RISKS ASSOCIATED WITH MEASURES TO ENHANCE BIODIVERSITY IN EUROPEAN SCOTS PINE FORESTS

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SUMMARY

The conservation and enhancement of biodiversity in European forests is an integral part of sustainable forest management, making it necessary to incorporate into forest operations measures specifically designed to meet the needs of wildlife. Scots pine forests make up a considerable proportion of the total forest area of western Europe and so could make an important contribution to forest biodiversity within the European Union. Pine silviculture is often based on clearfelling and replanting regimes, which produce relatively simple stand structures and landscapes. However, the growing appreciation of the importance of biodiversity has led to modifications being made to forestry methods, including changes to silvicultural systems, the retention of broadleaved trees, provision of deadwood as wildlife habitat, sensitive open ground and edge zone management, prescribed burning, and encouragement of natural regeneration. In many instances, these measures have been adopted with only limited attention given to their potential ecological and economic risks, such as pests and pathogens, habitat loss and fragmentation for some species, and future operational problems. This paper reviews the range of measures which have been proposed and, in some instances, adopted; and assesses the risks associated with them.

KEY WORDS: Biodiversity
Scots pine
Pests
Pathogens
Open space
Deadwood
Prescribed burning

INTRODUCTION

Helsinki and beyond - international commitments to sustainable forestry

Forestry Ministers committed the European forest industry to a shift towards sustainable forest management with the Ministerial Conference on the Protection of Forests in Europe (Ministry of Agriculture and Forestry in Finland, 1994), as part of a wider, global concern at the impacts of forestry on biodiversity. In response, many European countries have developed action plans and strategies aimed at conserving biodiversity and practising sustainable forest management, e.g. *Les Indicateurs de gestion durable des forêts françaises* (French Ministry of Agriculture and Fisheries, 1995); *Action plan for biological diversity and sustainable forestry* (Swedish National Board of Forestry, 1996); *The UK Forestry Standard* (Forestry Commission, 1998).

The purpose of this paper is to examine measures which can be used to enhance the biodiversity of Scots pine forests, and to assess any potential risks associated with these. Scots pine forests make up a considerable proportion of the total forest area of western Europe and, as such, make an important contribution to European forest biodiversity. For a full description of the status of Scots pine forests in Europe, see Mason and Alia (this volume). The paper does not deal with genetic diversity, concentrating primarily upon measures to enhance biodiversity in terms of species richness.

WHAT MEASURES ENHANCE FOREST BIODIVERSITY?

Increasing tree species richness

Conifer forests, and particularly those of plantation origin, may be enhanced through the addition or increased proportions of other species, especially broadleaves. For example, the UK Forestry Standard (Forestry Commission, 1998) states that the landscape and habitat diversity of conifer forests should be enhanced through strategically sited broadleaves, favouring the use of native trees or shrubs, and increasing their proportion to at least 5%.

There are numerous methods for achieving this, including the retention of broadleaved trees and shrubs along riparian zones and forest margins, encouraging natural regeneration of associated species where this occurs, or supplementary planting where necessary.

Increasing structural diversity and uneven-agedness

Many stands of Scots pine are even-aged and relatively simple in their structure (see Kint *et al.,* this volume). Depending on the spatial scale under consideration, there can be biodiversity benefits from promoting greater structural diversity, i.e. increased vegetation cover in different strata of the forest stand, and greater horizontal patchiness, both of which can be encouraged through silvicultural intervention.

Management and preservation of associated habitats

In addition to the forested area, there are numerous associated habitats such as forest edges (including the tree-line), riparian corridors, wetlands, heathlands, and grasslands, which confer biodiversity value at the forest or landscape scale. These areas provide niches for a range of species not found in the forest itself, and their communities may have particular conservation value, e.g. bog pool communities in unplanted mire systems.

Deadwood

A key component of stand structure, acting as a key substrate for many associated species, is dead and dying wood. This is a key factor for biodiversity in forest ecosystems, particularly when a range of forms of deadwood are present (Harmon *et al.,* 1986). Snags and fallen logs form the base of a food chain including microbes, invertebrates, small mammals, and birds (Perry, 1994). For example, the presence of large, decaying broadleaved trees indicates habitat suitability for hole-nesting birds such as woodpeckers (Angelstam and Mikusinski, 1994) as well as other taxa which utilise cavities, e.g. woodland bats (Mayle, 1990).

Fire

Fire plays an important role in the dynamics of Scots pine forests in the boreal and Mediterranean zones of Europe. At the stand-scale, fire may provide valuable niches for fire-dependent species, effectively truncating vegetation succession. At the landscape-scale, forest fires create a mosaic of different-aged forest stands (Shugart, 1984). The regenerating vegetation shows great heterogeneity both on a fine scale, among different sites within one burn, and on a regional scale, among different burns (Miles, 1987).

SPECIFIC MEASURES FOR ENHANCING BIODIVERSITY IN SCOTS PINE FORESTS - BENEFITS AND POSSIBLE RISKS

Encouraging broadleaved trees and associated vegetation

Benefits

The inclusion of broadleaved trees in Scots pine forests is likely to have benefits for biodiversity, particularly in terms of their associated invertebrate species (Kennedy and Southwood, 1984). Aspen, *Populus tremula,* and birch, *Betula* spp. are known to have particular value for biodiversity, e.g. supporting large numbers of invertebrates in boreal and hemi-boreal forests (Patterson, 1993; Worrell, 1995). In fact, the value of *Betula* spp. for wildlife is high for most taxonomic groups and, when mixed into conifer stands, it is likely to increase their diversity considerably (Patterson, 1993). One way in which it does this is through soil improvement, allowing more demanding herbaceous woodland plants to colonise (Miles, 1981). *Populus tremula* has a similar reputation to *Betula* spp. as a species with leaf litter which breaks down quickly, contributing to nutrient cycling and helping to maintain or increase the nutrient status of forest soils (Almgren, 1990). In addition, *Populus tremula* is recommended for retention in Scandinavian forests to provide habitat for both insects and woodpeckers (Hågvar and Sorensen, 1976; Aulen 1991).

Risks

Browsing by mammals is also likely to be influenced by the proportion of broadleaved trees and shrubs present in Scots pine forests. This has been shown for moose, *Alces alces*, which can considerably alter the expected development of young pine stands. In studies by Heikkilä and Härkönen (1996), browsing pressure on young *Pinus sylvestris* was shown to increase where these were growing with an abundance of preferred broadleaved species, e.g. rowan, *Sorbus aucuparia*, willow, *Salix* spp., *Populus tremula* and *Betula* spp. This is also related to the improved forest structure which results from tree species diversification (see later section).

An interesting example is provided by the relationship between *Populus tremula* and pine twist rust, *Melampsora pinitorqua*.. Despite its ecological benefits, the occurrence and encouragement of *Populus tremula* may present a direct and serious risk, since it acts as an alternate host to the rust, *Melampsora pinitorqua*, which affects mainly *Pinus sylvestris*. The damage it causes, although localised, can be quite serious, e.g. distortion of the stem and multiplication of leaders. It occurs where *Pinus sylvestris* and *Populus tremula* are in close proximity, since the sporidia do not normally travel more than approximately 200 metres (Murray, 1965). This rust is a widespread problem throughout Europe, with epidemic occurrence at the southern edge of the natural range of Scots pine in Italy, i.e. in the mountains of western Liguria; with further outbreaks in central and southern Italy. However, it has rarely been found above elevations of 1000 m (Naldini Longo *et al.,* 1991). The disease can be partially controlled by cutting *Populus tremula* suckers early in the summer, and by felling aspen trees in the immediate neighbourhood of *Pinus sylvestris* plantations. Early advice (Murray, 1965) was to avoid planting *Pinus sylvestris* on sites with *Populus tremula* present.

Changing silvicultural system

Structural complexity

The vertical structure of *Pinus sylvestris* forests can be poor, with few layers and limited understorey cover of shrubs (Ferris-Kaan *et al.,* 1998). Opportunities to enhance this exist through the application of stand-scale silvicultural operations such as cleaning, respacing, underplanting, thinning, brashing, pruning, and more innovative ideas such as girdling of trees to create snags (Kerr, 1999). However, there has been little research specific to Scots pine which has looked at the effects of silvicultural techniques on biodiversity.

In all forest types, structural complexity provides the basis for much of the diversity in species, habitats and processes (Franklin, 1992), and differential canopy opening can have positive effects on a range of elements of biodiversity, including vascular plants (e.g. Hannerz and Hånell, 1993). Vertical structure can be significantly improved through the introduction of silvicultural systems, such as group regeneration systems, which break up uniform canopy cover, and the application of different systems at a range of scales can also improve spatial heterogeneity (Kerr, 1999). This can have additional benefits apart from enhancement of biodiversity, with patchworks of different tree species and vegetation types in Mediterranean regions producing different inflammability levels, reducing the spread of fires (Vélez, 1990b). However, where understorey vegetation is highly flammable, any increase in cover can cause problems of providing continuity with the canopy, thereby giving enhanced risks of crown fires.

Enhancing the proportion of old stands

One measure which can add to the structural diversity of a stand is to retain a proportion beyond normal felling age, in order to produce «old-growth» habitat, as recommended for upland conifer forests in Britain (Peterken *et al.,* 1992). Proposals regarding how much of the forested area should be retained in this manner vary, e.g. Peterken *et al.* (1992) and Peterken (1996) advocate partitioning a plantation forest into a dual, long- and short-rotation system, i.e. 16.7% of the stands being retained on a 100 year rotation, with 83.3% on a 40 year rotation. Pennanen (1996), in a model for natural forest management for Finnish forests, recommends that 15% of the forest stands should be permanently set aside from felling, preferably as a mosaic of patches and groups of old trees.

Encouraging varied woodland structure, and hence increasing niche availability, is considered fundamental to insect community diversity (Young, 1992); while factors such as canopy depth, stem density, windblow, wind-snapped trees, shrub and understorey development are of equal importance to birds (Currie and Bamford, 1982). Insect pest outbreaks in old-growth are rare, as it supports higher predator populations (Sterling and Humphrey, 1999). Stands retained for an extended period beyond the normal felling age could provide refugia for predator species, from within which they can expand as prey numbers increase, thus acting as biological control agents. For example, red wood ants, *Formica aquilonia* and *F. rufa*, may act as key insect predators, without which many herbivorous insects become damaging to forest trees (Skinner and Whittaker, 1981).

The absence of these older stands reduces the carrying capacity of the habitat for birds and mammals characteristic of mature forest ecosystems. Old-growth stands (*sensu* Oliver and Larson, 1990) are characterised by a high degree of structural diversity, large quantities of standing and fallen deadwood in various stages of decay, and continuity of habitat for a range of species of high conservation importance (Sterling and Humphrey, 1999). Included are some saprophytic fungi with highly specialised niche requirements, such that the provision of deadwood in managed forests is insufficient to mimic the conditions found in old-growth Scots pine forests (Sippola and Renvall, 1999). The retention of living trees also helps to provide continuity between the previous and future generations, thereby providing refugia for specialist species with poor powers of dispersal.

However, vertical structure is highly influenced by herbivory (Putman *et al.,* 1989), and it is possible that a greater area of retained trees would lead to increased cover and food availability for large herbivores such as deer and moose, possibly leading to increased damage to crop trees in the young and intermediate developmental stages. An increase in proportions of younger-aged stands and openings with early successional species is likely to benefit deer, enabling them to achieve higher population densities (Gill *et al.,* 1996). This does not necessarily lead to an increase in levels of damage to trees, since any system which encourages natural regeneration can produce an excess of suitable browse material; the result of this can be to minimise damage. Reimoser and Gossow (1996) consider that continuous cover silviculture results in a more balanced system with less impact on forest vegetation by browsing deer.

Managing forest edges and open areas

Well-structured forest edges, with associated tree and shrub species, can be attractive for wildlife (Ferris-Kaan, 1991). The formation of an ecotone between forest and open conditions supports a range of taxa which utilise both habitats and specialists which are restricted to this zone. Consequently, both the number of species and their population density can be greater in the ecotone than in the communities flanking it (Ratcliffe, 1991).

Open areas within Scots pine forests may also have importance for biodiversity. Because of the range of sites types on which Scots pine grows, open habitats can comprise many different plant community types. Of particular importance at a European level is lowland *Calluna* heathland, and measures are being undertaken to restore this from forestry in some countries, e.g. Belgium, the Netherlands, and U.K. Minimum standards for open space are now being incorporated into forest management, e.g. between 10-20% (Forestry Commission, 1998), and detailed design guidance also exists, e.g. Bell (1998).

However, the creation of open space can present a threat to species reliant upon forest interior habitat, such as the European pine marten, *Martes martes* Linn. In Britain it is regarded as one of the mammals most vulnerable to fragmentation. This is due to its relatively low population density, slow breeding, fairly poor dispersal capabilities and close affinity to semi-natural habitats (Bright, 1993). In Swedish studies, densities of *Martes martes* were found to be 2-3 times higher within an intact old-growth forest reserve than in the surrounding landscape of commercially managed, fragmented forest (Bjärvall *et al.,* 1977). Therefore, a silvicultural system which creates open space, e.g. clearfelling and replanting, may be detrimental for this species.

In addition, forest edges in Scots pine forests may be attractive habitats for pest insects such as the larger pine shoot beetle, *Tomicus piniperda*, and in this respect they can have economic importance (Peltonen *et al.,* 1997). Studies on the effect of forest edges on the distribution and impact of defoliating Lepidoptera (e.g. Bellinger *et al.,* 1989) support this view. Furthermore, creating contrasting edge habitats is believed to benefit several of the smaller deer species such as roe, *Capreolus capreolus*, and muntjac, *Muntiacus reevesii* (R.M.A. Gill, pers comm). Reimoser (1994) found that if differentiated habitat types bounded by sharp edges were in close proximity, then deer would become over-abundant, giving rise to damage. Furthermore, new edges have been shown to lead to enhanced windthrow (C.P. Quine, pers comm).

Retention of deadwood

Benefits

Deadwood is an important component of forest ecosystems, influencing biological, physical and chemical processes (Harmon *et al.,* 1986; Samuelsson *et al.,* 1994). It provides a substrate for a wide range of organisms, particularly fungi and invertebrates; decay cavities for nesting sites or shelter for many vertebrates; and safe sites for seedling germination or growth of bryophytes.

Forest managers are encouraged to increase the volume and type of deadwood in managed forests, in order to provide suitable niches for dependent wildlife. Kaila *et al.* (1997) have shown that simple and cheap practices, such as leaving all the wood that is of little or no commercial use (deadwood, old or damaged trees) may provide breeding habitats for a number of saproxylic invertebrate species adapted to disturbed conditions, which have been declining in Scandinavia and elsewhere. However, at the same time, a number of the beetle species occurring frequently in the interior of closed forests do not find suitable habitat in clear-fell areas, and so retention of old-growth may be more important.

The retention of deadwood of broadleaved species may have direct benefits to wildlife, but can also have indirect benefits to successful *Pinus sylvestris* establishment and growth. Väre (1989) has shown that mycorrhizal infection of roots was significantly higher where the trees were growing among decaying birch logs. These saplings were also significantly taller compared to others which had no roots in the proximity of decaying wood.

Risks

While retention of dying and deadwood can provide valuable niches for endangered saproxylic insects, it can also provide breeding habitat for pest species. One example is the larger pine shoot beetle, *Tomicus piniperda*, a species unique among European bark beetles in causing considerable growth losses to *Pinus sylvestris* (Långström and Hellqvist, 1991). The beetles breed in fresh pine wood such as storm-felled trees and moribund trees. Their populations are kept in check mainly through the removal of suitable breeding material (Davies and King, 1977), and in studies where freshly dead and dying *Pinus sylvestris* have been left on site, local populations have been shown to build up over time (Czokajlo *et al.,* 1997). With the potential to cause 20-45% volume growth loss over periods of up to 10 years (Långström and Hellqvist, 1991), such a build up is clearly an unacceptable risk. Furthermore, the lesser pine shoot beetle, *Tomicus minor*, is known to be a very effective carrier of blue stain fungi in Scandinavia, preferring to attack standing dead and dying pines rather than fallen trees (Winter and Evans, 1990).

In Mediterranean *Pinus sylvestris* forests, the risk of insect outbreak as a result of deadwood retention is considered to be unacceptable. The risks of damage to live trees in the vicinity, by both *T. piniperda* and *T. minor*, as well as the six-toothed bark beetle, *Ips sexdentatus,* are considered to be too great (I. Cañellas, pers. comm.). Långström (1984) has shown large variations in attack density and brood production by both *Tomicus* species between individual pine trees, thereby greatly reducing the possibility of producing an accurate estimate of risk.

The threat of fire in Mediterranean *Pinus sylvestris* forests also mean that little deadwood is retained. The costs of removing deadwood or harvesting residue are approximately 15 ECU per hectare, which equates to 25-30% of the total costs of harvesting operations (I. Cañellas, pers. comm.).

Prescribed burning

Background

The importance and regularity of forest fires in the ecology of boreal and Mediterranean *Pinus sylvestris* forests is reflected by the large number of organisms (especially invertebrates and fungi) that are directly or indirectly dependent on or favoured by fire (Esseen *et al.,* 1992). Many of these organisms have shown population declines, and this general pattern is believed to result from efficient fire suppression measures during the twentieth century (Penttilä and Kotiranta, 1996).

The mean interval of forest fires has been estimated at between 80-120 years in Fennoscandia (Zackrisson, 1977; Engelmark, 1984), although there has been considerable variation. Studies in southern Finland (Tolonen, 1983) suggest a frequency of 70-700 years, and dry *Pinus sylvestris* forests have burned more often (on average 50-90 years) than moister, Norway spruce, *Picea abies* L. (Karst.)*,* dominated forests (90-160 years). Whether natural or anthropogenic, past fires were mainly surface fires of low to moderate intensity (Sannikov and Goldammer, 1996). They played a significant role in creating multiple-aged stands with a large number of old pines and a considerable number of dead standing and lying trees (Östlund *et al.,* 1997).

Contemporary forest management in boreal *Pinus sylvestris* forests does not create the extensive areas dominated by broadleaved trees, that formerly were common on more productive sites after fire (and which harbour a particular fauna). It also fails to provide habitat for fire-adapted plants and animals. Granström (1996) argues that a system of prescribed burning is necessary in order to maintain fire structured forests within reserve areas, recognising that fire-adapted fauna are mobile and able to locate rather small burns (Wikars, 1992). In contrast, fire-adapted plants are virtually sedentary as they are mainly seedbank species, and hence burning will have to be repeated over time on the same piece of land (Granström, 1996).

However, in Mediterranean *Pinus sylvestris* forests, prescribed burning is not specifically practised to meet any biodiversity objectives, and this may be partly explained by the susbstantial areas of both natural and man-made fires which occur annually (I. Cañellas, pers. comm.). About 50,000 fires sweep through 700,000-1,000,000 ha of Mediterranean forest each year (Vélez, 1990a), and this causes enormous economic and ecological damage. Where burning does occur, it may be in order to improve the quality of pasture for cattle production, through the elimination of shrub-dominated vegetation. This is also recognised as a means of protection against forest fires, an operation costing approximately 9 ECU per hectare per year in Spain (I. Cañellas, pers. comm.). In areas where there are conflicts between forestry and grazing, it is increasingly recognised that controlled grazing should be encouraged rather than prohibited, reducing fine fuel accumulation (Vélez, 1990a).

Silvicultural Benefits

Prescribed burning is recognised as one way of simulating the effects of forest fires in improving conditions for natural regeneration of forest stands and, occasionally, to open up the landscape (Lähde *et al.,* 1999). A serious obstacle to natural regeneration is the build up of a thick layer of raw humus, and on dry- to fairly-dry sites with *Vaccinium*-dominated vegetation, this can lead to replacement of *Pinus sylvestris* by spruce. Under such circumstances, burning has been shown to be the most effective means of establishing productive stands of pine (Viro, 1974). It is important that the ground is dry at the time of burning, in order to ensure that grass roots are killed off effectively. If not, then grass will quickly recolonise the site, making scarification necessary (Weslien and Wennström, 1997). Burning helps to thin the humus layer, improving the thermal conditions of the site and, in conjunction with cultivation, prescribed burning is considered ecologically beneficial. However, it has often been replaced by mechanised soil preparation (Parviainen, 1994).

Silvicultural Risks

If planting is carried out following burning, there is a high risk of seedlings being attacked by pine fire fungus, *Rhizina undulata*, and pine weevil, *Hylobius abietis. Rhizina* shows a close association with fire sites, due to the dormancy of its spores being broken by short-term heating to temperatures between 38-45^{0}C, followed by saproptrophic spread on coniferous roots (Butin, 1995). On most sites, therefore, the restock areas should be left unplanted for at least two years (Callan, 1990; Weslien and Wennström, 1997). Failure to observe this delay can lead to high levels of mortality, e.g. 90-100% in replanted *Pinus sylvestris* stands in northern Germany (Hartmann and Schmid-Molholm, 1997).

Post-fire Recovery

An important consideration in the use of prescribed burning is the post-fire recovery period, since the direct effect of fire is likely to be destructive for many species groups, e.g. fungi (Parmeter, 1977; Pugh and Boddy, 1988; Wicklow, 1988). Gorshkov and Bakkal (1996) studied the rate of post-fire recovery of species diversity in a range of *Pinus sylvestris* forest site types in the Kola Peninsula in Russia, and found a difference in the recovery dynamics of separate components of the community from 5-15 years (dwarf shrubs and herbs) to 120-140 years (tree stratum), with moss-lichen cover stabilising after 90-140 years. They also found that plant species richness recovered 30 years after fire, regardless of forest site type, in contrast to stabilisation of forest structure (i.e. 3-5 times earlier).

Studies of the impact of fires on understorey vegetation of martime pine, *Pinus pinaster*, forests indicate that species richness and diversity increase after burning (Rego *et al.,* 1991). Initially, the vegetation is made up of surviving individuals, but new species then progressively colonise the microsites opened by the fire. Finally, these species then gradually decline as the pre-fire community re-establishes on the site. It has been concluded that fire does not dramatically alter species composition (Trabaud and Lepart, 1980).

In boreal *Pinus sylvestris* forests, post-fire recovery is linked to the soil moisture status. On dry sites, the recovery of vegetation takes place slowly and a lot of nutrients released on burning may be leached away (Mälkönen and Levula, 1996). In contrast, on moist sites, very favourable nutrient conditions after burning stimulate the fast development of ground vegetation, which maintains nutrient cycling (Lindholm and Vasander, 1987). Furthermore, prescribed burning decreases microbial biomass and changes the microbe population, although the subsequent decrease in soil acidity and vegetation development gradually promote microbial activity (Fritze *et al.,* 1993).

Impact of Fire on Fungi and Invertebrates

Fire disturbance also provides large inputs of new, mainly competition-free resources and substrates for a range of taxa to use, insects in particular (Weslien, 1997). Pugh and Boddy (1988) have described this as «enrichment disturbance», and recognise that it can operate both on small and large scales. However, these resources are more typically over a large scale and are usually quite homogenous (Penttilä and Kotiranta, 1996).

This can have a number of effects on insect and disease conditions; the susceptibility of trees to insect and disease outbreaks is sometimes increased and sometimes decreased by fire. Conifers weakened by fire may become more susceptible to attack by bark beetles, and yet on the other hand, the mosaic of vegetation types and age classes created by frequent fire may act as a deterrent to the spread of large-scale epidemics (Lorimer, 1990).

The increased environmental homogeneity resulting from fire disturbance is thought to account for the observed decrease in diversity of fungal communities (Zak, 1991). While this may hold true at the local or stand scale, if the effect of fire is patchy at the landscape scale it is likely that fungal communities become more diverse (Zak and Wicklow, 1980).

In a study of the short-term effects of prescribed burning on wood-rotting fungi, Penttilä and Kotiranta (1996) found that both fungal species diversity and evenness in community structure had decreased considerably as compared with the pre-fire community. The greatest losses in species numbers occurred in moderately and strongly decayed trees, in coniferous trees and in very strongly burned trees. In contrast, the fungi associated with non-decayed and slightly-decayed trees, deciduous trees and slightly burned trees seemed to have survived the fire quite well; in these groups the species numbers had increased slightly as compared with the pre-fire community. *Heterobasidion annosum*, considered the most important decayer of commercial wood in Finland (Niemelä, 1994), disappeared after the fire. Other studies (e.g. Froelich and Dell, 1967) have found that fire reduces the level of infection of *Heterobasidion*. However, from a biodiversity perspective, the effects of fire were detrimental: no fruitbodies of threatened polypore species or other «old-forest species» were found again after the fire. Those species favoured by fire were mainly ruderal species able to utilise new, competition-free resources created by fire, and species found in dry, open conditions outside areas influenced by forest fires. Nevertheless, it is possible that a longer-term view is needed, since polypores in particular seem to have a lag phase in their fruitbody production after fire (Froelich and Dell, 1967), and it is probable that fruiting will be more abundant with time.

The strongest evidence for the effects of fire on invertebrates comes from North America, where the policy of fire suppression has actually given rise to greater intensity and frequency of attack by some pest species (McCullough *et al.,* 1998). This is well demonstrated by spruce budworm, *Choristoneura fumiferana*, which has increased in areas where fire suppression has led towards monocultures of fir or spruce/fir rather than the mosaic of pioneer species such as *Populus tremula, Betula* spp., jack pine, *Pinus banksiana,* and black spruce, *Picea mariana* (McRae *et al.,* 1994). Furthermore, there are many fire-dependent species, particularly among the Coleoptera (Carabidae, Buprestidae, Cerambycidae, Cucujidae, Lathridiidae), which are becoming rarer as fire suppression regimes increase, especially in the USA (Dajoz, 1998).

Ecological Conflicts

It is now recognised in Sweden that the use of fire may give rise to conservation conflicts (Linder, 1998). This is due to the long period that has elapsed since the last fire, creating habitats where red-listed species dependent on late successional structures are now present. For example, prescribed burning can remove most of the harvesting residues which may remain on the site, which provide a valuable deadwood habitat, as well as the ground vegetation (Mälkönen and Levula, 1996). Fire has also been shown to kill the mature «granny» pines under some conditions in northern Sweden (Linder *et al.,* 1998). It seems highly probable that if fire is prescribed in such stands, then red-listed species will be disadvantaged, or even eradicated. However, there is growing interest in attempting to mimic natural disturbance regimes in forests, and it is debatable whether this strategy should be subordinate to the conservation of single-species.

DISCUSSION

Resolution of conflict between costs and benefits

The biodiversity benefits of many of the measures proposed remain unquantified, with ecological intuition playing a large role. This is a problem hampering progress towards a realistic cost-benefit analysis, and serves to highlight the need for further research and monitoring. Costs, however, may be defined in either ecological, silvicultural (operations, timber quality and supply, etc.), genetic, and economic terms.

Costs and benefits between managing for biodiversity and timber production have been examined in a number of studies (e.g. Holland *et al.,* 1994). Lindenmayer (1999) suggests that restoration of structural and floristic complexity in managed forests could require more complicated survey and planning. This is likely to add to the costs of production in the short-term. In the United Kingdom, restructuring of even-aged plantations managed by patch clearfelling systems in order to enhance non-wood production functions, has been estimated to cost around an additional 10% (McIntosh, 1989). However, failure to maintain key ecosystem processes may lead to long-term losses of forest productivity, resulting in significant long-term economic costs.

The trade-offs between production costs and biodiversity benefits are tensions which may be examined using integrated economic and ecological modelling, e.g. Hyde (1989). It is possible to use linear programming to develop models to evaluate the benefits against the costs, using indices characterising stand species diversity, basal area diversity and vertical crown diversity (see Kint *et al.,* this volume). In order to maximise each diversity index, there will be some loss of timber production; to the order of 17-20% in studies of mixed spruce-fir-pine forests in Utah, USA (Holland *et al.,* 1994). Conversely, to maximise timber production, basal area and vertical crown diversity decline by roughly 10%, while species diversity declines by 24% (Holland *et al.,* 1994).

Hodge and Peterken (1998) have attempted to analyse the cost of strategies for the retention of deadwood in British forests. Based on ecological intuition and knowledge of the minimum requirement of certain invertebrate indicator species, a minimum recommendation of 5 m^3 ha^{-1} deadwood (snags and fallen logs) greater than 15 cm diameter has been proposed. This is likely to represent between 0.25% and 1% of standing volume in economically mature conifer stands. In order to minimise the revenue foregone by retentions on this scale, lower value stems should be retained. There is no conflict here, since these tend to be more valuable for wildlife, e.g. those with forks or branching into which decay fungi can gain easier access.

The introduction of significant areas of long-rotation plantations would alter the size distribution and total amount of wood produced, and would reduce the rate of financial return (Peterken *et al,* 1992), unless the value of the retained stems increased dramatically. Lämås *et al.* (1996) have estimated the economic loss due to extended rotation age in *Pinus sylvestris*-dominated forests in northern Sweden, and suggests that their model can be used to find stands for which the loss in present net value is low.

CONCLUSIONS

Research has so far concentrated on establishing the positive relationships between forest management for biodiversity and the responses of various taxa, e.g. increased deadwood has been shown to enhance the species richness of macrofungi in *Piuns sylvestris* forests in lowland Britain (Ferris *et al.,* in Press). However, often anecdotal evidence suggests that there are costs associated with such measures. These costs are not solely due to reduced revenues, and there needs to be more attention focused on costs due to the increased ecological and silvicultural risks. In some cases where this has been done, a decision needs to be taken by the forest manager, to weigh up the pros and cons of each particular action. At present, there is no system to enable this to be done in an informed way, and it seems unlikely that any system could take account of the unique set of circumstances in existence at each forest site; generalisations could, under these conditions, be misleading.

Nonetheless, for European *Pinus sylvestris* forests, a number of management options which may favour biodiversity have been extensively researched, e.g. increasing the proportion of broadleaved trees, management of forest edges and open space, changes to the silvicultural system, the retention of deadwood, and prescribed burning. It is possible to draw on this work to appraise the risks and benefits (Table 1), although it is an inescapable fact that the key factor in most cases will be the economic argument. The development of a reliable system for evaluating the revenue foregone as a consequence of biodiversity measures is an important area for future research.

TABLE 1

ECOLOGICAL BENEFITS AND RISKS TO THE FOREST OF MEASURES TO ENHANCE BIODIVERSITY

Beneficios y riesgos ecológicos para el bosque de las medidas para aumentar la biodiversidad

Measure		Biodiversity benefit	Risk to forest
Tree Species Diversity			
	Birds	++	None
	Mammals	+	++ (browsing)
	Invertebrates	+++	+ (defoliators)
	Flora	++	None
	Lichens	+	None
	Fungi	++	+
Structural Diversity			
	Birds	+++	None
	Mammals	++	None
	Invertebrates	++	+
	Flora	++	None
	Lichens	++	None
	Fungi	++	+
Edges & Open Areas			
	Birds	++	None
	Mammals	++	++(browsing)
	Invertebrates	++	+
	Flora	++	None
	Lichens	+	None
	Fungi	+	None
Deadwood Retention			
	Birds	++	None
	Mammals	++	None
	Invertebrates	+++	+(e.g. bark beetles)
	Flora	+	None
	Lichens	++	None
	Fungi	+++	+(e.g. pathogens)
Prescribed Burning			
	Birds	+	None
	Mammals	+	None
	Invertebrates	++	+
	Flora	+	None
	Lichens	+	None
	Fungi	++	+

(scoring system: + minor benefit/low risk, ++ beneficial for some species or groups/moderate risk, +++ highly beneficial/high risk).

ACKNOWLEDGEMENTS

We would like to thank Hugh Evans for comments on an earlier draft of this paper, and colleagues working with us in the E.U. Concerted Action FAIR1: CT95-1447 «Silviculture and Biodiversity of Scots pine Forests in Europe» for sharing their ideas with us.

RESUMEN

Riesgos asociados con las medidas para aumentar la biodiversidad en los bosques europeos de Pino silvestre

La conservación y el aumento de la biodiversidad en bosques Europeos es una parte integral de la gestión sostentable del bosque, siendo necesario incorporar en las operaciones forestales medidas específicamente diseñadas para cubrir las necesidades de la fauna. Los bosques de Pino silvestre constituyen una proporción considerable del área total de bosque de Europa occidental y podrían hacer una contribución importante a la biodiversidad de los bosques dentro de la Unión Europea. La selvicultura del pino se basa frecuentemente en las cortas a hecho y las plantaciones posteriores, que producen estructuras del rodal y paisajes relativamente simples. Sin embargo, la creciente consideración de la importancia de la biodiversidad ha conducido a modificaciones en los métodos de la silvicultura, incluyendo cambios en los tratamientos selvícolas, la retención de pies de frondosas, la provisión de madera muerta como hábitat de la fauna, la gestión sensible de las zonas de borde y de los terrenos abiertos, quemas prescritas, y el favorecer la regeneración natural. En muchos casos, estas medidas se han adoptado prestando una limitada atención a sus riesgos potenciales ecológicos y económicos, tales como plagas y patógenos, la fragmentación y pérdida de hábitat para algunas especies, y futuros problemas operacionales. Este trabajo revisa la gama de medidas que se han propuesto y, en algunos casos, adoptada; y se evalúan los riesgos asociadas con ellas.

PALABRAS CLAVE: Biodiversidad
Pino silvestre
Plagas
Patógenos
Espacio abierto
Madera muerta
Quemas prescritas

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