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**Changes in forest landscape structure in southern Finland
in the late 1900's**

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Academic Dissertation

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ABSTRACT

Finnish forest landscapes have experienced major changes during recent decades, and currently they are often regarded as fragmented. The assessment of historical landscapes and the quantitative documentation of supposed changes are matters that have largely remained unexplored, even though a thorough understanding of the historical processes and the existing pattern would be urgently needed for scientifically-based and sound management as the history and current landscape patterns may place long-term constraints on both ecological and economic objectives of future management options. The aim of this research was to define and quantify the changes in forest landscape structure caused by logging and road construction in private and state forests in southern Finland from the 1950's to the 1990's.

Areas varying between 14 000 and 20 000 ha in different parts of southern Finland were selected for study. Both private and state-owned forest were included, but the proportion varied somewhat between locations. The forest cover was analysed using aerial photographs from successive decades, the forests in each area being classified into no-canopy and closed-canopy forest types. A more detailed analysis and classification employing several development stages was performed in one of the areas.

The results show that the continuous cover of mature forests in the 1950's has been broken up by extensive logging, and that major changes in the landscape structure had already taken place before the 1970. The continuous closed-canopy forest still forms the matrix, however, and its connectivity has slightly improved since the 1970's, and mean size of no-canopy patches has slightly decreased. The interior forest areas are currently small, and edge-influenced forests make up a large portion of the landscape. Roads have had a greater impact on forest fragmentation than has harvesting, and increased harvesting can be attributed to road construction to some extent.

Ecological allocation of harvests, the use of alternative harvesting methods, and local restoration of the important structures of natural forests are needed in order to reverse the effects of past management, but even so the current pattern of Finnish forests may be retained as a legacy for decades in the Finnish forests. Forest management and the planning of forest operations should be carried out at the landscape level in order to enhance the maintenance of historical patterns of landscape cover.

Key words: clear-cut, fragmentation; edge effects, landscape change; landscape pattern; logging; roads

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Tomi, Iikka, Hilla and Veikko – you are the dearest.

LIST OF ORIGINAL ARTICLES

- I Löffman, S. and Kouki, J. 2001. Fifty years of landscape transformation in managed forests of southern Finland. *Scandinavian journal of forest research* 16: 44-53.
- II Löffman, S. and Kouki J. 2003. Scale and dynamics of a transforming forest landscape. *Forest ecology and management* 175: 247-252.
- III Löffman, S. 2006. Long term changes in two boreal landscapes with different management histories. Manuscript submitted.
- IV Löffman, S. 2006. Effect of road construction on harvesting intensity and forest fragmentation in southern Finland. Manuscript submitted.

Satu Löffman participated in planning the research, was responsible for conducting data analysis and was the main author in papers I and II.

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1 INTRODUCTION

1.1 Background

As in so many other countries at present, there is a great concern in Finland for maintaining and restoring the biodiversity of human-altered environments. The country is committed under international agreements to sustainable use of its environmental resources (Ministry of Agriculture and Forestry, 2000). Forests cover 86% of its terrestrial area (Finnish Forest Research Institute, 2004), and the assessment and monitoring of forest species and their habitats is an important part of sustainable management. Several national programmes for assessing environmental, cultural and social values for sustainable management have therefore included forests as an important target (e.g. Ministry of Agriculture and Forestry, 1999).

The sustaining of natural biodiversity focuses on species and their habitats, and aims at preserving viable populations of species in their current distribution areas. An important contributor to this branch of science has been landscape ecology, which emphasises the importance of spatial patterns in ecological processes. The emergence of modern landscape ecology was gradual, but the first guidelines were summarised in the agenda of a 1982 workshop of Northern American ecologists (Risser et al., 1984). Since then research has proceeded on many fronts with a large proportion of the studies concentrating on human-induced changes in landscapes and the consequent effects on species. The destruction and fragmentation of natural habitats has put many species at risk of extinction locally or even globally (Turner, 1996, Hill & Caswell, 1999).

Changes in forest landscapes have been recognised in Finland, but documentation and quantification of these changes has been rare. Sigurdson (1999) showed that 55% of old-growth forests in a forested area in Kuhmo had been lost. Some studies have analysed the landscape structure of the Russian forests close to the Finnish border (Burnett et al., 2003). Those areas have similar environmental conditions but a different management history. Comparisons have pointed to a clearly different landscape pattern from that prevailing in Finland, with differences in forest composition, age and patch size distribution (Siitonen et al., 1995, Saarinen et al., 2001, Kalliola et al., 2003) attributable to the intensified forest management practised in Finland since the 1950's, whereas the Russian areas have remained largely uncut during this time. Forest management affects both stand and landscape structure, in terms of tree species, age classes and the availability of dead wood (Esseen et al., 1997, Kouki et al., 2001), and these changes have in turn been related to changes in the abundance of many animal, plant and fungus species (Rassi et al., 2001).

1.2 Historical patterns and premises for forest management in Finland

The Finnish ecosystems are fairly young, having developed after the last glacial period, only about 10 000 years ago. The forests are seemingly homogeneous due to the small number of dominant tree species, largely pine (*Pinus sylvestris*), spruce (*Picea Abies*) and two birch species (*Betula spp.*). The proportions of forests, forested and open wetlands and waterways vary by region. The early inhabitants probably had a minor effect on the forests until the commercial opportunities arising from the 1700's onwards provided the impetus for more systematic logging and tar burning (Östlund, 1995). At the same time the forests

were also widely utilised for slash and burn cultivation (Heikinheimo, 1915). The results of these activities led to the first assessment of the country's timber resources and raised a concern for their future quantity and quality. The first assessments, in the mid-1800's, indicated a deficiency in large timber, and even in small wood in large parts of southern Finland (Berg & Leikola, 1995). Although the methods and intensity of observation are not accurately recorded, the results of these first assessments were supported by the first two national forest inventories, carried out in 1921 - 1924 (NFI1) and 1936 -1938 (NFI2), which showed that the forests had been extensively logged in southern Finland, although some variation existed between sub-regions (Ilvessalo, 1943). The proportion of forests entirely or mostly untouched by man was nevertheless still 51% in the southern parts of the country and 81% in the north.

For a long time the forests were harvested by means of selective cuttings, by extracting all trees above a certain minimum diameter. This regime was swiftly abandoned around 1950, however, as selective cuttings were seen to detract from both the commercial and genetic value of the remaining trees (Leikola, 1986). Instead, the even-age system with clear-cutting and optimised rotation times was adopted. Also, forest management came to be characterised by the regeneration of under-productive stands, punctual thinnings, and effective natural and artificial regeneration combined with machine tilling. This management was carried out by opting for the so-called "normal forest scheme", in which each age class from clear-cut areas to mature stands would be equally represented. The effect of this management is clearly seen by examining the NFI6 and NFI9 age distributions (Figure 1), in which there is a shift towards younger age classes and a more even distribution of age classes relative to the NFI1 and NFI3 assessments. The statistics also show a clear decrease in the percentage of old forests.

Legislatively, the Finnish government has guided forest management in the direction of intensified timber production since the enactment of the first Forest Improvement Law in 1928, and this intensive forest management has been promoted with many national campaigns and programmes from the 1950's onwards and with direct subsidies to forest owners, the aim being to achieve an increase in the allowable cut (Juurola et al., 1999). The investments subsidised have included road construction and drainage, for example.

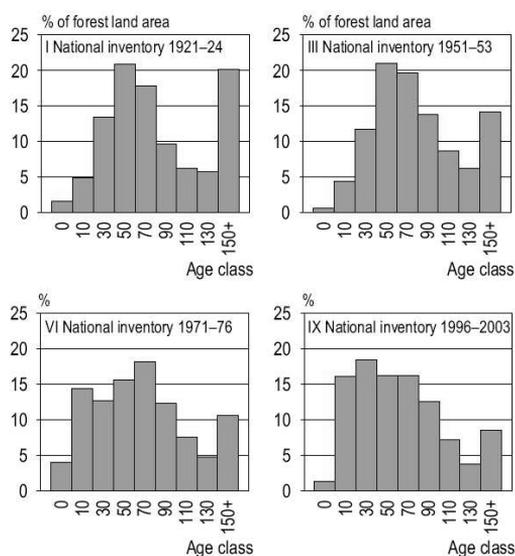


Figure 1. Trends in age class distribution in the Finnish forests, according to National Forest Inventories from the 1920's to the 1990's. Source: Finnish Forest Research Institute.

Historically, forest management has been largely based on the contemporary perception of natural forest dynamics. The site classification in terms of forest productivity, for example, originates from the early work of Cajander (1926) and describes the sites by reference to typical species in the understorey vegetation. The target of using the natural succession as a means of intensifying timber production may nevertheless have hampered the objective assessment of what the natural dynamics and natural characteristics of the forests concerned actually are (e.g. Lähde et al., 1991). It is only recently that the natural dynamics of boreal forests have been described with scientific validity. Recent research has strengthened the view that landscapes cannot be managed with reference to a fixed equilibrium, but rather it is essential to incorporate their natural variability into management (Kuuluvainen, 2002a, Wiens et al., 2002)

Disturbances in natural forests, such as forest fires, high winds, floods, ice, snow, herbivores and burrowing animals, may cause fragmentation and other modifications to the forest landscape structure (Esseen et al., 1997, Nilsson & Ericson, 1997). Natural disturbances, occurring on different scales, contribute to the biodiversity of forest ecosystems, whereas scale-limited human disturbances tend to simplify the forest and forest landscape structure (Kuuluvainen, 2002b). Changes in biotope composition and pattern are brought about in many ways by intensive forestry. The fragmentation of old-growth and mature forests is due to the shorter rotation period used in managed forests and the economically optimal rectangular age distribution, by comparison with a natural succession and disturbance regime (Utterer et al., 1996, Kurki et al., 1998). In addition, intensive management in Fennoscandia has favoured even-aged coniferous stands, where as multilayer stands, deciduous stands and mixed deciduous stands have become rare and isolated, or have even disappeared (Utterer et al., 1996, Östlund et al., 1997). Forest fires have been pointed in particular as supporting many structural properties of individual stands and landscapes (Angelstam, 1997, Wallenius et al., 2002). Natural fires are considered important for maintaining a diverse age structure between and within individual stands and for regulating the species composition of forests. The information collected on natural forest dynamics has been further integrated into simulation models, which have been especially valuable for developing landscape-level disturbance and succession models (Pennanen & Kuuluvainen, 2002, Pennanen et al., 2004).

1.3 Spatial and temporal analysis of landscapes

Landscape patterns can be either analysed in simulation studies, when subjected to different disturbance rules, or by describing real landscapes (Andrén 1997). Simulation studies provide information on the properties of given indices and landscape metrics (Turner et al., 1989, Gustafson & Parker, 1992, Hargis et al., 1998), and have also led to some important hypotheses and results in attempts to link spatial patterns with ecological processes (With & King, 1997). Competent simulation models can only be built on a knowledge of real-world phenomena, however, and thus the examination of real landscapes and landscape dynamics cannot be replaced (McGarigal & Cushman, 2002).

The real world data are usually based on remote sensing material that has been appropriately classified. Both aerial photographs and satellite images have been widely used to produce spatial data for temporal analysis (e.g. Dunn et al., 1991, Hall et al., 1991, Kadmon & Harari-Kremer, 1999, Korpela, 2006). The use of aerial photographs is limited by their temporal coverage, which may lead to possible loss of some pivotal change in the

landscape structure. Also they may be too limited in quality to allow certain cover types to be distinguished. Satellite images provide repeated coverage of the same area with fairly high temporal resolution, but their potential for tracking historical changes is limited to the early 1970's and thereafter. Other types of data include forest inventories, (old) forest maps and management plans and other historical records. Axelsson and Östlund (1997), for example, used old forest surveys, forest maps, records of fire history and other historical information to compare a forest landscape structure in 1914 and 1982.

Landscapes are commonly viewed as patch mosaics of cover types and analysed using a set of landscape indices, many of which were developed in the 1980's and have their roots in information theory and fractal geometry. O'Neill et al. (1988) introduced three indices of landscape pattern, namely dominance, contagion, and fractal dimension. Turner (1990) developed a spatial analysis program called SPAN that added several new indices to the analysis. Later, a diverse group of indices were integrated into a program called FRAGSTATS (McGarigal & Marks, 1995, McGarigal et al., 2002), while another software package offering a wide suite of indices is the *r.le* programs embedded in the GRASS environment (Baker & Cai, 1992). The development of computational methods has been one of the major achievements in landscape ecology.

Landscape indices can be divided into three groups (McGarigal, 2002): patch-level metrics, which are measures of individual patches and their spatial character or context, class-level metrics, which are measures of one patch type (class) existing in the landscape and can either be derived by averaging or calculated separately, so that they reflect an aggregate property of the patches in a particular cover type, while landscape-level metrics are measures extending over the whole of a landscape and are similarly derived by averaging or calculated using all the patches in all classes. The indices can be further divided into those describing landscape composition, e.g. the proportions and numbers of different cover types, or configuration, e.g. patch size distribution, patch shape and the interspersed of patches. (Haines-Young & Chopping, 1996, McGarigal, 2002).

Despite the wide use of different indices, they have a number of shortcomings, too. There is no single index that can adequately describe a particular landscape, and usually a suitable combination must be selected (eg. Hulshoff, 1995). Many of the indices fail to uniquely describe the spatial characteristics of a landscape (Baskent & Jordan, 1995), and measurements are also sensitive to the grain (smallest observable unit or patch size), extent (total area of the landscape) and classification (number of discrete classes, for example) used in the analysis (Turner et al., 1989, Wu, 2004, Garcia-Gigorro & Saura, 2005). Furthermore, many of the indices are correlated (Riitters et al., 1995, Hargis et al., 1998), and it may not be easy to relate their values to ecological processes (Tischendorf, 2001). Thus, the values are not absolute measures of landscape properties and are rarely useful by themselves, although they serve better for comparing alternative landscape patterns (Gustafson, 1998a).

Hanski (2005) distinguishes three ecological components of landscape change: 1) changes in the amount of a habitat, 2) changes in the configuration of a habitat, and 3) changes in the quality of a habitat. This classification is neutral in the sense that it does not speak in terms of decreases or increases, improvements or deteriorations. Habitat loss for some species is habitat creation for others. The spatial description of landscapes has been most effective in describing the first two components of habitat alteration, although the spatial component of habitat quality has also been recognised (Uutera, 1998, Hessburg et al., 2000, Kouki et al., 2001).

2 OBJECTIVES OF THE RESEARCH

The word fragmented is commonly used of the Finnish forest landscape, because of its mosaic appearance, with managed stands of different ages. The assessment of the historical landscape-level changes is a matter that has largely been ignored, however, in spite of the fact that a thorough understanding of the existing pattern and how it was created is needed for effective decision-making, since this may place long-term constraints on both the ecological and economic objectives of future management (Franklin & Forman, 1987, Gustafson & Crow, 1998, Baskent, 1999).

Since the advent of landscape ecology, numerous studies in different parts of the world have been carried out in order to describe landscape patterns and their past changes, but these are not directly applicable to Finnish conditions. Finnish forestry has its own unique history, and the changes to be expected should reflect the prevailing management combined with the natural conditions, both abiotic and biotic. It is reasonable to assume that the unique history will have some influence on the temporal and spatial patterns and processes of landscape changes in Finland.

This research aims at providing a multifaceted view of the changes affecting Finnish forests and their causes. More precisely, the objective was to define and quantify the changes in forest landscape structure that took place in the private and state forests of southern Finland from the 1950's to the 1990's, the period of intensive forest management.

The individual papers concentrate on specific issues concerning various aspects of landscape change and related phenomena. The objectives of the four studies were:

- I to define and quantify temporal changes in the Finnish forest landscape relative to a general model of habitat transformation,
- II to assess the effect of observation scale on the detectability of temporal changes in landscape structure,
- III to assess changes in forest composition as represented by different edge areas and the interior forest, and
- IV to assess the effect of roads on landscape fragmentation.

Private and state forests were analysed separately in order to discover any differences that might be caused by the different management regimes pursued by the two forest owner groups. Some effect of ownership on a landscape pattern may be expected because the average size of private forest holdings in Finland differs considerably from that of state-owned forests. Consequently, forest operations may have been applied to areas that differ substantially in size.

3 MATERIAL AND METHODS

The selection of areas for analysis was based on the idea of finding forest landscapes that were representative of southern Finland. The original goal was to find pairwise continuous areas of ca. 10 000 km² on both private and state land, but the proportion of state-owned forests is very low and they are mostly scattered around in small fragments, especially in the southern parts of the country. Thus the selection of areas was limited to places where sufficient continuous state land was to be found. Finally forests in Äänekoski, Evo, Lohikoski, Nurmes, and Reisjärvi were selected, although the proportion of state and private ownership still varied between the locations (Table 2, I).

National Land Survey (NLS) in Finland possesses large archives of aerial photographs taken from the beginning of the 20th century onwards, thus providing repeated coverage for most parts of the country. The idea here was to make use of these archives to analyse temporal changes in forest landscape structure. Eventually it was decided to search for photographs from the 1940's and 1990's and from an additional period in between these two in order to be able to analyse the timing and intensity of the changes more precisely. A compromise was also needed regarding the dates of the desired photographs, however, especially in the earliest period, the 1940's, which was finally represented by photographs taken between 1936 and 1954. The second period, the 1970's, was represented by photographs taken between 1969 and 1974, and the third period by ones taken between 1991 and 1997. They were all black and white photographs, on original scales that varied between 1:20 000 (early years) and 1: 60 000 (later years).

The digital aerial photographs were rectified and geocoded to the Finnish basic coordinate system, and resampled to a ground resolution of 5 m. Two datasets were created using the original images:

Dataset 1 The data were automatically classified into no-canopy and closed-canopy areas, where the no-canopy class included clear-cut, seedling and sapling stands and natural openings such as treeless mires and rocks, while the closed-canopy class covered the young, middle-aged, mature and old forest development stages. Lakes, agricultural land and roads were added to the classified images for the third period from the NLS Topographic Database, and digitised manually from the computer screen for the earlier periods. The processing of the dataset is described more precisely in I and the dataset is used in I, partly in II and in III.

Dataset 2 A more detailed classification, including separate classes for mature stands, young stands, saplings, clear-cut stands and open mires was digitised manually from the computer screen for the Nurmes area. This dataset is used in III.

In the first paper (I) the degree of forest transformation was analysed using the patch density, mean patch size, largest patch index, edge density and mean nearest neighbour in the closed-canopy class, while in II the effect of scale was analysed using randomly sampled squares of different sizes. The scale effect was analysed by calculating the averages for the largest patch size, mean patch size, area-weighted mean shape index and edge density for each square size sampled in the no-canopy class. In III the forest landscape structure was analysed using the proportions of the classes, number of patches, median patch size, mean patch size, largest patch size, border length and mean shape index. The

edge effect was simulated with varying edge widths and the number of multiple edges (overlapping edges), assuming a 100 m edge width, was analysed. The probability of incident cuttings was analysed in IV using smoothing splines and the effect of road construction and harvesting on landscape isolation using a landscape division index (Jaeger, 2000). The equations used for the calculation of landscape metrics are summarised in Table 1.

CLASS METRICS	EQUATION	UNITS	RANGE
Number of patches	$NP = n_i$	-	$NP \geq 0$
Class proportion	$\% LAND = \frac{\sum_{j=1}^i a_{ij}}{A}$	Percent (%)	$0 \leq LPI \leq 100$
Patch density	$PD = \frac{n_i}{A}$	1 per hectare (1/ha)	$PD \geq 0$
Mean patch size	$MPS = \frac{\sum_{j=1}^{n_i} a_{ij}}{n_i}$	Hectares (ha)	$MPS \geq 0$
Largest patch size	$LPI = \max_{j=1}^{n_i} a_{ij}$	Hectares (ha)	
Largest patch index	$LPI = \max_{j=1}^{n_i} \frac{a_{ij}}{A} \times 100$	Percent (%)	$0 \leq LPI \leq 100$
Mean shape index	$MSI = \frac{\sum_{j=1}^{n_i} \frac{0.25 p_{ij}}{\sqrt{a_{ij}}}}{n_i}$	-	$MSI \geq 1$
Area weighted mean shape index	$AWMSI = \frac{\sum_{j=1}^{n_i} \frac{0.25 p_{ij} a_{ij}}{\sqrt{a_{ij}} \sum_{j=1}^{n_i} a_{ij}}}{\sum_{j=1}^{n_i} a_{ij}}$	-	$AWMSI \geq 1$

Table 1. Summary of the class metrics used in the analysis of landscape structure.

Edge length (Border length)	$EL = \sum_{j=1}^{n_i} p_{ij}$	Meters (m)	$BL \geq 0$
Edge density (Border density)	$ED = \frac{\sum_{j=1}^{n_i} p_{ij}}{A}$	Meters per hectare (m/ha)	$ED \geq 0$
Mean nearest neighbour distance	$MNN = \frac{\sum_{j=1}^{n_i} h_{ij}}{n_i}$	Meters (m)	$MNN > 0$
Landscape division index	$DIVI = 1 - \sum_{i=1}^{n_i} \left(\frac{a_{ij}}{A} \right)^2$	-	$0 < DIVI < 1$

i = type of patch (class i)

j = a patch in class i

a_{ij} = area of patch j in class i

p_{ij} = perimeter of patch j in class i

h_{ij} = edge to edge distance from patch j in class i to the nearest patch also in class i

All metrics adapted from McGarigal & Marks (1995) except for DIVI from Jaeger (2000)

Table 1. Summary of the class metrics used in the analysis of landscape structure (continued).

4 RESULTS AND DISCUSSION

4.1 Spatial and temporal changes in landscape structure

General temporal changes in the areas were assessed using a set of landscape indices. All four papers (I, II, III, IV) testify to major changes in landscape structure that had already taken place by the 1970's. This is demonstrated by the loss of continuous forest cover (I), increase in the area of open patches and younger development stages (II, III), loss of interior mature forest (III), increased length of patch borders (I, II and III), and increased length of the road network (IV). The loss of closed-canopy forests did not continue at same rate between 1970's and 1990's, instead the regrowth of the earlier harvested areas reversed the forest transformation process in most of the landscapes studied (I), assisted by the afforestation of agricultural areas and improved forest growth on drained mires. Due to continuous harvesting, however, the age structure in the class of closed-canopy forest must have changed from older to younger, as shown in III, where the proportion of mature forest is shown to have continued to decrease throughout the period studied, so that the initial figure of over 90% diminished to less than 20%, whereas there was a large increase in the proportion of young stands, from practically 0 to 40–50%.

The results are in accordance with the historical national forest inventory data and reflect actual forest management operations within the time frame. Forest transformation process has been fairly similar in both state and private forests. In all the studies (I, II, III, IV), the landscape pattern of the first period was clearly different from that in the more recent landscapes. The loss of closed-canopy forest did not continue from the second period to the third period, but instead the fragmentation was reversed in many areas, although this effect was not very pronounced. Part of the restoration of the closed-canopy forest cover was based on the afforestation of mires after drainage. The counter effect to this, however, is that it may have also accelerated forest harvesting by inducing a potential increase in the future maximum allowable cut according to which forest harvesting was planned (Kuusela, 1956).

The scale of the landscape pattern was examined only in the private forest landscape in Nurmes (II), where a square kilometre was able to capture the structure brought about by no-canopy holes in the landscape, while the structure caused by clear-cutting was mostly captured by squares of 3 km × 3 km. The landscapes of the 1970's and 1990's were very similar, but there was more small-scale variation in the former. Thus, whereas the landscape of the 1990's had a more regular pattern of small, dispersed clear-cut patches, considerable variation was found in the 1970's. The area-weighted mean shape index of no-cover patches was clearly higher in the harvested landscapes of the 1970's and 1990's than in the original landscape.

The connectivity of the closed-canopy forest was slightly improved from the 1970's situation, since there were less large clear-cut areas in the 1990's. Instead, there were more small, dispersed clear-cut patches that created very fine-grained landscape structures. As forest harvesting continued, however, the age structure in the closed-canopy forest moved from mature forests to younger development classes. The increase in the length of borders between stand development classes continued in both private and state forests, and the larger proportion of edge-influenced areas in private forests in the Nurmes area can be attributed to the larger number of smaller clear-cut areas, which are probably more

dispersed in the landscape. In the Nurmes area, a 100 m edge on both sides of all forest patch borders produced a proportion of edge areas amounting to less than 30% at the beginning (III), which increased to 53% in state forests and 66% in private forests by the 1970's and was 61% in the state forests and 76% in the private forests in the 1990's. The proportion of multiple edges also increased in the landscapes, and the patches became so small that the edge effect was a combination of several neighbouring or nearby patches.

In addition to harvesting, the construction of a dense road network contributed to forest fragmentation (I, IV), dividing the forest areas into smaller patches. Also, the harvesting intensity was higher near roads (IV), the increased harvesting probability extending at least 200 to 300 metres from a newly built road. The effect of roads was especially clear in the 1990's landscapes, although detectable in the 1970's as well.

Forest management influenced many structural features of the landscape, including mean patch size, mean shape, edge density and forest interior. Other authors have associated the abundance of small patches, higher edge densities and a decrease in interior area with changes caused by forest management (Mladenoff et al., 1993, Spies et al., 1994, Reed et al., 1995, Reed et al., 1996, McGarigal et al., 2001). The naturally open mire patches in the present original landscape were mostly small, simple-shaped holes in the continuous closed-canopy forest, whereas the cut areas created large, complicated shapes. The actual managed stands were probably more simple in shape, but as they were connected to each other the shapes became more complex.

The fragmenting effect of roads was greater than that of forest harvesting in most landscapes, especially in the third period, the 1990's. This result is consistent with those of other fragmentation studies (Reed et al., 1996, Tinker et al., 1997, McGarigal et al., 2001). The current road densities are not exceptionally high compared with other forested areas, 25.2 m/ha in the Rocky Mountains (Reed et al., 1996) and 12 m/ha in the Northern Great Lakes Region (Saunders et al., 2002), nor is the rate of road construction exceptional, as a three-fold increment in road density was reported in the San Juan Mountains, Colorado (McGarigal et al., 2001) accounting for most of the changes in mean patch size and edge density. A dense network of forest roads is a major contributor to edge effects, but it also causes direct habitat loss. The road density in Finland is currently so high that additional roads would only have a minor effect on landscape isolation.

The change in forest composition is similar to that observed in other Fennoscandian forests. In Sweden, older forests dominated at the beginning of the century, with 83% of the forests classified as old-growth ones in the 1910 inventory but only 3% in that of the 1980's (Östlund et al., 1997). During the pre-industrial period the overall structure of forests was only marginally affected compared with the widespread use of clear-cutting that emerged in the 1940's (Esseen et al., 1997). On Russian side of the Finnish border, the forest landscape remained largely untouched during this period and exploitation has only just begun now, although it is progressing rapidly. Dramatic changes are expected, and the 90% cover of old forests could disappear almost completely in 80 years, with only small blocks of old forest remaining within mires and protected reserves (Burnett et al., 2003).

4.2 State versus private forests

Although the changes in state and privately owned forests followed the same general pattern, the private forests were originally more fragmented by agricultural areas (I). More large clear-cut areas were generally to be found on state land (I, III), but the largest single clear-cut patches on private land in the Nurmes area, for example, were 231 ha in the 1970's and 143 ha in the 1990's, compared with 70 ha and 90 ha, respectively, on state land (Table 1, III).

The larger number of smaller clear-cut stands in the private forests of the Nurmes area resulted in more edge areas than in the state forest (Table 3, III). In the first period there was a greater edge area in the state forests, but these were mostly edges between two natural land cover types, forest and mire, and not the outcome of logging. The area of multiple edges also increased throughout the period studied, the proportion always being higher in the private forests. Assuming a 100 m edge effect, the proportion of edge area was 63% (ca. 8% multiple edges) in the private forests and 53% (ca. 6% multiple edges) in the state forests in the 1970's and 76% (ca. 30% multiple edges) in the private forests and 61% (ca. 17% multiple edges) in the state forests in the 1990's. The relative increment in multiple edge area from the 1970's was substantial compared with the total edge area. Such a change prevailed towards the end of the period studied.

The average road density was higher on private land in all periods, although the range of density values was larger, too (Figure 2, IV). The relative standard deviation in road density decreased from period 1 to period 3 on both the state and private land. The increase in road density in the state forests was fairly consistently from less than ca. 3 m/ha at the beginning to ca. 6 m/ha in the 1970's and about 12–15 m/ha in the 1990's. Only the road densities in the Evo state forest landscape were distinctly higher in the 1970's (14 m/ha) and in the 1990's (20 m/ha). On private land the road density increased from 1–7 m/ha in the 1940's to 3–15 m/ha in the 1970's and 5–25 m/ha in the 1990's, and the differences in road density between individual landscapes at the end of the period studied were many times greater than in the initial situation. In three landscapes on state land, and 4 landscapes on private land the harvested patches appeared significantly closer to new roads both in the 1970's and in the 1990's. The Reisjärvi area had quite a different pattern, however, as the harvested areas were significantly further away than the unharvested areas. The effect of new roads on harvesting intensity was more prominent on state land than on private land, especially in the 1990's. The temporal changes viewed pairwise, state and private, for each of the five areas showed that the effects of roads and open areas on landscape division were very similar in each pair. This is probably due to environmental conditions or similarities in management between state and private lands in the same locality. The effect of road construction compared with forest harvesting varied between landscapes and periods. In the 1940's the contribution of roads and harvested areas varied between individual landscapes, but in the 1970's and 1990's landscapes the contribution of roads was dominant. The Reisjärvi area, where open mires are a natural dividing agent, was a clear exception to the general pattern.

The spatial and temporal changes detected in the state and private forest areas were very similar (I, III, IV), which reflects the prevailing view of forest management within the given time frame, as the same management principles were applied throughout the country both in state and private forests. The most important feature was that the forests were regenerated almost exclusively by clear-cutting. The resulting change was already clearly in

evidence by the 1970's, although the results suggest that the changes may have been more rapid and even more intense on state land (III, IV).

Crow et al. (1999) noted that in Wisconsin, United States, that landscape complexity was connected with the physical environment, so that comparisons between public and private ownership could not be handled independently of that factor. They also acknowledged that most marginal lands could be in state ownership, because they fail to support farming. In Finland, too, state lands tend to be in areas of lower productivity, or in remote areas that are difficult to reach, which probably explains why the changes have been more drastic in the state forests. These were not used until systematic cutting started in the 1950's, whereas the private forests had been used previously, because they were more accessible and closer to settlements. The early national inventories also report that most of the timber was available in areas where the forests were in state ownership (Ilvessalo, 1943).

The remote location of the selected areas may have biased the sample of private landscapes, since in other areas the proportions of agriculture and waterways may be more prominent, so that the forests are more fragmented. Although the effect of other land use types was minimised here by selecting areas that were predominately forest land, a slightly higher percentage of agricultural area was still found on the private lands. The general change pattern concerning forest cutting is still expected to be valid in other areas, too, but the effect of harvesting on forest fragmentation may have been even stronger in landscapes where the proportion of the forest cover had originally been lower, as indicated by the Reisjärvi area. The fragmentation of forests in intensely agricultural areas, e.g. in the south-western parts of Finland, may nevertheless have special characteristics that it would be of interest to study further, mainly on private lands.

Wear and Flamm (1993) found that infrastructural variables (distance from a road, distance from product markets) and environmental variables (elevation and slope) generally had a more significant influence on the harvesting decision of private forest owners than in the case of public lands. This is slightly in contradiction with the present results, since the effect of a new road tended to increase logging, especially in state forests, where management covers larger areas and the construction of new roads can be directed to areas where timber harvesting is planned in the near future. In private areas a new road may cross forest areas that belong to several owners with different goals for management or schedules for logging.

It is quite possible for private forest owners to have different interests with regard to how they manage their forests, and this may be a source of landscape heterogeneity. Uuttera et al. (1998) found a more versatile within-stand forest structure in private forests than in state forests regarding both the number of tree storeys and tree species, and the present results similarly show that private forests may contain more heterogeneous landscape structures. The size of managed patches was generally smaller in private forests, although in the Nurmes area, for example, both the largest clear-cut area and the largest mature forest patch were found on private land.

The intensity of road construction varied much more between individual private landscapes than between individual state landscapes. This may be due to special characteristics reflecting the economic or social function of forestry on a regional scale and how it relates to the intensity of forestry in a particular area. These regional characteristics probably have a historical influence, since the differences between the areas persisted and even increased through the period studied here. Conversely, road construction advanced in a similar manner in the individual state landscapes, indicating that similar management

strategies were adopted by the state regardless of geographical or other factors. The higher road densities found on private lands could be attributed to higher housing densities rather than forestry operations (see Hawbaker et al., 2005). Many of the rural roads may also lead to summer cottages, and may thus have scenic values that actually reduce the probability of harvesting in their vicinity. The importance of such roads for landscape ecology should nevertheless be assessed in connection with the phenomena under investigation (Hawbaker & Radeloff, 2004).

4.3 Implications for species

Most forest-dwelling species are likely to be adapted to the natural disturbances that maintain the vertical heterogeneity of stands and spatial variability of the landscape, but disruptions in stand and landscape characteristics by agriculture or forest management may be deleterious to these species. The loss of large natural patches has been pointed out as a serious defect in forested landscapes (Mladenoff et al., 1993, Crow et al., 1999). In Finnish landscapes closed-canopy forests still form the matrix and probably provide enough connectivity for many forest-dwelling species. A closed-canopy forest does not imply a natural forest, however, and thus the loss of mature natural forests and their natural structural properties may be more important than can be deduced from the present findings or those of other studies using a broad classification of forest cover types.

The proportion of mature forests has declined markedly, and the remaining old or mature forest patches may not serve populations in the best possible ways, as they are small, with an insufficient interior area. The quality of habitats for a particular species may change as a result of modifications in the location of patch borders, as the edge effect causes conditions near a patch border to differ from those in the interior parts of the patch (Ries et al., 2004). For forest interior species, the decline in population size can be greater than could be expected in the grounds of habitat loss alone, because of simultaneous fragmentation, i.e. the proportion of interior forest habitats is lower in a fragmented landscape (Bender et al., 1998). Also, poor-quality habitats near patch borders may increase the risk of mortality (Wiegand et al., 2005). A great abundance of multiple edges means that the probability of management operations occurring in the vicinity of a particular patch increases, and these operations may alter the habitat conditions of the nearby patches through edge effects. Furthermore, the probability of an edge effect will be increased because the actual managed stands will be smaller in area than the patches aggregated by development classes. Multiple edges may also increase the extent and distance of edge influence (Fletcher, 2005).

Critical thresholds for habitat proportions and connectivity may exist that regulate the persistence of given species in the landscape (Bascompte & Sole, 1996, Fahrig, 2002) but such thresholds can be difficult to define reliably in real landscapes (Huggett, 2005) and the task is further complicated by interacting variables. For some species the response times can be so long that it is difficult to relate the changes in species abundance to the changes in landscape structure (Gu et al., 2002, Hanski & Ovaskainen, 2002, Helm et al., 2006). From the management perspective, the recognition of a species threshold for a particular habitat type can help the realistic planning of restoration efforts, since the changes that have persisted in the landscape for a long time may seriously reduce the odds on the success of restoration operations (Schrott et al., 2005). Historical information on landscape changes can also be used to estimate the response of a particular species to future landscape changes

(Schrott et al., 2005), and to evaluate whether the actions planned are likely to increase or reduce the vitality of a given species in the future.

The current road density emphasises the importance of assessing the ecological value of the remaining roadless areas. Roads tend to be permanent features that can act as barriers to species movement and cause population isolation (Oxley et al., 1974, Mader, 1984, Swihart & Slade, 1984, deMaynadier & Hunter, 2000, Marsh et al., 2005), and they also create abiotic and biotic edge effects. Abiotic changes in water and nutrient flows and microclimate can affect the species composition along roadsides (Tikka et al., 2001, Koivula, 2005), increase predation (Bergin et al., 2000), facilitate invasion by exotic species (Buckley et al., 2003, Gelbard & Belnap, 2003, Pauchard & Alaback, 2004), and affect the occurrence of pathogens (Laine & Hanski, 2006). These changes may be further accelerated and expanded by increased human activities promoting forest harvesting and other forms of forest use in the area. All roads, even minor forest roads, have a negative impact on an area's wilderness value.

There has been a lot of discussion on the effect of habitat loss per se versus habitat fragmentation on population survival, and it seems that fragmentation is less critical in this respect (Harrison & Bruna, 1999, Fahrig, 2003). Only recently, along with the knowledge gained of the natural dynamics of forest landscapes, it has been possible to add the habitat quality component, which may refer to changes in the tree layer or the understorey vegetation and may be due to anthropogenic or natural disturbances, or to the natural succession. The quality component can include changes in the amount of dead wood available, suitable nesting holes, or the tree species composition of a patch. The flying squirrel (*Pteromys volans*), which is among the most intensively studied species in Finland, nests in mature spruce forests with an admixture of deciduous tree species (Selonen et al., 2001, Reunanen et al., 2002). Since it seems to be able to move around even in highly fragmented landscapes (Hanski et al., 2000), the sharp decline of its populations is mainly attributed to the loss of mature spruce forests and suitable nesting sites in managed forests. In the light of current information it seems that the decline will continue, because the forests generated in the 1960's are being managed in such a way that the tree species composition will never meet the requirements of the flying squirrel (Hanski, 2006). The absence of suitable nesting holes is another factor detracting from habitat quality in the case of this species.

Another species known to suffer from the loss of mature forests is the capercaillie (*Tetrao urogallos*) (Pakkala et al., 2003). Although this is among the best-studied species in the boreal and temperate forests, it is unclear to what degree its decline can be attributed to the disappearance of natural old forests, the reduction in the distribution and abundance of the bilberry (*Vaccinium myrtillus*), which is its main summer forage (Selås, 2000), or to the rise in the numbers of generalist predators that are better adapted to the altered structure and composition of the landscape (Kurki et al., 2000).

It is thus obvious that the effects of landscape transformation on species in general may occur because of spatial changes in the amounts and configurations of suitable habitat patches. The documentation of landscape change effects is nevertheless often complicated by simultaneous changes in the quality of the remaining patches. It is clear that both landscape characteristics and local stand characteristics must be considered when evaluating the effects of forest management on individual species (Mönkkönen & Reunanen, 1999, Kouki et al., 2001).

4.4 Implications for forest management

The effect of timber harvesting patterns on landscape structure have been examined in simulation studies either by changing the harvesting strategy (Franklin & Forman, 1987, Li et al., 1993, Gustafson, 1998b, Gustafson & Crow, 1998) or optimising it for some type of habitat or a particular species (Öhman & Eriksson, 1998, Kurttila et al., 2002, Kurttila & Pukkala, 2003, Öhman & Lämås, 2005). Different strategies of allocating harvest have a strong regulative effect on landscape structure, although patterns of past management are difficult and tardy to change.

It has been suggested that finding critical thresholds would be most helpful for forest managers in order to retain adequate amounts of suitable habitats for selected species. The use of critical thresholds in planning to protect selected species has also been criticised since, as pointed out, populations may have already declined above the detected thresholds. In addition, efficient management requires identifying which species are most vulnerable to habitat loss and what are their minimum requirements.

It has been proposed (Wickham, 1999) that areas where the proportion of largest patches relative to the proportion of anthropogenic cover is low could be prioritised for the re-introduction of forest, in order to maximise improvements in forest connectivity. Analogously, this would mean searching for areas in a forested environment where the largest forest patch is small compared with the logged area and concentrating restoration efforts there. This seems questionable, however, since the survival of species should in the first instance be ensured in the places where they already exist, since in areas where their populations have deteriorated badly even rapid interventions may not be able to save them.

Since the pattern created by forest cutting persists in a landscape for a long time, it may be easier to promote biodiversity within individual stands by increasing their naturalness. This would mean leaving enough dead wood, for example, or increasing its amount artificially. The vertical structure of the forest stands would benefit from letting the understorey develop naturally and not removing naturally regenerated deciduous species (Lähde et al., 1999). In addition, longer rotation times can be considered. Despite the effectiveness of stand-level management for biodiversity, landscape-level planning and the optimisation of alternative forest use options are badly needed and deserve much more attention than they have received so far. Besides traditional schemes of clear-cutting other alternatives should be actively researched including uneven-aged management to maintain continuous canopy cover. Harvesting in larger blocks (Lohmander & Helles, 1987) or harvesting progressively or in clusters (Franklin & Forman, 1987) would help to minimise the area of edge influence and maximise interior forest area. In some areas mimicking gap disturbance by group selection might be appropriate.

Finally, the necessity of such a dense forest road network should also be critically reassessed. Reducing road densities would be the best way to reduce their environmental impacts. The current management style and efficient wood procurement rely on a well maintained road network. Extending rotation times and concentrating harvesting activities in time and space could reduce the need for permanent roads as they could be replaced by temporary ones (Lindenmayer & Franklin, 2002).

5 CONCLUSIONS AND FUTURE RESEARCH NEEDS

Landscape change mirrors many aspects of our past and current society, and there is no doubt that humans have altered the landscape in many ways and will continue to do so. It would be an exaggeration to say that the situation in the 1940's represented pure, natural forests, because various sources indicate that the forests had been used for domestic and industrial purposes for centuries prior to that time, but it could reasonably be argued that the Fennoscandian forests were not intensively used or managed then, and that timber was only cut where it was available within reasonable reach.

The changes that have taken place during the recent period of intensive forest management have been rapid and uniform regardless of region. This places a lot of pressure on forest management, since the prevailing management style can be read off directly from the landscape structure. Management has proceeded with the very simple goal of achieving maximum timber production. Natural factors placed some limits on forest operations and their scheduling, but in the Finnish context, at least, the natural environment has been secondary in the process of allocating harvested stands in landscapes. Our concern for biodiversity and our understanding of the prerequisites for maintaining it nevertheless call for a new type of management. A lot of progress has been made recently in the ecological analysis of landscapes and of species response to landscape patterns, but there is still much to do in the areas of collecting relevant information on landscapes and species and incorporating that information into forest management and planning.

So far studies of landscape and stand structure have been based mostly on assessments of more natural forest areas beyond the borders of Finland, or on analyses of the few areas of natural forests remaining in Finland, together with simulation studies predicting forest landscape structures under natural disturbance regimes. It has been more difficult, however, to assess the influence of early timber use on forest structure. It would be interesting to know more precisely what the state of the forests was in the early 1900's and how it varied depending on the distance from human settlements and agricultural areas, for example. Early aerial photographs could be useful for this kind of research, combined with three-dimensional modelling in order to evaluate further attributes such as height and volume (Korpela, 2006). Unfortunately, interpretation of individual trees is often limited to the topmost layer, and even the assigning of a tree to species can be a problem when using black and white photographs. Potentially, however, old photographs enhanced by means of modern image processing techniques in combination with historical management data, forest inventory data and auxiliary information concerning land use could produce valuable information on the forest structure as it was before the commencement of intensive forest management. This type of data could also be used to reconstruct various cultural landscapes.

The national forest inventories in Finland are nowadays multisource ones, carried out using a combination of field measurements and satellite data (Tomppo, 1990). The accuracy of such data maybe too coarse for small-scale analysis, but the inventories do provide repeated coverage of nationwide information on an array of forest measurements, and the data can be analysed using spatial statistics to produce information on landscape characteristics and to monitor current state of forest landscapes and changes in them. Further improvements in data collection for forest inventories are to be expected with the development and introduction of laser scanning techniques (Naesset et al., 2004). The use of forest inventory data is especially interesting since the current line of research, analysing

landscapes as patch mosaics, has recently been challenged by the use of gradient data, in which a landscape is described using continuous variables, so that instead of saying that a forest is pine-dominated, for example, the proportion or volume of pine in each landscape element, typically a pixel, can be expressed precisely. This is exactly the kind of data that a multisource forest inventory can produce. Using selected variables, the precision with which the landscape is described can be greatly improved. This type of analysis would also considerably alter the analysis of edges because 1) it is recognised that fuzzy patch borders exist, and 2) the delineation of patch borders would be a secondary step conducted only when necessary or when it adds to the value of the analysis. Also, the preserving of the original data rather than being compelled to start with an appropriate classification scheme would make already collected data much more useful for a variety of purposes. So far the use of gradient data in landscape analysis has been only demonstrated (McGarigal & Cushman, 2005, Thessler et al., 2005), and there exists a need to develop the methodologies and applications further with special data references.

Regional planning is needed to ensure a variety of habitat types locally or to support certain specific species. This is especially important on private lands, although very challenging, too (Kurttila, 2001, Kurttila et al., 2006). The commitment of private forest owners to sustainable development depends mostly on the economic losses they need to accept or how justified the losses are. Research is needed into 1) how to integrate alternative silvicultural treatments into forest planning, 2) how to allocate forest operations in order to maximise their ecological and economic objectives, and 3) how to combine the management goals of a number of forest owners with landscape-level spatial goals. The simulation of different types natural disturbances and their local probabilities should probably also be addressed, because these can significantly alter the intended, ecologically and economically optimised pattern.

Last but not least, more data are needed on individual species, their preferred habitats and existing thresholds in relation to the structural properties of both landscapes and individual stands. Landscape statistics are of little intrinsic value without any connection with species.

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