone) and chronological (successional) (Pojar *et al.*, 1987). Within each zone or smaller sub-zone, sites are described in relation to the soil moisture and nutrient regimes with the variation being portrayed in an edatopic grid (Figure 12). Within the edatopic grid the range of site variation is delineated into site units, representing areas with equivalent physical properties which will support similar climax plant communities. Individual site units are often grouped for purposes of management into site series which might, for example, represent a range of sites that can be managed in a similar way, or have similar soil or vegetation characteristics (Figure 13).

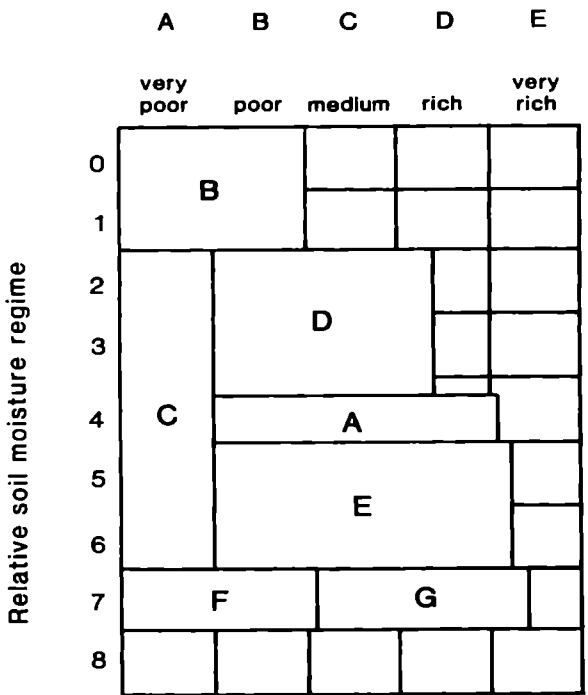


Figure 12 Edatopic grid displaying seven site units (from Mackinnon, 1992).

Prince Rupert Region Silviculture Stocking Standards

Site series	Target stocking (trees/ha)	Minimum stocking (trees/ha)	Preferred species	Acceptable species	Minimum spacing (m)	Regeneration period (yrs)
01	900	500	WH,DF	SS,LP	2.0	6
02	900	500	WRC	-	2.0	6
03	900	500	LP	WRC	2.0	6
04	900	500	SS	WH,DF,WRC	2.0	3
05	900	500	WH,DF	SS,WRC	2.0	3
06	900	500	SS	WRC	2.0	3
07,8,9	900	500	SS	WRC,WH	2.0	3
10	800	400	SS,WRC	-	1.0	6
11	A VOID HARVESTING					

WH = western hemlock

SS = Sitka spruce

WRC = western red cedar

DF = Douglas fir

LP = lodgepole pine

Figure 13 An example of silvicultural prescriptions related to site series for the Prince Rupert Region of British Columbia (adapted from Ministry of Forests, Province of British Columbia, Silvicultural Stocking Standards Manual).

Practical application

In the field, a forester will identify the site series after stratifying an area into homogeneous blocks with similar soil, site and vegetation features. Reference to field guides for the region in question will then provide site-related information on many aspects of management, all of which will be linked to the ecological potential of the site. Information on species choice, yield prediction, conservation and recreation values, etc., has been collated within the field guides, to enable appropriate management prescriptions to be made. In the context of biodiversity, this site-related information provides a means of encouraging ecologically sustainable forest management and defining appropriate levels of diversity.

Management planning and operations are simplified by combining many individual ecosystems into fewer, environmentally similar classes. The BEC system has become widely accepted in British Columbia, not through decree, but by virtue of its ability to make ecology workable and understandable. As a result, ecological awareness among forest managers has increased dramatically and

forest management practices have improved considerably.

The principles of the BEC system may be applied equally in Britain, although in the absence of naturally distributed vegetation cover the development of a similar classification system here would necessarily have to rely more heavily on soil and climatic factors.

Nevertheless, the value of the biogeoclimatic approach to site classification has been clearly demonstrated in British Columbia, and the potential for its adoption as a framework for biodiversity management in Britain deserves detailed investigation.

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Environmental planning in forest management

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Summary

The need for long-term planning in multi-purpose forest management is explained and a three level approach described. At a regional level, forests are set in context so that objectives can be determined for individual forests in a coherent way. The principles of forest design planning are discussed, with particular reference to the scale and degree of diversity, and the concept of sustained yield. The forest design planning process is outlined, and finally the third level of planning, the site management plan, is described.

Introduction

Environmental planning in forest management is fundamentally inseparable from forest planning. The forest itself is the environment. The forest is not only the trees, but also the understorey, the ground vegetation, the open spaces, soils, rocks and water, all of which are integral parts. Clearly, any action taken concerning the arboreal component of the forest, the trees, is not just of environmental consequence: it is in itself an environmental act. Silvicultural planning, production planning, road planning and recreation planning are all, therefore, aspects of planning the forest environment. This paper is in fact about forest planning, with a concentration on its environmental, rather than economic, social or cultural, consequences.

The need for planning is quite simply the requirement to understand the consequences of an action before embarking on that action. It has been described as both the launching point for the future and the process of learning from the past. Embarking on any action, whether it be felling a tree or creating a wildlife pond, without having fully thought through the consequences, runs a grave risk of failing to achieve one's objectives. This is true where there is a single management objective. It is even more true where there are multiple objectives. An action aimed at furthering

one of these will almost certainly affect the ability to achieve the others. With multiple objectives, it is all too easy to sub-optimize if decisions are taken without proper information and planning. Multi-objective forestry is now a firm policy in Britain. The aim is attractive and productive forests, which blend with the landscape, are rich in wildlife and are efficient to manage. If this is to be achieved, proper long-term planning, taking into account all the relevant issues and producing an integrated solution, is essential. Compiling a plan in itself requires clear thinking and logical action.

The need for planning in business-like forest management is all the greater because the consequences of actions may be a long way in the future. It is important to remember that every action that we take today will affect the options open to future generations. It is equally important to remember that to take no action, whether a conscious decision or through neglect or procrastination, will also affect the options open to future generations. The planning process must not, in itself, become a barrier to action. There are numerous examples of plans where the desire to collect ever more information and to predict every consequence to the last detail has led to stagnation and a failure to make any progress towards meeting objectives. Similarly, we must guard against the arrogance of the current generation: our expectation is that future generations should accept and implement our plans, in spite of our equally strong belief that the plans we prepare are better than any of the plans which have been prepared in the past. We have the right to plan only our own actions, and the responsibility to illustrate the consequent options which we believe will be open to future generations. We have no right to plan the actions of future generations, and we will waste a great deal of effort if we attempt to do so.

There is no single correct approach to forest planning. Any successful approach will almost certainly embody the principles described here.

This paper describes a three level approach: regional level, forest level and site level. Most attention is devoted to forest level planning, not because the other levels are less important, but because the concept of forest level planning requires greater explanation.

Regional level

The primary purpose of regional level planning is to determine the objectives that are appropriate to individual forests or parts of forest within the region. Current forestry policy is concerned with multi-purpose objectives. This does not mean that every objective should be given equal weight in every forest. The precise balance between different objectives, for example timber production and recreation provision, will vary from place to place, as well as from time to time. At the outset of the process, it is important to determine what the objectives are at that time. Foresters are often told 'First, decide your objectives', as if the forester has a free hand in making the choice. This view is perhaps encouraged by the 'shopping list' of objectives presented in the Woodland Grant Scheme application form. However, the forest manager usually has little choice in the selection of objectives: they are implicitly determined by the location, context, character and ownership of the forest. The advice should be 'Make the management objectives explicit'.

The precise objectives for any particular forest emerge from four sources.

Firstly, if public funding is involved, the national forestry policy objectives and guideline standards must be met. These include stated requirements such as watercourse or archaeological site protection and percentages of broadleaves, as well as objectives of wood production, improvement of landscape and enhancement of nature conservation. Even where public funding is not involved, legislative and regulatory requirements constrain the choice of objectives.

Secondly, there are the owner's particular objectives for managing the forest. For Forest Enterprise, these are the same as national policy objectives, whereas in the case of a private owner they might, for example, include sporting use, revenue requirements or capital appreciation.

The third source concerns other demands made on the forest apart from those expressed by the client. This might include the need to protect a

certain bird species, the demand for a particular type of timber or a particular type of recreation, or the desire to maintain a valued landscape. These demands may be expressed or emerge from local circumstances peculiar to the forest in question, although they may overlap with general policies.

The fourth factor concerns the ability of the forest to supply benefits to meet demands. It may be that there are physical limitations on its capacity, for example to supply certain kinds of timber, to carry some forms of recreation, or to support the habitat of a particular animal or bird, due to soil, location and climate.

Thus, it is a matter of determining, rather than selecting, the particular balance of management objectives appropriate to each forest in the region.

Experience indicates that it is unwise to attempt to produce an integrated plan for an entire forest district. The need for regular revision and amendment makes it an impossible administrative task to maintain. What is required at regional level is a series of subject-specific strategies or plans which can be individually revised, albeit that there will be connections between subject areas which require consequent amendment. The place for full integration is at forest level.

Three types of document which have proved valuable in Forest Enterprise forest districts are a conservation plan, a recreation strategy and a landscape assessment.

Conservation plan – identifies the general conservation value of the individual forests in the region, and the sites of importance to nature and archaeological conservation, both designated and undesignated.

Recreation strategy – identifies forests and parts of forests of current and potential recreation importance in relation to the geography of demand. A recreation strategy will also describe the location, intensity and scope of intended recreation provision and use.

Landscape assessment – identifies the relative visibility, prominence in the landscape and landscape character of the forests in the district.

This type of regional information provides the context in which to classify individual forests and parts of forests according to environmental sensitivity, and hence indicates the weight to be given

to environmental as against wood production objectives.

Forests can usefully be classified into four broad categories of sensitivity. The criteria by which these are judged are fairly easily stated and can be independently verified.

Very high	Includes special sites, e.g. sites of special scientific interest (SSSI), woodland parks, woodlands associated with historically important landscapes, highly visible areas in National Parks, National Scenic Areas, Areas of Outstanding Natural Beauty and Forest Parks.
High	Forests prominent in the views from major settlements, major highways, walking routes or important established viewpoints; those used intensively for recreation; forests of regional ecological importance.
Medium	Forests and woods prominent in the views from minor settlements, minor roads and moderately used walking routes; those partly used for recreation or light use throughout their area; woodlands of local ecological significance.
Low	Forests barely visible from any but the most minor roads or settlements, used little if at all for recreation, with little potential for development, and low ecological interest.

Having identified environmental sensitivity, the weighting required to be given to environmental objectives becomes apparent.

Forest level

Forest design plans apply to a whole forest, or to a distinct part, a landscape unit, of a larger forest. The forest design plan concerns the larger scale spatial layout of the forest, while smaller scale details are built in at the site management level. It is primarily concerned with the location and timing of tree felling, and the distribution of tree species and open space at regeneration, these being the principal tools available to the forest manager in controlling forest structure, and hence the current and future flow of forest products, both tangible and intangible.

The aim of forest design is to move the forest towards a condition in which it can continue indefinitely to meet its management, at the same time retaining sufficient versatility to meet unknown future demands.

The first step is to establish a vision of the desired future condition of the forest. The second is to work out how to get there. The two main factors required to describe desired future condition are structure and composition.

Forest structure is the spatial arrangement of different age classes (or more precisely, size classes) of trees and open spaces that make up the forest. In the case of patch clear felling, the predominant silvicultural system employed in Britain, forest structure is controlled by the size, shape and timing of felling coupes.

To achieve biological robustness, felling coupe size should relate to the scale of variation in the physical site characteristics: topography, soils and climate. Complex and intricate patterns of soil type indicate smaller patch sizes, compared to extensive areas of homogeneous soil type. Similarly, broken topography, which will also give rise to soil type variation, requires smaller patch sizes than rolling hills or uniform slopes. It is no coincidence that the desired pattern in ecological terms is also appropriate to harmonious landscape design: harmony requires that the pattern reflects the same scale of variation in physical site characteristics.

A further influence on patch size is the requirement for sustained yield. Foresters are well used to the concept of sustained yield for wood production. In this case, the unit of normality, the area from which a sustained yield is required, is the catchment of the processing mill. With modern transport systems, mill catchment areas are huge, and there is no necessary requirement that each individual forest has a normal structure. Regular annual timber yields may, however, be desirable for continuity of employment or to provide steady income to the single forest owner.

The concept of sustained yield applies equally to forest products other than timber. Hence the aim should be a forest structure which supplies a continuous yield of water quantity and quality, of wildlife value, of scenic quality and of recreational value. The unit of normality, the area from which a sustained yield is required, is different for each of these products. The smaller the unit area of normality, the smaller the patch size required for it to contain patches of the full range of age

classes. Thus for water quality, the unit of normality is the catchment. Felling and restocking of a large proportion of the catchment will cause sudden changes in water quality. The aim should be to have a normal forest structure within the catchment so that an equal area is felled each year. Net change within the catchment from year to year is then zero. With landscape, the unit of normality is the view: landscape quality will only be sustained if the forest structure is such that the whole area of forest within the view is not felled at one time. In the case of wildlife, the unit of normality is certainly no larger than the whole wood or forest: every wood or forest needs the complete range of young, middle aged, mature and very old trees, not necessarily in equal proportions, if it is to maintain its wildlife value continuously. The actual unit of normality for wildlife depends on its mobility – its ability to move from one patch to another – which varies with species. For recreation, it is the visited area. Recreation value cannot be maintained unless this area contains a range of young to very old trees, again not necessarily in equal proportions.

In many cases, the patch size indicated by landscape-ecological considerations will permit a normal structure to be developed within the required unit of normality. One exception will often be recreational value, which may require a more intimate scale of structural diversity than the physical characteristics of the site should indicate. Another may be ancient woodland, particularly where it contains plants that are slow colonisers. Resolution of the conflict in scale in these cases is best achieved, we suggest, by considering a silvicultural system other than clear-felling.

Choosing too small a scale of structural diversity is just as damaging as too large a scale; too small, and habitats will be fragmented, with the visual appearance moth-eaten; too large, and it will be biologically impoverished, and visually monotonous.

Next we come to the degree of diversity. How much diversity is appropriate in terms of numbers of species and habitats? The objective might be to maximise biodiversity within the physical and chemical limitations of the site. But is this a sufficient answer? Just because a hectare of brown earth could support 200 different species of tree, it does not follow that it should. Throughout the history of man's evolution, we have concentrated the productive process on selected elements of the biosphere, reducing biodiversity to those elements which we find useful. There is therefore a balance to be struck between maximising production and

maintaining biodiversity. There is, however, an almost universal consensus that the level of biodiversity in our upland forests needs to be higher. We need to let at least some light through the cellulose factory of the Sitka spruce canopy to make room for other life forms. We know the direction in which we have to move, even if we do not know how far. A proportion of other conifer species, a broad-leaved element, a network of open spaces and glades, and a range of age classes including old trees, dying trees and dead wood are necessary components of biodiversity.

There is a logical, precautionary argument for aiming at a level of biodiversity which mimics natural ecosystems. These have developed over a long period and have a proven record of their degree of ecological stability. It may be that some other level of biodiversity can be equally, or even more, stable, but it would take a long time to prove the case. It is also likely that attempting to maintain an unnatural level of biodiversity, whether unnaturally high or low, will require significant and continued intervention to counter the natural direction of development, resulting in a high cost of management. It makes sense therefore to mimic the levels of diversity found in natural systems. All natural forests contain varying degrees of diversity appropriate to their location. Those in more extreme situations, such as at high elevation or latitude or on poor, infertile soils, tend to be less diverse in structure and number of species present than those at lower elevations or on more fertile soils. We should aim for a similar trend in managed forests.

Too high a level of diversity for a given site is unsustainable, is inefficient, lacks a sense of unity and is visually chaotic. Too low a level of diversity and the forest is ecologically unstable, biologically impoverished and visually monotonous.

Forest design planning begins with the assembly and analysis of site information. The information required and its relevance will vary from place to place, but in general the following checklist will be useful.

Physical

- surface geology
- topography (contours)
- drainage and water courses
- soils
- windthrow hazard classification

Production	- growing stock inventory - economic felling ages - access roads and harvesting systems
Conservation	- sites of geological/geomorphological interest - sites of archaeological/historic interest - important extensive habitats - sites of specific wildlife conservation value
Recreation	- actual and potential recreational use - footpath routes – actual and potential
Visual	- viewpoints inside and outside the forest - elements of visual diversity - visual problems with the existing forest layout
Other factors, e.g.	- powerlines - water supply catchments

This information is used to conduct a ‘constraints and opportunities’ analysis, picking out those factors which reduce freedom of action, such as poor soils, high windthrow hazard class, physical obstructions, protected sites. The opportunities are the converse of constraints – good soils, low windthrow hazard class etc. It is useful to show these, sieved out on to a map and also illustrated on perspective sketches.

The other analysis required is a landscape pattern analysis. This comprises the visual forces in landform, ecological patterns determined from existing vegetation or from previous landuse, archaeological sites, soil, drainage and other elements of diversity such as rock outcrops. From this, appropriate patch sizes and shapes will begin to emerge.

An initial sketch design can then be prepared, comprising a coupe design and a restocking design. The coupe design will show a pattern of interlocking patches or areas, related in shape and scale to the landscape pattern. These coupes need not necessarily be clearfelling coupes, but could be the outline of an area to be treated by some other silvicultural system appropriate to the site and

the specific objectives. Some will be earmarked as long-term retentions, to be retained until biological maturity.

The restocking design uses the coupe layout to determine how each should be treated after felling, including what open ground is to be left unplanted, and the species to be replanted.

The initial sketch proposals are then tested against the constraints and opportunities analysis. Adjustments to the initial ideas can then be made, to ensure that the eventual plan will be workable and capable of being implemented. At this stage also, the economic felling ages of the original forest can be matched with the coupe pattern, to determine the optimum felling sequence, while ensuring that at least one age class difference is maintained between adjacent coupes. Some adjustment to coupe boundaries will usually be necessary – and achievable without unduly compromising the desired landscape pattern – to avoid unacceptably high revenue losses, for example from excessive premature felling.

In highly visible, landscape-sensitive areas, whether prominent hillsides or woodlands on the skyline, the coupe design is best prepared in perspective and translated to plan. In rolling or flat terrain, the coupe design should be prepared in plan. It should still be tested in perspective from key viewpoints.

In lowland woods, coupe shape and scale is likely to be determined as much by consideration of recreational use and conservation requirements as by landform and external landscape scale. In long established woodlands, there is likely to be an existing pattern of structural diversity, and consideration needs to be given to its appropriateness and whether there is a need for change, to break up some areas or to reduce fragmentation.

One of the largest categories of forest in overall area terms is the even-aged spruce forest in the uplands, on rolling terrain with infertile, poorly drained, windthrow prone sites. Freedom of action is severely constrained by wind, and, with few opportunities for lengthening rotations, structural diversity can be achieved only by premature felling. The existence of wind-firm edges is essential to successful coupe design in moderate to high hazard classes. Edges formed by roads, rides, watercourses and major age class or yield class changes should be surveyed and mapped prior to the design stage. In Glentworth (Dumfries and Galloway) this has been developed into a survey of ‘harvestable units’: areas bounded by windfirm

edges and capable of being harvested and extracted as independent units. A variety of these units are then combined by the designer into felling coupes.

Windfirm edges can be created by making severance cuts, felling a 20 metre swathe of trees along desired coupe boundaries. Severance cuts must be made well before critical height, the height at which trees become liable to windthrow, during the period of negligible or low risk. The 'window of opportunity' for these operations is extremely short in the high hazard classes: for example, a yield class 12 Sitka spruce stand reaches moderate risk after 17 years in hazard class 6; 19 years for yield class 16 in hazard class 5 (Quine and Gardiner, 1992). Forest design plans therefore must be prepared at an early stage if stand structure is to be manipulated. In these forests, it is unlikely that the desired future condition can be achieved in a single rotation. Where windthrow hazard is high, it is usually over-ambitious to try to introduce more than two age classes at the end of the first rotation. Effective structural diversity requires a minimum of 10 years between adjacent coupes, and attempts to introduce a third age class will almost certainly be at the expense of this age separation. Any opportunities for longer rotations offered by locally more windfirm sites are immensely valuable for increased structural diversity. Heavy thinning while still in the negligible to low windthrow risk period (for example, before 28 years old for yield class 12 Sitka spruce on hazard class 4 sites) will extend the length of time these stands can be retained.

With any of these approaches to forest design planning, the proposed design plan must then be tested against the management objectives for the particular forest, and against national policy guideline requirements. Does the proposed structure achieve or move sufficiently towards identified requirements for sustained yields of wood products, water quality, wildlife value, scenic quality, recreation value and so on? There may be several options that need to be tried to find out which best achieves the necessary balance.

The final forest design plan will contain:

- a written statement explaining the background;
- the analyses of the survey documents, so that consultees and managers can understand the reasons why the design plan has turned out the way it has;
- photomontages illustrating coupe design and restocking design from main viewpoints;
- a coupe plan, typically at 1:10 000 scale (or exceptionally at 1:25 000 or 1:5000), showing felling periods for clear felling systems, or management units for other silvicultural systems;
- a restocking plan at the same scale, showing what species are to be planted where, together with the open spaces and retained stands;
- an appendix, containing the basic survey information in case reference needs to be made to it at some time in the future.

Site level

The third level of planning is the site management plan. This is a crucial level because it controls the day-to-day activity of the harvesting and restocking operations. A site management plan is prepared for each felling coupe, before a chainsaw is brought near the first tree. In preparing the site management plan, the forester conducts a detailed survey of the site, identifying the small scale, site-related features which will not necessarily have been picked up in the forest design plan. Information collected – wet areas, rock outcrops, old trees, nest sites – should be recorded on a large scale plan, say 1:5000 to allow sufficient definition. A harvesting plan and a restocking plan are then prepared. The harvesting plan will show the detailed shaping of the edges of the felling coupe, the sequence and direction of felling, extraction routes, method of extraction and planned stacking areas of sufficient capacity, well away from watercourses. It will also identify which living and dead trees are to be retained. It will identify sensitive sites and state the precautions required to ensure their protection. In this way, potential problems should be identified in advance and solutions provided.

The restocking plan will show ground preparation, drainage layout and planting design. It will show the detailed design of edges, the shaping of open spaces, and areas reserved for the natural regeneration of broadleaves. It will also identify sensitive sites and the measures required for their protection or enhancement.

The site management plan ensures that everyone involved in the harvesting and restocking opera-

tion knows what is required for that particular site. It can also be used as a basis for contract specification, to ensure that the contractors meet all their obligations regarding site protection.

Conclusion

British forestry is facing a marvellous opportunity at the present time. An increasing area of post-war plantations are reaching the end of the first rotation. At the same time, there is a firm political commitment to multi-purpose

forestry. With the right planning, these plantations can be turned into true forests: forests which will meet the need of the present day as well as having a legacy of value to future generations.

Reference

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Valuation of forest resources and evaluation for conservation

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Summary

In the last decade there has been a move away from intensive forestry (where the main objective was timber production) and towards the concept and establishment of multi-use forests (catering for recreation and wildlife conservation as well as timber production). Such a move has demanded changes in forest management practices and at the same time there has been considerable discussion and research into methods for valuing non-timber products and evaluating biodiversity. That is, there are two main aspects of relevance which should be distinguished from each other: one aspect is concerned with methods of valuing non-timber products and the other is concerned with identifying which components of the forest biodiversity have the greatest conservation importance. This paper briefly describes some of the methods which have been used to value non-timber products such as water catchments and the forest as a recreational resource. The paper then goes on to consider evaluation of biodiversity for conservation. Management for conservation of wildlife should be undertaken in as effective a manner as possible and therefore an important stage in designing management for conservation would be to identify species and areas of greatest conservation interest. Evaluation of wildlife and natural communities is now widely used to help identify areas of conservation interest and to help establish priorities. A conceptual method of evaluation for use in plantation forests is proposed.

Introduction

Integrated management of forest resources for timber, recreation and conservation is likely to be concerned with the following: valuation of timber

products; valuation of non-timber resources; and evaluation of the biodiversity in order to identify conservation priorities. Here we are concerned with valuation of non-timber products and evaluation of biodiversity for conservation. These two topics are very much related in the context of managing forests for biodiversity but they should be distinguished from each other.

When assessing the total economic value of forest resources it is logical to think in terms of *use values* and *non-use values*. Summarised in Figure 14 are further subdivisions which may be usefully examined when determining this total economic value and which will be discussed in the sections that follow. The total economic value includes several types of social benefits (Turner, 1990) yet the process of valuing all the social utilities of forests (such as watershed protection and recreation) is not easy. The Forestry Commission has sponsored considerable research into valuing forest recreation (Benson and Willis, 1992) and it seems that not all of the wide range of resources which forest ecosystems provide have previously been valued adequately. One may assume, therefore, that the actual value of forest resources is far higher than the value which direct 'use valuation' procedures have so far determined. In addition, these natural resources are unlikely to have been managed in an economically efficient manner since economic indicators fail to reflect all the costs and benefits of their use (Turner, 1990).

There is considerable interest in conservation of biodiversity (that is the variety of nature at all levels from genes through species to ecosystems). Despite the growing concern at the loss and degradation of biodiversity, we have to face the fact that it is impractical to conserve all aspects of

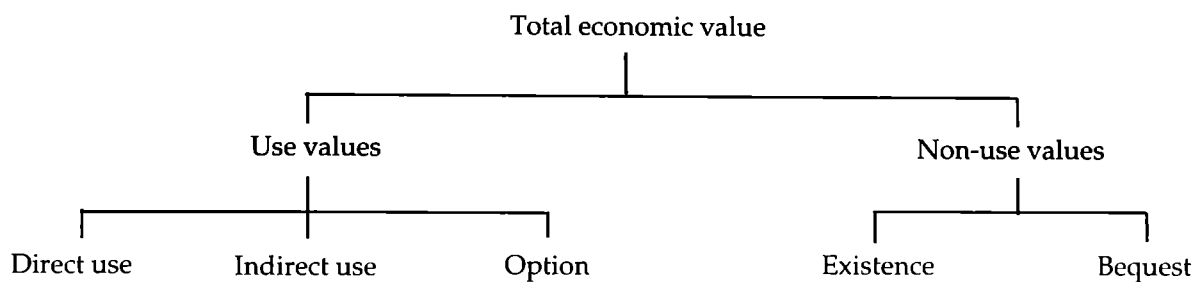


Figure 14 The total economic value of forest resources.

biodiversity to the same extent. That is true on a global scale and is also probably true at the scale of a single forest. We have to make choices and we have to decide priorities: for example, which species and which areas of the forest should be the objective of conservation programmes? How do we identify the areas of greatest conservation interest? These are questions which are the basis of evaluation and especially ecological evaluation.

Many methods have been developed for evaluation of species and habitats. Some of these are based largely or entirely on ecological criteria (e.g. species composition, species rarity, species richness, species diversity, number of endemics), hence the expression ecological evaluation. Natural and semi-natural areas have been the main concern of evaluation and ecological evaluation and therefore criteria such as naturalness and representativeness have been applied and quantified. There is no reason why plantation forests should not be the concern of evaluation; indeed, this is very timely when there is growing interest in the development of multi-use forests and management of those forests for biodiversity, recreation, environmental benefits and timber.

There is, however, a need for the development of an evaluation method for plantation forests, a method which can provide a basis for identifying the areas of greatest conservation interest. Such an evaluation could also form the basis of a management programme. In this short paper, the concept of valuation of non-timber products is outlined. Methods for evaluation of biodiversity which have been developed for use on woodlands and forests are then described and this is followed by a suggested conceptual plan for evaluation of forest biodiversity for conservation.

Values and benefits of nature

Concern about the future of biodiversity is increasing and there are more and more initiatives directed at its conservation. There have been international initiatives (e.g. the Biodiversity Convention), national legislation for protection of natural areas and species, and many local initiatives such as the creation of community woodlands. With so much attention being directed at conservation, the question might be asked, why the interest in conservation? Or, what is the importance or value of biodiversity? There are ethical reasons for conserving biodiversity, but there are also many material advantages in doing so (Table 2).

Conservation of biodiversity does not just mean protection or preservation. True, a major and popular objective of conservation has been to protect biodiversity from disturbance and any form of exploitation. However, there is an alternative view that by carefully managed exploitation conservation of biodiversity is more likely to succeed. In other words, if there is a commercial interest in a species (that species being seen as a resource), then it is more likely that the species and its habitat will be managed and protected than if the objective was simply to conserve the species for no other reason than its intrinsic value.

Natural and living resources such as mineral deposits and forests have a market value. But what of the indirect values of biological resources: is it possible to value a mangrove swamp, a chalk grassland or even a population of butterflies? In recent years there has been a rapidly growing interest in how to value the natural environment and biodiversity. One approach to types of biodiversity values and the application to wetlands is shown in Table 3.

Table 2 Reasons for conserving nature (after Spellerberg and Hards, 1992)

Ethical reasons such as:

- 1. heritage value
- 2. cultural value

Provides enjoyment, for example:

- 1. leisure activities ranging from bird watching to walking
- 2. sporting activities such as orienteering and diving
- 3. aesthetic value by way of seeing, hearing or touching wildlife
- 4. enjoyment of nature depicted in art

Resource for food, materials, research inspiration and education (utilitarian), for example:

- 1. genetic resource
- 2. source of food
- 3. source of organisms for biological control
- 4. source of pharmaceutical products
- 5. source of materials for buildings
- 6. source of materials for making goods
- 7. source of fuel for energy
- 8. source of working animals
- 9. for scientific research
- 10. educational value
- 11. inspiration for technological development

Contribution to maintenance of the environment (ecosystems and climates), for example:

- 1. role in maintaining CO₂–O₂ balance
- 2. role in maintaining water cycles and water catchments
- 3. role in absorbing waste materials
- 4. role in determining the nature of world climates, regional climates and microclimates
- 5. indicators of environmental change
- 6. protection from harmful weather conditions by providing wind breaks or flood barriers

Valuation of forest resources

Use and non-use values

Within forest ecosystems there is a great diversity of tangible commodities which may be of use to humans. For example, the direct exploitation of forest resources for human use and human survival generates a direct use value for forest resources. These may include recreation and related experiences in addition to the commercial timber production. In a study by Hanley and Ruffell (1992) the following 17 reasons were given for visiting a forest: walking, picnic or barbecue, dog walking, special feature such as reptillary, visit the forest, views and scenery, break in journey, visit area in general, entertain children, cycling, see water feature, peace and quiet and

fresh air, boating and fishing, visit forest centre, watch wildlife, photography, other explanations.

Indirect use values are related to the functional services provided by forested ecosystems which indirectly support human populations or property, or permit economic activity to continue. Watershed protection is one example of such indirect use associated with forested catchments (as are ground water recharge and flood control). Many of these indirect use values may benefit populations (human and non-human) far outside the boundaries of the forested site.

Option values relate to an individual's desire to be able to use the forest in the future, as well as to be able to study further to gain more information about the ecological system. This may, in turn,

Table 3 An economic approach to types of biodiversity values and an application to wetlands (from Spellerberg, 1992)

Value type	Sub-type	Example
<i>Use values</i>		
(A) Direct	Consumptive Productive Non-consumptive	Home consumed forest fruits Plant breeding Tourism
(B) Indirect		Ecological process
(C) Option values		Potential value of medicinal drugs
<i>Non-use values</i>		Existence value of certain species
Application for wetlands^a		
<i>Use values</i>		
(A) Direct		Fuelwood, fish, wheat, rice, soya-beans, cowpeas, shrubs, grasses, bird viewing, water transport, etc.
(B) Indirect		Groundwater recharge/discharge, flood and flow control, shore-line/bank stabilisation, sediment retention, nutrient retention, etc.
(C) Option values		Environmental functions threatened with conversion, etc.
<i>Non-use values</i>		Birds, wetland ecosystems, etc.

^a Modified from Aylward, 1991.

allow further uses to be developed. It may also be the option simply to visit the site and thereby use the forest. Option values are, therefore, good indicators of the conservation value attached to forest resources.

As would be expected, it is possible to identify non-use values of forests and there are two aspects: existence values and bequest values. Existence values might be where an individual wishes to assure the availability of a good or service for others or for future generations. As such, forested ecosystems may be valued by humans living far away from the forest itself. The value is based purely on the fact that such places exist, whether people actually choose to use or visit the forest or not. Bequest value is the value of

leaving a legacy to future generations of the existing diversity of forest resources.

Amenity valuation

As long ago as 1967, Helliwell devised a method to determine the amenity valuation of woodlands; a version of the method was published by the Arboricultural Association in 1990. The method has two objectives: firstly, to provide a format for the assessment of the amenity value of a woodland and, secondly, to give planners and managers a basis for including precise and consistent amenity values in the calculations. Briefly, six standard factors are identified for each woodland and for each of these there is a score of up

to four points. The scores for all the factors are then multiplied together to give an assessment of the amenity value of the woodland. The six standard factors are: size of the woodland, position in the landscape, viewing population, presence of other trees, composition and structure of the woodland, and compatibility in the landscape. This method, which has been endorsed by the Tree Council, provides a clear and straightforward method for amenity evaluation of woodlands.

Notional monetary values

In 1973, Helliwell suggested a basic method for valuing wild species based on four factors: the number of associated species, abundance in Britain, conspicuousness and material value. Scores are awarded for each factor (1, 2, 4, 8, 16 or 32). For example the value of Oak (*Quercus robur*) as a species in Britain would be calculated as follows:

32 (many associated species) x 16
(very common) x 4 (conspicuous)
x 4 (some material value).

Helliwell then went on to suggest that the score be multiplied by the arbitrary figure of £10 000 to give a shadow price. This approach, however, ignores aesthetic values and the values of the functions of many species of wildlife in open ecological systems. Furthermore, subjective judgements are made in the process of assigning the scores. Why use these notional monetary values? After all, do they help to place a meaningful value on wildlife?

Preventative expenditure and replacement costs

This approach places a value on a change in environmental quality or the loss of a function or service provided by the ecological system. The approach is easily employed where the environmental changes affecting the forest ecosystem involve physical effects which are easily observed. However, the approach assumes that it is possible to replace a certain ecosystem or prevent certain actions occurring. The approach also assumes that the existing system is in an optimal condition and that the recreation of a particular forested system is possible. Would the Department of Transport have been able to recreate Oxleas Wood several miles from its original site?

Damage costs avoided

This approach is based on the concept of the value of an environmental good or service being equal to the costs of property or other damage which would

occur if the service did not exist. This approach is relatively easily applied. However, where the damage cannot be valued in terms of market prices (i.e. non-use related benefits) the value will be underestimated.

Hedonic travel costs

This approach uses travel cost (the amount of money spent travelling to that resource) as a proxy for the value of that resource. This approach has been applied most frequently to evaluation of the recreational value of the forest resource (see for example, Benson and Willis, 1992).

The approach is useful because it relies on observed behaviour, providing a degree of objectivity which other methods do not have. However, the data requirements are considerable. Comparability of values are also questionable, although Sorg and Loomis (1985) provide an approach to comparisons. The approach takes no account of the experience for which the travel was undertaken and also it is assumed that the people know how much they will enjoy the trip when they initially decide to take the trip. In addition some trips may be multi-purpose and so the valuation would be an overestimate of the true value.

Contingent valuation

This method, which is widely accepted in the USA, was first used on Forestry Commission sites in 1987 (Hanley, 1989). Social survey techniques are used which are in the form of bidding games. Individuals are asked how their behaviour would be altered contingent on a new hypothetical situation. After an introduction and a full explanation of the purpose of the questionnaire participants are asked:

1. How much they would be willing to pay to achieve an improved situation.
2. How much compensation they would require to accept a reduction from their current situation.
3. How much they would be willing to pay for current circumstances.

It has been suggested that such an approach suffers from strategic behaviour bias, information bias and instrument bias (Sorg and Loomis, 1985) although if the questionnaire is constructed properly these biases may be minimal.

The greatest advantage of this method is that it results in a maximum willingness to pay value. It

is also useful in that it allows option, bequest and existence values to be derived. However, it is a hypothetical approach and subject to the above biases. In addition a considerable amount of work would be required to complete these surveys.

Energy equivalent method

This approach assumes that there is a fixed relationship between the energy embodied in a product and its market price. The total amount of energy captured by a system is used as the indicator of the potential for that system to do useful work for the economy.

Whether energy prices (fossil fuels) may be used as a measure of economic value is questionable. The approach also requires an estimate for the primary productivity for the forest system. The approach is regarded as a comprehensive valuation technique. However, an overestimation of the values may result if some of the forest systems and services are not of value to society.

The gross expenditures method (GEM)

This approach works on the principle that the value of hunting (fishing, etc.) is at least equal to the total expenditure by hunters on travel, food, lodgings and equipment. However, according to Serg and Looms (1985), the actual incurred costs are not the appropriate measure of value because if hunting was not available in the forest the money could be saved or spent elsewhere. GEM, therefore, does not recognise the net value of the natural resource (the hunted animals) over and above the expenditures. In fact a declining population of a certain hunted species (salmon for example) may increase the gross expenditure as hunters spend more during the hunt.

Evaluation for conservation

Evaluation for conservation has been the subject of much research (Spellerberg, 1992) designed for particular taxonomic groups and some designed for particular habitats. A common element of many aspects of conservation is the need to identify areas of conservation interest and perhaps also the need to prioritise those needs. Evaluation for conservation is concerned with identifying conservation needs and priorities. The following are the kinds of questions which may be addressed (from Spellerberg, 1992):

- If single species are the target of conservation then what are the best criteria for selection of those species?
- Saving the last remaining fragments necessitates a choice. That being so, should the choice be based on taxonomic and ecological criteria as well as utilitarian and intrinsic values?
- Should the overall aim be to maximise biodiversity wherever possible or is it better to combine utilitarian and conservation interests?
- Should there be established criteria for selection of protected areas?
- Would it be sensible to direct immediate efforts towards centres of endemism, i.e. those regions with large numbers of endemic species?
- It is impractical to protect all biotic communities from pollution, disturbance and damage. Therefore what are the most effective methods for identifying the most sensitive biotic communities?

Evaluation for conservation can be undertaken at different levels. For example there are methods which have been developed to evaluate the conservation needs of a particular taxonomic group such as flowering plants. One particularly good example is the method used as a basis for the *British red data book: vascular plants* by Perring and Farrell (1983) in which the authors have selected a few criteria which are then used to calculate a threat number (0–15). The higher the number the greater the conservation needs.

Here we are more concerned with evaluation methods at the habitat or community level, especially those of relevance to woodlands and forests.

Ecological evaluation of woodlands and forests

Ecological evaluation of woodlands and forests has mostly been designed with natural and semi-natural woodlands, rather than plantation forest, in mind. For example Goodfellow and Peterken (1981) have devised a method for survey and evaluation of woodlands based on the woodlands in Norfolk. The method has four stages: preliminary site selection, ground survey of selected woods, ranking, and assessment and selection. The preliminary site selection provisionally assigned the woodlands to one of four categories (ancient woods supporting some semi-natural stands, ancient woods entirely stocked with plantations, secondary woods originating before 1830,

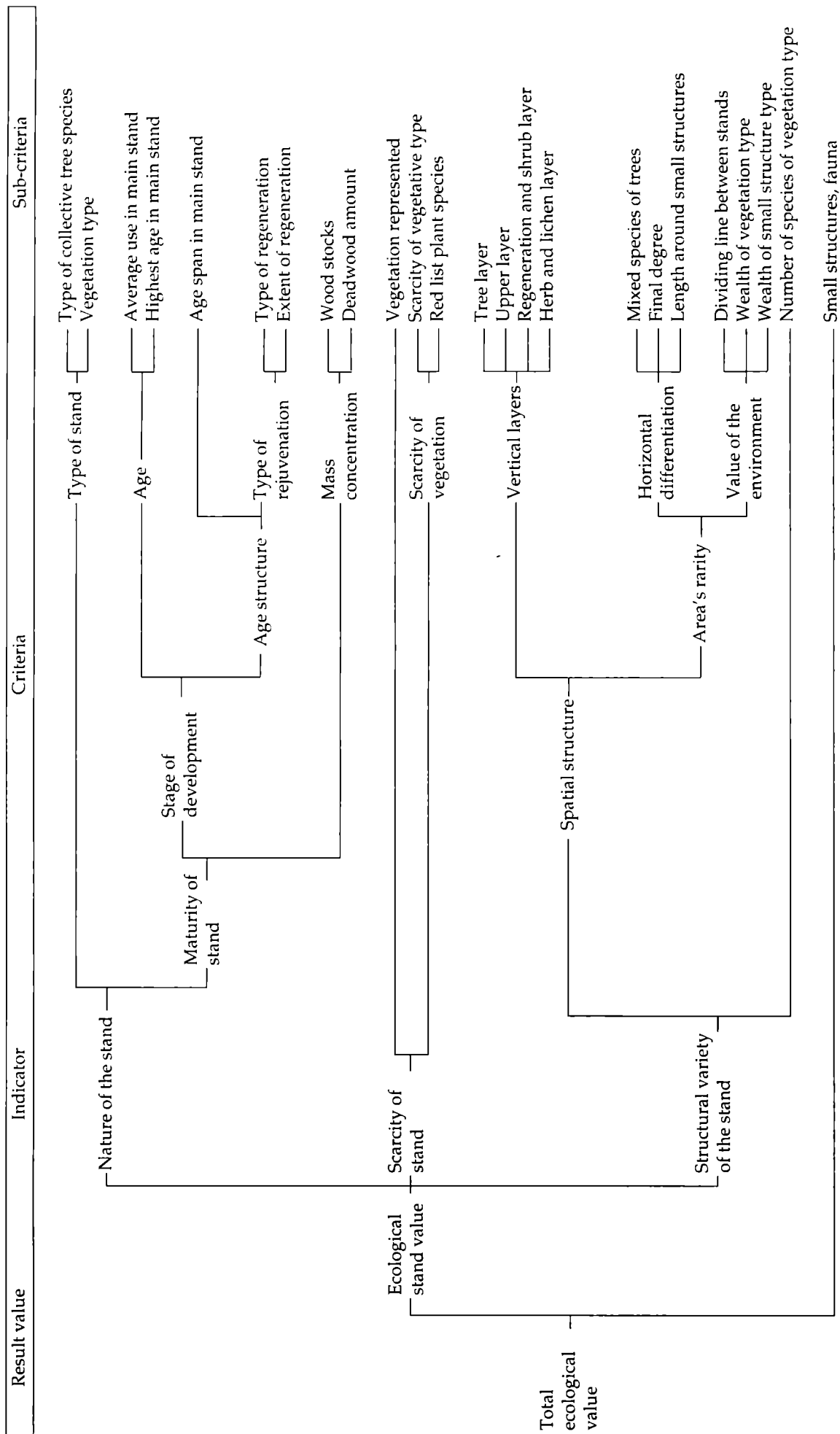


Figure 15 Ecological value analysis: value variables and their connections (from Ammer and Utschick, 1988).

secondary woods originating after 1830). For the survey, features such as composition of each stand, woodland vascular plants, rides, banks and ditches, etc. were recorded. The evaluation was undertaken initially by way of a league table according to the number of a selected list of woodland vascular plants, divided into two groups (the first with introduced species and planted species, the remainder all native species). Ranking also took into consideration various site features, bonus points being added for glades, rides, open water, continuity of coppice management, records of rare invertebrates and adjacent semi-natural habitats. Points could also be subtracted for rampant introduced species such as rhododendron and concrete rides. This method of evaluation could usefully be adapted for plantation forests, as a basis either for comparing plantations or for identifying areas of conservation interest within a single plantation forest.

More recently, Ammer and Utschick (1988) have undertaken an extensive programme of research on the survey and evaluation of woodlands in Germany. Site descriptions are based on age structure, soil types, topography and other environmental variables (see Pyatt, pages 28–31). The evaluation method is aimed at forest stands and takes particular account of the following:

- structures such as rocky outcrops, ponds, and faunal surveys;
- rarity of the habitats;
- proximity to natural communities and structures;
- structural variety within the stands.

Proximity to natural communities, rarity and variety are the three basic elements but each of these is further divided into sub-criteria (Figure 15). The results are expressed as scores and presented in the form of maps.

Elements of these two methods seem to provide an excellent basis for a conservation evaluation method for forests in Britain. The emphasis on the importance of initial surveys and inventories is understandable but of course such inventories need to be updated. In the apparent absence of any ecological evaluation method for forests and in the light of recent forest biodiversity initiatives it seems timely that such a method is devised.

Proposed ecological evaluation for forest biodiversity

Whereas some ecological evaluation methods could be designed to identify the relative conservation of whole forests, this method (Figure 16) is designed to identify the priorities, species and areas of conservation importance within a managed forest. The surrounds of a forest and the extent of isolation (or degree of connectivity with other forests and habitats) is however very important. Because some features will have a relatively higher importance in some regions, this method needs to be simple and versatile so that it can be used for a wide range of forests in different geographic regions.

It is suggested that the evaluation be based almost entirely on surveys and inventories and subsequent subjective judgements of the relative conservation value of any particular feature. That is, no scoring method is proposed. The survey and inventory need to be undertaken at different scales (see Gill and Bell, pages 35–41) and should take particular note of those features which could be managed in support of biodiversity, e.g. area of forest, number and extent of forest compartments, age and species of trees in stands, the nature of the forest surrounds, and the number and characteristics of rides, glades and other unplanted areas; the presence of protected species or species identified as being at risk by means of red data books. Features such as habitat structure, age class structure, open spaces, dead and dying wood, tree species richness, understorey vegetation, riparian and wetland areas are important for biodiversity and many can be quantified (Spellerberg and Sawyer, 1993).

Discussion

Forests provide a diversity of resources and services, many of which are difficult to express in monetary terms. An economically efficient utilization of the full suite of forest resources is therefore unlikely to occur but that is no reason why timber and timber products should be the only resources used. Indeed there are those who would suggest that using forests for timber only is an under-utilization of the economic potential. However, the determination of a value for a specific resource using alternatives to the established economic criteria does remain a challenge to ecologists, especially to those ecologists who wish to accommodate current ideas about environmental economics in

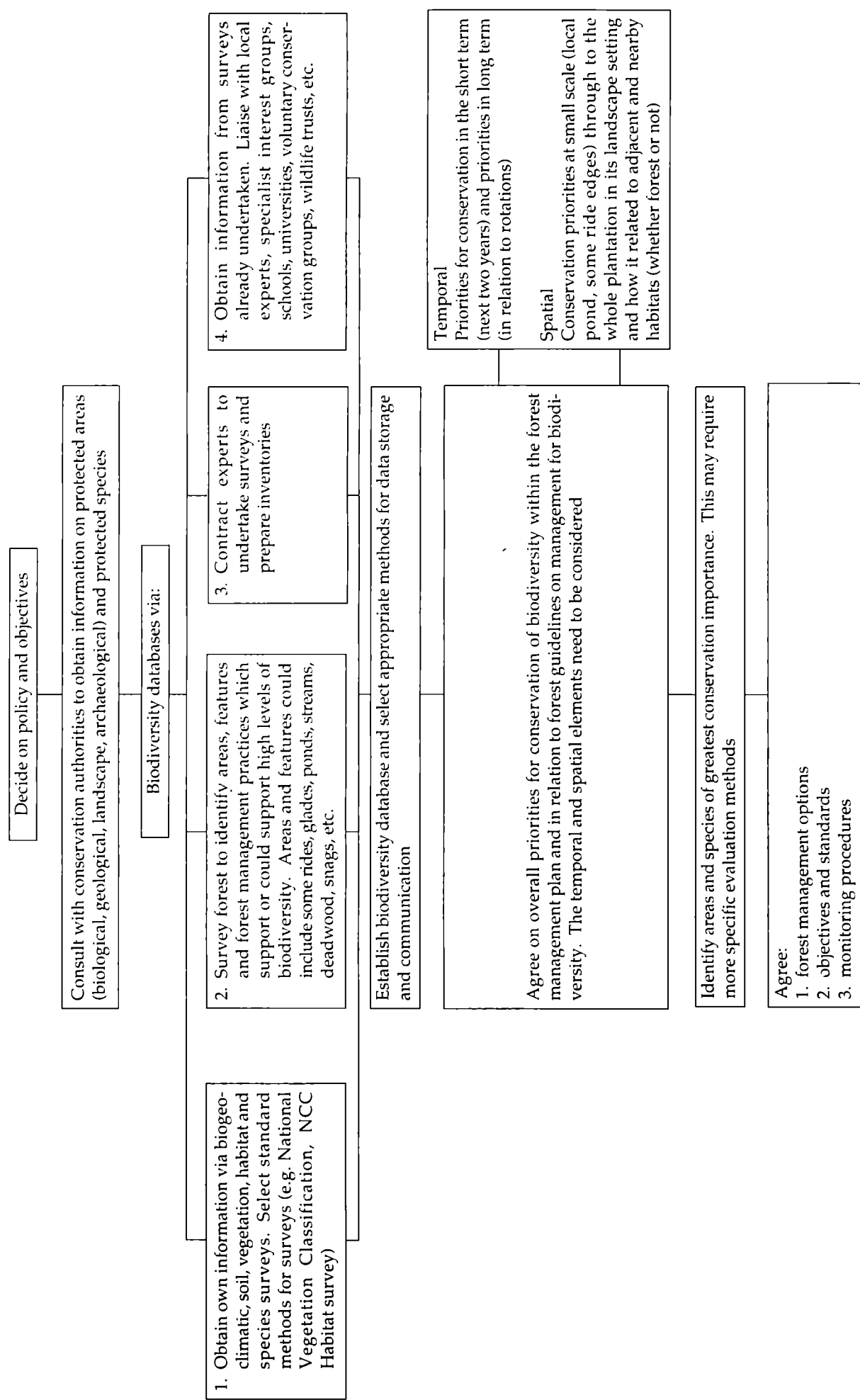


Figure 16 Evaluation of forests for biodiversity.

their approach to ecosystem management. However, the difficulties of evaluating forest resources in terms of man-made capital may be the crux of the problem. Valuation of natural resource capital (interactions, functions, structures and composition) may have to be incorporated within the economists thinking if they are not to be continually wrong in their short-term analyses. In fact the economics will always be wrong while they fail to take account of changes in natural resource stocks and fail to validate the costs to future generations of compromising future choices.

The growing acceptance of multi-use forests and the accommodation of wildlife in plantation forests has now been well established in many countries including Australia, the USA and Scandinavia. In Britain, nature conservation is now recognised as an intrinsic part of modern forestry (Morton Boyd, 1987). Consequently there are now opportunities for an interdisciplinary approach to management of forest resources. Interdisciplinary and collaborative efforts should ensure that there is less conflict between interests. For example, such an approach may help to ensure that the best possible opportunities are used to establish areas for conservation of biodiversity while at the same time managing the forest effectively for timber production. Ecological evaluation of the forest as a means of identifying the areas of greatest conservation interest (and potential interest) is an important aspect of this work but such evaluations must be based on comprehensive inventories.

Forest plantations are dynamic and they change within short spaces of time. It is important that management for biodiversity takes into consideration the changing forest habitats. It would seem advantageous, therefore, to establish a monitoring programme which could be used to appraise the effectiveness of the management undertaken for biodiversity.

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The Forestry Commission and Biodiversity

The Forestry Commission has a responsibility to implement Government policy on biodiversity in Britain's forests. In 1992, the Forestry Authority Biodiversity Initiative was established, with two principal objectives:

To identify methods for improving biodiversity in managed forests.

To develop standards of biodiversity for managed forests.

In response to this Initiative, the Forestry Authority Research Division has introduced a Biodiversity Research Programme. This Programme brings together a wide range of existing research and involves a series of new multi-disciplinary projects, aimed at:

- Developing monitoring protocols and collecting baseline information on selected species or taxa, and structural and habitat diversity in stands of different ages in major UK forest types.
- Identifying biodiversity criteria and indicators for managed forests at the stand and landscape scale.
- Establishing sites for the validation of chosen biodiversity indicators, implementing recommended forest management practices, and long-term monitoring of biodiversity in managed forests.
- Identifying and recommending practical standards by which to appraise biodiversity in managed forests.