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Natural disturbances dynamics as component of ecosystem management planning

Abstracts and short papers from the workshop of the SNS network ”Natural Disturbance Dynamics Analysis for Forest Ecosystem Management” in Geysir, Iceland, 11.-15. October, 2003

Edited by Aðalsteinn Sigurgeirsson and Kalev Jõgiste

1. INTRODUCTION

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The network “Natural disturbance dynamics analysis for forest ecosystem management” is a group of researchers from Nordic Countries, Baltic States, Netherlands and Russia. The group is one of the networks under the SNS, the Nordic Forest Research Cooperation Committee. The activity of the group has been going on during two years. Firs meeting was hold in Hiiumaa, Estonia August 1.- 4. 2002.

The research activity of the scientists is focused on the disturbances in forests. The evaluation of the impact of disturbances is one of the major tasks. The main aim is to gain new knowledge in the area of natural disturbance regimes and forest ecosystem processes. An ecosystem understanding integrates physical and chemical processes with an understanding of the adaptations of individual organisms. An understanding of these ecosystem processes in a social and economic context requires both a multidisciplinary and an interdisciplinary approach. The challenge is to bring a wide range of subject matter experts together to develop a shared understanding of the ecosystem processes. The resulting understanding will lead to improved forest resource utilization and planning.

Traditional forest management focused on a few commodities. The resulting forests raise concerns with respect to the resilience and sustainability of the forest ecosystem. These concerns of ecosystem resilience are particularly relevant in the context of global climate change. Changing social and economic conditions further challenges traditional forest management. Society expects a broader range of forest values to be sustained. This research explores the opportunity of studying natural disturbance processes and the response of forests to disturbance to create sound management plans for our future forests.

The meeting in Iceland was successful. A seminar session with interesting presentations and two days of excursion were very productive and inspiriting. Many questions were addressed during the workshop. The beautiful nature of Iceland has evoked exiting ideas about the management. The hard work of Icelandic foresters serves as good example of solving problems coming from history and developing sustainable management for future.

This volume contains the abstracts and short papers by the workshop participants. The conference was hosted by the *Icelandic Forest Research.* Thanks to our colleagues!

The support of SNS, the Nordic Forest Research Co-operation Committee, in providing workshop funding is gratefully acknowledged.

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2. DYNAMICS OF DEAD WOOD IN EUROPEAN BEECH FORESTS IN RELATION TO NATURAL DISTURBANCES

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Background

Dead wood provides habitats for maintaining biodiversity, plays an important role in forest productivity, and secures long-term carbon storage. Recently, dead wood has become one of nine Pan-European indicators of sustainable forest management and biodiversity conservation. However, there is a lack of knowledge about the dynamics, i.e. the quantities, temporal fluctuations and spatial distribution of dead wood in forest reserves of Europe and how such information could be transformed into practical management guidelines for sustainable forest management (Hahn and Christensen 2003). In this paper we focus on the dynamics of dead wood in beech forest reserves in relation to disturbance regimes.

Natural disturbances are crucial factors influencing the amount of dead wood in forests in both time and space. In European beech forests, both abiotic and biotic disturbance types are common. The most frequent abiotic disturbance is wind, primarily strong storms during the winter. Other abiotic disturbances are sudden changes in groundwater levels causing water-logging, and ice- and snow-breaks (Standovar 2001). Biotic disturbances are primarily pathogenic fungi and insects. Each disturbance type has its own characteristics regarding frequency and intensity, providing large differences in the spatial and temporal distribution of dead wood.

Wind disturbance

Disturbances by wind occur in all parts of the beech forest range, but are most dominant in the Atlantic beech forests of Northwest Europe. Normal wind speeds pick out trees, which are already weakened by fungi, insects or ageing, whereas strong wind storms or hurricanes also hit trees, which are in good health conditions. The trees are damaged in a variety of ways including breakage of major branches, complete stem snapping and uprooting (Wolf et al. 2003). The proportion of the different damage types differs considerably according to soil type, groundwater level, and windspeed. Uprooting is for example much more frequently connected to wind disturbances, and in many beech forests uprooting is more or less confined to strong winds. Wind disturbance is operating at the large scale, being infrequent, but with high intensity and impact, therefore creating large, multiple-tree gaps and thereby also high fluctuations in the dead wood levels. The creation of large quantities of dead wood in short time leads to a temporally uneven distribution of decay phases and a spatially clumped dead wood distribution.

Insect and fungal disturbance

Disturbance patterns of fungi and insects in European beech forests act at the single-tree level. The patterns of large-scale lethal attacks by insects or fungi as known from coniferous forests are very rare in beech forests. Often single-tree death creates standing dead trees or, alternatively weakening of the wood leading to breakage at the base of the tree. The disturbance by fungi and insects in beech forests is a major driving force in the building of the fine-grained mosaic, which is typical for most beech forests in central Europe (Emborg et al. 2000). Dead wood generated by the death of single trees is often spatially well-dispersed with all decay phases being present at the same time, thus securing a continuous, steady but low input of dead wood.

The disturbance regime also influences the snag-log ratio in forest reserves. The snag proportion is often higher in montane Fagus-Abies forest types than in lowland mixed deciduous beech forests (Christensen et al. 2003). This is due to the lower risk of severe windthrow in the Fagus-Abies forests, but also the nature of fir trees to produce snags and stumps rather than fallen logs.

Dead wood and forest management

The differences in the patterns created by different disturbance types are important when understanding the patterns of dead wood in beech forests. In the management of dead wood in beech forests it is crucial to take the knowledge of natural disturbances into consideration. In table 1 we give some of the main characteristics of dead wood, which should be considered when discussing natural dead wood volumes as guidelines for sustainable forest management.

Table 1. Main dead wood characteristics of two different disturbance types in natural beech forests

Example	Single-tree death of weakened trees caused by *Fomes fomentarius*	Multiple-tree death of both weakened and vital trees caused by strong windstorm
Frequency	High frequency	Low frequency
Intensity	Low intensity	High intensity
Spatial distribution	Even / random	Uneven / clumped
Temporal distribution	Short intervals, low volumes	Infrequent, occasional high volumes
Snag/log proportion	High proportion of snags	High proportion of logs
Decay class distribution	Even distribution to decay classes	Uneven distribution to decay classes

Conclusion

In conclusion, dead wood production in natural beech forests is a combination of two or more disturbance types, causing a mixture of single-tree and large scale dead wood patterns. This means that dead wood volumes vary in both time and space. Therefore, a range rather than a fixed threshold value should be applied as a guideline, aiming at a target value, but accepting certain fluctuations around this value within a period.

References

Bigler, J. 2002. Stormfald og dets betydning som dynamisk faktor for urørt skov i Danmark. Master thesis. The Royal Veterinary and Agricultural University, Denmark.

Christensen, M., Hahn, K., Mountford, E., Odor, P., Standovar, T., Rozenbergar, D., Diaci, J., Wijdeven, S., Meyer, P., Winter, S., and Vrska, T. 2003. Dead wood in European beech forest reserves. Manuscript in prep.

Emborg, J., Christensen, M. and Heilman-Clausen, J. 2000. The structural dynamics of Suserup Skov, a near-natural temperate deciduous forest in Denmark. Forest Ecology and Management 126:173-189.

Hahn, K. and Christensen, M. 2003. Dead wood in European forest reserves – a reference for forest management. EFI proceedings. In press.

Standovar, T. and Aszalós, R. 2001. Natural gaps in beech forests in Hungary. Natural canopy gap characteristics in European beech forests. NatMan working report 6. 20-24. Available at http://www.flec.natman.kvl.dk.

Wolf, A., Friis Møller, P., Bradshaw, R. H.W. and Bigler, J. 2003. Storm damage and long-term mortality in a semi-natural, temperate deciduous forest. Forest Ecology and Management. In press.

3. DISTURBANCE STUDIES AND QUANTITATIVE METHODOLOGY: EXAMPLES OF PERMANENT WINDTHROW SAMPLE PLOTS

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Introduction

The methodological considerations are essential part of planning the studies of disturbances. The great variability and fast changes in biota and environment in the course of disturbances push researchers to use unconventional variables. Heterogeneity is indicated as most important underlying concepts of the contemporary paradigm of ecosystem management and conservation by many authors (Kuuluvainen 2002).

The scale of different phenomena creates substationally contrasting approaches in disturbance studies. The mechanism of disturbances can be followed in mechanical details using time resolution as high as micro- or nanoseconds. On the other hand the processes of regeneration can be followed through several decades or hundreds of years.

The permanent sampling is an important methodological aspect. The processes in forest ecosystems can be approached adequately by measuring time series. The approach of “space to time” substitutes contains considerable amount of unexplained variability. The monitoring design is a key factor determining the understanding of chemical-physical background of disturbances.

An additional lead factor in disturbance studies is the naturalness of events. The man-made dynamics of forest encompassed many facets of disturbances. The mimicking of disturbances is even understood as the management regime for particular conservation areas. Present paper suggests some quantitative characteristics to study the dynamics of forest ecosystems after windthrow.

Material and methods

The study of windthrow was carried out in Northeast and Southeast Estonia. In the summers of 2002 and 2003 we established 18 sample plots in four variants to the protected forests devastated by localised windstorms. The study areas were placed in the eastern part of Estonia, ten plots in Tudu Forest District (59º11´ N 26º52´ E) and eight plots in Halliku Forest District (58º43´ N 26º55´ E). The windstorm occurred in Tudu in the summer of 2001 and in Halliku in the summer of 2002.

Four plots were established to the areas totally destroyed by wind, five plots were partially damaged, five plots were logged after windthrow and four plots were left as control plots. All trees at one totally damaged plot were marked with permanent labels. Thus we obtained four variants for further observations.

The size of the study plot was 40×20 meters (0.08 ha). We used co-ordinate system to determine the position of each tree. In the case of laying trees we used co-ordinates of both ends of log to map the tree. For each tree we described the position, species, the type of a damage (uprooting or stem breakage) and decay rate. We measured breast height diameter of standing trees and snags, from stumps we took root collar diameter. The diameters of laying trees were taken from both ends. We also measured the heights and widths of root plates. In the autumn of 2003 the additional vegetation composition descriptions were composed (the percent of projection coverage was estimated by species).

Results and Discussion

The study plots were placed mostly in Norway spruce (*Picea abies* (L.) Karst.) dominated stands. Secondary species were European aspen (*Populus tremula* L.*)*, Silver birch *(Betula pendula* Roth*)* and black alder (*Alnus glutinosa* (L.) Gaerth*).* On damaged plots in Tudu forest district more common secondary tree species was European aspen, as in the same time the black alder was more abundant in damaged plots in Halliku Forest District. Silver birch was dominant on harvested plots in both Forest Districts – about half of the trees were birches.

The total wood volumes were lower on sample plots in Halliku Forest District, varying between 233 and 355 m^3 per hectare (except at one partially destroyed plot, where it was 541m^3 because of higher stand density and bigger proportion of big diameter black alders). The wood volumes on damaged and control plots in Tudu forest district were 388-661 m^3 per hectare. Volumes of removed timber on harvested areas were smaller – 238-303 m^3 per hectare. The bigger wood volumes in Tudu were probably caused by higher stand density and bigger proportion of European aspens. Laasasenaho (1982) equations were used in calculating wood volumes.

The size of a tree is very important characteristic that influences the windfirmness of a single tree. Several studies (Peterson 2000, Vygodskaya et al. 2002, Arévalo et al. 2000) have emphasized the windfirmness of bigger trees. Data analysis of our partially destroyed study plots showed damage occurrence more or less in equal proportion in all diameter classes, although the damage was a bit less among trees with smaller breast height diameter. Only one diameter class, 8-12 cm, had over two times bigger number of damaged trees.

There were also specific differences in the nature of damage among species. For example, in the case of Norway spruce uprooting and stem breakage occurred in all diameter classes. European aspens with bigger breast height diameter were uprooted more commonly and smaller aspens were mainly stembroken. At the same time black alders mostly broke and only few of them were uprooted.

The determination of dead wood positions enables the analysis of seedbeds in future. A remarkable fact is the size of disturbed ground area by uprooting of root plates: in Halliku Forest District 5.54% of total sample plot area was affected by root plate uplift. This fact emphasizes the importance of spatial approach.

The conditions for vegetation growth vary greatly due to availability of resources and space to grow. The pits of upmounded root systems are providing contact to mineral soil. At the same time the water regime is influenced to a high degree. Furthermore, the falling particles of soil from root plates often destroy the regeneration in pits.

At least in the first stage, the big portion of ground area is covered by fallen tree stems. The decaying logs are particularly important for shade tolerant species (Cornett et al. 2001). The amount of dead wood first decreases after heavy disturbance as death of further trees takes long time in the course of succession (Kuuluvainen 2002).

The decay rate of the fallen stems is usually driven by species composition of dead wood. Aspen logs decay more rapidly than those of spruce (Hytteborn and Packham 1987). Thus the regeneration pattern depends on the substrate for establishment and earlier forest composition creates basic conditions in this regard.

The physiological patterns can be included in simulations of stand development. So called process based models offer high resolution for dynamic description of recovery after disturbances. For example coarse woody debris determines the soil chemistry properties and affects the nutrition at least of juvenile trees.

Present work suggests the variables to describe the spatial aspects of initial conditions of regeneration. The empirical model to describe and predict the regeneration patterns includes the growth characteristics (e.g. height increment) as response variables. Environmental factors, particularly the spatial distribution of seedbed conditions, are independent variables for regression models constructed in the course of further analysis.

The damage pattern of one storm may have a directional component, but as numerous storms with different wind direction overlap, the correspondence with aspect is lost (Frelich 2002). Turbulence caused by eddies may change the falling direction of trees in a small area and this is obviously the case in the present study.

One of the crucial points in predictions for development scenarios is the species composition of survived trees. The size of the seed pool depends directly on the distance to seed source. From this point of view the spatial scale of damages must be quantified for regeneration estimations. Advance growth creates important support (in certain conditions main source) for the formation of next forest generation.

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References

Arévalo, J.R., DeCoster, J.K., McAlister, S.D., Palmer, M.W. 2000. Changes in two Minnesota forests during 14 years following catastrophic windthrow. Journal of Vegetation Science 11: 833-840.

Cornett, M.,W., Puettmann, K.J., Frelich, L.E., Reich, P.B. 2001. Comparing the importance of seedbed and canopy type in the restoration of upland *Thuja occidentalis* forests of Northeastern Minnesota. Restoration Ecology, 9: 386-396.

Frelich, L.E. 2002. Forest dynamics and disturbance regimes. Studies from temperate evergreen-deciduous forests. Cambridge University Press. Cambridge.

Hytteborn, H., Packham, J.R. 1987. Decay rate of *Picea abies* logs and the storm gap theory: a re-examination of Sernander plot III, Fiby Urskog, Central Sweden. Arboricultural Journal, 11: 299-311.

Jõgiste, K., Ilisson, T. 2003. Stand conditions after heavy storm windthrow in North-east Estonia. In: Ruck, B., Kottmeier, C., Mattheck, C., Quine, C., Wilhelm, G., (eds.) Wind effect on trees. Proceedings of the international conference, University of Karlsruhe, Germany, September 16-18, 2003. pp. 289-294.

Kuuluvainen, T., 2002. Natural variability of forests as a reference for restoring and managing biological diversity in boreal Fennoscandia. Silva Fennica, 36: 97-125.

Laasasenaho, J. 1982. Taper curve and volume function for pine, spruce and birch. Communicationes Instituti Forestalis Fenniae, Vol. 108.

Peterson, C.J. 2000. Damage and recovery of tree species after two different tornadoes in the same old growth forest: comparison of infrequent wind disturbances. Forest Ecology and Management, vol 135: 237-252.

Vygodskaja, N.N., Schulze, E.-D., Tchebakova, N.M., Karpachevskii, L.O., Kozolov, D., Sidorov, K.N., Panfyorov, M.I., Abrazko, M.I., Shaposchnikov, E.S., Solnzeva, O.N., Minaeva, T.I., Jeltuchin, A.S., Pugachevskii, M.Y. 2002. Climatic control of stand thinning in unmanaged spruce forest of the southern taiga in European Russia. Tellus. Series B, Vol. 54(5): 443-461.

4. MONITORING OF NATURE RESTORATION AREAS IN ESTONIAN FORESTS

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Background

During the Estonian Forest Conservation Area Network project (Viilma et al. 2001) 136 additional areas were selected for establishment of new nature conservation areas or for adding to existing valuable forest areas. These additional areas formed 102 nature conservation areas in Estonia. This work initiated EU Life-Nature project “Protection of priority forest habitat types in Estonia” concentrating on developing of 20 more complicated areas of them. Forest species, especially species habiting in old forests, can be preserved by nature restoration activities in certain cases. These activities increase also the overall nature value of forest conservation areas. The aim of restoration is almost always to make necessary initial conditions for predictable natural processes and changing of heavily human influenced ecosystem to more natural condition. Significant part of the project concentrates on nature restoration activities on areas with initial low nature value and where natural recovery is too long process. Also nature restoration measures are planned on areas where natural processes are heavily suppressed or fully excluded by human activities. Planned activities include restoration of water regime, special cuttings and establishing missing structural forest ecosystem elements needed for biodiversity indication. This includes several measures what are imitating natural disturbances in forest ecosystems.

Nature restoration activities have many risks including the risk of total loss of the ecosystem. This is more probable with extensive and stronger gradient activities as controlled forest burning or creating large quantities of deadwood in short time interval. This means that forest restoration activities should be carefully planned and done on right time, location and method. There is no long-term experience in this field in Estonia and therefore all restoration activities should be applied on very small extent, as results could be unpredictable. In this context all restoration activities are seen as research experiments what should have long-term and continuous monitoring.

Monitoring method

Nature restoration will be done mainly in man-made forests where stand structure will not develop "natural ecosystem like appearance" in time frame within one forest generation. Also forest restoration activities will be cariied out in even-aged monocultures.

Monitoring of restoration areas is continuous long-term research enabling to evaluate the success of restoration activities and, if necessary, to plan some additional activities needed for success. Also the monitoring enables to develop the methods of nature restoration. The overall goal of the monitoring is to follow the changes and processes in stand after restoration activity is done. It is important to distinguish between normal development in stand and direct influence of restoration measures. Therefore similar areas without restoration measures should be monitored also.

The main method for nature restoration monitoring is forest stand monitoring by permanent sample plots (PPS) accompanied with scoring of stand nature value (Korjus 2002) and species-specific special inventories. PPS are established by the method of Prof. Andres Kiviste (Kiviste & Hordo 2002) before restoration activity is done and are re-measured after restoration activity is done and then re-measured with interval of three years. Gradually this interval will be prolonged up to ten years. PPS are high-accuracy large circular forest sample plots with tree position data recorded. Restoration responses will be monitored on tree, stand and restoration area levels. Tree level monitoring reflects the growth, death and decomposition of the tree but also species occupying the tree on different development stages. Tree level monitoring is also important on cases when trees are treated differently within the stand and these different treatments should be observed. Stand level monitoring enables to observe different stand components and their changes but also changes in forest stand composition and structure. Restoration area level enables to monitor the change of overall naturalness of area.

Some other aspects are regarded on establishment of PPS:

1. Monitoring of nature restoration should only be possible with long-term and regularly re-measured sample plot series.
2. Sample plot should be positioned in stand by pre-selection the location on the map and typical to whole stand. From process monitoring viewpoint such positioning could be considered as random location.
3. Trees are measured with their position coordinates and numbered to enable re-measurement of the same trees.
4. Sample plots should be big enough to monitor processes within the stand. Too small sample plots may reflect the processes inadequately.
5. In the neighbourhood of restoration area should be retained untouched areas similar to restoration area to establish control sample plots for evaluation of efficiency and necessity of the activity on the restoration area.

Biodiversity in a given area is usually evaluated through surveys of species richness in different taxonomic groups (Terradas et al. 2003). Species-specific special inventories include ground vegetation monitoring, insect monitoring, lichen monitoring and fungi monitoring. Ground vegetation is monitored on PPS by circular plots with radius 0.6 m. Normally the number of the plots is 16 per one PPS. In certain cases 1*1 m rectangular sample plots are combined with the circular plots. All species on sample plots are be determined, but also their height, vitality, development stage and amount is recorded. On fungi and lichen monitoring the observations on living and dead trees, tree roots and sumps are added. All lichen species are determined. Insect monitoring is especially valuable and proper to evaluate changes on restoration areas because of insect's species and habitat richness, fast life cycle, distribution patterns and lot of monitoring methods available. Species determination on different lichen, fungi and insect monitoring is done in special laboratories in most cases.

Results and conclusions

The establishment of PPS for monitoring nature restoration areas was started in the autumn 2003 and is still going on. Restoration activities are just in planning stage now and no activities have been done yet. The guidelines for forest restoration (Jõgiste et al. 2002) have been prepared. All nature restoration activities can be done according to conservation and management plan for nature conservation area.

Preliminary selection of restoration areas has been done from forest stand databases. Selected stands were cultivated, middle-aged, normal or too dense, coniferous pure stands on mineral soils. These pre-selected stands were evaluated in nature for final selection of stands and possible activities. 44 restoration areas with permanent sample plots were established and this number will increase. Re-measurements will be done after restoration activities in 2004.

There are still missing areas for water regime restoration. Influences of closing the drainage systems in forests should also be monitored and subsequent PPS established.

As a conclusion, we state that nature restoration is in most cases more than simple "cosmetics of stands for demonstration their naturalness" but careful and profound planning and realisation of restoration activities and correct monitoring of these areas is a key for successful restoration.

References

Jõgiste, K., Kuuba, R., Kalda, A., Parmasto, E., Jüriado, I., Õunap, H., Kiviste, A., Korjus, H., Viilma, K., Öövel, J., Amos T. 2002. Metsade looduslikkuse taastamise juhend. Tartu, 131 p., http://www.metsahoiu.ee

Kiviste, A., Hordo, M. 2002. Network of permanent forest growth plots in Estonia. – Metsanduslikud Uurimused/Forestry Studies, XXXVII: 43-58. (In Estonian, summary in English).

Korjus, H. 2002. Inventorying natural values in forest stands – Metsanduslikud Uurimused/Forestry Studies, XXXVII:. 59-71. (In Estonian, summary in English).

Viilma, K., Öövel, J., Tamm, U., Tomson, P., Amos, T., Ostonen, I., Sørensen, P., Kuuba, R. 2001. Estonian Forest Conservation Network. Final Report. Tartu: Triip Grupp, 95 p + 306 p.

Terradas, J., Salvador, R., Vayreda J., Lloret, F. 2003. Maximal species richness: an empirical approach for evaluating woody plant forest biodiversity. – Forest Ecology and Management, 189: 241-249.

5. NATURAL AND ANTHROPOGENIC DISTURBANCES IN FINNISH FORESTS: SOME IMPLICATIONS FOR FOREST MANAGEMENT AND RESTORATION

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Introduction

In Finland a long history of forest utilization, recent intensive forest management, and suppression of natural disturbances, such as forest fires and gap dynamics, have led to a reduction in forest structural complexity, volume of dead wood and overall biodiversity of forests (Working group…2000, Siitonen 2001, Kuuluvainen 2002, Rouvinen et al. 2002a). Especially in southern Finland also many protected areas have experienced a period of extensive utilization before becoming protected. This situation is a threat to biodiversity, since in addition to the small area of protected forest (presently only ca. 1% in southern Finland), the value of reserves as a habitat for naturally occurring species has been reduced.

Dead wood provides a striking example of the impact of human utilization on forest characteristics and biodiversity. Dead trees provide habitat for a large number of species, ca. 5000 in Finland (Siitonen 2001). In southern Finland in managed forests there is only 2-10 m^3/ha of coarse woody debris, while in natural forests the average is 60-90 m^3/ha of dead wood. This means that the average volume of coarse woody debris has decreased by 90-98%.

Judged from general species-area models this could in the long run mean a disappearance of up to 50 % of the species depended on dead wood (Siitonen 2001).

Natural fire regime and human impact

Although the fire regime without human impact is not observable any more in Finland, it is evident that the natural fire regime has always been affected by the cooler and moister summers compared with more continental areas in Russia and Canada. It is well known from dendrochronological studies that in the 17th-19th centuries *Pinus sylvestris* dominated forests have typically burned frequently, with 30 - 60 year intervals (Zackrisson 1977, Lehtonen 1997). However, during this period human activity, such as slash-and-burn cultivation, significantly increased fire frequencies. Recent studies indicate that prior to significant human influence fire return intervals have been much longer, in northern forests up to several hundreds of years (e.g. Pitkänen et al. 2002, Wallenius 2002). This is also geographical variation so that lightning ignitions are much more frequent in the southern part of the country compared with the northern part (Larjavaara et al., unpubl.).

In general, the natural fire regime would most likely be a mixed severity one (Pennanen 2002). Pitkänen (1999) estimated that before settlement in eastern Finland about half of the fires have been stand-replacing. Typically a significant proportion of large fire-resistant *Pinus* trees would survive fires and form a persistent and more or less continuous cover of overstory trees (Pennanen 2002). The dynamics of natural *Picea* dominated forest landscapes, that are associated with moist and fertile lowland areas, would typically be driven by small-scale gap dynamics due to autogenic disturbance agents (Kuuluvainen et al. 1998, Engelmark 1999). However, in this type of forest fires or severe storms could occasionally cause severe and widespread destruction (Sirén 1955, Syrjänen et al. 1994). There are no reliable estimations of fire size distributions in natural conditions. However, because of the humid climate and landscapes fragmented with peatlands and water bodies, the average fire sizes would evidently be significantly smaller compared with those in continental areas of the boreal zone.

In Finland the past utilization of forests has been both extensive and intensive. The past human influence on forests in Finland can roughly be divided into four main, temporally partly overlapping phases: (1) the phase of hunting, gathering and small-scale slash-and-burn cultivation, (2) the phase of slash-and-burn cultivation and tar burning (1700-1900), (3) the phase of selective logging (1870-1950) and (4) the phase of industrial forest utilization (1950->) (Kuuluvainen et al. 2004).

Since the Second World War, the structure of forests has strongly been shaped by intensive and extensive forest management aimed at high and sustained production of valuable timber. Forest management is typically based on compartments of 0.5-10 ha as basic operational units. Silvicultural treatments, such as thinning, have been widely used to create homogeneous single-species even-aged stands, to fully utilize the sites' wood production potential. The dominant harvesting method is clear-cutting, covering ca. 2/3 or the harvested area, with retention trees left in groups of 10-20 trees. In Scots pine stands seed tree cuttings are done (1/3 of harvested area). At landscape level the management ideal has been a fully regulated even-aged forest, where each stand age class covers an equal area. In regeneration, whether by natural seeding or by planting, soil preparation has been and is used extensively. First plowing was common, but currently lighter scarification methods are favored. Although forest management has profoundly changed the structure of forests over the past decades, the regeneration material has always been local and exotic species have not been used in Finnish forestry.

The new forestry law (since 1997) sets ecological and social sustainability, preservation of biodiversity and sustainable yield of forests as equally important goals. As a consequence, forest management methods were modified during the 1990s. New management practices include setting aside habitats of special importance for forest biodiversity (so-called key habitats), retention of living and dead trees in harvesting, and favoring prescribed burning and deciduous admixture.

However, at the moment we do not know how efficient these new methods are in preserving biodiversity. Accordingly, it has recently been suggested that the natural forest should be used as a reference when developing forest management methods aiming at biodiversity restoration and maintenance (Kuuluvainen 2002). This approach suggests that there is a need to diversify cutting treatments. In practice this would mean a shift from current clear-cutting dominated harvesting to management where a range of partial harvesting methods, inspired tree mortality patterns in natural forests, are applied (Kuuluvainen 2002).

Current research findings suggest that restoration actions at a wide scale are needed both in protection areas and in managed forests to better sustain native biodiversity (Hanski 2000, Kuuluvainen 2002, Working group…2003). This is the case especially in southern Finland, where forest utilization has been most intensive and long lasting, and where only ca. 1% of forest is protected. To be effective restoration actions must consider the roles of both protection areas and managed forests. A major challenge of forest management is to use knowledge of the ecology of natural forests to develop management practices that would restore some of the habitat availability found in natural forests. A future challenge, involving restoration as an essential element, is to create a comprehensive strategy for biodiversity conservation that considers both protection area networks and managed forests and covers the whole country.

Conclusions

It is evident that in areas that have been strongly affected by past and more recent forest utilization extensive restoration of both managed and protected forest ecosystems will be needed to sustain their natural level of biological diversity (Working group… 2000, Kuuluvainen et al. 2002). Restoration is urgently needed in particular to accelerate the formation of dead wood and other structural features resembling those of natural forests in order to enhance the conservation function of both protected and managed forests.

References

Engelmark, O. 1999. Boreal forest disturbances. Teoksessa: Walker, L.R. (toim.), Ecosystems of Disturbed Ground, s. 161-186. Ecosystems of the World 16. Elsevier Amsterdam.

Hanski, I. 2000. Extinction debt and species credit in boreal forests: Modelling the consequences of different patterns of forest restoration. Annales Zoologici Fennici 37: 271-280.

Kuuluvainen, T., Syrjänen, K. & Kalliola, R. 1998. *S*tructure of a pristine *Picea abies* forest in north-eastern Europe. Journal of Vegetation Science 9: 563-574.

Kuuluvainen, T. 2002. Natural variability of forests as a reference for restoring and managing biological diversity in boreal Fennoscandia. Silva Fennica 36(1): 97-125.

Kuuluvainen, T., Aapala, K., Ahlroth, P., Kuusinen, M., Lindholm, T., Sallantaus, T., Siitonen, J. & Tukia, H. 2002. Principles of ecological restoration of boreal forested ecosystems: Finland as an example. Silva Fennica 36(1): 409-422.

Kuuluvainen, T., Jäppinen, J.-P Kivimaa, T., Rassi, P., Salpakivi-Salomaa, P. ja Siitonen, J. 2004. Ihmisen vaikutus Suomen metsiin [Human impact on forests in Finland].In: Kuuluvainen, T., Saaristo, L., Keto-Tokoi, P., Kostamo, J., Kuuluvainen, J. Kuusinen, M., Ollikainen, M. ja Salpakivi-Salomaa, P. (toim.). Metsän kätköissä – Suomen metsäluonnon monimuotoisuus. [Hidden in the forest – biodiversity of forests in Finland]. Edita. (in press).

Lehtonen, H. 1997. Forest fire history in north Karelia: dendroecological approach. University of Joensuu, Research notes 59: 1-23. Joensuu.

Pennanen, J. 2002. Forest age distribution under mixed-severity fire regimes – a simulation-based analysis for middle boreal Fennoscandia. Silva Fennica 36(1): 213-231.

Pitkänen, A. 1999. Paleoecological study of the history of forest fires in eastern Finland. University of Joensuu, Publications in Sciences 58.

Pitkänen, A., Huttunen, P., Jungner, H. & Tolonen, K. 2002. A 10 000 year local fire history in a dry heath forest in eastern Finland. Canadian Journal of Forest Research 32: 1875-1880.

Rouvinen, S., Kuuluvainen, T. & Karjalainen, L. 2002a. Coarse woody debris in *old Pinus sylvestris* dominated forests along a geographic and human impact gradient in boreal Fennoscandia. Canadian Journal of Forest Research 32: 2184-2200.

Siitonen, J. 2001. Forest management, coarse woody debris and saproxylic organisms: Fennoscandian boreal forests as an example. Ecological Bulletins 49:11-41.

Sirén, G. 1955. The development of spruce forest on raw humus sites and its ecology. Acta Forestalia Fennica 62: 1-363.

Syrjänen, K. Kalliola, R., Puolasmaa, A. & Mattson, J. 1994. Landscape structure and forest dynamics in subcontinental Russian European taiga. Annales Zoologici Fennici 31: 19-34.

Wallenius, T. 2002. Forest age distribution and traces of past fires in a natural landscape dominated by *Picea abies*. Silva Fennica 36(1): 201-211.

Working group on the need for forest protection in southern Finland and Ostrobothnia. 2000. Forest protection in southern Finland and Ostrobothnia. The Finnish Environment 437, 284 p. (in Finnish with English summary).

Working group on restoration. 2003. Restoration in protected areas: report by the working group on restoration. The Finnish Environment 618, 220 p. (in Finnish with English summary).

Zackrisson, O. 1977. Influence of forest fires on the North Swedish boreal forest. Oikos 29: 22-32.

6. SHORT FACTS ON SWEDISH FIRES WITH EMPHASIS ON FIRE HISTORY

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Most of Sweden lies within the boreal and boreo-nemoral zones. In these parts of our country, fire has, up until recently, played an important role in shaping the forests. Out of the total forested area of 23 milj ha, only a few hundred thousand hectares forest belongs to the temperate deciduous zone where presumedly the rich hardwood forest (*Fagus, Quercus, Fraxinus, Ulmus, Tilia, Acer*) has been less influenced by fire in the past. Although most of our forests are shaped by fires it is not until the recent decade forest managers and public has become aware of this. Research about fire or rather research *related* to fire has increased dramatically in the last years, largely catalyzed by a growing concern for environmental issues.

At present only a fraction (on the order of 0.017-0.0017% annually) of the forest land burns annually, from a few hundred hectares in wet years to a few (max c 5 000 ha) thousand hectares in a dry summer. Most of the ignitions are today human caused but lightning ignitions can cause a substantial amount of the fires in dry years when periods of high pressure are followed by thunderstorms with little or no rain following. The number of lightning ignitions follow a N-S and W-E gradient where the highest amount occur in the southeastern summerdry part of the country (in the order of 0.2 ignitions per 10 000 ha and year) and the lowest lightning ignition frequency in the high altitude northern forests (a factor 0.1 or less than in the southeast) (Granström 1993).

The burned area per year (eg. “fire frequency”) is today very different from the past situation due to effective indirect and direct fire suppression. The network of forest roads is extremely dense, even in the north and allows for early attack by fire fire-fighting crews. According to fire history studies (Kohh 1975, Zackrisson 1977, Engelmark 1984, Niklasson and Granström 2000) fires were occurring at 50-150 y interval in the north and down to 20 y in the south (Page et al. 1997, Niklasson and Granström 2000, Niklasson and Drakenberg 2001). A very rough national “ average fire frequency” amounts to about 1.7 % annually burned area, equivalent to about 58 y return intervals. Fire suppression becomes effective around 1860-1880 over most of the country and since then burned area diminished to a seemingly steady level since the 1950s. In the southern part suppression seems to have started less uniform and generally earlier. The reasons behind the rapid decline in fire frequencies is somewhat debated although there are evidence for strong human impact on the fire regime from some regions (Niklasson and Granström 2000, Granström 2001). The rapid growth of organized forestry and expanding timber industry over most of the country in the late 1800s should have had a major effect on this process.

The long time of fire suppression in combination with an increasingly intense industrial forestry has had negative consequences for many species. While the loss of old trees must be ascribed more to forestry than fire suppression, the absence of fire has pushed several hundred of fire-adapted and fire-requiring species, predominantly invertebrates, from being common to rare or even extinct in the country (Ahnlund and Lindhe 1992, Ehnström, Långström et al. 1995, Wikars 1997). A few of these species are strictly dependent on fire *per se* while the major part of this group depend on structures and processes that mainly fires provided in the past such as:

openness/sun-exposure, dead wood, damaged trees with lowered vitality, fire scars, burnt ground. Another strongly negative effect from the combination of forestry and fire suppression is the lack of seral stages dominated by deciduous trees such as *Betula, Populus, Salix.* The reproduction from seeds of *Populus* and *Salix* is strongly promoted by fires and is now a rather rare event. The flagship species white-backed woodpecker *Dendrocopus leucotos* is now on the verge to extinction in Sweden being confined to older deciduous-dominated forest, typically of fire origin. Only a hundred years ago this bird was common all over the country.

Although the awareness has increased dramatically among foresters and public about fire, this has so far very little been turned into action when it comes to using fire as a tool. Although the structures, substrates and effects of fire has influenced the design of alternative management regimes (Angelstam 1998) the incorporation of fire is hampered by a lack of practitioners, anxiety for loosing control of burns and a lack of resources. In fact, according to the Swedish certification criteria under the Forest Stewardship Council as much as 5% of the annual clear-cut area should be burned. This is hardly accomplished at present, and these burnings typically lack from a species-oriented view resulting in superficial burning of the organic layer but high tree layer mortality (pers obs, Granström 2001). The burning for regeneration purposes had a renaissance in the period 1950-1970 (annually on the order of 10 000 ha) but ended rather abruptly due mainly to rapidly growing labor costs and rationalization of management systems.

Fire research in Sweden is mainly concentrated to Umeå in the north with studies on succession, fire behaviour, and fire history, plant-plant interactions, ecosystem functioning and paleoecology. Uppsala has a strong tradition in enthomology and in southern Sweden as a whole paleoecological research has been done in the past but very little other research. Fire history studies have just started and pilot studies in fire behaviour/flammability. A lot of the research in other fields of ecology can be ascribed to fire or has fire a common denominator but this is out of the scope with this short overview.

A conceptual model is presented as a guide to the maintenance and restoration of ecologically sustainable boreal forest (Angelstam 1998). The model is based on the hypothesis that self-sustained forest ecosystems can be (re-)created, and their biodiversity developed, if forest management can simulate the composition and structure of boreal forest landscapes by introducing and maintaining disturbances leading to naturally dynamic spatial and temporal patterns of forest regeneration. The major explanatory variable in the model is the effect of wildfire on sites with different fuel characteristics and climates found in the European boreal forest. Four levels of fire intensity are distinguished, based on mean fire frequencies. These range from extremely low in some wet tall-herb sites or sites at high altitudes or latitudes with a humid climate, where fire is absent or rare, to dry lichen-rich sites where fire occurs often. The model is called ASIO, after the words Absent, Seldom, Infrequent and Often, indicating the four levels. Three main disturbance regimes are distinguished in the European boreal forest, based on the complex interactions between probabilistic (e.g. mean fire intervals at different site types) and random events (e.g. where and when a fire occurs): (1) gap-phase *Picea abies* dynamics; (2) succession from young to old-growth mixed deciduous/coniferous forest; and (3) multi- cohort *Pinus sylvestris* dynamics. The model stems mainly from studies in Fennoscandia, but some studies from outside this region are reviewed to provide support for a more general application of the model. The model has been implemented in planning systems on the landscape level of several large Swedish forest enterprises, and is also used as an educational tool to help private land owners with the location and realization of forest management regimes. Finally, the model can be used to develop an administrative system for the monitoring of biodiversity in boreal forest.

Analysis of fire statistics revealed that there are steep gradients in the distribution of lightning-caused fire ignitions in Sweden (Granström 1993). The highest ignition density was found in the southeastern provinces of Kalmar and Ostergotland, ca. 0.23/10 000 ha/yr. From there, densities generally declined both to the north and to the west, with a density averaging ca. 0.05 in the six northernmost provinces, and an equally low density in the southwestern province of Halland. For both northern and southern Sweden, lightning ignitions peaked in early July, but in the south the season for ignitions started 2-3 weeks earlier and ended 2-3 weeks later. The geographical gradients in lightning ignition density correspond to the average precipitation during summer. The patterns of lightning ignition densities may also indicate gradients in natural fire frequencies. This hypothesis is supported by the distribution of certain fire-adapted plant species.

References

Ahnlund, H. and Lindhe, A. 1992. "Hotade vedinsekter i barrskogslandskapet- några synpunkter utifrån studier av sörmländska brandfält, hällmarker och hyggen." Entomologisk Tidskrift 113(4): 13-23.

Angelstam, P. K. 1998. "Maintaining and restoring biodiversity in European boreal forests by developing natural disturbance regimes." Journal of Vegetation Science 9(4): 593-602.Ehnström, B., B. Långström, et al. (1995). "Insects in burned forests-forest protection and faunal conservation (preliminary results)." Entomologica Fennica 6: 109-117.

Engelmark, O. 1984. "Forest fires in Muddus National Park (northern Sweden) during past 600 years." Canadian Journal of Botany 62: 893-898.

Granström, A. 1993. "Spatial and Temporal Variation in Lightning Ignitions in Sweden." Journal of Vegetation Science 4(6): 737-744.

Granström, A. 2001. "Fire Management for Biodiversity in the European Boreal Forest." Scandinavian Journal of Forest Research Suppl. 3: 62-69.

Kohh, E. 1975. "Studier över skogsbränder och skenhälla i älvdalsskogarna. (A study of fires and hard pan in the forests of Älvdalen)." Svenska Skogsvårdsföreningens Tidskrift 34: 481-512 (in swedish).

Niklasson, M. and B. Drakenberg 2001. "A 600-year tree-ring fire history from Kvill National Park, southern Sweden- implications for conservation strategies in the hemiboreal." Biological Conservation 101: 63-71.

Niklasson, M. and A. Granström 2000. "Numbers and sizes of fires: long term trends in a Swedish boreal landscape." Ecology 81: 1496-1499.

Page, H., M. Niklasson, et al. 1997. "Die Feuergeschichte des Nationalparkes Tivedens in Schweden. Eine kulturhistorische und dendrochronologische Untersuchung." Forstarchiv 68(2): 43-50. English abstract

Wikars, L.-O. 1997. Effects of forest fire and ecology of fire-adapted insects. Department of Zoology, Uppsala University: 35.

Zackrisson, O. (1977). "Influence of forest fires on the North Swedish boreal forest." Oikos 29: 22-32.

7. NATURAL FOREST DISTURBANCES IN LITHUANIA

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Abstract

The paper presents summarized results of studies and gives the perspectives of new research topics in Lithuania, namely: (1) Wind damages in Norway spruce stands; (2) Common ash (*Fraxinus excelsior* L.) dying; and (3) The outbreaks of Scots pine stands defoliating pests.

Keywords: Norway spruce, wind damages, *Fraxinus excelsior* dying, Scots pine, defoliating pests.

Introduction

The effect of various natural disturbances (strong winds, frosts, droughts, diseases and pests) on Lithuanian forest ecosystems is evident. These disturbances cause a decrease in forest productivity. Under strong effect a considerable amount of wood production is lost. Sometimes natural disturbances influence the processes of artificial and natural regeneration.

This paper presents summarized results of some studies and gives the perspectives of new research topics, namely: (1) Wind damages in Norway spruce stands; (2) Common ash (*Fraxinus excelsior* L.) dying; and (3) The outbreaks of Scots pine stands defoliating pests.

Results and discussion

Wind damages in Norway spruce stands

The specific data on wind damages of Lithuanian forests is found only since the fourth decade of the 1900’s (LMM, 1992). Strong storms that damaged significant areas of forests were recorded in 1930-1932. Later the storms damaged forests in 1956 (wind breaks and windfalls caused the lost in wood production of nearly 3 mill. m^3), in 1961 (0.8 mill. m^3), and in 1967 (3.5 mill. m^3). The storms to a less extent and with considerably lower intensity of damages occurred in 1969, 1981, 1983, 1991, and in 1993 as well. The outbreaks of bark beetles are often observed in the stands damaged by storms. For instance, in 1994-1996, an extensive dying of Norway spruce stands caused by *Ips typographus* is related to the storm damages in 1991 and 1993.

In some parts of Lithuania unintensive wind damages generally are observed every year. Therefore the total wind breaks and the windfalls over longer period can be of the same extent as the forest damages caused by strong but infrequent storms.
Summarizing the results of conducted studies (Лабанаускас 1973, Miksys 1998a, 1998b), it can be stated that:

- Norway spruce (*Picea abies* L.) stands are the most sensitive to wind damage. The stands of the other tree species are damaged only by strong storms.
- Forest sites determine the intensity of wind damages in Norway spruce stands: the lowest stability of Norway spruce is observed in more wet and fertile forest sites (Fig. 1).
- The stands of low stocking level are damaged by wind to the great extent, especially after non-clear final cuttings (occasional, selection etc.), and occasionally even after the thinnings (Fig. 2). Therefore it is recommended to avoid non-clear final cuttings in the Norway spruce stands growing in forest sites of high moisture and fertility.
- The stands formed from the older (over 60-year-old) second storey of Norway spruce trees have low sustainability and are more often damaged by wind. The ones formed from the younger 30-40-years-old Norway spruce trees have the higher adaptive capacity and are more resistant to wind damages.
- The uneven-aged Norway spruce stands are more resistant to wind damages than these of even age. Therefore it is recommended to form more uneven-aged Norway spruce stands, especially in the protected forest areas.
- The even-aged Norway spruce plantations because of uniform storey structure have the same wind resistance as the even-aged Norway spruce stands of natural origin.

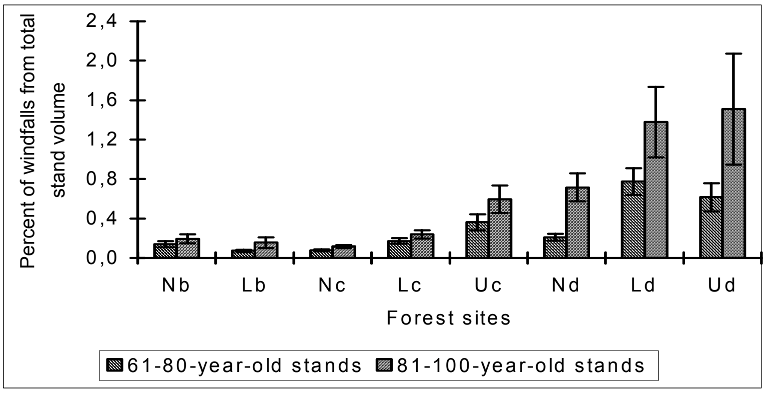

Fig. 1. Wind damages of Norway spruce stands in different forest sites according Lithuanian classification. Hydrotops: N – mineral normally moistured; L – temporarily overmoistured; U – permanently overmoistured mineral soils. Trophotops: b – oligotrophic; c – mesoeutrophic; d – eutrophic mineral soils

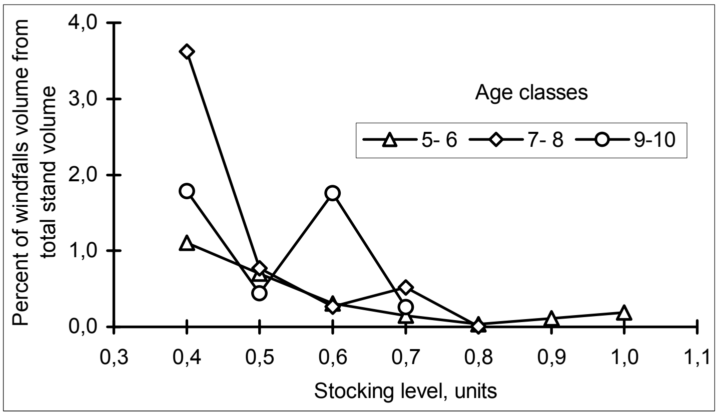

Fig. 2. The intensity of wind damages in Norway spruce stands with different stocking level. Forest site according Lithuanian classification Ld – temporarily overmoistured eutrophic mineral soils

Ash decline

The ash (*Fraxinus excelsior* L.) 10-12 years ago was one of the most healthy tree species in Lithuania. However, during the last 5-6 years, the condition of ash stands decreased considerably: the mean defoliation increased 2-3 times and reached up to 35-40% (Ozolincius and Stakenas 1999, Ozolincius 2002). The ash decline is observed in more than 60% of all the ash stands area. In the regions where the ash stands comprise more than 10% of the total forest area only around 2% of these stands are undamaged (Juodvalkis, Vasiliauskas 2002).

Apparently there is a complex of factors that may cause ash decline. In the future it is necessary to clarify several hypothesis of this new phenomenon.

Some hypothesis of the main predisposing factors determining ash declineare given below:

- *Climate warming and new pests and diseases related to it*. There are some presumptions that pests and diseases habitats are expanding from the south to the north. For instance, the phytoplasmal diseases those are spreaded out by *Cicadellidae* family *Thysanoptera* order of insects can be one of the reasons for ash decline. For the meantime they are indicated as the cause of ash decline (the trees immunity system is weaker) most often in North America. The viruses can be another infecting factor. One of them – *Arabis masaic nepovirus* – via found in the Poland, neighbour country of Lithuania. Besides, the determination of chloroplasts DNA (Heuertz 2002) shows that the ash have re-emigrant to Lithuania with the glaciers from the Balkans (Hungary, Greece). Therefore, if the origin ash population was adapted to particular diseases and pests, apparently the present ash genetical diversity is too narrow to assure the stability of ash stands.
- *The influence tropospheric ozone*. The elevated ground-level ozone concentrations arise in the forest due to more intensive photochemical reactions and slower wind speed. There is an opinion that this is the reason for a much better condition of open-grown ash trees comparing to those growing in the stands.

In addition to above*, the extreme climatic conditions and especially the fluctuation of ground water level caused by droughts* could stimulate ash dying. It was determined that the Seleninov hydrothermal coefficient has some correlation with the tree condition (defoliation) (Ozolincius and Stakenas 2001). Besides, the ash trees of the worst condition are found in the overmoistured forest sites.

Fungal diseases (especially *Armillaria*), *Sleosporium* sp., and *Nectria galligena* are often indicated among the direct factors having influence on the condition of ash trees. It is supposed that *Armillaria* could be distributed by some of the blight species (e.g. *Cryptococcus fagisuga*) that damage the bark of the trees.

The outbreaks of Scots pine stand defoliating pests

The damages of Scots pine stands by defoliating pests comprise more than 30-45% of all the forest damages in Lithuania (the data of Lithuanian Forest Sanitary Protection Service). The outbreaks of Scots pine stand defoliating pests (they are especially common during the last 10 years) mainly appear in the southern part of Lithuania, where sandy Arenosols and Scots pine stands prevail. *Panolis flammea, Dendrolinus pini, Diprion pini, and Lymantria monacha* are causing the greatest damages of the Scots pine stands (Table 1). It is indicated that the period of fluctuation of their outbreaks during the last decades is as follows: *Diprion pini* – every 3 years, *Dendrolinus pini* – 6 years, *Lymantria monacha* –7 years (Fig. 3), and *Panolis flammea* – every 17-18 years.

Table 1. The outbreaks of Scots pine stand defoliating pests in Lithuania

Defoliating pest	Total area of the outbreaks (thou. ha)	The greatest outbreak in the last decade	The period of fluctuation
Panolis flammea	44	2000	17-18 years
Dendrolinus pini	28	1995	6 years
Diprion pini	14	1993	7 years
Lymantria monacha	11	1993	3 years

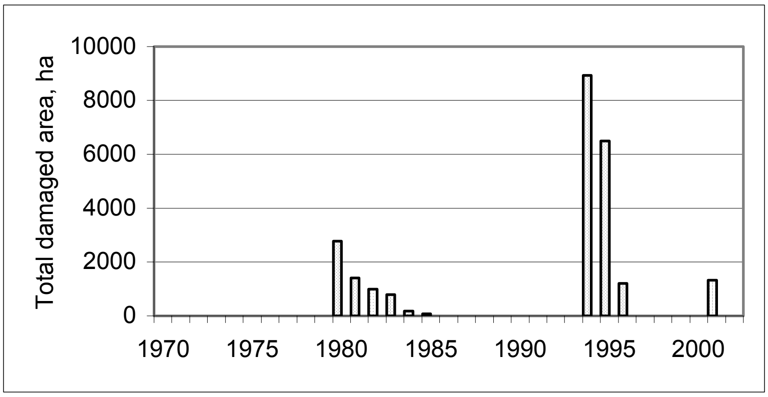

Fig. 3. The fluctuation of the outbreaks of Lymantria monacha (The data of Lithuanian Forest Sanitary Protection Service)

The investigations in the outbreaks of Scots pine stands defoliating pests revealed that: (1) if during 2 years the defoliation of Scots pine trees comprises 90-100%, the trees are dying; (2) after 3-4 years the outbreaks of *Panolis flammea* are followed by bark beetles (Ziogas 1997). For the meantime such effects/consequences can be avoided most effectively only by spraying insecticides by airplanes.

References

Heuertz M. 2002. Population genetic structure in common ash: a focus on south-eastern European genetic resources. – Ph.D. thesis. Univeriste Libre de Bruxells, 73-96.

Juodvalkis A., Vasiliauskas A. 2002. Uosynų dziuties apimtys ir jas lemiantys veiksniai (The factors that may cause ash dying). Musu girios (Our Forests), No 8: 5-7 (in Lithuanian).

LMM, 1992. Lietuvos misku ukis (Lithuanian Forestry). Vilnius, “Mokslas”. – 207 p. (in Lithuanian, with Russian, English and German abstracts).

Miksys V. 1998a. Eglynų atsparumas vejo poveikiui: augimvieciu salygu ir medynu skalsumo itaka (Spruce stands resistance to wind damage: influence of stand stocking level and conditions of forest site). Miskininkyste (Forestry), Vol. 1 (41): 44-58 (in Lithuanian, with English, Russian abstracts).

Miksys V. 1998b. Eglynu atsparumas vejo poveikiui: medynu rusines sudeties itaka (Spruce stands resistance to wind damage: influence of stand species composition).

Miskininkyste (Forestry), Vol. 2 (42): 24-34 (in Lithuanian, with English, Russian abstracts).

Ozolincius R. 2002. Uosynu dziuvimo hipotezes (The hypothesis of ash dying). Musu girios (Our Forests), No 7: 2-4 (in Lithuanian).

Ozolincius R., Stakenas V. 1999. Regional forest monitoring. Forest monitoring Level I. In: Monitoring of Forest Ecosystems in Lithuania (Editor-in-chief – R.Ozolincius). Kaunas, “Lutute”, pp. 82-149.

Ozolincius R., Stakenas V., 2001. Influence of sulphur deposition and drought stress on forest condition in Lithuania. Baltic Forestry, Vol. 7, No 2 (13): 18-23.

Ziogas A. 1997. Misko entomologija (Forest Entomology). Kaunas-Akademija. – 235 p. (in Lithuanian).

Лабанаускас Б. 1973. Создание ветроустойчивых лесонасаждений (Creation resistant to wind damages forest plantations). Каунас. – 20 с. (in Russian).

8. STUDIES OF NATURAL DISTURBANCES IN DANISH FOREST ECOSYSTEMS - THE EFFECT OF WINDTHROW AND GAP FORMATION ON N AVAILABILITY AND SOIL WATER CHEMISTRY

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Abstract

The effect of natural disturbances was studied in different forest ecosystems in Denmark after a severe storm in December 1999. Damages were worst in the southern part of Jutland where large areas of mono-cultural coniferous plantations were windthrown. In more heterogeneous forests disturbances were restricted to uprooted trees and formation of gaps. The two studies discussed in this paper comprise the effect of windthrow on soil water chemistry in coniferous plantations and the effect of gap formation on biogeochemical processes and microclimatic conditions in a semi-natural beech-dominated forest.

In the windthrow study different logging practices, soil preparations, and planting were applied on the study areas. In the first years after windthrow stem harvest increased nitrate losses in stands with present ground vegetation, but showed no effect in stands without ground vegetation. However, the effect of logging has to be evaluated over a longer period of time. Generally, the establishment of vegetation cover seemed to decrease concentrations and losses of nitrate.

The effect of gap formation in a semi-natural forest was most pronounced on soil moisture content which remained close to field capacity in the gap due to reduced water uptake by plants and interception to the canopy. Increased soil temperatures in the southern and eastern parts of the gap were partly explained by the insulating effect of ground vegetation and regeneration. Soil NO_3-N concentrations and mineralization rates revealed the weakest response to the gap. Nitrate losses were increased in the gap, but were generally quite high also under the closed canopy. The overall conclusion is that the effect of a gap is strongly modified by structural variations of the forest and a rapid growth of regeneration and edge trees, and that nitrate losses are not only governed by disturbances, but also by the overall N status of a forest ecosystem.

Keywords: Canopy gap, microclimate, mineralization, natural disturbances, nitrate leaching, soil water chemistry, windthrow

Introduction

Dynamics of natural forest ecosystems have come into focus of Danish research on forest ecosystems. While managed forest ecosystems have been studied intensively for many years, natural forests or natural disturbances of forest ecosystems are only scarcely investigated. This is partly due to the few numbers of natural forests left in Denmark and the seldom occurrence of natural large-scale disturbances in this region. However, the increasing interest in nature-based forestry, the role of forests for biodiversity, and the importance of ground water protection has resulted in more research projects dealing with these topics (e.g. Larsen 1995; Christensen and Emborg 1996; Madsen and Larsen 1997).

During a severe storm on 3 December 1999 large areas of monocultural coniferous plantations were windthrown, while more heterogeneous forests only suffered form uprooted trees and formation of gaps. The damages of this storm event offered the opportunity to study the effect of natural disturbances on different kinds of forest ecosystems. In a semi-natural mixed deciduous forest several investigations were carried out in and around a small canopy gap created naturally during the storm. Another study focused on changes in soil water chemistry in coniferous plantations which were windthrown during the storm. The present paper gives a short overview of these projects, the investigations carried out at the different study sites, and the general conclusions of the results so far.

Windthrow studies

Methods and materials

The most severe damages of the storm occurred in the southern part of Jutland. Forest stands in this region are predominately coniferous monocultures, and they are exposed to high nitrogen (N) depositions from agricultural activities. For this reason, throughfall and soil water chemistry had been monitored in several coniferous plantations for at least three years prior to the storm. After windthrow it was decided to continue measurements in order to follow the development in leaching losses of nitrate (NO_3^-) and other elements. Furthermore, different logging practices, soil preparation, and establishment of new culture were applied on parts of the study area. Data on site conditions, N deposition, and NO_3^- leaching from two stands prior to the storm are published in Hansen (2003).

Three coniferous stands of silver fir (*Abies alba*), Norway spruce (*Picea abies*), and Sitka spruce (*P. sitchensis*), respectively, were included in the study. The silver fir had a dense ground cover and regeneration (ash (*Fraxinus excelsior* L.), beech (*Fagus sylvatica* L.), mountain ash (*Sorbus aucuparia*), and silver fir), whereas the two spruce stands had closed canopies and no ground vegetation prior to the storm. The Sitka spruce stand was left untouched after windthrow. Of the Norway spruce and Silver fir stands, half of the area was left untouched, and from the other half stems were removed, but slash remained at the sites. All three stands were left for natural regeneration, except for one part of the logged area in the Norway spruce stand. This part was later on planted with a mixture of tree species with and without soil preparation, respectively. Volumetric soil moisture content along a 90 cm deep profile was obtained monthly from Time Domaine Reflectrometry (TDR) (Topp et al. 1980). Soil solution was collected below approximate rooting depth (90 cm) using porous PTFE (teflon) suction cup lysimeters with a suction of about 500-600 HPa. In each treatment three groups with three lysimeters were placed randomly. Samples were bulked within each group giving three replicate samples per treatment. Soil solution and precipitation were collected monthly and analysed for cations (NH_4^+, Na^+, Ca^{2+}, Mg^{2+}, Al^{n+}) and anions (NO_3^-, Cl^-, SO_4^{2-}), DOC, DON, pH, and conductivity. Vegetation cover was measured annually by local frequency. Net N mineralization was measured in the Silver fir stand in the second year after windthrow. Throughfall was measured prior to the windthrow, whereas wet deposition was measured after windthrow. Nitrogen deposition was calculated using monthly precipitation volumes.

Results and discussion

Prior to windthrow increased nitrate-nitrogen (NO_3-N) concentrations in soil solution in the Silver fir stand indicated that this site was saturated with N (Aber et al. 1989, 1998). Leaching losses of NO_3-N decreased to levels recorded prior to windthrow after two years in the Silver fir stand, but remained on an increased level in the Norway spruce and Sitka spruce stands. However, it seemed that NO_3-N losses peaked in the second year after windthrow. Similar to other studies carried out after clear-cutting, it was found that NO_3-N concentrations in soil solution were strongly correlated with ground vegetation cover and thus the N demand of plants (Bartsch et al. 1999, von Wilpert and Mies 1995, Emmett et al. 1991). Especially when ground vegetation and regeneration had already been present prior to windthrow, NO_3-N concentrations were on a lower level than in stands without or with only slowly developing ground vegetation. Furthermore, concentrations decreased in the first two years after windthrow, while they increased in the stands with absence of ground vegetation as the vegetation cover was damaged during logging. Logging and replanting practices had no effect on NO_3-N concentrations in soil solution in stands with little ground vegetation, but stem harvest significantly increased leaching losses of NO_3-N as compared to the untouched area in the stand with a well developed and fast growing ground vegetation. The latter is explained by slightly higher NO_3-N concentrations in soil solution and by an increase in soil water flux in the harvested area. Increased water yield following removal of biomass are reported by e.g. Bosch and Hewlett (1982), Baker (1986) and Hornbeck et al. (1997). Generally, losses of NO_3-N after windthrow were not on extremely high levels, even in this region with high N deposition. However, removal of the canopy cover had decreased N deposition from about 25-40 kg N ha^{-1} yr^{-1} to under 15 kg N ha^{-1} yr^{-1}. Results have shown so far that differences in leaching of NO_3-N were larger between sites than between harvesting treatments within each site.

Gap studies

Methods and materials

The effect of a naturally created canopy gap (ca. 18 m diameter) was studied in a semi-natural forest located on Zealand, in the western part of Denmark. The mixed deciduous forest, dominated by beech (*Fagus sylvatica* L.) and ash (*Fraxinus excelsior* L.), has the status of a strict non-invention forest and shows the typical structure of a natural forest ecosystem (Emborg et al. 1996). Several studies conducted in the forest investigated the role of the gap for biogeochemical and hydrological processes, natural regeneration, microclimate, and fungi in a natural forest ecosystem. Focus was put on differences between the gap and the surrounding forest. In the following, a short introduction to the research project on biogeochemistry is presented. Detailed results will be published in Ritter and Bjørnlund (subm.), Ritter et al. (subm.), and Ritter and Vesterdal (in prep.).

Measurements were carried out in the gap and under the canopy along two perpendicular transects. Soil temperature, soil solution chemistry, soil moisture content, soil temperature, and litter decay were measured during the first 3 years after gap formation in December 1999. Soil temperature was measured every 10 minutes at a depth of 5 cm in the mineral soil by P100 sensors, and the average of two hours was recorded using the DL2e data logger system (sensors and logger: Delta-T Devices Ltd., Cambridge, UK). Soil solution was collected at 90 cm depth and volumetric soil moisture was measured along a 90 deep profile as described for the windthrow studies. Mass loss and N release from foliar beech litter were investigated in November each year by the litterbag technique (Wieder and Lang 1982). Net N mineralization and nitrification rates in the upper 10 cm of mineral soil together with soil N concentrations were investigated during the second year after gap formation (Januar 2001-February 2002). Soil samples were collected using the sequential soil core method (De Boer et al. 1993) and analysed as described in Ritter and Bjørnlund (subm.). Throughfall was sampled during the third year after gap formation (October 2001-2002) using plastic funnels of 14 cm diameter placed 1 m above the forest floor. Soil solution and throughfall were analysed for cations (NH_4^+, K^+, Na^+, Ca^{2+}, Mg^{2+}, and Al^{3+}), anions (NO_3^-, Cl^-, and SO_4^{2-}), DOC, pH, and conductivity. All sampling and measurements were carried out on a monthly basis, except for soil temperature and litter decay. Leaching fluxes of NO_3-N were estimated by using the water balance model WATBAL (Starr, 1999; Starr, 2003, in prep.).

Results and discussion

The formation of a canopy gap altered the investigated parameters only moderately. The clearest effect of the gap was on soil moisture which was usually maintained close to field capacity in the gap throughout the year. Under the canopy soil moisture decreased to about 50% of field capacity during the growing season. This is in consistence with other gap studies (Bauhus and Bartsch 1995, Gray et al. 2002). However, in parallel with growth of regeneration, soil moisture levels tended to decrease also in the gap during the growing season (Ritter and Vesterdal in prep.). A similar effect of vegetation cover was found on soil temperatures which were highest south of the gap centre and up to several meters into the understory east of the gap, thus in areas with only sparse ground vegetation cover (Ritter et al. subm.). In other studies in mature deciduous and coniferous forests increased soil temperatures were found the northern part of the gap which received the highest amount of direct radiation (Bauhus 1996, Wright et al. 1998, Gray et al. 2002). Hence, variations in soil properties, forest structure, and ground vegetation cover seemed to modify the effect of the gap.

An effect of the gap was also found on soil ammonium-N (NH_4-N) concentrations. They generally dropped at the onset of the growing season in the second year, but more in the gap than under the closed canopy, and remained lower within the gap than in the closed forest throughout the growing season. This may reflect the increased N demand of the regenerating trees in the gap (Johnson 1992) resulting in a competition between microorganisms and roots for available NH_4^+ (Jonasson et al. 1996). However, net N mineralization and nitrification rates showed no effect of gap formation, only clear seasonal trends and large spatial variations (Ritter and Bjørnlund subm.). This is in contrast to gap studies in managed forests reporting both increased and decreased mineralization rates in gaps (Mlandenoff 1987; Bauhus and Barthel 1995; Bauhus 1996). Mean NO_3-N concentrations in soil solution tended to be higher in the gap than under the closed canopy, as also found in gaps and other small scale disturbances in forest stands (Bartsch et al. 1999, Bartsch 2000, Hobara et al. 2001). However, because of high spatial variations, differences were only significant in few months of the year. The generally high levels close to and even above the threshold value of NO_3-N in drinking water and high leaching fluxes both from the closed forest and in the gap indicated that a heterogeneous forest structure and untouched forest ecosystem do not guarantee low N losses the ground water.

Conclusion

The windthrow studies have shown so far that fast growth of ground vegetation helps to keep leaching losses of NO_3-N at a low level. If ground vegetation or even regeneration have been present prior to windthrow, it is suggested to protect this vegetation cover. Logging practices should be applied carefully in order to enhance natural establishment of a new forest cover. However, the effect of logging has to be studied for a longer time period after windthrow since the removal of biomass also means the removal of N pools. Furthermore, results have to be compared with losses from other clear-cut areas in Denmark, and it is necessary to follow the development in NO_3-N concentrations until or even after the establishment of a new forest cover.

The gap study revealed that an overall effect of a canopy gap on biogeochemical and microclimatic parameters was rapidly modified by growth of regeneration and edge trees. This was most clearly for soil moisture contents, soil temperature regimes, and soil NH_4-N concentrations. Net N mineralization and nitrification were presumably more governed by the spatial variation in soil properties and microclimatic conditions in this highly heterogeneous forest than by changes due to the canopy opening. Furthermore, high concentrations of NO_3-N in soil solution and high losses of NO_3-N both from the gap and under closed canopy indicated that the overall N status of a forest ecosystem and not only the extent and frequency of disturbances have to be considered when discussing the role of forests for ground water quality.

Acknowledgements

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References

Aber, J. D., Nadelhoffer, K. J., Steudler, P., Melillo, J. M. 1989. Nitrogen saturation in northern forest ecosystems. BioScience, 39: 378-386.

Aber, J.D., McDowell, W., Nadelhoffer, K., Magill, A., Berntsen, G., Kamakea, M., McNulty, S., Currie, W., Rustad, L., Fernandez, I. 1998. Nitrogen saturation in temperate forest ecosystems: hypothesis revisited. BioScience 48: 921-934.

Baker, M. B. 1986. Effect of ponderosa pine treatments on water yield in Arizona. Water Resources Research 22: 67-73.

Bartsch, N. 2000. Element release in beech (*Fagus sylvatica* L.) forest gaps. Water, Air and Soil Pollution 122: 3-16.

Bartsch, N., Bauhus, J., Vor, T. 1999. Auswirkungen von Auflichtung und Kalkung auf das Sickerwasser in einem Buchenbestand (*Fagus sylvatica* L.) im Solling. Forstarchiv 70: 218-223.

Bauhus, J. 1996. C and N mineralization in an acid forest soil along a gap-stand gradient. Soil Biology & Biochemistry 28: 923-932.

Bauhus, J., Barthel, R. 1995. Mechanisms for carbon and nutrient release and retention in beech forest gaps. II: The role of soil microbial biomass. Plant and Soil 168-169: 585-592.

Bauhus, J., Bartsch, N. 1995. Mechanisms for carbon and nutrient release and retention in beech forest gaps. I. Microclimate, water balance and seepage water chemistry. Plant and Soil 168-169: 579-584.

De Boer, W., Hundscheid, M. P. J., Schotman, J. M. T., Troelstra, S. R., Laanbroek, H. J. 1993. In situ net N transformation in pine, fir, and oak stands of different ages on acid sandy soil, 3 years after liming. Biology and Fertility of Soils 15: 120-126.

Bosch, J. M., Hewlett, J. D. 1982. A review of catchment experiments to determine the effect of vegetation changes in water yield and evapotranspiration. Journal of Hydrology 55: 3-23.

Christensen, M., Emborg, J. 1996. Biodiversity in natural versus managed forest in Denmark. For. Ecol. Manage. 85: 47-51.

Emborg, J., Christensen, M., Heilmann-Clausen, J. 1996. The structure of Suserup Skov, a near-natural temperate deciduous forest in Denmark. Forest & Landscape Research 1: 311-333.

Emmett, B. A., Anderson, J. M., Hornung, M. 1991. Nitrogen sinks following two intensities of harvesting in a Sitka spruce forest (N. Wales) and the effect on the establishment of the next crop. Forest Ecology Management 41: 81-93.

Gray, A. N., Spies, T. A. 1997. Microsite controls on tree seedling establishment in conifer forest canopy gaps. Ecology 78: 2458-2473.

Hansen, K. (Ed.), 2003. Næringsstofkredsløb i skove – ionbalanceprojektet. Forest & Landscape Research, No 33-2003, 300 pp.

Hobara, S., Tokuchi, N., Ohte, N., Koba, K., Katsuyama, M., Kim S.-J., Nakanishi A., 2001. Mechanism of nitrate loss from a forested catchment follwing a small-scale, natural disturbance. Canadian Journal of Forest Research 31: 1325-1335.

Hornbeck, J. W., Martin, C. W., Eagar, C. 1997. Summary of water yield experiments at Hubbard Brook Experimental Forest, New Hampshire. Canadian Journal of Forest Research 27, 2043-2053.

Johnson, D. W. 1992. Nitrogen retention in forest soils. Journal of Environmental Quality 21 (1), 1-12.

Jonasson, S., Vestergaard, P., Jensen, M., Michelsen, A. 1996. Effects of carbohydrate amendments on nutrient partitioning, plant and microbial performance of a grassland-shrub ecosystem. OIKOS 75, 220-226.

Larsen, J. B. Ecological stability of forests and sustainable silviculture. For. Ecol. Manage. 73: 85-96.

Madsen, P., Larsen, J. B. 1997. Natural regeneration of beech (*Fagus sylvatica* L.) with respect to canopy density, soil moisture and soil carbon content. For. Ecol. Manage. 97: 95-105.

Mladehoff, D. J. 1987. Dynamics of nitrogen mineralization and nitrification in hemlock and hardwood treefall gaps. Ecology 68(5), 1171-1180.

Starr, M. 1999. WATBAL: A model for estimating monthly water balance components, including soil water fluxes. In: Kleemola S., Forsius M. (Eds.), 8th Annual Report 1999, UN ECE ICP Integrated Monitoring. The Finnish Environment 325: 31-35. Finnish Environment Institute, Helsinki, Finland.

Topp, G. C., Davis, J. L., Annan, A. P. 1980. Electromagnetic determinatin of soil water content: measurements in coaxial transmission lines. Water Resour. Res. 16: 574-582.

von Wilpert, K., Mies, E. 1995. The influence of stand structure and tree species on mineral cycling. In: Nutrient uptake and cycling in forest ecosystems. European Commission. Luxembourg. Viii, 276 pp.

Wieder, R. K., Lang, G. E. 1982. A critique of the analytical methods used in examining decomposition data obtained from litter bags. Ecology 63: 1636-1642.

Wright, E. F., Coates, K. D., Bartemucci, P. 1998. Regeneration from seed of six tree species in the interior cedar-hemlock forests of British Columbia as affected by substrate and canopy gap position. Canadian Journal of Forest Research 28: 1352-1364.

9. GAP DYNAMICS AND DISTURBANCES IN FORESTED ECOSYSTEMS

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Abstract

The components of gap dynamics, consisting of spatial and temporal factors and of magnitude, are also applicable to disturbance dynamics. Comparing research results in gap and disturbance dynamics on different geographical locations is a useful tool for the investigation, prediction and implementation of the impact of different disturbance types in several forest ecosystems. The exact description of key terms, concepts and methodologies of research are very important for these comparisons. A digital database, containing details on research parameters like location, plot size and disturbance frequency, could form an interesting guide for future research.

Keywords: gap dynamics, natural disturbances, disturbance regimes, ecosystem management

Introduction

This article is based on the theoretical framework for a study of the importance of windthrow in a pine forest ecosystem in Estonia.

In this research, as in many studies of disturbance mosaics, an attempt was made to relate the gap patterns in the plots with forest disturbances. Understanding of gap dynamics and disturbances is of importance for various reasons. Risk management in silviculture is often mentioned in ecosystems experiencing frequent disastrous disturbance events. Also for the implementation -or even imitation- of disturbance dynamics in sustainable ecosystem management improved insights in the driving forces of these processes are required (e.g. Johnson et al. 1999, Payette et al. 1990). As a starting point, this article will distinguish the components of both gap and disturbance dynamics, based on theories in available literature.

Components

The **spatial component** of gap dynamics consists of the size and the patterns of gaps. Often the gaps appearing in a forest stand are not of equal size nor is there any regular pattern present. However, areas often show a tendency to a certain gap size or regularity. Relating this phenomenon to the disturbance types addressing the area, the characteristics of the disturbance agents could explain the spatial distribution of the gaps present (Armstrong 1999, Quine 1999).

Closely related to the patterns in the spatial component is the **temporal component** of gap dynamics. This component is usually expressed as the frequency of the occurrence of gaps in time. Frequency can be considerably varying: even within stands gaps can appear with intervals of decades to millennia. However, when the cause of gap formation can not be determined, the frequency signature can seem rather chaotic. According to Seymour et al. (2002), size of gaps caused by disturbances and frequency of occurrence are related. More catastrophic disturbances, causing large sized gaps, in general occur with large intervals, while small-scale disturbances have a higher frequency (e.g. Lorimer et al. 1988, Rogers 1996).

The **component of magnitude** is composed of intensity, being the physical forces to which the forest stand is subject, and severity, which is the actual damage done, often expressed as the loss of biomass. Each species and site reacts differently to a certain disturbance intensity, which could explain some of the differences in severity. Therefore, gap size, recovery rate and patterns are depending on the disturbance types, but also on surrounding parameters like the horizontal and vertical stand structure and the site conditions (Blackburn et al. 1988, Canham et al. 2001).

Distinguishing and studying different biotic and a-biotic disturbance types separately could help to understand the interaction of disturbances and complex patterns of gaps existing in forest stands. For example, small-scale disturbances by storms, addressing an area regularly, can cause an accumulation of woody debris. This can be the fuel wood for large-scale forest fires occurring on an infrequent basis.

Comparing the results of disturbance research in different geographical sites is an important tool to quantify the effects of disturbance types to different forest ecosystems. A comparable research approach, including definition formulation and a close description of the site conditions will facilitate the interpretation considerably.

Of course one should realise that not each gap is caused by disturbances. More gradual processes, like senescence and global climate change, are also influencing gap formation and gap closure or expansion (Ban et al. 1998, Bergeron and Leduc 1998). Furthermore, natural and human disturbances often interfere, complicating the picture increasingly.

Research and analysis

When considering research approach and definition formulation in gap dynamics research, it is important to define the scale of the research, the study plots and the boundaries set to gap size. Reporting the shape of the gap, in terms of minimum width and the way of assessment of gap shape, is essential when studying the gap overlays of several disturbance types or in case of a specific disturbance type like windthrow (Quine 2003). Furthermore, methods to report gap boundaries should be described accurately as these are not univocal between studies (e.g. Runkle 1985, Brokaw 1982, Brandani et al. 1988). While studying gap occurrence frequency, the methods applied to date gaps and to determine the gap formation agent are varying (Lorimer and Frelich 1989, Lorimer et al. 1988). Additionally, in long-term research it is important to set the monitoring frequency. Investigating magnitude, the severity and intensity of disturbance types should not be related without considering and describing the biotic and abiotic site characteristics. The implementation of natural disturbances in ecosystem management also means that the impacts of human disturbances, especially in terms of silvicultural activities, are adequately clear.

In conclusion, for the comparison of research results in gap and disturbance dynamics, it is very important to describe key terms, concepts and methodologies as exact as possible.

Suggestions for the SNS network 'Natural disturbances dynamics as component of ecosystem management'

The SNS Network resources are obviously aimed at the exchange of ideas and research experiences. During the Hiiumaa and Geysir meetings, in 2002 and 2003 respectively, suggestions were made to extend the results and effects of this scientific network. Subjects for research were highlighted for co-operation in projects. Furthermore, analogue to the existing Database on Forest Disturbances in Europe (DFDE, see for more information *http://www.efi.fi/projects/dfde*), a disturbance dynamics research database could be developed. Besides results, investigated parameters like gap size, perimeter, pattern, disturbance frequency and intensity, size of the area researched *etc.* could be included in this digital database. In this way, existing literature can be categorised and form a guide for future research in forest disturbance dynamics.

Eventually, experiments with management schemes in ecosystem management can be adapted to the frequency, patterns and severity of the natural disturbance regime in an area. The availability of data and information on sites with corresponding characteristics is relevant in case existing data resources are insufficient. However, we should beware of the risk of simplifying reality. Researching different separate disturbance types is just a first step in understanding the complex synergism of natural processes and human activities in forested ecosystems.

References

Armstrong G. W. 1999. A stochastic characterisation of the natural disturbance regime of the boreal mixed-wood forest with implications for sustainable forest management. Canadian Journal for Forest Research, 29: 424-433.

Ban Y., Huacheng X., Bergeron Y. and Kneeshaw D. D. 1998. Gap regeneration of shade-intolerant *Larix gmelini* in old-growth boreal forests of north eastern China. Journal of Vegetation Science 9. Special features in vegetation science 15: 529-536.

Bergeron Y. and Leduc A. 1998. Relationships between the change in fire frequency and mortality due to spruce budworm outbreak in the south eastern Canadian boreal forest. Journal of Vegetation Science 9. Special features in vegetation science 15: 493-500.

Blackburn P., Petty J. A. and Miller K. F. 1988. An assessment of the static and dynamic factors involved in windthrow. Forestry, 61: 29-43.

Brandani A., Hartshorn G. S. and Orians G. H. 1988. Internal heterogeneity of gaps and species richness in Costa Rican tropical wet forest. Journal of Tropical Ecology, 4: 99-119.

Brokaw N. V. L. 1982. The definition of treefall gaps and its effects on measures of forest dynamics. Biotropica 14, 2: 158-160.

Canham C. D., Papaik M. J. and Latty E. F. 2001. Interspecific variation in susceptibility to windthrow as a function of tree size and storm severity for northern temperate tree species. Canadian Journal for Forest Research, 31: 1-10.

Johnson N. C., Malk A. J., Szaro R. C. and Sexton W. T. 1999. Ecological Stewardship - A Common Reference for Ecosystem Management. Vol. I. Elsevier Science Ltd.

Lorimer C. G. and Frelich L. E. 1989. A methodology for estimating canopy disturbance frequency and intensity in dense temperate forests. Canadian Journal for Forest Research, 19: 651-663.

Lorimer C. G., Frelich L. E. and Nordheim E. V. 1988. Estimating gap origin probabilities for canopy trees. Ecology, 69: 778-785.

Payette S., Filion L. and Delwaide A. 1990. Disturbance regime of a cold temperate forest as deduced from tree-ring patterns: the Tantaré Ecological Reserve, Quebec. Canadian Journal for Forest Research, 20: 1228-1241.

Quine C. P., Humphrey J. W. and Ferris R. 1999. Should the wind disturbance patterns observed in natural forests be mimicked in planted forests in the British Uplands?. Forestry, 72: 337-358.

Quine C. P. 2003. Wind-driven gap formation and gap expansion in spruce forests of upland Britain. Proceedings of International Conference ‘Wind Effects on Trees’. September 16-18. University of Karlsruhe. Germany.

Rogers P. 1996. Disturbance ecology and forest management: a Review of the Literature. US Department of Agriculture. Forest Service. Intermountain Research Station. 16 p.

Runkle J. R. 1985. Disturbance regimes in temperate forests. In: Pickett S. T. A. and White P. S. (eds.). The ecology of natural disturbance and patch dynamics. Academic Press. New York. 17-33.

Seymour R. S., White A. S. and deMaynadier P. G. 2002. Natural disturbance regimes in north eastern North America - evaluating silvicultural systems using natural scales and frequencies. Forest Ecology and Management, 155: 257-267.

Ulanova N. G. 2000. The effects of windthrow on forests at different spatial scales: a review. Forest Ecology and Management, 135: 155-167.

PROGRAMME OF THE WORKSHOP

Natural disturbances dynamics as component of ecosystem management planning

Monday October 13, 2003
Short presentations, plus questions and discussion
Morning session (moderator Adalsteinn Sigurgeirsson) 10.15-12.00

1. Welcome notes by Icelandic Forest Research.
   Dr Adalsteinn Sigurgeirsson 10.15
2. Disturbances in forest ecosystems in Iceland.
   Gudmundur Halldórsson 10.30
3. The history of native Icelandic woodlands.
   Olafur Eggertsson 10.55
4. Marine aerosol; a disturbance factor in coastal forests in Iceland
   Thorbergur Hjalti Jónsson 11.20
5. The modelling approach and quantitative analysis in disturbance studies.
   Dr. Kalev Jõgiste 11.45

Afternoon session 13.30-18.00 (moderator Kalev Jõgiste)

6. The problem of scale: Past and Present fire disturbances in Sweden.
   Dr. Mats Niklasson 13.30
7. Gap dynamics of forested ecosystems and disturbances.
   M.Sc. Floor Vodde 13.45
8. Some new research ideas about dead wood dynamics in beech forests.
   Katrine Hahn 14.00
9. Disturbances and the frame of network.
   Dr. Timo Kuuluvainen 14.15
10. Windthrow studies in Denmark.
    Eva Ritter 14.30
11. Forest disturbances in Lithuania.
    Kestutis Armolaitis 14.45
12. Discussion 15.00

Coffee break 15.00-15.30

Group work “Planned activities and prospective of the network”

1. Introduction to group work.
   Dr. Kalev Jõgiste 15.30

2. Group work 15.40-16.30

   1. What are our chances to apply for EU funding for bigger network and what (network) activities should be priority?

   2. Are we able to initiate the organisation of conference in 2005? We have discussed with representatives of several organisations the possibility to have a bigger event in co-operation with IUFRO. The preliminary topic could be the “Forest disturbance modelling”. We should negotiate about the IUFRO regulations.

3. Presentations of group works (Moderator Henn Korjus)

   16.30

Coffee break 17.15-17.30

4. Discussion 17.30

Also: We would like to discuss the possibility for compilation of paper set giving idea about of the state-of-the-art of disturbance studies.

5. Closing session 17.45
6. Concluding remarks 18.00

WORKSHOP "NATURAL DISTURBANCES DYNAMICS AS COMPONENT OF ECOSYSTEM MANAGEMENT PLANNING"

Geysir, Iceland, October 11-15, 2003

Saturday, 11.10.2003

Arrival at Keflavík airport, 1 hour S of Reykjavík
16.00
Arrival at Hotel Geysir, dinner
19.00

Sunday, 12.10.2003

8.30	Breakfast
10.00	Field excursion to Thingvellir, Mosfell, Gullfoss, the National Forest at Haukadalur
20.00	Dinner

Monday, 13.10.2003

8.30	Breakfast
10.00	Start of the workshop
10.15	Presentations
12.00	Lunch
13.30	Planning the network "Natural disturbances dynamics as component of ecosystem management planning"
Concluding remarks	
20.00	Dinner

Tuesday, 14.10.2003

8.30	Breakfast
10.00	Field excursion to Thjorsardalur, Rangarvellir, Fljótshlid
19.00	Arrival in Reykjavík, Dinner

Wednesday, 15.10.2003

5.00	Breakfast
6.00	Drive to Keflavík airport

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