

Aspects of Biodiversity of Scots Pine Forests in Europe

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Abstract

Management of Scots pine, the most widely distributed conifer in the world, was often based on clearfelling and replanting regimes, resulting in a rather poor biodiversity value. However, there is nowadays a general expectation to increase biodiversity by applying a more complex silviculture.

Although present knowledge of genetic factors is insufficient to draw firm conclusions on the desirable level of genetic diversity, it seems unlikely that current silvicultural practices will limit genetic diversity in Scots pine

Native pinewoods are rare in Europe, but have a flora and fauna of high conservation value. Therefore they must be recognized as a priority habitat under the European Commission's Habitat Directive. The high conservation value of native stands is a function of their old-growth structure that provides a rare habitat.

A number of measures should be taken in all types of Scots pine forests to enhance biodiversity. Firstly, old growth habitats should be promoted. Foresters have to accept that a small percentage of the pine resource should be managed upon much longer rotations. Secondly, retention of deadwood should be encouraged. Dead and dying wood are key components of stand structure and act as key substrates for many associated species, such as microbes, invertebrates, small mammals and birds. Furthermore, a complex stand structure should be promoted at both the horizontal and the vertical scale. A small scale forestry, group regeneration systems, natural regeneration, introduction of broadleaves and stronger thinnings are strongly recommended. Availability of quite precise niches significantly increases biodiversity value. Conservation of isolated populations, found under extreme environmental conditions, is an absolute need. Populations endangered either by their small size or by environmental stresses, hybridization with other species or human interference should be primarily conserved. Forest edges support a range of taxa, and open habitats can comprise many different plant community types. A large number of organisms are directly or indirectly dependent on or favoured by fire.

However, enhancing biodiversity provokes also some risks. Generally, browsing is considered as a moderate risk. Introduced species, such as aspen, act as an alternate host to the rust. Open species can present a threat to the European pine marten, dying and deadwood can provide breeding habitats for pest species (*Tomicus piniperda*), burning increases the risk of seedlings being attacked pine fire fungus (*Rhizina undulata*) and forest edges may be an attractive habitat for pest insects.

An extension of existing growth models is needed to incorporate biodiversity issues in forest management planning. Distance dependent individual tree growth models should be developed. Sets of indicators for biodiversity must integrate compositional, structural and functional

attributes. Attributes such as species richness, species abundance, species diversity, horizontal and vertical distribution, tree age, tree size, stand diversity, architectural complexity, genetic variants and deadwood are needed for the establishment of biodiversity indices. Assessment of functional phenomena needs the knowledge of the driving biotic environmental factors.

Key words: *Scots pine, biodiversity, genetic diversity, silvicultural measures, risks, assessment.*

1. Introduction

Scots pine (*Pinus sylvestris* L.) is the most widely distributed conifer in the world. Its range spreads over a distance of 14,000 km, from 8°W in Spain to 141°E in Russia and from latitude 70°N to 37°N (Boratynski, 1991; Nikolov and Helmisaari, 1992)

In the European Union the extent of Scots pine exceeds 28 million hectares, representing at least 20% of the commercial forest area. This is about 20% of the world distribution of the species. Nearly 80% of this area is to be found in two Scandinavian states, Finland and Sweden (Mason & Alia, 2000).

There was a major expansion in the area of Scots pine forests in Europe during the twentieth century. The role of Scots pine forests in Europe varied from pioneer plantation species on degraded agricultural land to a dominant component of a native forest ecosystem. **Pine silviculture was often based on clearfelling and replanting regimes**, which produced relatively simple stand structures and landscapes. **However, there is nowadays a general expectation of a move away from simple management systems towards a more complex silviculture based upon greater use of natural regeneration and with a greater diversity of tree species and ages.** This move is likely to be more pronounced in Central and Western Europe where the economic returns from current management are low.

This evolution is in line with the pan-European criteria for sustainable forest management agreed in Resolution L2 of the Lisbon conference about "the maintenance, conservation and appropriate enhancement of biological diversity in forest ecosystems". **A set of guidelines emphasizes herewith the following aspects of positive benefit to biodiversity:**

- a preference for natural regeneration;
- use of species and provenances that are adapted to site;
- management practices that promote a diversity of stand structures and species, where appropriate;
- maintenance of traditional management where this has created valuable ecosystems;
- tending and harvesting to be carried out so that no lasting damage is caused to the ecosystem;
- infrastructure to be designed so that damage to the forest ecosystem is minimized;
- achievement of a balance between grazing pressure and the forest ecosystem;
- maintenance of standing and fallen deadwood as well as of rare tree species;
- protection and/or restoration of key biotopes.

The conservation and enhancement of biodiversity in European forests is obviously an integral part of sustainable forest management, making it necessary to incorporate into forest operations measures specifically designed to meet the needs of wildlife. However, there is still a tremendous need of information on the different levels of forest ecosystems and management systems. Therefore, this paper wants to focus on some aspects of an important coniferous tree species, with a high biodiversity potential, Scots pine. Aspects of general genetic diversity of this widespread species will be presented, together with a comparison of biodiversity aspects in native pinewoods and pure plantations. Most appropriate measures to enhance biodiversity, such as structure complexity, old growth, deadwood, etc., will be discussed. Risks linked with these measures will also be presented. And finally the need and problems of biodiversity assessment is stressed.

2. Genetic diversity of Scots pine forests

Genetic diversity is the basis for ecological biodiversity. It is the first part of the trilogy of biodiversity standard components "genes, species and ecosystems".

The vast natural range of Scots pine encompasses a great diversity of ecological and physiographic conditions, which have caused the species to evolve and fragment into a larger number of local variants or "races". This is very clearly shown in numerous provenance experiments (Wright and Bull, 1963; Giertych, 1979).

A complicating factor in discussing genetic diversity, however, is the traditional "free movement of germ plasm" in Central Europe. Large areas of pine plantations were established with non-local material, which origin cannot be traced.

In a pine forest one can observe differences among trees in size, stem form, branching habit, etc. All these differences are not due to genetic factors alone. For example, in a mixture of age classes, older trees are usually taller and bear more cones than young ones. Branchiness, stem form, and diameter are strongly modified by the ambient soil fertility and the growing space available.

Fortunately, for some thirty years **it has been possible to reveal the genetic variation at the molecular level**, initially by means of isoenzymes, and more recently by means of various DNA markers. Numerous studies on the genetic structure of Scots pine have consistently **shown large genetic variation in all characteristics studied** (Forrest et al., 2000; Koski, 2000).

Although Scots pine is one of the most thoroughly investigated conifers as regards the reproductive systems and genetic factors, Koski (2000) concludes that **our knowledge is insufficient to draw firm conclusions on the desirable level of genetic diversity in pine forests** and the impacts of human activities on this. The pattern of variation is noteworthy. **As a rule the within population (stand) variation comprises more than 90% of the total genetic variation within a region.** And in terms of isoenzyme variation, Muona and Harju (1989) reported that **more than 98% of overall variation in Central Finland occurred within stands.** Koski (2000) especially compared the genetic value of natural and artificial regeneration. The starting point was that the maintenance of genetic diversity is generally accepted as an

unconditional goal of forest management. One can sometimes find the statement that genetic diversity of forest trees can be maintained only by means of natural regeneration, and in extreme opinions only in unmanaged forests. As a consequence there is a common belief, that artificial regeneration would always lead to loss of genetic diversity. Genetically improved reforestation material originating from clonal seed orchards has an especially bad reputation. Koski (2000) reviewed this argument and concluded that there is little evidence that greater use of artificial regeneration will result in decreased genetic diversity at the stand level. This conclusion appears to hold whether the planted seedlings are derived from wild stands, from seed stands or from seed orchards. However, in the last instance, the orchard should contain at least 40 clones to provide the heterozygosity expected of natural populations. Consequently **there is no reason to postulate that artificial regeneration per se would systematically erase genetic diversity.** Both in managed natural regeneration and in artificial regeneration the genetic quality of the reproductive material is decisive.

Moreover, Koski (2000) also stresses that despite nearly a century of research into the genetics of Scots pine, there is still limited knowledge of genetic diversity in adaptive traits such as cold hardiness or drought tolerance. Bastien and Alia (2000) too found that genetic diversity measured by isozymes or molecular markers is a poor indicator of genetic variability for adaptive traits.

Present knowledge shows that it seems unlikely that current silvicultural practices will limit genetic diversity in Scots pine. Current national and international guidelines take genetic values into account. The same holds true at advanced tree breeding programs, where long breeding populations are being established for the maintenance of variation. In Finland, all approved seed sources provide as large genetic diversity as natural regeneration. Clonal seed orchards are alleged to produce genetically uniform material. Plus trees were selected for their superior phenotype in growth and stem form. Grafts of plus trees are used in clonal seed orchards (Koski 2000).

3. Biodiversity of native Scots pine forests versus planted Scots pine forests

Native pinewoods are rare in Europe. In Great Britain there are 241,000 hectares of Scots pine forests, but native pinewoods are found only in Highlands of Northern Scotland where they cover an estimated 16,000 ha occurring in 84 sites. Areas of indigenous pinewood typically occur on infertile lithologies with strongly leached podsollic soils. All native pinewoods have been influenced by man to varying degrees. Moreover, grazing by deer, sheep and cattle and extensive felling have simplified the structure and species composition. **These native pinewoods are recognized as a priority habitat under the European Commission's Habitat Directive** (Forestry Commission, 1994). Besides there is still a smaller area of semi-natural pinewoods that occur within the natural range of Scots pine and are believed to result from natural regeneration from planted stands of trees of native origin. In Germany the proportion of natural pine forest in the potential natural vegetation is less than 3% (Beck, 2000).

Native pinewoods have flora and fauna of high conservation value. In Scotland they include the endemic bird species Scottish crossbill (*Loxia scotica*), other rare bird species together with

a group of vulnerable or endangered invertebrate and bryophyte species (Mason & Alia, 2000). In Sweden a survey of 1487 threatened plant, animal and fungus species found that 10 % were primarily associated with Scots pine forests (Berg et al., 2000).

Until recently, studies of pinewood biodiversity tended to focus upon the native pinewoods because of their specialized flora and fauna and there have been fewer investigations in planted pinewood. A comparative study of Mason (2000) reveals however some interesting results.

Summers et al. (1995) claim that there are **no mammal species specifically associated with Scots pine** in Britain. However, planted Scots pine forests are a favoured habitat for the red squirrel (*Sciurus vulgaris*) where stems in the late stem exclusion and understory reinitiation phase (see Mason 2000), growing at a density of ca. 500 trees ha⁻¹, allow free movement through the canopy without having to come to the ground. Trees of this size also provide larger cones than the smaller ones associated with the older trees in mature pinewoods.

Scotland's Pine stands in the early exclusion stage have some 16-20 **breeding bird species** and there are about 30 breeding species in the transition to understory reinitiation (Mason, 2000; Summers et al., 1995). These figures compare with some 45 species breeding in the "old-growth" native pinewood habitat. The three key pinewood bird species, capercaillie (*Tetrao urogallus*), crested tit (*Parus cristatus*) and Scottish crossbill are found most regularly in this latter stage. Capercaillie populations are generally higher in native pinewoods than in younger planted stands (17-24 birds km⁻² against 8 km⁻²) where the limited factor appears to be a suitable chic habitat (Moss & Picozzi, 1994). The crested tit favour native pinewoods with rotten stumps and a diverse field layer but have also been recorded breeding in Scots pine plantations of 20 years and older. The Scottish crossbill breeds almost exclusively in native pinewoods where they frequent the older more open stands, often in trees 160-200 or more years old. Limiting factors in the spread of this species into younger natural forest or plantations include tree age, density and cone size.

Many invertebrate species in native pinewoods also occur in Scots pine plantations, including many common and vagrant species also found in other habitats. Humphrey et al. (1999) found no difference in species richness or diversity for both syrphids (hoverflies) and carabids (ground beetles). Vertical stand structure showed the best correlation. Richness and diversity was greatest in young stands with a high field layer vegetation. **However, the rare invertebrate species tend to be associated with the deadwood habitat and open structure characteristic of the "old-growth" stage.** For example, three rare species of wood ant depend upon an open woodland structure as sunlight is needed to warm their nests to help mature the brood. They are commonest in the old-growth pinewoods but are also found in younger Scots pine plantations although they are restricted to the sunny margins of the stands. They are very susceptible to clear felling which removes their food supply and nest building materials.

According to Pitkin et al. (1995) the **most characteristic plant species of Scotland's native pinewoods are dependent on an intact field and shrub layer but have also all been found in mature, undisturbed Scots pine plantations in the understory reinitiation phase.** These plants are potentially vulnerable to disturbance through forestry operations or natural catastrophe. Disturbance caused through clearfelling, heavy thinning, intensive soil cultivation and herbicide applications of large areas can damage the viability of plant populations. A key

aspect in the survival of such plant species would be to maintain tracts of mature, undisturbed woodland and to provide connectivity to areas where new pinewoods are developing. Similar conclusions apply to bryophyte, lichen and fungal communities associated with British pinewoods.

Anders and Hofmann (1997) stress that the number of species found in pine ecosystems can vary strongly due to the combination of different ecological factors. Solely from the number of species, it is not possible to assess the ecological quality, stability or naturalness. It is necessary to compare the number of species with potential natural conditions and to assess the vegetation qualitatively and quantitatively considering the site conditions.

In Germany pure pine plantations predominantly occur on sites of natural hardwood forests. But **the planting of artificial pine forests on former broadleaved sites led to ecological changes**, such as (Beck, 2000):

- Increase of light penetration in the pole stage. The open structures allow more light to enter the canopy and increase heat exchange. In combination with a high crown interception, the latter leads to drier ground conditions as whole.
- Poorly decomposition of the litter of the pine stands, due to an unfavorable C/N ratio, generally results in raw humus or in a raw humus/mull intermediate with distinct properties of the upper horizon.

Anders & Hofmann (1997) found that such ecological conditions result in the following changes in plant species composition:

- loss of more nutrient and moisture demanding species;
- higher coverage of mosses;
- promotion of colonization and development of acidophilic, light and temperature demanding species with low nutrient demands;
- increase of acidophilic species on intermediate and fertile sites following induced N-deposition.

Mason (2000) concludes that the **distinction traditionally drawn between the native pinewoods and the plantation pinewood resource is artificial**. Indeed, while native Scots pine stands appear to have distinctive communities compared to plantations, a number of rare species, once thought to be restricted to native pinewoods, occur in plantations in the later stages of stand development. **The high conservation value of the native stands is a function of their old-growth structure that provides a rare habitat** in what is otherwise a comparatively young forest area. **The amount and distribution of deadwood are critical features for the pine ecosystem biodiversity.**

The presence of key species is a function of development stage and site type rather than of pinewood category. However, most of the notable species are critically dependent upon the characteristic structure of "old growth" native pinewoods, such as: a lower stocking density, more large trees, greater variation in diameter, a clumped distribution of stems and appreciable quantities of standing and lying deadwood.

An overall conclusion of this review might be that **the integration of timber production with enhanced biodiversity does not require the abandonment of silvicultural practices.** Anyway a relatively small percentage of pinewood stands should be managed upon much longer rotations than those which prevailed in the past. It should also be stressed that it is still debatable how biodiversity should be measured as a whole. In this respect the results of Vaisanen et al. (1993), indicating that **total species richness and abundance is actually greater in managed stands, but at the expense of rarity value,** are noticeable. Managers have to compromise between the two aims of increasing total biodiversity and of encouraging relatively rare species with potentially high specialized requirements. Knowledge of niche availability, especially the "quality" of niches may eventually allow choices to be made on a rational ecological basis, particularly at quite local scales (Evans & Jukes, 2000).

4. Measures enhancing forest biodiversity

Many authors note that **a number of measures are being taken in all types of Scots pine forest to enhance biodiversity** (Lust et al., 1998; Beck, 2000; Evans & Jukes, 2000; Ferris & Pritchard, 2000; Koski, 2000; Lust et al., 2000; Mason, 2000; Mason & Alia, 2000). The measures may cover a lot of issues, such as:

- the promotion of increased deadwood habitat and of old growth;
- lengthening of rotation period;
- stand transformation;
- change of silvicultural systems;
- favouring of native broadleaved trees;
- encouragement of natural regeneration;
- increase of structural diversity;
- greater use of prescribed burning;
- attention to special habitats;
- managing forest edges and open areas;
- specific niche provision;
- management and preservation of associated habitats;
- designation of natural reserves;
- avoidance of artificial measures, monitoring of impact;
- etc.

The above measures correspond to a large extent to the well known FSC (Forest Stewardship Council) standards, which are now shaping the future of forestry in many European countries. **In Sweden the FSC agreement includes the following environment and biodiversity standards** (Egnell, 2000):

1. **At least 5%** of the productive forest area is exempted from management other than the measures required to preserve and support the natural biological diversity of the habitat.
2. **Soil scarification** is to be limited to sites where this measure is required to achieve good regeneration.

3. Owners of larger landholdings are to take all reasonable measures to **burn an area** corresponding to at least 5% of the regeneration area of dry and mesic areas during a 5 year period. "Larger landholdings" are not defined in the standards but they are written for large forest companies with landholdings of hundreds of thousand hectares.
4. **Natural regeneration**, for example under shelterwood and seed tree systems, is used where this method will result in good regeneration of species of tree adapted to the site and to management goals.
5. **Trees with high biodiversity** value should be protected in all measures, and not felled. Cleaning and thinning are carried out in a way that protects, to a reasonable extent, potentially high biodiversity value trees.
6. When felling for regeneration, enough wind resistant trees of various species with good chances of developing into large, **old trees** during the next rotation period should be left, with the aim of incorporating at least 10 such trees per hectare into the next forest generation.
7. **Dead wood**, except for small felling residues, is to be protected from forest management unless there is a documented risk of the mass reproduction of insect pests.
8. **Standing dead wood**, such as high stumps of common deciduous and coniferous trees should be created during thinning and regeneration felling.
9. The **origin of seedlings and seeds** is to be documented. Provenances adapted to the site are to be used within the framework of the National Board of Forestry regulations and general guidelines.
10. If their natural presence allows, **broadleaved trees** are to be protected when cleaning and thinning so that they make up at least 5-20% of the stand, including the immediately surrounding area (depending on region, soil conditions, site quality and the total proportion of broadleaved trees on the forest holding).

Today, implementation of sustainable forest management prescriptions is almost considered as a normal forest management measure, especially in public forests. Ferris & Pritchard (2000) remark however, that 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**. Risks vary both with the measure under consideration and with the particular Scots pine ecosystem being considered. For example, prescribed burning appears more acceptable in boreal pine forests where it may assist with natural regeneration and lead to an increase in broadleaved occurrence. By contrast, in Mediterranean pinewoods, it may imperil the long-term development of the pine ecosystem.

Canellas et al. (2000) concluded, that, in the light of the lack of definitive information on the effect of different silvicultural treatments on the biodiversity of Scots pine stands and on the even less exact information on their effects on the functioning of the forests, it is necessary to set up experiments specifically aimed at clarifying uncertainties in this interesting and complex area, which has such important economic and ecological implications.

4.1. Promoting old growth habitat

As already stressed above, Mason (2000) concluded that the high conservation value of native pinewoods stands is a function of their old-growth structure that provides a rare habitat. Old growth stands have a number of characteristic structures compared to younger plantations including lower stockings, large quantities of standing and fallen deadwood in various stages of decay, more variable diameter distributions and the presence of large trees (Oliver & Larson, 1990; Ferris & Pritchard, 2000).

Sterling & Humphrey (1999) point out that old growth stands provide continuity of habitat for a range of species of high conservation importance. **Structural diversity, as found in old-growth stands, increases niche availability and is considered as fundamental to insect community diversity** (Young, 1992). Questions arise however about the risk of insect pest outbreaks. To this, Sterling & Humphrey claim that such risks are rare in old-growth, as it supports higher predator populations. Stands retained for an extended period beyond the normal felling age could provide refugia for predator species, from which they can expand as prey numbers increase, thus acting as biological control agents. In this respect Skinner and Whittaker (1981) refer to red wood ants, which may act as key insect predators, without which many herbivorous insects become damaging to forest trees. The presence of old-growth increases the carrying capacity of the habitat for birds and mammals, characteristic of mature forest ecosystems. Factors such as canopy depth, stem density, windblow, wind-snapped trees, shrub and understorey development determine bird populations (Currie & Bamford, 1982). Some saprophytic fungi have highly specialized 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 & Renvall, 1999).

If we accept that provision of old growth structures should be a major objective of future pinewood management, then Mason (2000) arises **the questions how much old growth should there be and how should we manipulate younger stands to achieve desired structures** so that the old growth habitat can be maintained over time? It should be clear that managing the existing old-growth stands in isolation from the other parts of the pinewood resource is doomed to failure in the long-term. Many trees in the old stands are near biological maturity, there has been limited regeneration, and the stems themselves may be vulnerable to catastrophic damage at some point in the future. Therefore **a percentage of planted stands in the stem exclusion and understorey reinitiation phases need to be identified and managed as future old-growth habitats**. These stands need to be of a sufficient scale to provide the desired habitat, i.e. they need to be identified **at the scale of tens or even hundreds of hectares** rather than reserved at a level of a hectare or less. Mason (2000) stresses that the main means of developing the desired features will be through thinning to speed the progress through the understorey reinitiation stage. He recognizes however, that a suitable method and intensity of thinning to promote old growth in pinewoods has not been well defined. Nevertheless he claims that there is little evidence to suggest that variable density thinning in younger stands will hasten the establishment of desired structures and some concern that this may reduce the timber value of the residual stand. A more effective approach may be to maintain standard thinning practice

until the stand is well beyond 100 years of age and starts to show signs of break-up. At that stage thinning should favour clumps of large trees with wide spaces between the clumps which will form the framework of the future old-growth habitat. **Once this transition stage has been reached, then any attempt at systematic timber production would be foregone, but until break-up pine stands should be managed to produce high quality and high value timber.** The other critical aspect of thinning practice in designated old-growth stands is to make sure that dead and dying trees are not removed but are left as biological legacies to initiate the continuity of deadwood habitat.

Pennanen (1996) does not share Mason's vision about the scale of the old-growth stands. Instead of favouring rather big areas, he pleads for a mosaic of patches and groups of old trees. At the same time he recommends that 15% of the forest stands should be permanently set aside from felling. However, Ferris & Pritchard (2000) state that proposals regarding how much of the forested area should be retained in this manner vary.

Anyway, the main general conclusion regarding enhancement of biodiversity in Scots pine stands is, that **foresters have to accept that a small percentage of the pine resource should be managed upon much longer rotations** (UKWAS, 2000).

4.2. Retention of deadwood

The presence of native pinewoods together with old-growth stands proved already significantly the importance of deadwood. Indeed, **dead and dying wood are key components of stand structure and acting as key substrates for many associated species**, particularly when a range of forms of deadwood are present (Harmon et al., 1986; Samuelsson et al., 1994). Deadwood influences biological, physical and chemical processes. It provides a substrate for a wide range of organisms, particularly fungi and invertebrates, decay cavities for nesting sites or shelter for many invertebrates and safe sites for seedling germination or growth of bryophytes. Kaila et al. (1997) have shown that simple and cheap practices, such as **leaving all the dead 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. 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 have also indirect benefits to successful Scots pine 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.

Geudens (see Lust et al., 2000) investigated in Belgium invertebrate populations on Scots pine deadwood on one side and of the exotic tree species black cherry on the other side. One of the objectives was to find out how the exotic species must be controlled, i.e. what is the value of

exotic deadwood in a pine stand? Between April and October 1996, 9,135 arthropod individuals were trapped from 0,73 m³ deadwood of Scots pine and 5,315 from 0,46 m³ deadwood of black cherry. The composition of the xylobiont invertebrate populations was broadly similar for the two tree species:

	Diptera	Coleoptera	Hymenoptera	Araneida	Other
Scots pine	72%	11%	11%	4%	2%
Black cherry	54%	14%	15%	10%	7%

The greater share of the flies (Diptera) in Scots pine wood is due to a very high density of Diptera individuals in the stumps of Scots pine: 1,700 individuals per 100 dm³. Most beetle species were found in Scots pine wood (36 species), more than in black cherry wood (21 species), while most species were found when Scots pine and black cherry were growing in combination (38 species).

This survey provided interesting conclusions for the management of Scots pine forests invaded by the exotic tree species black cherry:

1. The dead wood of the two tree species together contains more beetle species than of a single one of them. This suggests that the ongoing conversion into mixed stands of conifers and hardwoods (pine, oak, birch) will be positive for the diversity of xylobiont beetle species.
2. Concerning the control of black cherry the following can be stated:
 - The liquidation of this species creates high amounts of dead wood in a very short time. In the first place this only stimulates generalist xylobionts. Highly specialized, rare species are unlikely to profit from this on the long term.
 - The total and large scale extraction of all this dead wood as fuel wood on the other hand leads to the periodic disappearance of the habitat "dead hardwood" from this forests. It will take several decades before a considerable amount of deadwood of hardwood species will recur.

Perry (1994) too stresses **snags and fallen logs form the base of a food chain, including microbes, invertebrates, small mammals and birds**. The presence of large, decaying broadleaved trees indicates habitat suitability for hole nesting birds such as woodpeckers (Angelstam & Mikusinski, 1994) as well as other taxa which utilize cavities, e.g. woodland bats (Mayle, 1990).

4.3. Structural complexity

As already stressed above, structural features of the forest, such as the presence of old-growth and the amount and distribution of deadwood, are critical factors for the forest ecosystem biodiversity. In all forest types, structural complexity provides the basis for much of the diversity in species, habitats and processes (Franklin, 1992). Therefore a complex forest structure should be developed. Instead of the simple regular structure of monocultures, **a complex stand structure should be promoted at both the horizontal and the vertical scale.**

Mason (2000) distinguishes between four essential stand development stages with Scots pine in Great Britain: stand initiation stage (until 20 y), stem exclusion stage (20-80 y), understorey reinitiation (80-140 y) and old growth (>140 y). 85% of the Scots pine stands belong to the stem exclusion stage, 9% to the understorey reinitiation stage and only 2% to the old growth stage. Of course such a distribution does not provide a satisfying biodiversity. In this respect Lust et al. (1998, 2000) plead for a **more small-scale forestry**, as an instrument to reach the goal. Also Ferris-Kaan et al. (1998) claim that the vertical structure of Scots pine forests can be poor, with few layers and limited understorey cover of shrubs. To enhance this, they see opportunities 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). But Ferris & Pritchard (2000) add to this, that there has been little research specific to Scots pine which has looked at the effects of silvicultural techniques on biodiversity. Nevertheless, Kerr (1999) underlines that vertical structure can be significantly improved through the **introduction of silvicultural systems, such as group regeneration systems**, which break up uniform canopy cover, and that the application of different systems at a range of scales can improve spatial heterogeneity. Beck (2000) too refers to alternative silvicultural systems developed to increase the degree of naturalness in existing pine plantations. In general these models aim at the same time to decrease costs, produce large and high quality timber, increase the structural diversity, increase the proportion of broadleaved species and of continuous cover forest systems.

Structural diversity is undoubtedly **encouraged by natural regeneration** (Lust & Geudens, 1998; Lust et al., 2000). Natural regeneration leads to slower changes, it tends to produce mixed stands with a more irregular structure, natural genetic variety can be better maintained and a natural distribution of tree species in relation to soil types is favoured. In accordance with Maddelein (1991) Lust et al. (2000) point out that natural regeneration of Scots pine can easily occur in three different ecological situations:

1. regeneration of Scots pine under canopy of Scots pine;
2. regeneration of Scots pine under its own canopy, after removal of soil vegetation and the organic soil profile;
3. regeneration of Scots pine on clear-cut areas.

However Canellas et al. (2000) warn also, that **there is still very little information about the influence of these silvicultural operations on ecosystem diversity**, or about the duration of any effects. In the northern part of the Sistema Central in Spain it was found that, **at the age of 50-60 year, Scots pine stands originating either from natural regeneration or from direct seeding were similar in every respect in the floristic composition of their understorey**. By studying the pinewoods of the Sierra de Guadarrama it was also impossible to demonstrate any floristic differences between the understorey of pinewoods subjected to intensive silviculture with clearcutting followed by regeneration seeding, compared with stands growing in similar ecological conditions where regeneration is promoted by moderate felling in uniform shelterwood systems. **These results are, however, at least partially contradictory** with the findings of Lust & Muys (1998) in Flanders. Here, the presence of herb species was very limited in traditional Scots pine stand. In all, only nine herb species were counted at the study site. Species of the

primary succession, an oak-birch forest, were still common 30 years ago, but have now disappeared. The dominating herb species is presently *Deschampsia flexuosa*, sometimes covering over 80% of the soil. Besides this grass species, only three herb species are present in more than 50% of all plots. Most species are located on particular microhabitats, such as tree stumps, branches and mineral soil brought to the surface by uprooted trees.

The introduction of broadleaves in Scots pine stands is also accepted as a normal measure to enhance Scots pine forest biodiversity. But the share of broadleaves is still under discussion, as it is depending on many factors. 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%. In Flanders, on the contrary, it is pleaded to convert the Scots pine plantations to the original oak-birch forests, wherein the share of Scots pine is limited (Lust et al., 2000).

Kennedy & Southwood (1984) stated already that the **inclusion of broadleaved trees** in Scots pine forests is likely to have **benefits** for biodiversity, **particularly in terms of their associated invertebrate species**. Aspen and birch are known to have particular value for biodiversity, e.g. supporting large numbers of invertebrates in boreal and hemi-boreal forests (Patterson, 1993; Worrel, 1995). The value of birch spp., when mixed into conifer stands, is likely to increase their diversity considerably. One way in which it does this is through soil improvement, allowing more demanding herbaceous woodland plants to colonize. Aspen has a similar reputation to birch spp. as a species with a 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, aspen is also recommended for retention in Scandinavian forests to provide habitat for both insects and woodpeckers (Aulen, 1991).

In order to increase biodiversity some authors plead for an appropriate thinning, mainly for a strong and/or differentiated thinning. **Thinnings should allow that any understorey and middle storey layers should be maintained** (Lust et al., 2000). They must create differential canopy openings, which have positive effects on a range of elements of biodiversity, including vascular plants (Hannerz & Hanell, 1993).

4.4. Niche availability and preservation of associated habitats

At a landscape level, the area occupied by the host tree species, the length of time that species have been present, and the taxonomic relatedness of the host to other trees, all influence invertebrate biodiversity. At a stand level, tree size and above-ground architecture affect the number of niches available. Invertebrate-host interactions at the tree level reflect the precise habitat and nutritional requirements of the potential invertebrate colonisers. However, **the biodiversity value of the tree will be dependent on many ecological factors at the range of spatial and temporal scales**. Evans & Jukes (2000) stress, that, **ultimately, the availability of quite precise resources will determine reproductive success by invertebrates and, thus,**

one of the features that must be considered in assessing the biodiversity of invertebrates in a given forest system is the availability in relation to those resources.

However, the few studies, such as that of Hunter (1977), that have considered species richness in relation to niche availability indicate that both local and regional information on biodiversity can be considered at the micro scale of the niche itself. This is particularly true for those species that have precise habitat requirements that may be represented only within the tree ecosystem itself. Preferences for niche are not a simple function of the total availability of a particular niche on a tree. Surprisingly, there is little information on the quantitative relationships between invertebrate colonisers and niche availability at the tree level. However, some approximation to test the null hypothesis that presence of both diversity and abundance of species is direct function of the relative presence of niches in the tree as a whole can be made.

The direct interactions between invertebrate colonisers, their food plants and other biotic and abiotic factors include elements linked to a wide range of spatial scales. Such invertebrate-host interactions occur at the tree itself, where invertebrates will encounter and use a range of niches or micro-habitats. At a simplistic level, the presence of any invertebrate species depends on its net reproductive rate once it has encountered a potentially suitable niche and has committed itself to feeding and reproduction on the selected host.

Availability of niches is dependent on characteristics of the population of putative host plants. This involves both size and plant structure, so that trees, by virtue of their dominant physical position in the environment, offer the greatest potential for colonisation, retention and support of invertebrates. Such structural complexity is one component among many that determine invertebrate biodiversity at the invertebrate-host interface (Strong et al., 1984).

Size is an important attribute of trees. Long lived old-growth forests will contain bigger trees, both in terms of absolute height and, more significantly, in overall biomass and structural complexity, than younger forests. Vaisanen et al. (1993) found, that the species composition of xylophagous Coleoptera in Finnish forests is significantly different in primaeval compared with managed Scots pine, even though the total numbers of xylophagous species was greater in managed forests. The authors showed that **the main differences were in availability of much larger quantities of deadwood and the larger trunk diameters in the primaeval forests.** They also pointed out, that the larger trees offered more heterogeneous habitats for Coleoptera, particularly those with specialist requirements. Thus the fauna of primaeval forests differed almost completely from those of managed forests, but both carried high biodiversity.

Plant architecture usefully defines both the size of a plant and the availability of niches on that plant (Lawton, 1983). Trees can be divided into many niches potentially capable of supporting invertebrate colonisers with specific feeding or reproductive requirements. The degree of subdivision of the tree will depend on invertebrate requirements rather than on strict physical partitioning of the tree itself. For example, bark feeding insects can be separated into groups on the basis of degradation of bark. Thus, although it is convenient to split the tree into easily described physical niches, there is a considerable degree of complexity associated with each category.

In addition the concept of feeding guilds can also be considered in relation to the diversity of invertebrates that actually colonise the tree. Many authors recognise the major guild categories proposed by Moran and Southwood (1982), in which they categorised chewers, sap-suckers, epiphyte fauna, scavengers, insect predators, other predators, parasitoids, ants and tourists.

Evans & Jukes (2000) present a schematic classification on the major niche provision in a Scots pine tree, including estimated proportionate volume in each niche.

NICHE	COMPONENTS	PROPORTION (%)
Crown foliage	Buds	6
	Needles	
	Cone/seeds	
Branchwood		12
Stem	Bark	10
	Wood	60
Stump and root	Litter and detritus	12
	Soil	

Increasing interest is now also being given to the **conservation of isolated populations which are found under extreme environmental conditions** for the species and which have potential importance in the maintenance of genetic diversity. Discontinuities in the distribution of Scots pine are found at the southern and western extent of its natural range. These areas were refugia during glaciations explaining their high genetic distinctiveness. The most important populations are found in Austria (local provenances adapted to high altitudes in the Central Alps), Finland and Sweden (arctic populations), France (Massif Central, Alps, Pyrenees), Spain (Sierra Nevada), Northern Scotland up to 20 populations of critically small size (<100 trees) (Mason & Alia, 2000; Boratynski, 1991).

The main factors affecting the existence of endangered populations are their small size (Scotland and Spain), **environmental stresses** (Hungary, Spain, by drought), hybridisation with non-native seed sources (France, Haguenu population; Scotland), **hybridisation with other species** (France, Angles population), lack of regeneration through over-grazing (Spain, Scotland) and **human interference** (Hungary). In countries of Central and Northern Europe, endangered populations of Scots pine are not normally considered as a concern.

Evans and Jukes (2000) conclude that **an issue for managers is the need to distinguish between increasing total biodiversity and the encouragement of rare species with particular niche requirements** which may also be dependent upon the continuity of that niche within the landscape. Knowledge of niche availability, especially the quality of the niches may eventually allow choices to be made on a rational ecological basis, particularly at quite local scales.

4.5. Secondary measures such as forest edges, open areas and prescribed burning

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 utilize 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 site 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. 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)

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). Forest fires played a significant role in creating multiple-aged stands with a large number of dead standing and lying trees (Östlund et al., 1997).

Prescribed burning is recognized 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). Granström (1996) argues that a system of prescribed burning is necessary in order to maintain fire structured forests within reserve areas, recognizing that fire-adapted fauna are mobile and able to locate rather small burns (Wikars, 1992). Burning also helps to thin the thick layer of raw humus, which is often a serious obstacle to natural regeneration, by improving the thermal conditions of the site.

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 (Wicklow, 1988). Gorshkov and Bakkal (1996) 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 stabilizing after 90-140 years. Studies of the impact of fires on understorey vegetation of maritime pine forests indicate that species richness and diversity increase after burning (Rego et al., 1991).

5. Risks related to measures to enhance biodiversity

The conservation and enhancement of biodiversity 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. However, 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. Ferris and Pritchard (2000) assessed some risks associated with this range of measures.

The authors consider **browsing as a moderate risk** of enhancing biodiversity. It comes up by increasing tree species diversity and specific management of edges and open areas. Besides there is a low risk for defoliators by invertebrates and for bark beetles and pathogens by retaining deadwood.

Browsing by animals is likely to be influenced by the proportion of broadleaved trees and shrubs present in Scots pine forests. This has e.g. been shown for moose, *Alces alces*, which can considerably alter the expected development of young pine stands. Heikkilä and Härkönen (1996) reported on browsing pressure on young Scots pine stands, where these were growing with an abundance of preferred broadleaved species, e.g. rowan, *Sorbus aucuparia*, *Salix* spp., aspen, *Populus tremula*, and *Betula* spp.

An interesting example was already provided by Murray (1965) concerning the relationship between *Populus tremula* and pine twist rust, *Melampsora pinitorqua*, which affects mainly Scots pine. Despite its ecological benefits, the occurrence and encouragement of **aspen** may present a direct and serious risk, since it **acts as an alternate host to the rust**. The damage, although localized, can be quite serious, e.g. distortion of the stem and multiplication of leaders. It occurs where both species are in close proximity, since the sporidia do not normally travel more than approximately 200 meters. The disease can be partially controlled by cutting aspen suckers early in the summer and by felling aspen trees in the immediate neighbourhood of Scots pine plantations.

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 (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 (Björvall et al., 1997).

Retention of **dying and deadwood can also provide breeding habitat for pest species**. An example is shown by Langström and Hellqvist (1991), viz. the larger pine shoot beetle, *Tomicus piniperda*, a species unique among European bark beetles in causing considerable growth losses to Scots pine. The beetles breed in fresh pine wood such as storm-felled trees and moribund trees. With the potential to cause 20-45% volume growth loss over periods of up to 10 years, such a risk is clearly unacceptable. In Mediterranean Scots pine forests, the risk of insect outbreak as a result of deadwood retention is also considered to be unacceptable.

Besides, the **threat of fire** in these forests also mean that little deadwood is retained (Ferris & Pritchard). Generally, 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°C (Butin, 1995). On most sites, therefore, the restock areas should be left unplanted for at least two years (Weslien and Wennström, 1997).

Failure to observe this delay can lead to high levels of mortality, e.g. 90-100% in replanted Scots pine stands in northern Germany (Hartmann and Schmid-Molholm, 1997).

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 an economic importance (Peltonen et al., 1997). Studies on the effect of forest edges on the distribution and impact of defoliating Lepidoptera support this view (Bellinger et al., 1989).

6. Biodiversity assessment

There is an increasing demand to include biodiversity assessments as an additional management input within silvicultural decision making. However, **none of the existing forest models in use to support forest management planning was explicitly designed to consider the biodiversity of forest ecosystems and how this may change under different silvicultural treatments** (Lexer et al., 2000). Indicators are a prerequisite for considering biodiversity in forest management planning because they allow the impact of stand treatments on biodiversity to be assessed and evaluated (Rennings, 1994). **Sets of indicators for biodiversity must integrate compositional, structural and functional attributes.**

Lexer et al. (2000) give an overview of the attributes of ecosystem composition, related indicators, numerical indices and data required for their calculation.

- The most common compositional method of defining diversity at the ecosystem level is the **species**.
- The simplest species-related diversity measure is **species richness**, defined as the number of species. Species density refers to the number of species per area unit.
- **Species abundance** combines the number of species with the number of individuals.
- **Species diversity** is another important biodiversity indicator. It consists of two fundamental components: a) the number of species in the community (species richness), and b) the relative (proportional) abundance or degree of dominance of individuals among different species. The latter is usually referred to as evenness or equitability and measures the extent to which species are equally represented in a community.

A prominent species diversity index is the Shannon index. Another well known dominance indicator is Simpson's index. Other similar indices of species diversity include Brillouin index, McIntosh index, Berger-Parker index and the sequential comparison index (Magurran, 1988).

Indices of structural diversity quantify the arrangement of structural elements of a system in space.

- A set of measures for **horizontal and vertical distribution** of structural elements is given by Lexer et al.(2000) and Kint et al. (2000). Horizontal indicators require data on the coordinates of each tree, the species and the tree density. The establishment of profiles of vertical structure needs species, height of each individual and leave area per canopy stratum or individual tree.
- **Tree size** differentiation is determined by diameter, height and coordinates of each tree.

- Diversity of age is given by **tree age**.
- **Stand diversity** relates to species diversity, distance differentiation, height differentiation and crown differentiation. It can be characterized by the proportion of species with minimal and maximal abundance, the minimal and maximal distances between neighbouring trees, minimal and maximal height and minimal and maximal crown width.
- **Architectural complexity**, defined as the number of structural categories (e.g. bark, buds,...) into which the vegetation at each site could be divided, reflects types of micro-habitats occupied by invertebrates (Magurran, 1988).
- At the **genetic level** the presence of genetic variants within a given species induces additional phenotypical variability. A commonly used indicator of genetic structure is heterozygosity at genetic marker loci (Nei, 1987). It needs data on alleles and genotypes.
- **Deadwood** is an important component of structural diversity, particularly in old-growth primeval and natural forests (Lähde et al., 1999). It is a resource and habitat for algae, fungi, lichens, mosses, an immense number of invertebrates and even some vertebrate organisms that are cavity specialists. Standing dead trees and fallen logs have to be distinguished ecologically because these two types of deadwood differ in terms of their origin, decaying processes and colonization (Stöcker, 1998). Additionally, dimensional diversity considerably enhances species diversity. Stöcker (1998) argues in favour of standing dead trees with large diameters, because the benefits of such material for invertebrates cannot be replaced by fallen logs. Deadwood is usually quantified on a volume per unit area basis in different decay and diameter categories.

Functional phenomena can be divided into an array of processes such as photosynthesis, transpiration or the flow of energy and matter through a forest ecosystem, **which are mainly driven by the biotic environment** (e.g. sunlight, temperature). In this context silvicultural interventions can be considered as a controlled disturbance to favour certain parts of the population (e.g. individuals, tree species, etc.) or processes (e.g. tree growth) within a forest ecosystem. In addition the functional component of biodiversity includes all forest that affect diversity at the genetic level. These forces may act either at the molecular level or at the population level.

Lexer et al. (2000) conclude their study on models for biodiversity assessment with some statements, a.o.:

- With very few exceptions, prominent parameters for biodiversity assessment such as deadwood and understorey vegetation are currently not included in forest models.
- Mortality from windthrow and other abiotic and biotic agents substantially affects stand structure and composition. Such disturbances are rarely, if ever, included in forest models.
- Patch models have to be extended so that better representations of stand structure and management options are provided (Lindner et al., 1997)
- No current forest model considers attributes of genetic biodiversity. Regarding this genetic component of biodiversity, a first attempt to integrate processes into real-structure forest models is provided by Degen and Scholz (1996).

The main conclusion is that an extension of existing growth models is needed to incorporate biodiversity issues in forest management planning. Probably the most promising approach lies in the **development of the family of distance dependent individual tree growth models** because these explicitly address horizontal and vertical structural diversity of forest stands. A major limitation is the lack of information on genetic diversity.

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