

# Impacts of natural disturbances on the development of European forest resources.

Application of model approaches from tree and stand levels to large-scale scenarios

Mart-Jan Schelhaas



ALTERRA  
WAGENINGEN UR





Impacts of natural disturbances on the development of European forest resources: application of model approaches from tree and stand levels to large-scale scenarios

MISSION: Alterra is the main centre of expertise on rural areas and water management in the Netherlands. It was founded 1 January 2000. Alterra combines a huge range of expertise on rural areas and their sustainable use, including aspects such as water, wildlife, forests, the environment, soils, landscape, climate and recreation, as well as various other aspects relevant to the development and management of the environment we live in. Alterra engages in strategic and applied research to support design processes, policymaking and management at the local, national and international level. This includes not only innovative, interdisciplinary research on complex problems relating to rural areas, but also the production of readily applicable knowledge and expertise enabling rapid and adequate solutions to practical problems.

The many themes of Alterra's research effort include relations between cities and their surrounding countryside, multiple use of rural areas, economy and ecology, integrated water management, sustainable agricultural systems, planning for the future, expert systems and modelling, biodiversity, landscape planning and landscape perception, integrated forest management, geo-information and remote sensing, spatial planning of leisure activities, habitat creation in marine and estuarine waters, green belt development and ecological webs, and pollution risk assessment.

Alterra is part of Wageningen University Research Centre (Wageningen UR) and includes two research sites, one in Wageningen and one on the island of Texel.

Impacts of natural disturbances on the  
development of European forest resources:  
application of model approaches from tree and  
stand levels to large-scale scenarios

Mart-Jan Schelhaas

ALTERRA SCIENTIFIC CONTRIBUTIONS 23

ALTERRA, WAGENINGEN UR

2008

The research presented in this thesis was conducted at Alterra Green World Research in Wageningen, The Netherlands and the European Forest Institute in Joensuu, Finland.

This work was also published as *Dissertationes Forestales* 56, available at <http://www.metla.fi/dissertationes>. The work was supervised by Prof. Timo Pukkala from the University of Joensuu, Finland.

ISBN 978-90-327-0356-1

## Abstract

Natural disturbances can significantly affect the sustainable production of forest services. Until now there has been no concise overview of the damage such disturbances have caused to European forests, and their role in projection models has often been ignored. This dissertation aims to contribute in filling those gaps. A literature review in Paper I revealed that from 1950 to 2000 the annual average timber volume damaged by disturbances was 35 million m<sup>3</sup>: 53% by storms, 16% by fire, 8% by bark beetles and 8% by other biotic factors.

A natural disturbance module was added to a large-scale scenario model, which was then applied to Switzerland and Austria. For Switzerland, it was found that the inclusion of natural disturbances significantly affected the development of growing stock, both under current and changing climatic conditions (paper II). In Austria, climate change doubled the expected damage by bark beetles by the end of the century (paper III). Adaptation through replanting with different tree species after clear-felling had only a small mitigating effect, since older forests are the most vulnerable.

To study how silvicultural regimes affect the wind damage risk, a wind damage module was added to an individual-based forest simulator. The explicit inclusion of shelter and support from neighbouring trees enabled both individual tree and whole stand stability to be simulated in detail (papers IV-V). Silvicultural regimes leading to relatively low tree height to stem diameter (h/d) ratios experienced the least damage. Low h/d-ratios could be obtained by maintaining low stand densities in even-aged stands or by favouring trees with a low h/d ratio when thinning in uneven-aged stands. It is concluded that the inclusion of disturbances in projection models of different scales offers great possibilities for exploring alternative scenarios and associated risks, for example for adapting to expected future climate change.

**Keywords:** abiotic damage, biotic damage, wind damage, bark beetles, simulation model

## Acknowledgements

Finally the moment is there to write one of the last pieces of this dissertation: the acknowledgements! Indeed a sign that the end of the journey is near. Sometimes I still can't believe it's really happening. Combining work, a family and a PhD called for creative working hours and places. Paper IV was finished during a parental leave. The last parts of paper V and the summary were finished in hotel rooms in Bulgaria, in between extensive lunches, dinners and breakfasts. I submitted it from the university in Sofia. I started putting everything in the right format at the airport of Vienna late noon, and finished it after midnight at the railway station of Utrecht, waiting for the train that would bring me home.

In the long list of people to thank, my "boss" Gert-Jan Nabuurs undoubtedly deserves the first place. You have given me many opportunities over all these years and it was you who first launched the idea of doing my PhD on natural disturbances. You introduced me to the world of science, modelling and travelling around the globe. Besides work I enjoyed your company in many other activities, like playing squash, running, mountain biking and snowball fights in- and outside the sauna. Yes, the sauna evenings are a thing to remember, like our attempt to attain 100% humidity in sauna. And no matter how busy work was, there was always time for a laugh!

My supervisor at the University of Joensuu was Timo Pukkala. Usually I would come to your office once a year, explaining that the progress I made was again less than I expected, and invariably I would come with a new optimistic estimate for the future. Thanks for your patience over the years and your guidance in writing the articles and the summary. Another person at the University that I owe great thanks is Heli Peltola. After our first contact you got involved more and more, and without you this dissertation would have been much harder to finish. You were always ready to answer my sometimes very detailed questions about HWIND, provide lengthy comments on manuscripts and doing practical things for me that are easy if you are in Finland, but impossible if you are somewhere else. And it was you that came up with the idea of introducing tree interactions in the fourth article. I'm still very happy about that!

The connection to Finland started in 1998 with a scholarship at the European Forest Institute (EFI), and many shorter and longer visits followed. EFI is a very nice place to be, with an interesting mix of people, nationalities and cultures. The working environment was always very good, mainly because of all the nice people. I will not try to list all of them, so just thanks to everybody! Just two exceptions: I had great times sharing the office with my "big brother" Ari Pussinen. We had big discussions on EFISCEN, and you were a master in drawing graphs and pictures, and explaining them. Although not directly involved in the dissertation work, you always had a clear view on where to go and what to do. They sure were helpful! A very special word of thanks goes to the Schuck family: Andreas, Annette, Lukas, Judith and Finn. Andreas



introduced me to cross-country skiing, plop fights and music styles I didn't know they existed. Annette "forced" me to speak German, took me to concerts at the university, and we shared many board games and walks. The children adopted me as a kind of second daddy ("Papa Mart-Jan") and we had great fun together. Yes, I can surely say that you provided me with a second home in Finland.

Also at Alterra many colleagues were involved directly or indirectly in this work. During the literature review, the people at the library were very helpful in finding even the most obscure references. They even had to upgrade me to "special user" due to the large number of publications I was borrowing. Koen Kramer allowed me to use, modify and expand the ForGEM model. Bert van der Werf assisted in programming, model design, bug hunting and statistical issues. The many discussions on forest management and windthrow with Sander "hee bedankt" Wijdeven were helpful in getting the model and manuscripts in shape. In ForGEM matters, also Isabel van der Wyngaert was a good sparring partner. Thanks to all of you! And of course thanks to all the other colleagues for relaxing and humorous coffee breaks, sometimes badly needed to get out of the work for a moment. With this respect I surely have to mention Sandra Clerkx, always ready for a chat, a joke, a gossip, chocolate, whatever as long as it is not work related. Thanks for sharing the office with me, respecting the "ear-plug" times and joining me on Fridays at 16:30 visiting the "magnet" downstairs.

Then there were many colleagues outside Alterra and EFI. Thanks to all those people who helped me with finding and translating local sources of information for the first paper. During the many travels, many people tried their best to let me feel at home and showed me around. I appreciate that a lot! Also many thanks to Rupert Seidl. Your scholarship proposal came as a welcome surprise to me. Besides the efficient cooperation, we had nice discussions on work and non-work related topics. I hope we can continue to collaborate in future! Although much of the work in this dissertation was done in the evening hours, some project time was spent on it as well. Particularly CAMELS, EFORWOOD and ADAM were projects that were closely related to the work in this dissertation. Furthermore, Alterra provided me the opportunity to follow courses for the study plan and resources to finish the work. Of course there was life besides the PhD work as well. I spent lots of time with Detmer behind the computer, "working" together on "reports", "articles" and other "useful" things. Those times I could really forget about anything else! Also having my weekly daddy-day helped in putting everything back into the right perspective. And whenever I needed to travel, my and Sandra's parents were always ready to step in. I don't know how we could have managed without you. My sincere thanks! And this brings me to the last person I really need to thank: my wife Sandra! Thank you for being my companion all this time and for your support, patience and love.

# Contents

Abstract	5
Acknowledgements	6
1 Introduction	10
1.1 Development of forest resources and the role of natural disturbances	10
1.2 Assessment of forest resource development and risks of natural disturbances	12
1.3 Aims of this thesis	13
2 Natural disturbances in the european forests in the 19 <sup>th</sup> and 20 <sup>th</sup> centuries: a review (paper I)	14
3 Model approaches and applications (papers II-V)	15
3.1 Large-scale scenario modelling including damage by various natural disturbances (paper II)	15
3.2 Large-scale scenario modelling including damage by bark beetles (paper III)	17
3.3 Stand-level modelling for assessment of risks from wind damage (papers IV-V)	17
4 Results	18
4.1 Large-scale scenario modelling application including damage by various natural disturbances (paper II)	18
4.2 Large-scale scenario modelling application including damage by bark beetles (paper III)	19
4.3 Stand-level modelling application for wind risk (papers IV-V)	20
5 Discussion and conclusions	22
5.1 Evaluation of the large-scale scenario model including the overall disturbance module (paper II)	22
5.2 Evaluation of the large-scale scenario model including the bark beetle damage module (paper III)	22
5.3 Evaluation of the stand-level application for assessment of risks from wind damage (papers IV –V)	23
5.4 Conclusions	24
6 References	26

## *Papers*

I	Natural disturbances in the European forests in the 19 <sup>th</sup> and 20 <sup>th</sup> centuries	33
II	Adding natural disturbances to a large scale forest scenario model and a case study for Switzerland	59
III	Modelling bark beetle disturbances in a large scale forest scenario model to assess climate change impacts and evaluate adaptive management strategies	81
IV	Introducing tree interactions in wind damage simulation	117
V	The wind stability of different silvicultural systems for Douglas fir in the Netherlands: a model based approach	146

# 1 Introduction

## 1.1 Development of forest resources and the role of natural disturbances

European forests are among the most intensively managed forests in the world. As a result, although only 5% of the global forest area is located in Europe (193 million hectares, excluding the Russian Federation), European forests account for 23% of the global round wood removals (FAO, 2007). Removals have increased: from just over 400 million m<sup>3</sup> in 1990 to nearly 500 million m<sup>3</sup> of timber in 2005 (FAO, 2007). In addition, removals of non-wood forest products are estimated to be over half a million tonnes per year (FAO, 2007). Meanwhile, other forest functions are also becoming more important: for example, the area designated primarily for nature conservation has tripled (from 6.6 million ha in 1990 to 20.3 million ha in 2005: FAO, 2007) and now accounts for 10.5% of the total forest area. A further 19.5 million ha (10.1%) has been assigned a primarily protective function (protection of soil, water, air and infrastructure, for example). Another important function of the forest, especially in more urbanised regions, is recreation. UN-ECE (2005) has estimated that the total number of person-visits to European forests was roughly 3.6 billion per year. Further, forests can play a significant role in combating climate change. Carbon stocks in European forests amounted to 19.5 Pg C in the 1990s, and have been increasing by 0.14 Pg per year (Nabuurs et al., 2003). Not surprisingly, forests can be an important source of bioenergy (EEA, 2006), and the higher oil prices and greater interest in bioenergy are likely to increase the demand for woody biomass from forests (IPCC 2007).

Despite their intensive use, the forest resources in Europe are growing. The forest area is currently increasing by about 760 thousand ha per year (0.4%; FAO, 2007) and the average growing stock volume increased from 124 m<sup>3</sup> ha<sup>-1</sup> in 1990 to 141 m<sup>3</sup> ha<sup>-1</sup> in 2005 (FAO, 2007). These trends are projected to continue for at least several decades to come (Schelhaas et al., 2006). Despite these positive trends, there are other developments that are cause for concern. One of these is natural disturbances (FAO, 2007). These disturbances can cause unforeseen loss of living forest biomass and reduce the value of the timber or forest stand. They play a significant role in forest development and also affect the projections of the development of forest resources. Consequently, disturbances are highly relevant factors in the sustainable management of forest ecosystems (FAO, 2007). However, information on the historical and current role of disturbances in European forests is very patchy and their role in forest projection models is usually ignored.

Risks, such as natural disturbances, can theoretically be described by the surface area of a triangle (Kron, 2002), where the sides represent hazard, exposure and vulnerability. Any change to one of the sides of the triangle will lead to a

corresponding change in risk level. Hazard could be seen as a probability of occurrence, which in forestry is often related to the climate. For example, fire hazard depends for a large part on weather conditions, as it is increased by the factors high temperature, lack of precipitation and presence of wind. Another hazard factor is the presence of an ignition source. Most of the fires in Europe are caused by humans, either intentionally or through negligence. Only very few fires are ignited by lightning. Another hazard – that of wind – is entirely related to the climate. European countries with Atlantic coastlines generally experience higher wind speeds, with Ireland and the United Kingdom in particular having a rather severe wind climate. Also, mountainous regions generally experience a more severe wind climate due to the topographical influences.

Compared to fire and wind hazards, the biotic hazard is more difficult to characterise because of the great diversity of agents of biotic disturbance. Many insects are favoured by warm and dry conditions (Speight and Wainhouse, 1989), but fungi favour more moist conditions (Coakley et al., 1999). Low winter temperatures may reduce insect survival rates (Leather et al., 1993), but may on the other hand be favourable for the synchronisation of life cycles with host plant species (Buse and Good, 1996). In general, biotic hazard seems to be less in more extreme climatic conditions (Jactel et al., 2007). Further, biotic hazard depends on the dispersion capacity of the agent as well as on the presence of natural enemies. The latter might be related to many factors.

The exposure side of the triangle can be expressed in terms of the values that are at stake. In principle these values includes all forest functions (such as protection, timber production, biodiversity, landscape, amenity), but many of them are difficult to quantify. Therefore, in this thesis only the two variables that are easiest to quantify are taken into account as a proxy for exposure of forest to risk: forest area and wood volume. Vulnerability expresses how easily a forest is damaged by the agent under consideration. It can usually be linked to the actual state of the forest. For example, coniferous species generally burn more readily than broadleaved species (Meyer, 2005) and fire spreads more easily in young and dense stands (Vélez, 1985; Brown and Smith, 2000). A forest's vulnerability is also heightened by the presence of flammable material on the forest floor (litter and humus, for example).

Tree species composition is also important for vulnerability to wind. Coniferous species are considered more vulnerable to wind damage than broadleaved species, with Norway spruce (*Picea abies* (L.) Karst) being regarded as particularly vulnerable (Schütz et al., 2006). However, Norway spruce has often been used to afforest sites that have unfavourable soil conditions and are waterlogged (Böhm, 1981). Though rooting depth is an important factor in vulnerability to wind, many other factors play a role as well. Wind vulnerability is a very complex field of research, often resulting in conflicting findings. The tree and stand variables most important with regard to tree stability are tree height and the ratio of tree height to stem diameter at breast height (h/d ratio). Taller trees are more exposed to the wind than shorter

ones, for example. Because stand or tree age is highly correlated with tree height, age is often used as a proxy for height.

Vulnerability to biotic agents is again difficult to characterise and largely depends on the specific agent under consideration. Some agents are specific to certain tree species or stages of stand development. Others are generalists, damaging trees regardless of whether they are young or old. Overall, broadleaved tree species seem to have more associated insect species than conifers (Southwood, 1961; Brandle and Brandl, 2001). However, tree species react differently to damaging agents. For example, broadleaves can react immediately to defoliation by forming new leaves and shoots, whereas defoliation in conifers can remain visible for several years. In general, trees will be more vulnerable to biotic agents if they are more exposed to stress (drought, extreme temperatures, pollution, damage from logging, wind or fire, etc.). Under such circumstances, many secondary agents are able to cause severe damage. In such cases, it often proves difficult to pinpoint one cause of tree mortality, which is why the term “complex” damage is often used.

## **1.2 Assessment of forest resource development and risks of natural disturbances**

In order to sustain several different functions simultaneously, large areas of forests should be managed as multi-purpose forests. The increasing pressure from society on the forests to fulfil a diversity of functions calls for targeted policies and careful planning and management. However, it is not easy to oversee impacts and consequences of management and policy decisions on the whole range of forest functions. Thus, there is a clear need for numerical analysis tools; currently, however, only a few are available.

In general, mathematical models can be used to objectively quantify risks and can show the long-term implications of selected actions. Furthermore, such detailed tools are valuable for deriving vulnerability ratings over a large range of possible situations; such ratings are difficult or even impossible to obtain from field studies. One spin-off of the increased interest in multi-purpose forestry and nature-oriented management has been the advances made recently in modelling growth and yield (Hasenauer, 2006). Detailed tree-level simulation models are needed to cope with uneven-aged and mixed species forests. Some of these models have been extended to include specific disturbances.

In recent years, substantial progress has also been made in the modelling of the mechanisms of wind damage to tree stands. Peltola et al. (1999) and Gardiner et al. (2000) have developed mechanistic models to estimate, for a given tree, the critical wind speed needed for stem breakage or uprooting. In these models, the stand under study is represented, however, by one “average” tree, assuming even-aged, mono-species stands with relatively little variation in height or diameter. In stands with higher variability among trees, such as in unmanaged or uneven-aged stands, a

different approach is needed. Ancelin et al. (2004) recently extended this approach and evaluated the risk of wind damage for all trees in a particular stand. All these recent models evaluating the stability of individual trees largely ignore interactions with neighbouring trees. However, an important factor in stand stability is crown contact between trees. Not only can trees dissipate absorbed wind energy through crown contact (Milne, 1991), they can also physically support each other (Quine et al., 1995). One of the reasons for increased wind damage risk after thinning is that trees receive less support from their neighbours (Ruel, 1995). Thus, this modelling approach could still be further refined, making use of the improved shelter and support mechanisms between adjacent trees in a stand. At the same time, introducing such a mechanistic wind damage module into a stand simulator would make it possible to evaluate the risk of wind damage over time in relation to forest growth and dynamics as affected by silvicultural management.

In policy decisions, scenario projections can play an important role when forecasting the long-term impacts of alternatives in forest policy and management (Nabuurs and Päivinen 1996, Nabuurs 2001) and how they influence the fulfilment of forest functions. At the European scale the most widely used tool for such projections is the simulation model EFISCEN (European Forest Information SCENario model). It has, among other things, been used to study forest resource development under scenarios of demand for timber (Schelhaas et al., 2006; Nabuurs et al., 2007), climate change (Nabuurs et al., 2002; Schröter, 2004) and changing management regimes (Nabuurs, 2001; Schelhaas et al., 2007a). Furthermore, it has been used to assess the possible contribution of forests to biomass supply for bioenergy (EEA, 2006). Expected climate change impacts have already been incorporated in EFISCEN via the use of growth modifiers (Nabuurs et al., 2002), which can be derived, for example, from detailed process-based models. However, EFISCEN still needs to be refined to take into account the effect of natural disturbances, such as fire, storm, snow damage and biotic damage in projections made for current or changing climate conditions.

### **1.3 Aims of this thesis**

The main aim of this thesis is to increase the understanding of the role of natural disturbances in the past and future development of European forest resources. Within this context, the special objectives of papers I-V were as follows:

- To present a comprehensive overview of past forest disturbances in Europe (Paper I).
- To extend the large-scale scenario model EFISCEN with a module that takes into account the influence of natural disturbances and that is able to handle hypothetical changes in disturbance regimes due to climate change. The extended model was parameterised for and applied to Switzerland, in order to study the behaviour and impact of the disturbance module (Paper II).

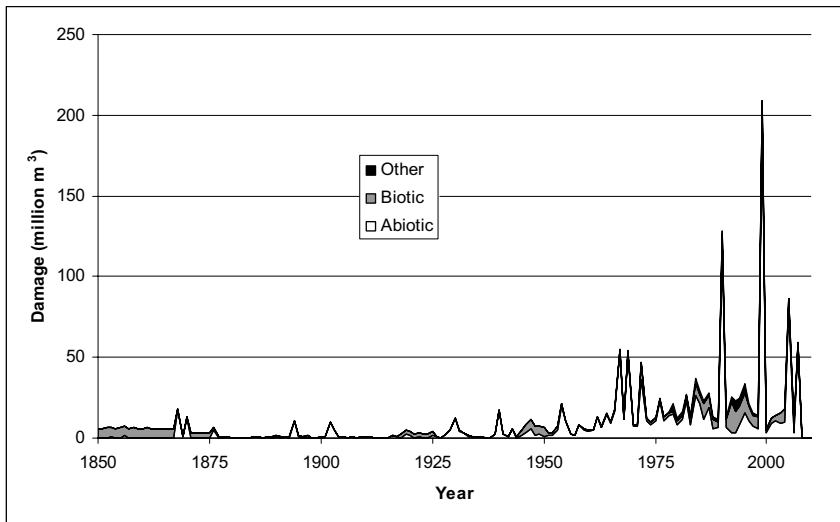
- To improve the EFISCEN disturbance module with respect to the simulation of bark beetle impacts under changing climate. This new module was applied to investigate the effects of adaptive management strategies on bark beetle damage in Austrian forests. A secondary aim of paper III was demonstrate the suitability of EFISCEN as a tool for upscaling the results of a more detailed model (Paper III).
- To develop a tool that can be used in quantifying the vulnerability of tree stands to wind damage under a range of stand conditions, in order to support forest management by predicting the consequences of different management actions at the stand level (Paper IV). The tool was used and demonstrated to evaluate the risk of wind damage for a variety of management regimes of Douglas fir at stand level (Papers IV and V).

## **2 Natural disturbances in the european forests in the 19<sup>th</sup> and 20<sup>th</sup> centuries: a review (paper I)**

Reports going back for more than 600 years show that natural disturbances are not a new phenomenon in Europe. However, early reports are scarce and no analysis can be done on these occurrences. Over time, the reports become more abundant, but not until after about 1950 are enough data available to allow missing data to be estimated. Over the period 1950-2000 an annual average of 35 million m<sup>3</sup> timber was damaged by disturbances; there was much variation between years. Storms were responsible for 53% of the total damage, fire for 16%, snow for 3% and other abiotic causes for 5%. Biotic factors caused 16% of the damage, with bark beetles being responsible for half of this. The bark beetle outbreaks were usually connected to severe windfall events, often in combination with adverse weather conditions in the following summers. For 7% of the damage no cause was given, or there was a combination of causes. The 35 million m<sup>3</sup> of damaged timber corresponds to about 8.1% of the total fellings in Europe and to about 0.15% of the total stem volume of growing stock.

Over the period 1961-2000, the average annual area affected by forest fires was 213,000 ha, which is 0.15% of the total forest area in Europe. Almost half (44.9%) of the total area affected by forest fire is in just two countries: Spain and Portugal. The total Mediterranean area (including France) accounted for 93.6% of the area burned.





**Figure 1.** Reported volume of timber affected by natural disturbances in Europe, 1850-2007 (Paper I).

Though Figure 1 shows that most types of damage seem to be increasing, this is partly an artefact of the improved availability of information. However, more complete time series at national and regional levels confirm the rising trend, at least for storms (Holmsgaard, 1986; SFSO and FOEFL, 1996; Mosandl and Felbermeier, 1999) and fires (CREAF, 1999; Xanthopoulos, 2000; Konstantinov, 2003; Meta, 2003). At the same time, no study has found evidence of storms being more frequent or more intense (Schiesser, 1997; Dorland et al., 1999; Können, 1999; Lässig and Mocalov, 2000). The most likely explanations for an increase in damage from disturbances are changes in exposure, such as increases in forest area and average volume of growing stock, and changes in vulnerability, such as increased average stand age and a larger proportion of conifers.

### 3 Model approaches and applications (papers II-V)

#### 3.1 Large-scale scenario modelling including damage by various natural disturbances (paper II)

In paper II, a disturbance module is added to the large-scale scenario model EFISCEN (European Forest Information SCENario model) whose core dynamics are based on a model developed by Sallnäs (1990). EFISCEN is especially suitable for regional or country level applications, in which national forest inventory data are used as input. The state of the forest is depicted as an area distribution over age and stem

volume classes in a matrix. A separate matrix is set up for each forest type, defined by region, owner class, site class and tree species. Processes such as growth, mortality, thinnings, final fellings and regeneration are simulated by transitions of the area to other volume classes. A detailed description of EFISCEN can be found in Schelhaas et al. (2007b).

In this research, the three most important disturbance agents in Europe were included in EFISCEN: fire, storm/snow and insects. All these agents are able to destroy a stand wholly (stand-replacing disturbances) or partially (non-stand-replacing disturbances). Both outcomes are simulated by movements of area through the matrix, in a way similar to that for final fellings (stand-replacing disturbances) and thinnings (non-stand-replacing disturbances). For each combination of forest type, disturbance agent and type of disturbance, a vulnerability matrix is set up. This matrix expresses the vulnerability of each cell in the matrix relative to all other cells in all matrices. The hazards for damage by storm/snow and fire are defined by lognormal distributions. The hazard for insect damage depends on the occurrence of storm damage and warm and dry years. Fire hazard is taken as a proxy for warm and dry years. Actual damage in a certain time step depends on the hazard (represented by random draws from the hazard distributions), vulnerability (as defined in the vulnerability matrices) and exposure (defined by the distribution of area over the matrix). Because of the stochastic character of the approach, Monte Carlo simulation is used.

In paper II, EFISCEN was applied to Switzerland to study the behaviour and impact of the newly developed disturbance module on projections of forest resources. For this purpose, a 60-year period was simulated, with and without the disturbances module. In both cases, the felling volume required (planned fellings) was kept the same. In the first case, the volume of unplanned fellings due to disturbances was kept constant, equal to the historical level. In the second case, the volume of unplanned fellings was determined by the disturbances module. Furthermore, a climate change scenario was applied, where the degree of disturbance and the size of the wood volume increment were increased simultaneously. The climate change assessment is based on the HadCM2 scenario (Mitchell et al. 1995), which assumes that in the period 1990 to 2100, the concentration of atmospheric CO<sub>2</sub> doubles. On average, the climate scenario predicted that mean annual temperature would be 1.5 °C higher in 2050 compared with 1990, and that during the same period, average annual precipitation would increase by between 5% and 15%. Associated changes in the size of the wood volume increment were derived from earlier detailed simulations with the TREEDYN3 model (Bossel, 1996; Sonntag, 1998, Kramer and Mohren, 2001).

### **3.2 Large-scale scenario modelling including damage by bark beetles (paper III)**

In paper III, EFISCEN was applied to all Norway spruce forests in Austria to study the development of damage due to the European spruce bark beetle (*Ips typographus* (Scol. Col.)). In this application, vulnerability and hazard were calculated outside the model and no Monte Carlo approach was applied. Two adaptive management scenarios were compared with a “business as usual” scenario. Adaptive management consisted of changing the tree species distribution to more closely resemble the potential natural vegetation (PNV). In the first adaptation scenario, tree species change was implemented immediately, whereas in the second, the species change was delayed by 20 years. All three management scenarios were evaluated under current and changing climate conditions. The climate change scenario was based on the B2 emission scenario of the IPCC (2000) as predicted with the climate model HadCM3 (Mitchell et al. 2004). The average temperature change for the last decade of the 21st century relative to the period 1990-2004 at was +2.4°C, averaged over whole Austria. Precipitation increased only slightly (+20mm), with limited decrease and increase in the individual provinces. The LPJ model (Schröter, 2004) was used to derive changes in wood volume increment under climate change, while the PICUS model (v 1.41, Lexer and Hönninger, 2001; Seidl et al., 2005; Seidl et al., 2007) was used to assess bark beetle hazard and vulnerability.

### **3.3 Stand-level modelling for assessment of risks from wind damage (papers IV-V)**

In paper IV, a wind damage module is added to the ForGEM (Forest Genetics, Ecology and Management) model, henceforth referred to as ForGEM-W. ForGEM is a forest model that simulates the growth and development of individual trees on a scale of up to several hectares (Kramer, 2004; Kramer et al., 2007). Most of the key processes are modelled essentially similar to those of SORTIE (Pacala et al., 1993; Pacala et al. 1996), except for the gap-type approach for light interception (Bugmann, 2001). Individual tree growth is driven by light interception. Intercepted light is transformed into photosynthates, which are allocated to specific parts of the tree. Crown expansion is influenced by competition from other trees. Regeneration is explicitly simulated by the processes of seed production, dispersal and germination.

The wind damage module uses a static mechanistic approach. It determines for a given mean hourly wind speed which trees will be damaged. Tree height and crown and stem shape determine how much wind drag the tree experiences. The weight of the displaced stem and crown add to the total turning moment at stem base, which is also affected by the shelter and support received from surrounding trees. The

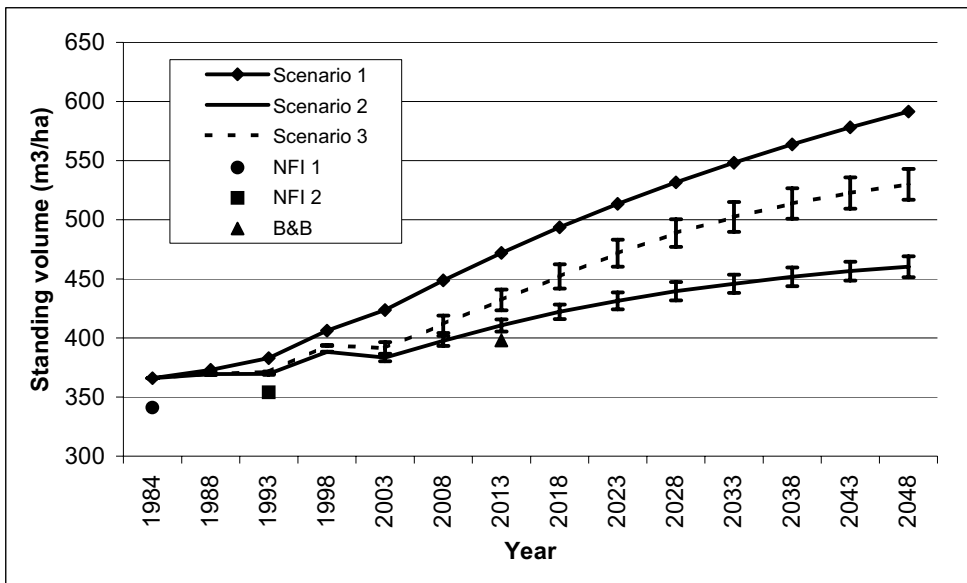
sheltering effects depend on relative heights of trees and the presence of foliage. Trees can also experience additional loading if hit by fallen trees (the domino effect), which affects the total turning moment, too. However, trees with a height of less than 5m are not expected to be uprooted or broken by wind, though they can be destroyed by other falling trees. Gaps in the stand are also explicitly taken into account in determining wind load and shelter zones. Trees are assumed to break or be uprooted if the maximum stem resistance or the maximum anchorage resistance, respectively, is exceeded. Both stem resistance and anchorage are functions of diameter at breast height (DBH).

In a case study, the ForGEM-W model was parameterised for Douglas fir (*Pseudotsuga menziesii* (Mirb.) Franco) in the Netherlands. Damage patterns were explored in three simulated 60-year old stands with different management history at two different wind speeds (paper IV). These situations were also used to conduct a sensitivity analysis of the model. Furthermore, six different management regimes were evaluated over a full rotation period, to compare their effectiveness in reducing wind damage (paper V).

## 4 Results

### 4.1 Large-scale scenario modelling application including damage by various natural disturbances (paper II)

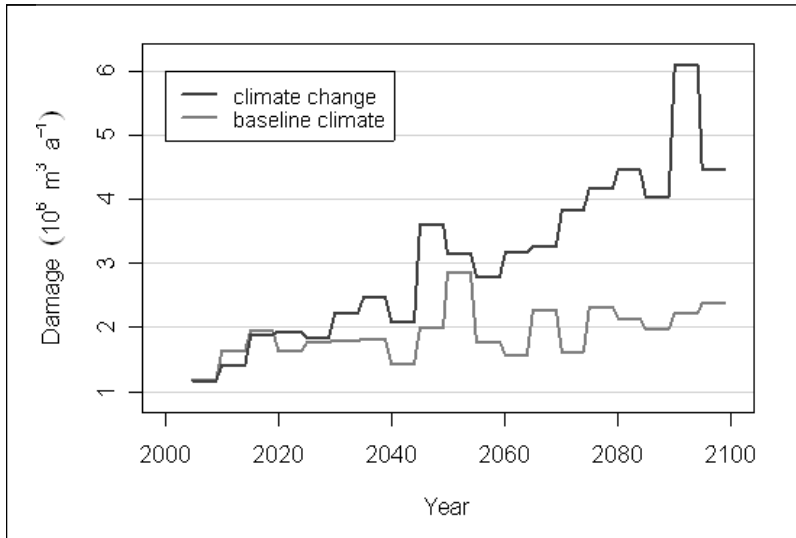
Without the natural disturbances module, EFISCEN projected that the growing stock volume in Switzerland would increase from 366 m<sup>3</sup> ha<sup>-1</sup> in 1984 to 592 m<sup>3</sup> ha<sup>-1</sup> in 2048. When the natural disturbances module was included, the growing stock increased to only 460 m<sup>3</sup> ha<sup>-1</sup> in 2048 (Figure 2). The difference is attributable to changes in the vulnerability of the forest, which cause the impact of natural disturbances to increase over time: from 2.8 m<sup>3</sup> ha<sup>-1</sup> in 2003 to 3.9 m<sup>3</sup> ha<sup>-1</sup> in 2048. Under a simulated climate change scenario, the frequency of disturbances was assumed to increase, which resulted in 25% more damage. However, the increment increased more than the damage done by disturbances, which resulted in a simulated growing stock volume of 530 m<sup>3</sup> ha<sup>-1</sup> in 2048. The increase in damage must mainly be attributed to an increasing average volume of standing timber.



**Figure 2.** Development of standing stem volume in Switzerland under 1) current climate and without disturbances (Scenario 1), 2) current climate with disturbances (Scenario 2) and 3) climate change and disturbances (Scenario 3), with standard deviation. Also shown are the standing volume as given in National Forest Inventory 1 (NFI 1) (Mahrer 1988), NFI 2 (Brassel and Brändli 1999) and corresponding stem volume (B&B) according to the projection by Brassel and Brändli (1999). The initial difference in standing volume is due to the fact that temporarily unstocked areas were not included in the EFISCEN dataset.

## 4.2 Large-scale scenario modelling application including damage by bark beetles (paper III)

EFISCEN projects that damage due to bark beetles in Austria will increase dramatically under the climate change scenario: from 1.33 million  $\text{m}^3 \text{yr}^{-1}$  in the period 1990–2004 to 4.46 million  $\text{m}^3 \text{yr}^{-1}$  in the period 2095–2099, with a peak of 6.09 million  $\text{m}^3 \text{yr}^{-1}$  (Figure 3). Under the current climate with business as usual management scenario, damage in the period 2095–2099 amounted to 2.38 million  $\text{m}^3 \text{yr}^{-1}$ . Average damage per hectare was found to be highest in pre-alpine Norway spruce stands, but simulated increases in accumulated damage under climate change were greatest in the inner Alps (+166%). The two adaptive management strategies (i.e. species change) revealed a considerable time-lag between the start of adaptation measures and a decrease in simulated damage by bark beetles. When EFISCEN simulations including the new bark beetle disturbance module were compared with 15 years of observed bark beetle damage for Austria there was also good agreement between the observed and predicted data at province level.



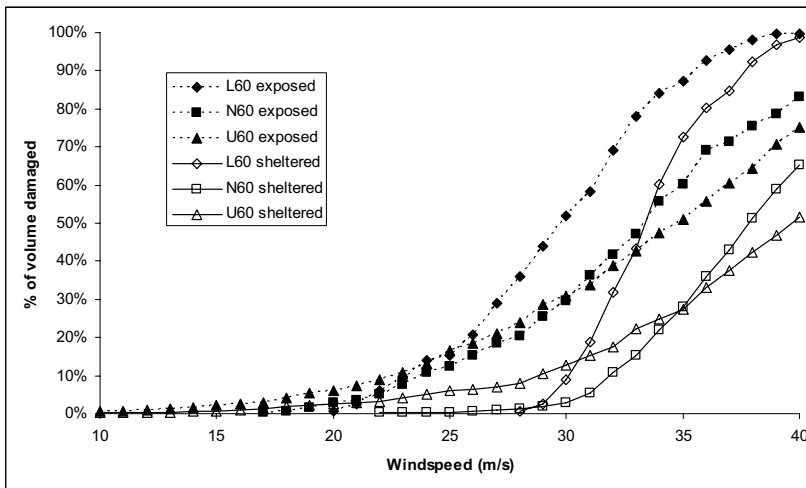
**Figure 3.** Development of total damage in Austria from bark beetles under current and changing climate conditions.

### 4.3 Stand-level modelling application for wind risk (papers IV-V)

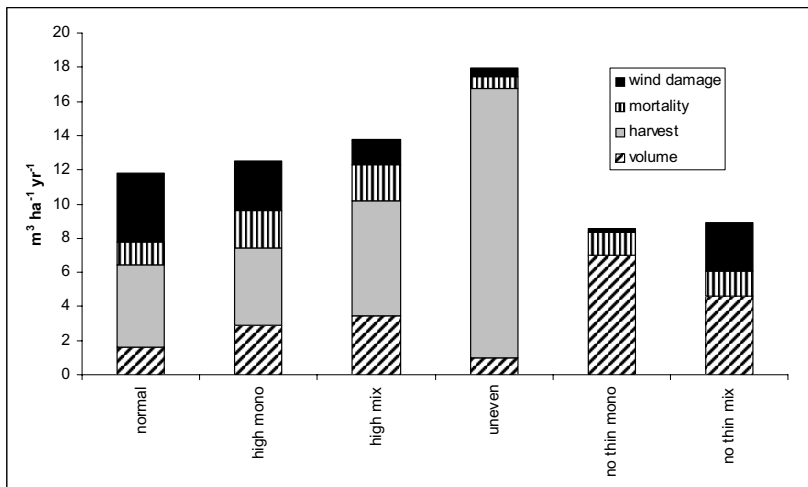
First, the wind damage patterns in three forest stands with different management histories were studied using the ForGEM-W model (paper IV). At the same wind speed, stands freshly exposed to wind showed considerably more damage than sheltered stands (Figure 4). In exposed stands, the damaged trees were located significantly closer to the upwind edge than undamaged trees. In sheltered stands, trees with a higher height to diameter ratio ( $h/d$  ratio) than average were most sensitive to wind damage, but lower individual tree stability in dense stands was clearly compensated for by the support of other trees. The wind speeds needed to cause damage approximated those of known windthrow events.

The wind damage module proved to be especially sensitive to the parameters determining resistance to uprooting and to the drag coefficient. Two of the tree characteristics to which the wind damage module proved to be very sensitive were tree height and diameter at breast height (paper IV). This was also found in paper V, in which it was reported that the management regimes that were most successful in avoiding wind damage were those that led to relatively low  $h/d$  ratios (Figure 5): for example, low  $h/d$  ratios could be obtained in even-aged situations by having a relatively low stand density throughout the rotation; in uneven-aged systems they could be obtained by favouring trees with relatively low  $h/d$  ratio in thinning. Admixture of a stable species (beech (*Fagus sylvatica* L.)) resulted in less damage only when the beech replaced Douglas fir. Admixture of beech in a Douglas fir stand with wide spacing increased the damage in Douglas fir,

because the  $h/d$  ratios of the Douglas fir increased in response to the increased competition from the beech.



**Figure 4.** Percentage of standing volume damaged at different wind speeds for exposed and sheltered stands of different tree densities at age 60 (L=low density, N= normal density, U=unmanaged)



**Figure 5.** Average gross annual stem wood increment and its components, averaged over the period needed to obtain an average diameter of 60 cm or over a period of 100 years, in different management scenarios. Normal: thinning from below, following yield table density; high mono: free thinning from above, monoculture Douglas fir; high mix: free thinning from above, equal mixture of Douglas fir and beech; uneven: uneven-aged management; 200 mono: monoculture of 200 Douglas fir trees, no thinning; 200 mix: 200 Douglas fir trees with admixture of 3800 beech trees, no thinning.

## **5 Discussion and conclusions**

### **5.1 Evaluation of the large-scale scenario model including the overall disturbance module (paper II)**

The EFISCEN model is designed to work on a large scale and should be able to be applied throughout Europe. Because of its general applicability, however, the model cannot take account of local circumstances, such as the complicated relief in Switzerland, in great detail. Another point worthy of note is the assumed distribution of the damage due to disturbances. In the Swiss case, these distributions were derived from historical records. However, a better method would be to couple the observed climate directly to disturbance damage, thus also enabling a better assessment system for future climate conditions. There is also great scope for improving the quantification of volume increment changes due to climate change. The approach used here must be seen as a what-if scenario to demonstrate the model's ability and the consequences of such a scenario.

The projections for Switzerland showed large differences in forest resource development depending on whether or not natural disturbances were included. This shows that ignoring natural disturbances in resource projections can lead to biased and over-optimistic views. Despite the sometimes rather crude assumptions in the natural disturbance module, the model projections correlated well with other projections (Brassel and Brändli, 1999). However, the model approach has various limitations and still has scope for improvement, as discussed above.

### **5.2 Evaluation of the large-scale scenario model including the bark beetle damage module (paper III)**

The simulations in Austria showed good agreement between the model and observations of stem wood increment levels, increases in the volume of standing growing stock and bark beetle damage. The model results were the outcome of combining the use of simulated forest structures in EFISCEN with an upscaling of PICUS model logic on bark beetle damage. The current implementation of bark beetle damage in PICUS was found to be very suitable to the requirements of such upscaling. Whereas the absence of year-to-year fluctuations in bark beetle populations damage has to be seen as a major limitation of the current approach in PICUS (cf. Seidl et al. 2007), the annually independent calculation fitted well for an upscaled application in the EFISCEN environment.

A major limitation of the approach presented for Austria is the implicit assumption of an average amount of material for bark beetles to breed in, not



dependent on wind or snow damage events. In view of this limitation, the projections have to be seen as indicative investigations of the climate dependencies of the herbivore–host relationship. Since several studies point at the possibility of increases in extreme weather events such as storms under climate change (e.g., Leckebusch and Ulbrich, 2004), the results presented in the scenario analysis have to be seen as conservative estimates.

The investigated adaptation scenarios of changing the tree species composition after clear felling proved to be ineffective over the time horizon considered. Even if the tree species change were implemented immediately, the first effects would not become visible until the second half of this century. Additional strategies are to be tested, for example involving increasing the harvest volumes in highly vulnerable areas, reducing rotation lengths, or introducing other tree species in existing stands.

### **5.3 Evaluation of the stand-level application for assessment of risks from wind damage (papers IV–V)**

As demonstrated in papers IV–V, the inclusion of a shelter and support mechanism in the wind damage module of ForGEM is an important improvement in the modelling of wind damage within a large range of different stand types. The approach enables the balance between individual tree stability and stand stability to be studied in much more detail. However, much work is still needed, not only to improve and calibrate the model, but also on other aspects of the wind damage module. Two particularly important issues which should be considered in greater detail in the future are the acclimatisation of trees to windy conditions, and the anchorage component. Within the tree growth model, special attention should be paid to competition processes, both above and below ground, and to the process of declining diameter growth with age. There are currently no data sets available that allow a validation of the wind damage module.

The applications of ForGEM-W clearly show how management can influence a stand's vulnerability and exposure and thus the risk of wind damage. At the landscape scale, vulnerability can be influenced by planning clear fellings in such a way that damage could be minimised at the newly created edges (Zeng 2006, Zeng et al. 2007). At an even larger spatial scale, an evenly distributed area over all age classes might be an effective way of limiting risks (Savill, 1983). At such large scales, EFISCEN can be a valuable tool to evaluate the development of damage patterns under different scenarios. Model results at a smaller scale, such as those obtained using ForGEM-W or PICUS, can also be useful in parameterising the disturbance module of EFISCEN.

## 5.4 Conclusions

The historical evaluation showed a tendency for damage from most disturbance agents to increase. This tendency can be explained in terms of changes in hazard, vulnerability and exposure. The future risk of disturbance is just as dependent on the same factors. Although it is impossible to accurately predict such stochastic events as disturbances, it is possible to evaluate the likely trends of each of the sides of the risk triangle and give an outlook on future disturbance risk.

An obvious change in hazard is posed by changes in climate. Although climate change will have different effects in different regions in Europe, there is a general tendency for higher temperatures and more variability in precipitation (Meehl et al., 2007). Although precipitation is expected to increase in some cases, it might be more than offset by expected temperature increases. This will probably lead to an increase in future fire hazard (Kurz et al., 1995; Gerstengarbe et al., 1999; Schelhaas and Moriondo, 2007). The same factors will be influential for biotic hazards. An increase in temperature is expected to increase the reproductive capacity and number of completed life cycles per year of important biotic disturbance agents such as bark beetles (Harrington et al., 2001; Bale et al., 2002). Moreover, climate change is likely to cause shifts in the outbreak ranges of insect species (e.g., Parmesan et al., 1999; Williams and Liebhold, 2002). Additionally, an increase in environmental stress factors for the host species, such as drought, may reduce host resilience to insect infestation (e.g., Wermelinger, 2004; Rouault et al., 2006).

It is still unclear how climate change will affect the European wind climate. Some studies project an increase in storminess (Zwiers and Kharin, 1998; Lunkeit et al., 1996, Meehl et al., 2007), while others expect a decrease (Lässig and Schönenberger, 2000). Apart from climate change, other developments can influence the hazard of specific agents. For example, the increased use of forests for recreation increases the fire hazard due to an increase of ignition sources, but at the same time the chances of early detection increase as well. Another example is globalisation: increased trade increases the probability of accidentally introducing exotic pest species.

In this research, exposure was defined in terms of forest area and wood volume. Both variables have increased considerably in Europe since at least 1950 (Kuusela, 1994). Due to the high proportion of relatively recent afforestations, the European forest can be considered as rather young. The observed trend of increasing growing stock is largely attributable to an increase in the average age of the European forest. In the coming decades, both the forest area and the growing stock are projected to continue to increase (Schelhaas et al., 2006). However, increased demand for biofuels could lead to more harvesting in existing forests and less land becoming available for afforestation due to the establishment of dedicated bioenergy crops.

Vulnerability to various disturbance agents is mainly dependent on the state of the forest, expressed in terms such as species composition, age class composition,

density/thinning status and distribution over diameter and height classes. In Europe, the state of the forest is largely determined by historical management actions. The current age class structure with a relatively large proportion of young forests is the result of heavy exploitation decades to centuries ago and more recent mass afforestation. Currently, increment figures largely exceed the volume harvested, so it is expected that in the future the age class structure will become more uniform. This will be facilitated by the Europe-wide shift towards more nature-oriented management (Nabuurs, 2001). One aspect of nature-oriented management is the favouring of broadleaves rather than conifers. Increased use of broadleaves and a tendency for older forest would reduce the vulnerability of forests to fires. However, much agricultural land is being abandoned, especially in the Mediterranean region, leading to larger contiguous forest areas where fire can spread more easily. Furthermore, the pioneer species that colonise this land might be conifers that burn easily.

A shift towards more broadleaved species will also decrease wind vulnerability. However, such changes will only take place very gradually. On the other hand, the current trend towards older forests is happening relatively fast and will continue for some time still. This change will probably be more influential in the shorter term and will negatively affect the overall wind vulnerability. It is difficult to predict how vulnerability to biotic disturbances will develop. Each tree species and each stand development phase has its own associated portfolio of biotic disturbance agents. A shift in tree species or stand structure will be favourable for avoiding certain pests, but might lead to increased occurrence of other agents.

We can conclude that hazard and exposure are likely to increase for most disturbance agents in the near future. The development of vulnerability is less easy to predict, but drastic decreases in vulnerability are not very likely in the short term. Therefore, it seems very likely that in the coming decades we will have to reckon with an increased role of disturbances. Changing disturbance regimes may impose adverse feedbacks on the sustainable provision of important forest services and functions (e.g. Ayres and Lombardero 2000, Dale et al. 2000). Adaptive strategies may help to reduce the expected impact of future disturbances. Such strategies will have to focus on reducing exposure and vulnerability. A reduction in exposure could be achieved by earlier harvesting, less growing stock and limiting forest expansion in risky areas. Vulnerability could be reduced by modifying species composition, changing the age-class distribution and adopting specific forms of management aimed at reducing vulnerability to prevailing disturbance agents.

However, as the results of paper III show, it can take a long time before the effects of adaptation measures become visible. Before then, forest managers can anticipate the occurrence of disturbances and, for example, increase the flexibility of their management plans and take measures that limit the consequences of disturbances. Regional and national authorities should also be prepared for the occurrence of damage and have contingency plans in place. One adaptation strategy has already been incorporated as an aspect of nature-oriented management: natural

disturbances are increasingly seen as part of the system and their occurrence has increasingly been accepted and seen as an opportunity rather than a loss.

To conclude, natural disturbances can significantly affect the sustainable provision of services that forests provide. Although unpredictable, they occur regularly and forest management should be prepared for them. All decisions taken concerning the management of the forest can affect the risk of disturbances. It is therefore important to understand the processes behind disturbances and to be able to indicate the effects of decisions on the risk of disturbance. At the stand scale, the coupling of tree growth simulators with disturbance models offers great possibilities for exploring alternative management regimes and their associated risks. Similarly, at the regional and national scales, forest resource scenario models are being used to support policy makers. Inclusion of disturbances in such models has a significant effect on the projections and thus possibly on decisions to be derived from them. The inclusion of disturbances also offers the possibility of exploring scenarios for adapting to expected future climate change.

## 6 References

- Ancelin, P., Courbaud, B. and Fourcaud, T. 2004. Development of an individual tree-based mechanical model to predict wind damage within forest stands. *Forest Ecology and Management* 203:101-121.
- Ayres, M.P. and Lombardero, M.J. 2000. Assessing the consequences of global change for forest disturbance from herbivores and pathogens. *The Science of the Total Environment* 262:263-286.
- Bale, J.S., Masters, G.J., Hodkinson, I.D. Awmack, C., Bezemer, T.M., Brown, V.K., Butterfield, J., Buse, A., Coulson, J.C., Farrar, J., Good, J.E.G., Harrington, R., Hartley, S., Hefin Jones, T., Lindroth, R.L., Press, M.C., Symrnioudis, I., Watt, A.D., Whittaker, J.B. 2002. Herbivory in global climate change research: direct effects of rising temperature on insect herbivores. *Global Change Biology* 8:1-16.
- Böhm, P. 1981. Sturmschäden in Schwaben von 1950 bis 1980. *Allgemeine Forstzeitschrift* 36, 1380.
- Bossel, H., 1996. TREEDYN3 forest simulation model. *Ecological Modeling* 90:187-227.
- Brandle, M. and Brandl, R. 2001. Species richness of insects and mites on trees: expanding Southwood. *Journal of Animal Ecology* 70:491-504.
- Brassel, P. and Brändli, U.-B.(Eds.). 1999. Schweizerisches Landesforstinventar. Ergebnisse der Zweitaufnahme 1993-1995. Birmensdorf, Eidgenössische Forschungsanstalt für Wald, Schnee und Landschaft. Bern, Bundesamt für Umwelt, Wald und Landschaft. Bern, Stuttgart, Wien, Haupt.
- Brown, J. K. and Smith, J. K. 2000. Wildland fire in ecosystems: effects of fire on flora. Gen. tech. rep. rmrs-gtr-42-vol. 2, U.S. Department of Agriculture, Forest Service. Available: [http://www.fs.fed.us/rm/pubs/rmrs\\_gtr42\\_2.html](http://www.fs.fed.us/rm/pubs/rmrs_gtr42_2.html).

- Bugmann, H. 2001. A review of forest gap models. *Climatic Change* 51:259-305.
- Buse, A. and Good, J.E.G. 1996. Synchronisation of larval emergence in winter moth (*Operophtera brumata* L.) and budburst in pedunculate oak (*Quercus robur* L.) under simulated climate change. *Ecological Entomology* 21:335-343.
- Coakley, S.M., Scherm, H., and Chakraborty, S. 1999. Climate change and plant disease management. *Annual Review of Phytopathology* 37:399-426.
- CREAF. 1999. CREAM 1988-1998, Center for Ecological Research and Forestry Applications - CREAM, Barcelona.
- Dale, V.H., Joyce, L.A., McNulty, S. and Neilson, R.P. 2000. The interplay between climate change, forests, and disturbances. *The Science of the Total Environment* 262: 201-204.
- Dorland, C., Tol, R.S.J. and Palutikof, J.P. 1999. Vulnerability of the Netherlands and Northwest Europe to storm damage under climate change: A model approach based on storm damage in the Netherlands. *Climatic Change* 43:513-535.
- EEA. 2006. How much bioenergy can Europe produce without harming the environment? European Environment Agency. EEA report 07/2006, Copenhagen. ISSN 1725-9177. 67 pp.
- FAO. 2007. State of the World's Forests 2007. Food and Agricultural Organization of the United Nations, Rome. 144 p.
- Gardiner, B., Peltola, H. and Kellomäki, S. 2000. Comparison of two models for predicting the critical wind speeds required to damage coniferous trees. *Ecological Modelling* 129:1-23.
- Gerstengarbe, F.-W., Werner, P.C., Lindner, M. and Bruschek, G. 1999. Estimation of Future Forest Fire Development in the State of Brandenburg. *International Forest Fire News*, No. 21, 91-93.
- Harrington, R., Fleming, R.A. and Woiwood, I.P. 2001. Climate change impacts on insect management and conservation in temperate regions: can they be predicted? *Agricultural and Forest Entomology* 3:233-240.
- Hasenauer, H. 2006. Sustainable forest management: growth models for Europe. Springer, Berlin, 398 p.
- Holmsgaard, E. 1986. Historical development of wind damage in conifers in Denmark In: Minimizing wind damage to coniferous stands, Proceedings of the workshop organized jointly by the Danish forest experiment station and the commission of the European Communities at Løvenholm Castle, Denmark, March 3-7, 1986:2-4.
- IPCC., 2000. Emissions Scenarios 2000. Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge. 570 pp.
- IPCC, 2007. Mitigation. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge UK
- Jactel, H., Vodde, F., Branco, M., Carnus, J.M., Gonzalez, J.R., Grodzki, W., Langström, B., Moreira, F., Netherer, S., Nicoll, B., Orazio, C., Piou, D.,

- Santos, H., Schelhaas, M.J. and Tojic, K. 2007. Prevalence of sanitary hazards in European forests. EFORWOOD Deliverable 2.4.2.
- Können, G.P.R. 1999. De toestand van het klimaat in Nederland 1999, Koninklijk Nederlands Meteorologisch Instituut, De Bilt. 64 pp.
- Konstantinov, V. 2003. "Bulgaria - Analysis of the Problems Connected to Forest Fires in Bulgaria." International Forest Fire News, 28.
- Kramer, K. (Ed.). 2004. Effects of Silvicultural Regimes on Dynamics of Genetic and Ecological Diversity of European Beech Forests; Impact Assessment and Recommendations for Sustainable Forestry. Alterra, Wageningen.
- Kramer, K., Mohren, G.M.J. (eds). 2001. Long term effects of climate change on carbon budgets of forests in Europe. Alterra Report 194. Alterra, Wageningen, the Netherlands. 209 p.
- Kramer, K., Buiteveld, J., Forstreuter, M., Geburek, T., Leonardi, S., Menozzi, P., Povillon, F., Schelhaas, M.J., Teissier du Cros, E., Vendramin, G.G. and van der Werf, D.C. 2007. Bridging ecophysiological and genetic knowledge to assess the adaptive potential of European beech. Ecological Modelling, submitted manuscript.
- Kron W. 2002. "Flood risk = hazard x exposure x vulnerability". In: Wu M. et al., (ed.), Flood Defence, Science Press, New York, 82-97.
- Kurz, W.A., Apps, M.J., Stocks, B.J. and Volney, W.J.A. 1995. Global Climate Change: Disturbance Regimes and Biospheric Feedbacks of Temperate and Boreal Forests. In: Woodwell, G.M., Mackenzie, F.T. (Eds), Biotic Feedbacks in the Global Climatic System, pp. 119-133.
- Kuusela, K. 1994. Forest resources in Europe, 1950-1990. Cambridge University Press, Cambridge, 154 p.
- Lässig, R and Mocalov, SA. 2000. Frequency and characteristics of severe storms in the Urals and their influence on the development, structure and management of the boreal forests. Forest Ecology and Management 135:179-194.
- Lässig, R. and Schönenberger, W. 1997. Was passiert, wenn man die Natur sich selber überlässt? Ergebnisse der Sukzessionsforschung auf Windwurfflächen. Laufener Seminarbeiträge 1/1997, pp. 67-74.
- Leather, S.R., Walters, K.F.A. and Bale, J.S. 1993. The Ecology of Insect Overwintering. Cambridge University Press, Cambridge, 255 p.
- Leckebusch, G.C. and Ulbrich, U. 2004. On the relationship between cyclones and extreme windstorm events over Europe under climate change. Global and Planetary Change 44:181-193.
- Lexer M.J. and Hönninger, K. 2001. A modified 3D-patch model for spatially explicit simulation of vegetation composition in heterogeneous landscapes. Forest Ecology and Management 144:43-65.
- Lunkeit F., Ponater M., Sausen R., Sogalla M., Ulbrich U. and Windelband M. 1996. Cyclonic Activity in a Warmer Climate. Contributions to Atmospheric Physics/Beiträge zur Physik der Atmosphäre 69, 393-407.

- Mahrer, F. (ed.). 1988. Schweizerische Landesforstinventar. Ergebnisse der Erstaufnahme 1982-1986. Swiss forest land inventory. Results of the first period 1982-1986. Eidgenössische Anstalt für das Forstliche Versuchswesen, Birmensdorf, Switzerland. Berichte no. 305.
- Meehl, G.A., Stocker, T.F., Collins, W., Friedlingstein, P., Gaye, A., Gregory, J., Kitoh, A., Knutti, R. and Co-authors, 2007: Global climate projections. Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, S. Solomon, D. Qin, M.Manning, Z. Chen, M.Marquis, K.B.Averyt, M. Tignor and H.L.Miller, Eds., Cambridge University Press, Cambridge, 747-846.
- Meta, M. 2003. "Albania - Update on the Forest Fire Situation." International Forest Fire News 28.
- Meyer, J. 2005. Fire effects on forest resource development in the French Mediterranean region – projections with a large-scale forest scenario model. Technical Report 16. European Forest Institute. 86 p.
- Milne, R. 1991. Dynamics of swaying of *Picea sitchensis*. Tree Physiology 9:383-399.
- Mitchell, T.D., Carter, T.R., Jones, P.D., Hulme, M. and New, M. 2004. A comprehensive set of high-resolution grids of monthly climate for Europe and the globe: the observed record (1901-2000) and 16 scenarios (2001-2100). Tyndall Centre Working Paper 55, 25 pp.
- Mitchell, J.F.B., Johns, T.C., Gregory, J.M. and Tett, S.F.B. 1995. Climate response to increasing levels of greenhouse gases and sulphate aerosols. Nature 376:501-504.
- Mosandl, R. and Felbermeier, B. 1999. Auf dem Weg zum naturnahen Wald. AFZ/Der Wald 910-914.
- Nabuurs, G.J., 2001. European forests in the 21st century: impacts of nature-oriented forest management assessed with a large-scale scenario model. Alterra Scientific Contributions 2, Alterra, Wageningen.
- Nabuurs, G.J. and Päävinen, R. 1996. Large scale forestry scenario models: a compilation and review. European Forest Institute, Working Paper 10, Joensuu, Finland.
- Nabuurs, G.J., Pussinen, A., Karjalainen, T., Erhard, M. and Kramer, K. 2002. Stemwood volume increment changes in European forests due to climate change –a simulation study with the EFISCEN model. Global Change Biology 8:304-316.
- Nabuurs, G.J., Schelhaas, M.J., Mohren, G.M.J., and Field, C.B. 2003. Temporal evolution of the European Forest carbon sink from 1950 to 1999. Global Change Biology 9(2):152-160.
- Nabuurs, G.J., Pussinen, A., van Brusselen, J. and Schelhaas, M.J. 2007. Future harvesting pressure on European forests. European Journal of Forest Research 126, 391-400.
- Pacala, S.W., Canham, C.D. and Silander, J.A. Jr. 1993. Forest models defined by field measurements: 1. The design of a northeastern forest simulator. Canadian Journal of Forest Research 23: 1980-1988.

- Pacala, S.W., Canham, C.D., Saponara, J., Silander, J.A. Jr., Kobe, R.K. and Ribbens, E. 1996. Forest models defined by field measurements: II. Estimation, error analysis and dynamics. *Ecological Monographs* 66:1-43.
- Parmesan, C., Ryrholm, N., Stefanescu, C. et al., 1999. Poleward shifts in geographical ranges of butterfly species associated with regional warming. *Nature*, 399:579-583.
- Peltola, H., Kellomäki, S., Väisanen, H. And Ikonen, V.-P. 1999. A mechanistic model for assessing the risk of wind and snow damage to single trees and stands of Scots pine, Norway spruce, and birch. *Canadian Journal for Forest Research* 29:647-661.
- Quine C., Coutts, M., Gardiner, B. and Pyatt, G. 1995. Forests and wind: Management to minimize damage, Forestry Commission Bulletin 114. HMSO, London.
- Rouault, G., Candau, J.N., Lieutier, F., Nageleisen, L.M., Martin, J.C. and Warzée, N. 2006. Effects of drought and heat on forest insect populations in relation to the 2003 drought in Western Europe. *Annals of Forest Science* 63:613-624.
- Ruel, J-C. 1995. Understanding windthrow: silvicultural implications. *Forestry Chronicles* 71:435-445.
- Sallnäs, O. 1990. A matrix growth model of the Swedish forest. *Studia Forestalia Suecica* No 183. Swedish University of Agricultural Sciences, Faculty of Forestry, Uppsala.
- Savill, P.S. 1983. Silviculture in windy climates. *Forestry Abstracts* 44:473-488.
- Schelhaas, M.J. and Moriondo, M. 2007. Bosbrand en klimaatverandering. *Vakblad Natuur Bos Landschap* 4(8):13.
- Schelhaas, M.J., van Brusselen, J., Pussinen, A., Pesonen, E., Schuck, A., Nabuurs, G.J. and Sasse, V. 2006. Outlook for the Development of European Forest Resources. A study prepared for the European Forest Sector Outlook Study (EFSOS). Geneva Timber and Forest Discussion Paper, ECE/TIM/DP/41. UN-ECE, Geneva. 118 p.
- Schelhaas, M.J., Cienciala, E., Lindner, M., Nabuurs, G.J. and Zianchi, G. 2007a. Selection and quantification of forestry measures targeted at the Kyoto Protocol and the Convention on Biodiversity. *Alterra report* 1508. 95 p.
- Schelhaas, M.J., Eggers, J., Lindner, M., Nabuurs, G.J., Pussinen, A., Päivinen, R., Schuck, A., Verkerk, P.J., van der Werf, D.C., and Zudin, S. 2007b. Model documentation for the European Forest Information Scenario model (EFISCEN 3.1). Wageningen, Alterra, *Alterra report* 1559, EFI Technical Report 26, Joensuu, Finland. 118 p.
- Schiesser, H.-H, Pfister, C. and Bader, J., 1997. Winter storms in Switzerland north of the Alps 1864/65-1993/94. *Theoretical and applied climatology* 58: 1-19.
- Schröter, D. (Ed). 2004. ATEAM final report 2004. Advanced Terrestrial Ecosystem Analysis and Modelling. Potsdam Institute for Climate Impact Research (PIK). 139 pp.
- Schütz, J.P., Götz, M., Schmid, W., Mandallaz, D. 2006. Vulnerability of spruce (*Picea abies*) and beech (*Fagus sylvatica*) forest stands to storms and consequences for silviculture. *European Journal of Forest Research* 125:291-302.



- Seidl, R., Lexer, M.J., Jäger, D. and Hönninger, K. 2005. Evaluating the accuracy and generality of a hybrid forest patch model. *Tree Physiology* 25:939-951.
- Seidl, R., Rammer, W., Baier, P., Schopf, A. and Lexer, M.J. 2007. Modelling tree mortality from bark beetle infestation in Norway spruce forests. *Ecological Modelling* 206:383-399.
- SFSO and FOEFL. 1996. Forestry and Timber Industry in Switzerland, Yearbook. Swiss Federal Statistical Office, Bern.
- Sonntag, M. 1998. Klimaveränderungen und Waldwachstum: TREEDYN3-Simulationen mit einer Analyse modellstruktureller Unsicherheiten. Verlag Mainz, Aachen. Dissertation, University of Kassel, Germany.
- Southwood, T.R.E. 1961. The numbers of species of insect associated with various trees. *Journal of Animal Ecology* 30: 1-8
- Speight M. R., Wainhouse, D. 1989. Ecology and Management of Forest Insects. Clarendon Press, Oxford, 374 pp.
- UN-ECE, 2005. European Forest Sector Outlook Study, Main report. ECE/TIM/SP/20. United Nations, Geneva.
- Vélez, R. 1985. Fire prevention in Aleppo pine forests. In: *Le pin d'Alep et le pin brutia dans la sylviculture méditerranéenne*, volume 1986 of Options Méditerranéennes. CIHEAM, 167–178. Available: <http://ressources.ciheam.org/om/pdf/s10/CI010861.pdf>.
- Wermelinger, B. 2004. Ecology and management of the spruce bark beetle *Ips typographus* – a review of recent research. *Forest Ecology and Management* 202:67-82.
- Williams, D.W. and Liebhold, A.M. 2002. Climate change and the outbreak ranges of two North American bark beetles. *Agricultural and Forest Entomology* 4:87-99.
- Xanthopoulos, G. 2000. "The 1999 Forest Fire Season." *International Forest Fire News* 22:76-84.
- Zeng, H. 2006. Influence of clear-cutting on the risk of wind damage at forest edges: A gis-based integrated models approach. University of Joensuu, Faculty of Forestry. *Dissertationes Forestales* 16.
- Zeng, H., Pukkala, T., Peltola, H. 2007. The use of heuristic optimization in risk management of wind damage in forest planning. *Forest Ecology and Management* 241:189-199.
- Zwiers, F.W. and Kharin, V.V. 1998. Changes in the extremes of the climate by CCC GCM2 under CO2 doubling. *J. Climate* 11:2200-2222.



# I Natural disturbances in the European forests in the 19<sup>th</sup> and 20<sup>th</sup> centuries

*Global Change Biology* 9: 1620-1633

Mart-Jan Schelhaas<sup>1,2,\*</sup>, Gert-Jan Nabuurs<sup>1,2</sup>, Andreas Schuck<sup>2</sup>

<sup>1</sup>) Alterra, Green World Research, PO Box 47, NL-6700 AA Wageningen, the Netherlands

<sup>2</sup>) European Forest Institute, Torikatu 34, FIN-80100, Joensuu, Finland

\*) Corresponding author:

Alterra, Green World Research

PO Box 47, NL-6700 AA Wageningen, the Netherlands

E-mail: [martjan.schelhaas@wur.nl](mailto:martjan.schelhaas@wur.nl)

Fax.: +31.317.419000

## Abstract

This paper, based on a literature review, presents a quantitative overview of the role of natural disturbances in European forests from 1850 to 2000. Such an overview provides a basis for modelling the possible impacts of climate change and enables one to assess trends in disturbance regimes in different countries and/or periods. Over the period 1950-2000 an annual average of 35 million m<sup>3</sup> wood was damaged by disturbances; there was much variation between years. Storms were responsible for 53% of the total damage, fire for 16%, snow for 3% and other abiotic causes for 5%. Biotic factors caused 16% of the damage, and half of this was caused by bark beetles. For 7% of the damage no cause was given or there was a combination of cause. The 35 million m<sup>3</sup> of damage is about 8.1% of the total fellings in Europe and about 0.15% of the total volume of growing stock. Over the period 1961-2000, the average annual area of forest fires was 213,000 ha, which is 0.15% of the total forest area in Europe. Most types of damage seem to be increasing. This is partly an artefact of the improved availability of information. The most likely explanations for an increase in damage from disturbances are changes in forest management and resulting changes in the condition of the forest. Forest area, average volume of growing stock and average stand age have increased considerably, making the forest more vulnerable and increasing the resources that can be damaged. Since forest resources are expected to continue to increase, it is likely that damage from disturbances will also increase in future.

**Keywords:** Disturbance, history, storm, fire, insects, damage

## Introduction

In intensively managed ecosystems like the forests in Europe, natural disturbance dynamics have been largely eliminated and the irregular and unpredictable occurrence of disturbance events has been replaced by a regular management regime of thinning, clearcutting and replanting. However, nature has not been totally mastered, as was demonstrated by the exceptionally severe storms in 1990 and 1999, which damaged respectively 120 and 180 million m<sup>3</sup> of wood (Lässig and Schönenberger, 2000, UN-ECE/FAO, 2000b). There are no previous records of storms with so much impact on European forest, although historians in France claim that in earlier times severe storms also severely damaged French forests (Doll and Riou-Nivert, 1991). Some people (Rhein-Zeitung, 1999) saw in the 1999 storm the first sign of a change in storm patterns, caused by the greenhouse effect. Climate change might have an impact on disturbances in the forest, either directly by influencing the storm frequency and intensity, and by the drier conditions favouring forest fires, or indirectly by weather conditions favouring insect pest infestations (McCarthy et al., 2001). Another reason for change is that management in Europe is now more nature-oriented, mimicking nature and being more tolerant of natural disturbances than used to be the case. Thus, the role of natural disturbances can be expected to increase in the future.

To date, there has been no historical overview of disturbances in Europe's forests. Of the few existing overviews of storm damage (Holmsgaard, 1982, Avram, 1983), the study by (Doll and Riou-Nivert, 1991) is the most comprehensive. Most others cover only the biggest events, appear to be incomplete, or focus on a certain region. Statistics at European level are available for forest fires (FAO, 1982, 1984, UN-ECE/FAO, 1996, 1998, 1999, 2000a, 2001), but these cover only the last 20 to 30 years, and have some time gaps. No statistics are available at European scale on biotic damage. However, certain countries, e.g. Germany, have very detailed records on the occurrence and degree of damage from insects and fungi (Waldschutzsituation, 1999, 2000). Though other European countries have information available on biotic damage, it is scattered over many published sources.

A comprehensive overview of forest disturbance would support analyses of forest disturbances. It could also provide a basis for modelling possible impacts of climate change (e.g. Schelhaas et al., 2002), allow disturbance regimes in different countries and/or periods to be compared, and could be used in smaller-scale analyses for underpinning new management guidelines. The historical review presented in this paper is the result of a study that had three aims:

1. to present a quantitative historical overview of the available data on natural disturbances in European forests
2. to estimate the total damage to European forests from natural disturbances
3. to discuss the possible causes of observed trends.

## Definitions

In this study 'natural disturbance' was defined as an event that causes unforeseen loss of living forest biomass or an event that decreases the actual or potential value of the wood or forest stand. The study focused on 30 countries in Europe, excluding the Newly Independent States that emerged after the breakdown of the Soviet Union<sup>1</sup>.

The study focuses on forest land. However, the definition of forest varies between literature sources, between countries and over time. Usually literature sources do not mention definitions, so it is impossible to apply any correction for that. Therefore, all results of this study must be interpreted with this weakness in mind. Especially in many Mediterranean countries the distinction between forest and other wooded land is difficult. Since fire is the most common disturbance in these countries, special care is taken to separate these categories in the case of fire. However, this separation may again vary between countries. Usually a crown cover of 10 or 20% is the minimal requirement for a classification as forest land.

Fires include all wildfires that affect forest land. Prescribed burning is not taken into account, unless it escaped to other forest stands that were not intended to be burned.

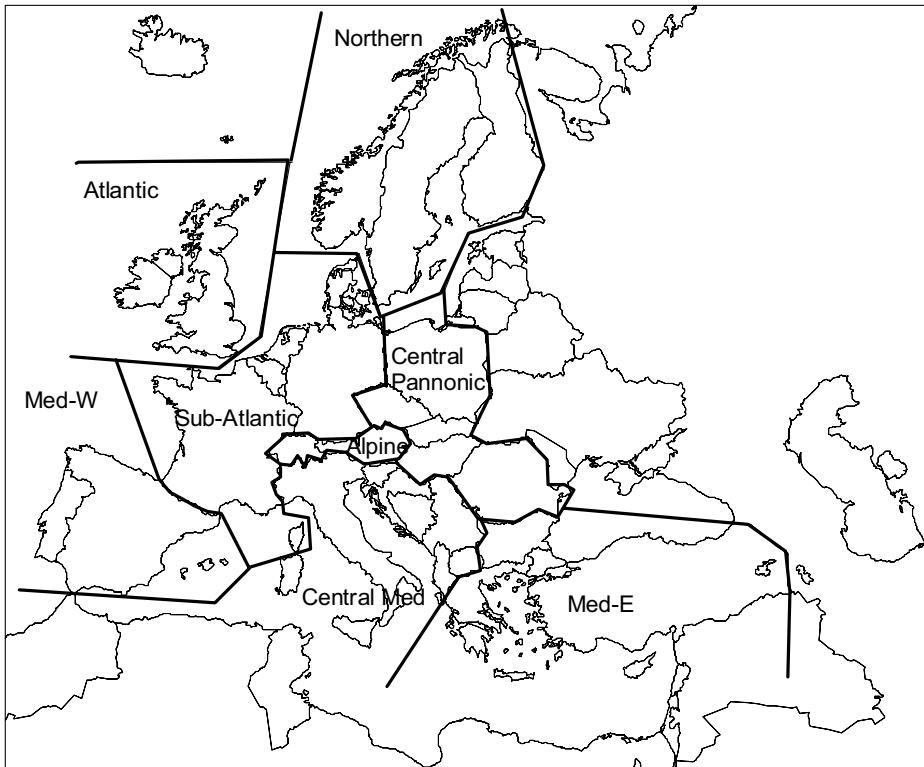
In this study, the term "damaged" is often used in connection with various disturbance types. With this term, we mean that the stand or stems have been affected in one way or another by the disturbance. The degree to which they have been affected can vary, ranging from fire scars to complete burning or uprooting. This does not say anything about the potential of the wood for further commercial use.

## Material and Methods

The basis of this study is a literature review. Information on volumes and areas damaged by disturbances was gathered from published and Internet sources. In order to make a pan-European estimate, Europe was divided into ecological zones, based on those used by Kuusela (1994) (Figure 1) but differing in the following ways: Germany and Denmark were assigned to the Sub-Atlantic region; while Poland, the Czech Republic, Slovak Republic, Hungary and Romania were put together in one group (Central Pannonic), since almost no data were available for the original Pannonic group (Hungary and Romania). For disturbances other than forest fires, all Mediterranean countries were put in one group, since information on such disturbances was very scarce for the whole Mediterranean area.

---

<sup>1</sup> Countries included were: Albania, Austria, Belgium, Bosnia Hercegovina, Bulgaria, Croatia, Czech Republic, Denmark, Finland, France, Germany, Greece, Hungary, Ireland, Italy, Luxembourg, The FYR of Macedonia, The Netherlands, Norway, Poland, Portugal, Romania, Slovak Republic, Slovenia, Spain, Sweden, Switzerland, Turkey, United Kingdom, and Yugoslavia.



**Figure 1.** Division of Europe into ecological zones

***Forest fires***

Time series of burned areas of forest land were compiled from the available data per country. For the years where only the burned area of forest and other wooded land was available, we estimated the burned area of forest land using the average ratio of burned area of forest land to burned area of forest and other wooded land. We used the time series of burned area of forest land to calculate the average area burned per country. The contribution of a given country to the total European forest fire area was calculated using Equation 1:

$$W_i = A_i / \sum_{i=1}^n A_i \tag{1}$$

where

- $W_i$  = weight of country i in the total European forest fire area
- $A_i$  = average forest fire area in country i
- n = number of countries

For each year the total forest fire area in Europe was calculated with Equation 2.

$$EF_j = \sum_{i=1}^{n_j} RF_{ij} / \sum_{i=1}^{n_j} W_i \quad (2)$$

where

$EF_j$  = estimated total forest fire area in year  $j$

$RF_{ij}$  = reported forest fire area in country  $i$  for year  $j$ , including only those countries that reported a forest fire area, and with the provision that the total contribution of these countries was at least 30%

$n_j$  = number of countries with fire statistics in year  $j$

The same approach was used for the number of forest fires.

The total wood volume damaged per annum by fire was estimated per country by multiplying the fire-affected area of forest land by the average volume per hectare that was damaged in a fire. The total wood volume damaged in Europe was then estimated using the same technique, but weighted with the area of forest land burned. The average volume of wood damaged by forest fire per hectare and per country was derived from the collected data. This yielded estimates for most Mediterranean countries, and for one Alpic and one Central Pannonic country. For countries in these zones with no known average damaged wood volume, the average damaged wood volume of the nearest country within its region was used. For the countries in the Atlantic, Sub-Atlantic and Northern zones, an average damaged wood volume of 25 m<sup>3</sup> ha<sup>-1</sup> was assumed, since no data on volume of wood were available for these countries.

### ***Damage from other causes***

The remainder of the available data was sorted into the following groups: storm damage, snow damage, other abiotic damage, damage from bark beetles, other biotic damage and other damage (the latter includes damage from combinations of biotic and abiotic causes and damage from unknown causes). This damage was generally reported in volumes of wood. The approach followed for each of these damage groups entailed first checking the data per country to remove duplicates, since some events were reported in more than one source. When literature sources referred to the total damage for a certain period, this amount was distributed evenly over that period, taking into account the known damage from other sources. In order to get an overview of the total reported damage in Europe, the damage per country was simply aggregated for each year.

Given the huge variation in information available per country, it is clear that this overview does not cover all forest damage from disturbances. To get a pan-European estimate we scaled up the reported damage. For this estimate we assumed

that the total wood volume damaged by a certain event was related to the total volume of growing stock. FAO statistics (FAO, 1948, 1955, 1960, 1963, 1976, UN-ECE and FAO, 1985, 1992) provided the total growing stock per country with intervals of 5-10 years. Since the information on disturbances before this period was not complete, we decided to limit our European estimate to the period 1950-2000.

We used linear interpolation to calculate the volume of growing stock for each country per year from the FAO statistics. Per ecological zone, we selected one or several countries with apparently complete disturbance series from 1950 to 2000 to represent the whole ecological zone. For these ‘representative’ countries, we used Equation 3 to calculate the proportion of total growing stock that was damaged per year

$$D_{jk} = \sum_{i=1}^{n_k} RD_{ijk} / \sum_{i=1}^{n_k} GS_{ijk} \quad (3)$$

where

$D_{jk}$  = the proportion of the growing stock damaged in the representative countries of zone k in year j

$n_k$  = number of representative countries in zone k

$RD_{ijk}$  = the damage reported in representative country i of zone k in year j

$GS_{ijk}$  = the total growing stock in representative country i of zone k in year j

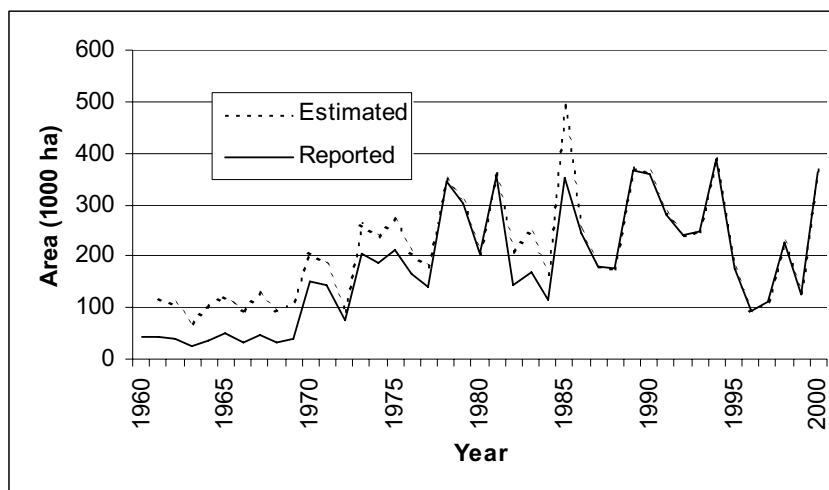
Multiplying this share by the total growing stock of the other countries in the ecological zone yielded an estimate of the missing damage for those countries. If the actual damage reported for those countries for that year were higher, this was used instead of the estimated amount.

## Results

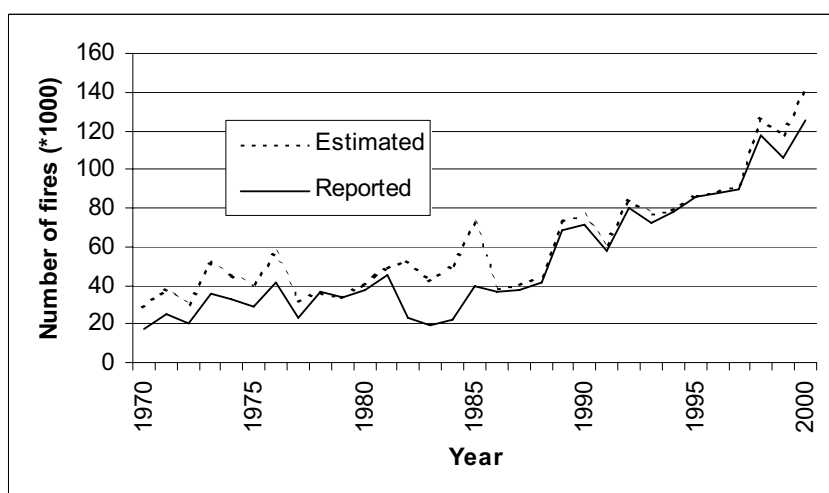
### *Forest fires*

Over the period 1961-2000, on average 178,000 ha of fire on forest land was reported (Figure 2). After correction for missing data, the total area is estimated at 213,000 ha, which is about 0.15% of the total forest area in Europe. The forest fire area has clearly increased over this time span. In the 1960s the estimated average was almost 114,000 ha, whereas in the 1980s it was over 280,000 ha. In the 1990s the average decreased slightly, to about 227,000 ha. Variation between years is very large; the absolute maximum was in 1985, with an estimated forest fire area of almost 500,000 ha. The number of forest fires also shows a clear increase, from an average of about 40,000 fires per year in the 1970s to more than 95,000 fires in the 1990s (Figure 3).





**Figure 2.** Annual burned forest area as reported in European countries for 1960-2000, and as scaled up for total Europe for 1961-2000.



**Figure 3.** Annual number of forest fires as reported in European countries and as scaled up for total Europe for 1970-2000.

Almost half (44.9%) of the total forest fire area is accounted for by two countries: Spain and Portugal (Mediterranean West). Excluding France, the total Mediterranean area accounts for 88% (Table 1), but when France is included this figure increases to 93.6%. In the Mediterranean area, about 0.3% of the forest area burns every year; the figure for Portugal and Spain is 0.6%. The contribution of each ecological zone in the number of fires is slightly different, with Mediterranean West again dominating

(40.5%), but Central Pannonic also having relatively many fires (13.1%) and Mediterranean East relatively few (6.3%). This is an indication that the average fire in the Mediterranean East affects a large area, and that the Central Pannonic region has many fires, but these affect small areas.

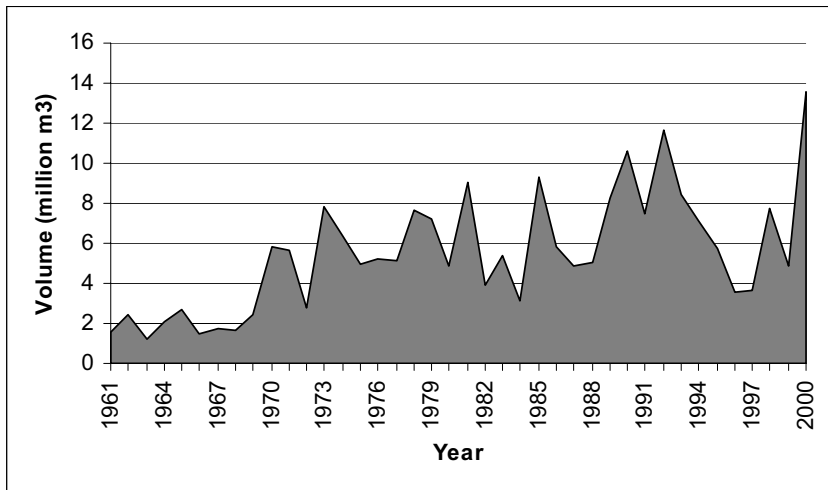
**Table 1.** Share of the ecological zones in the total forest fire area

Ecological zone	Share in total forest fire area (%)	Share in number of fires (%)
Northern	1.8	8.3
Central Pannonic	2.2	13.1
Alpine	0.4	0.5
Atlantic	0.3	1.2
Sub-Atlantic	7.3	12.7
Mediterranean West	44.9	40.5
Central Mediterranean	26.1	17.4
Mediterranean East	17.0	6.3

The average wood volume that is damaged by fire varies considerably per country, from 2 m<sup>3</sup> ha<sup>-1</sup> in Yugoslavia to 152 m<sup>3</sup> ha<sup>-1</sup> in Romania (Table 2). This probably reflects differences in severity of the fires between the countries and the type of forest where fire occurs, as well as the differences in the overall state of the forest. Another reason could be that some countries report only the wood that is really burned, while others report the saleable wood from trees that have been killed. These figures are usually based on only a few observations, so they are probably not very reliable. When these figures are combined with the forest fire areas per annum, we get an estimate of the wood volume damaged by fire annually in Europe: 5.5 million m<sup>3</sup> for the period 1961-2000 (Figure 4). Since the damaged volume is closely related to the forest fire area, the estimated damaged volume also increases over time. It is 2.3 million m<sup>3</sup> for the period 1961-1970 but rose to 7.4 million m<sup>3</sup> in the 1990s.

**Table 2.** Average wood volume damaged by fire for different countries and ecological zones

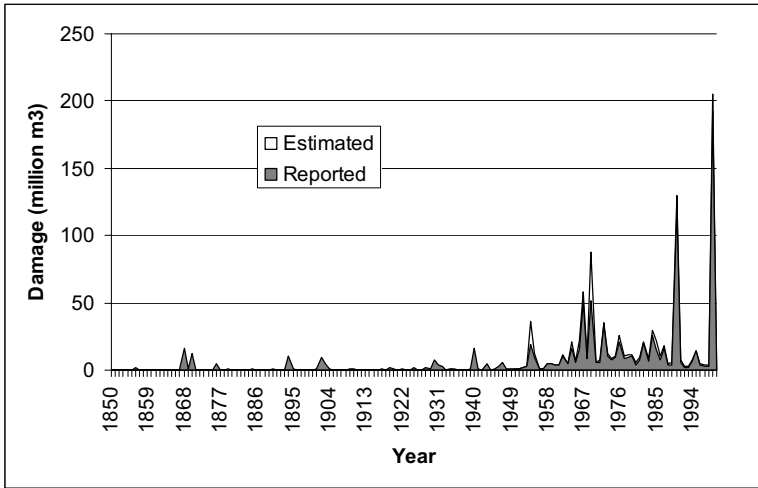
Ecological zone	Country	Wood volume damaged (m <sup>3</sup> ha <sup>-1</sup> )
Mediterranean West	Spain	6
	Portugal	43
Mediterranean Middle	Italy	35
	Slovenia	13
	Yugoslavia	2
Mediterranean East	Bulgaria	98
	Turkey	37
Central Pannonic	Romania	152
Alpine	Switzerland	91
Atlantic	Assumption	25
Sub-Atlantic	Assumption	25
Northern	Assumption	25



**Figure 4.** Volumes of wood damaged by fire for total Europe (1961-2000) as estimated from the total upscaled forest fire area and average wood volume damaged by fire, per country.

### ***Storm damage***

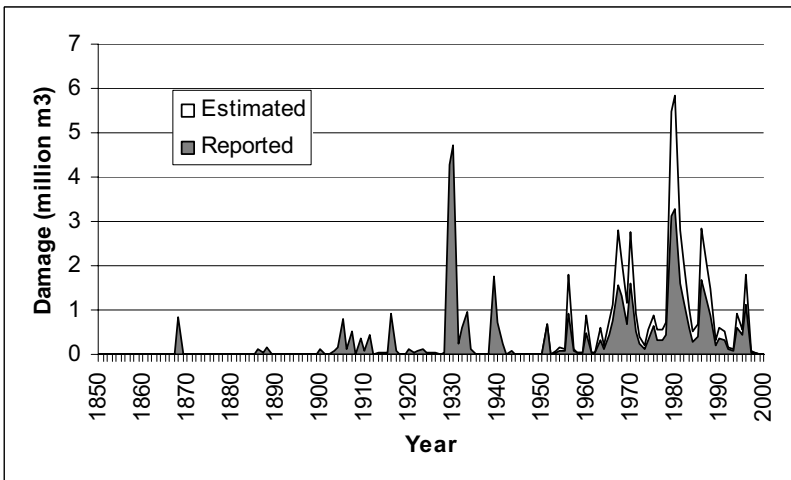
The reported damage from storms also seems to have increased since 1850, both in frequency and in magnitude (Figure 5). However, the reports of storm damage are incomplete at the European scale, so it is more likely that the increase in frequency of storm damage actually reflects the increasing number of reported events rather than a real increase in the number of events. Variation between years is again very large, with the storms of 1990 and 1999 as very exceptional, causing damage of 120 and 180 million m<sup>3</sup> of wood respectively. In between major storms like these, there are numerous accounts of localised damage, for example from whirlwinds and thunderstorms. Most of the damage from storms was reported in the Sub-Atlantic zone, the Alpine zone and the Central Pannonic zone, and especially in the more mountainous areas. The Atlantic region also reported damage, but not in such large amounts, although windthrow is a very regular event in this zone (Maxwell Macdonald, 1952). The Northern zone had a much lower frequency of accounts of storm damage than Central Europe. The estimated average annual storm damage over the period 1950-2000 is 18.7 million m<sup>3</sup> of wood.



**Figure 5.** Volumes of wood damaged by storms as reported in European countries for 1850-2000 and as scaled up for total Europe for 1950-2000.

***Snow damage***

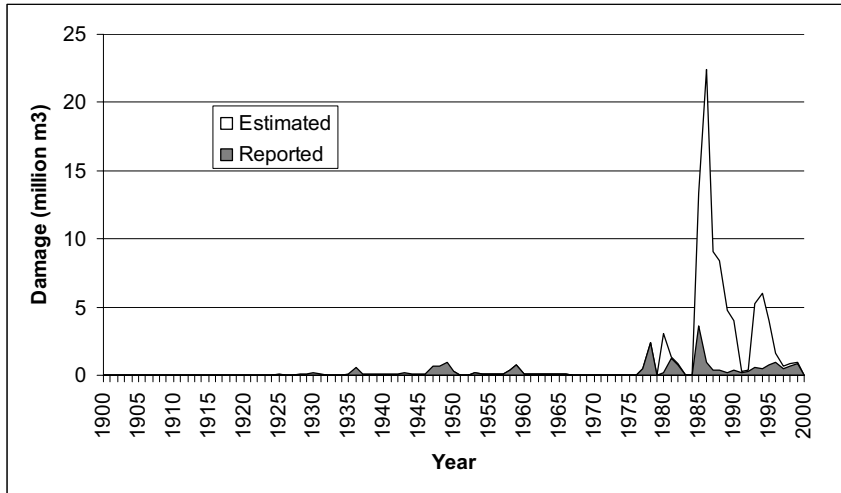
Snow damage is far less important than storm damage; it accounts for an estimated average annual amount of almost 1 million m<sup>3</sup> of wood per year over the period 1950-2000 (Figure 6). Once again, the apparent increase in frequency is most probably an artefact of increased reporting. There is great variation in the amounts reported, but no trend can be discerned. Most of the reported snow damage occurred in Germany, Austria, Czech Republic and Slovak Republic; there were incidental reports in other countries.



**Figure 6.** Volumes of wood damaged by snow, as reported in European countries for 1850-2000 and as scaled up for total Europe for 1950-2000.

### ***Other abiotic damage***

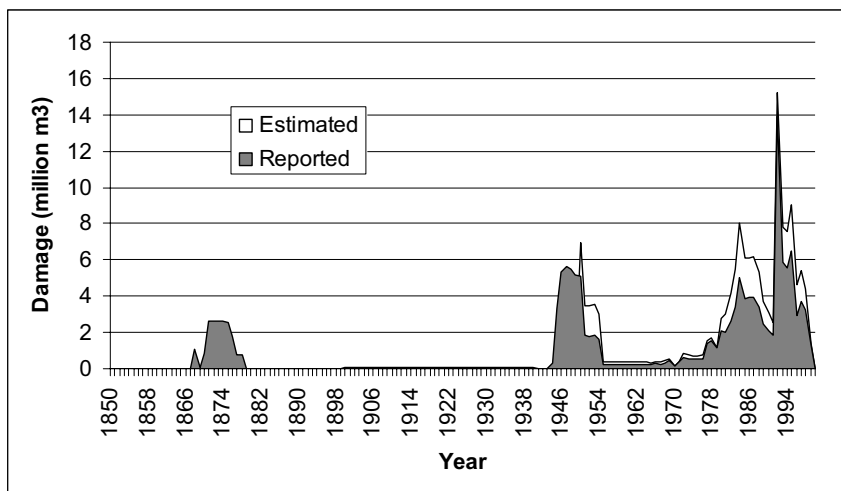
There are several other abiotic causes of damage besides fire, storm and snow, such as drought, waterlogging of the soil, and frost. However, in terms of volume they are minor in comparison to storm damage. The average annual amount reported over the period 1950-2000 is 0.4 million m<sup>3</sup> per year (Figure 7). The estimated amount is larger, 1.8 million m<sup>3</sup>, but this is largely an artefact of the estimation. The only information available for other abiotic damage in the Mediterranean area was for Slovenia, which is a very small basis for extrapolation.



**Figure 7.** Volumes of wood damaged by other abiotic causes, as reported in European countries for 1900-2000 and as scaled up for total Europe for 1950-2000.

### ***Bark beetle damage***

Damage from bark beetles, mainly *Ips typographus* (L.), is usually highly correlated with storm damage. Several severe outbreaks are known from historical records. The first one was in the Sumava Mountains and surroundings (Czech Republic) after the storms of 1868 and 1870, where the blown wood was not cleared out in time (Figure 8). There was another severe outbreak in Central Europe after the Second World War. Large amounts of damaged wood remained in the forest for a long time because of the shortage of labour. This provided suitable nesting material, and weather conditions were favourable for the fast development of the insect population (Schwerdtfeger, 1955, Pfeffer and Skuhavy, 1995, Zatloukal, 1998). In 1969 a severe storm in Scandinavia triggered a bark beetle outbreak that caused millions of cubic metres of damage (Annala and Petäistö, 1978), but since no exact figure could be found, this is not visible in Figure 8.

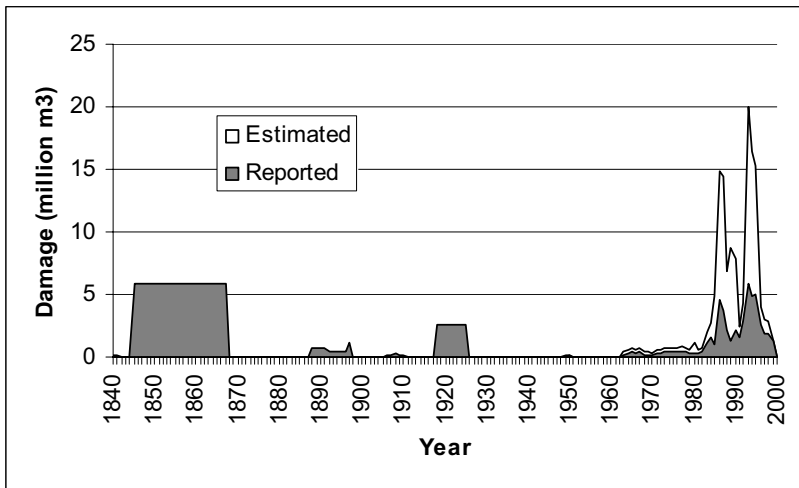


**Figure 8.** Volumes of wood damaged by bark beetles, as reported in European countries for 1850-2000 and as scaled up for total Europe for 1950-2000.

The insect damage in the 1980s was mainly in Poland and the Czech Republic, after damage from several storms. The outbreak of 1990-1997 originated from the storms of 1990 and was also favoured by several warm and dry summers (Pfeffer and Skuhravy, 1995). For the period 1950-2000 we estimate that the average wood volume damaged by bark beetles in total Europe was about 2.9 million m<sup>3</sup> per year.

***Other biotic damage***

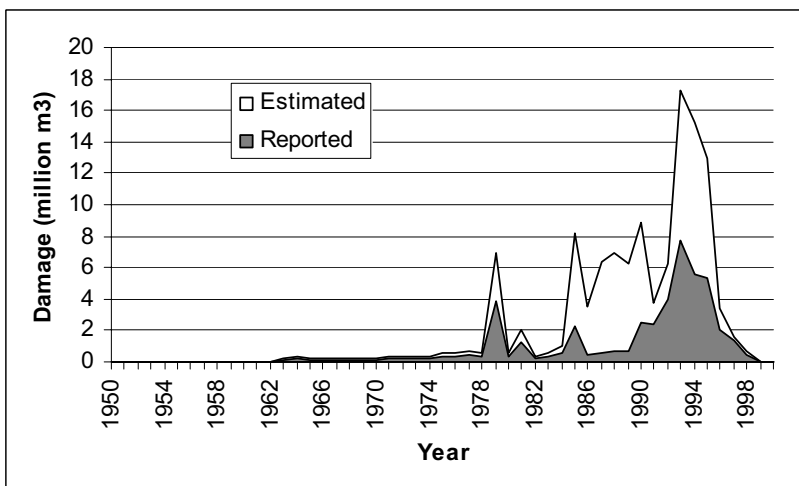
Damage from other biotic factors includes damage from all biotic factors except bark beetles. In this group too, several important outbreaks can be detected (Figure 9). The damage in the period 1845-1867 was caused by an outbreak of *Lymantria monacha* (L.) that spread throughout Europe, causing total damage of 135 million m<sup>3</sup> of wood (Plochmann and Hieke, 1986). Around 1890, another outbreak of *Lymantria monacha* occurred in Germany, followed by an outbreak of *Bupalus piniarius*. The peak in the 1920s was again caused by *Lymantria monacha*, but this time in the Czech Republic (Zari, 1981). The damage after 1965 was caused by several different species, and must be attributed to a general increase in the monitoring and reporting efforts of several countries. Most of the reports are from Central European countries, such as Poland, Germany, Slovak Republic, Czech Republic and Slovenia, all of which increased their monitoring efforts. This increase in monitoring suggests that such biotic damage occurs more often in these countries, but that does not mean that it does not occur in other countries as well.



**Figure 9.** Volumes of wood damaged volumes by other biotic factors, as reported in European countries for 1840-2000 and as scaled up for total Europe for 1950-2000.

### *Other damage*

This category contains reports of damage not specifying the cause, reports where the cause was not clear (often a combination of causes) and reports of total sanitation fellings, where no separation was made into damage from different causes. Over the period 1950-2000, the estimated average volume in this category was 2.3 million m<sup>3</sup> per year (Figure 10). Since the cause of this damage is unclear, it is not possible to explain any apparent trends or peaks.



**Figure 10.** Volumes of wood damaged due to other causes, as reported in European countries and as scaled up for total Europe for the period 1950-2000.

### ***Total damage***

If we aggregate all separate causes of damage for the period 1950-2000 we arrive at a total estimate of about 35 million m<sup>3</sup> damaged wood per year from disturbances. Given a total European felling for the countries concerned of about 430 million m<sup>3</sup> (UN-ECE and FAO, 2000), the damage represents about 8.1% of the felling volume. Compared to the growing stock of almost 24 billion m<sup>3</sup> (UN-ECE and FAO, 2000), the share of damage is only 0.15%. Slightly more than half of the damage (53% of the total) is caused by storms and 16% by fires. Snow and abiotic causes other than fire and storm account for 8%, which makes the total abiotic component 77%. Biotic damage accounts for 16% of the total damage, and half of this is caused by bark beetles. The causes of the remaining 7% are unknown or combinations of other causes.

## **Discussion**

The literature review revealed an overwhelming number of information sources, varying greatly in degree of detail, accuracy and time spans covered. In total, about 400 references were included in the literature review. In this study, only the results of the analysis are shown; the detailed information is available via internet (Schelhaas et al., 2001).

### ***Reliability***

When compiling this historical overview, several familiar problems turned up in relation to the literature sources. For example, sources often refer to only one owner category (mostly state forests), the years differ between sources (harvest years, calendar years, accounting years, etc.), the units are different (sometimes volume is reported as overbark, sometimes as underbark, sometimes only harvestable wood is reported, sometimes total wood, and mostly it is not clearly stated what exactly is presented). Furthermore, different sources reporting the same event may give very different statistics. It usually takes several years to clear storm damage, so the actual amount damaged is not known until much later after the storm, which also results in different estimates and reports (Rottmann, 1986).

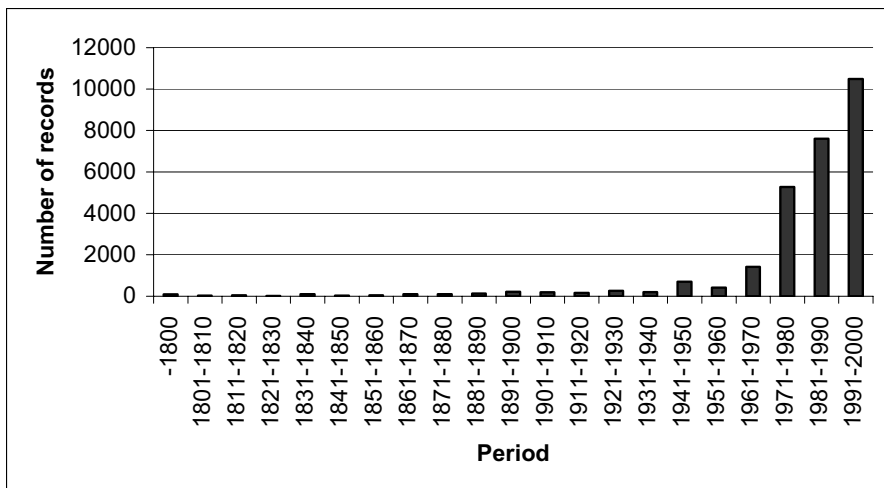
When more than one report on the same event was available, we used the most reliable or accurate information source, i.e. the figure that was best explained. When information was available in series, such as statistics for the area burned per year for a country, this series was used, in order to ensure consistency between the years. Underbark figures were converted to overbark, assuming that bark contributes 12% to the overbark volume. We used calendar years, and when other types of years appeared, we assumed them to be in the calendar year in which most months occurred. So a bookkeeping year from March 1999 to February 2000 was counted as 1999. Despite the careful examination of the information sources, differences,



inconsistencies and incompatibilities between sources will still remain, so the overall accuracy of the reported damage cannot be regarded as very good.

Moreover, the overview is probably not complete, since minor damage is usually not reported. Holmsgaard (1986) estimated that although a good overview exists for Denmark for the major storm damage to forest in the 20<sup>th</sup> century, only half of the total damage was included in his overview; the rest consisting of smaller windthrows. In the United Kingdom this is referred to as ‘endemic windthrow’ (Miller, 1985), as opposed to catastrophic events. However, no data on the magnitude of this endemic windthrow in the United Kingdom could be found. Detailed studies on a smaller area, such as a forest district, can give some insight into the magnitude of this smaller damage. However, such studies are very rare and difficult to extrapolate to national and European levels.

Another complicating factor is the increase in the number and detail of reports over time. Over the years more and more countries have started to collect statistics on various topics, and more recent reports are usually more easily obtainable than old ones. Public and professional interest has also stimulated reporting. Together, all these factors have led to reports increasing in number and detail over time. Figure 11 clearly shows an almost exponential increase of the number of records found per 10-year period from 1800 onwards.



**Figure 11.** Number of records found per 10-year period between 1800 and 2000.

The methods used for extrapolating to the European scale are rough and sometimes, as in the case of the Mediterranean area, were based on only a few data. Given the comments on the reported damage made in the preceding section, it is clear that the estimates presented can only be regarded as indications about the magnitude of the different types of disturbance. For example, our estimate of the total snow damage in

Europe is about 1 million m<sup>3</sup> of wood per year. According to Nykänen et al. (1997), the annual snow damage in the European Union is 4 million m<sup>3</sup> of wood, but they give no reference for the source of this figure. The difference between these two estimates indicates the range of uncertainty, and the caution we should use in interpreting the results.

For the forest fire area we estimated an annual average of 227,000 ha in the period 1991-2000. This estimate might seem to be relatively low in comparison with other figures, but it must be remembered that we focussed on the area of fires on forest land only, whereas usually the fire area of forest and other wooded land is referred to, which is substantially larger.

## ***Trends***

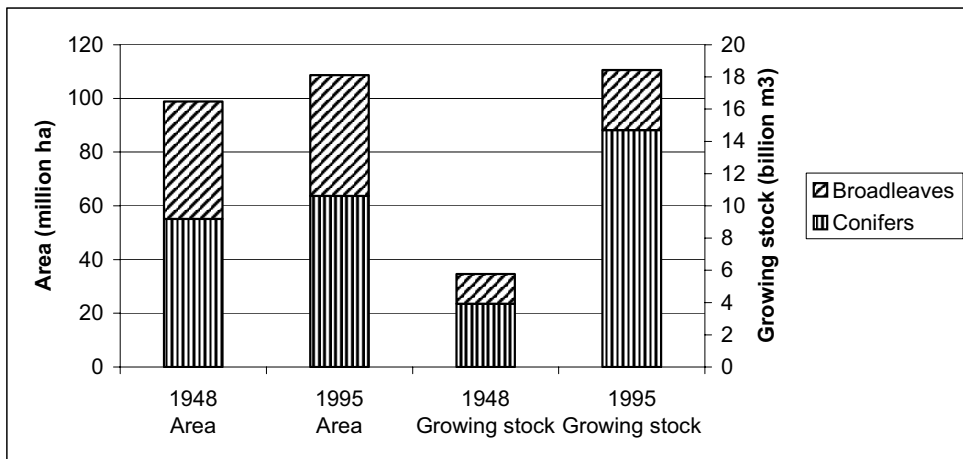
### *Storm*

Looking at the time series on storm damage, the inevitable conclusion is that damage from storms is increasing. As noted above, however, this conclusion is undermined by changes in literature sources over time. On the other hand, truly catastrophic events are likely to have been reported, because of their significant impact. Although too much uncertainty surrounds the figures for Europe to justify the conclusion that damage is increasing, more complete time series at national and regional levels also suggest this trend. For example, Holmsgaard (1986) demonstrates that storm damage in Denmark has increased since 1894, while Mosandl and Felbermeier (1999) show a sharp increase in sanitation fellings in Bayern (Germany) and in the Czech Republic since 1950. The amount of storm damage in Switzerland has also increased since 1868 (SFSO and FOEFL, 1996).

After the storms in December 1999, the question arose as to whether the frequency and intensity of storms had changed due to climate change and whether we could expect more storms and more severe storms in future. There have been various studies of the historical wind climate, and most conclude that although there are significant fluctuations between decades, there is no long-term trend in storminess (Dorland et al., 1999). Schiesser (1997) concludes that there were more storms in Switzerland before 1940 than in the second half of the century. Lässig and Mocalov (2000) found that neither the annual number of storms nor their mean wind speed increased over the period 1946-1996 in the Urals. Können (1999) concludes that in the Netherlands there was no increase in the number or intensity of storms in the 20<sup>th</sup> century. On the other hand, storms show a highly stochastic character, which makes it very difficult to detect a trend in a short time series.

If we assume that the storm climate has not changed, but that storm damage is increasing, we are left with the question of what has caused the increase in damage. Most studies point to changes in the state of the forest and to changes in forest management. After a long period of deforestation and forest overexploitation in most parts of Europe, reforestation started and a more regulated way of forest management was introduced. This process started at different times in different parts of Europe,

but the general pattern is the same. In this process of restoration, coniferous species were often used, because of their valuable wood, their ability to grow under unfavourable conditions on deforested and degenerated sites and the ease of artificial regeneration (e.g. Böhm, 1981). Spruce in particular was planted far beyond its natural range (e.g. Maurer, 1982). In this way, not only did the tree species composition shift from the more wind-firm broadleaved species towards more unstable conifers like spruce, but the forest area also expanded. For instance in Denmark, the total forest area increased from 200,000 ha in 1881 (Holmsgaard, 1982) to 442,000 ha in 1990 (Schelhaas et al., 1999), while the area of conifers increased from 50,000 ha in 1881 (Holmsgaard, 1982) to almost 300,000 ha in 1990 (Schelhaas et al., 1999). Since 1850 the area of state forest in Baden-Württemberg has increased by 16% and the share of spruce in the total area has increased from 20% to 40%, while the total share of conifers has increased from 42% to 63% (Lekes and Dandul, 2000). Not only has the increase in total forest area and the area of conifers played a role: developments in the growing stock and age class distribution have also influenced the stability of forest stands. During the last century the growing stock increased steadily in most European countries. From 1948 to 1995 the total area of conifers in 19 European countries with data for both years increased by almost 16%, while the total coniferous growing stock increased by 275% (Figure 12). Although differences may exist between the definitions used in the inventories for these two years, the general trend is very clear.



**Figure 12.** Total forest area, area of conifers and total volume of growing stock in the whole of Europe.

Lengthening the rotation periods, with the aim of obtaining trees with larger diameters, has caused the age class distribution of the forest to shift to the higher age classes. And because older trees are taller, forests are becoming more vulnerable to windthrow (Zimmermann, 1985, Mosandl and Felbermeier, 1999). For example, in the Czech Republic the age class distribution in 1920 showed almost no forest older than

80 years, but by 1995 such forest accounted for 30% of the total forest area. Møller (1957) attributes the increase in storm damage in Denmark to an increase in forests older than 45 years, which is the age at which windthrow starts to occur. UN-ECE/FAO (2000b) further mention that in France there is a long-term trend in converting coppice and coppice with standards into high forest, which also adds to the general vulnerability of the forests.

Besides these reasons, which were mentioned in most studies, some additional explanations were put forward. Emmer et al. (1998) state that centuries of large-scale spruce monocultures have negatively affected the forest soil, by changing the microclimate and the input of litter. This has led to deterioration of the sites, such as increased acidification of the soil, litter accumulation and declining nutrient cycling, which facilitates pest outbreaks and windthrow. Moreover, a large part of Central Europe suffers from heavy air pollution, which aggravates the decline in vitality (Emmer et al., 1998). Lekes and Dandul (2000) mention air pollution as a factor that causes root damage and thus increases the instability of the trees.

Another factor that influences the vitality and functioning of the roots is fungal infection. Some studies suggest that fungal infections of roots have increased over time, causing root systems to decrease in size and thus diminishing the anchorage of the trees (Holmsgaard, 1986, Eriksson, 1986). Damage due to harvesting activities can also contribute to an increase of fungal infections such as root and butt rot (Schmid-Haas and Bachofen, 1991).

Management activities in general can greatly influence the forest's resilience to disturbances. After a heavy thinning, stands are more vulnerable to damage from wind (Lohmander and Helles, 1987) and snow (Nykänen et al., 1997), but these effects disappear after a few years. Jalkanen and Mattila (2000) found that factors increasing a stand's susceptibility to wind damage included special cuttings, e.g. for ditches, power lines or roads, or sanitation fellings after damage. Newly clearcut areas also expose the edges of the remaining stands to the force of the wind. Careful planning of such activities could help to avoid damage. Hendrick (1986) notes that in Ireland, much planting has been done along open furrows, which encourages the root system to develop only on one side, which causes instability. Changes in management, e.g. in the thinning and regeneration regime, might influence the increasing trend, but it is difficult to generalise this on a European scale. However, the trend of increasing the mechanisation of forest work could have led to heavier and less frequent thinnings.

Another factor often mentioned in connection with severe wind damage is the occurrence of waterlogged soils during a storm. Waterlogged soils give less support to the roots and thus increase the susceptibility to windthrow (Laiho, 1987, Offergeld, 1986, Poeppel, 1994). In the Netherlands, although the storm climate does not seem to have changed, a significant increase in precipitation has been recorded (van Boxel and Cammeraat, 1999). If this trend is pan-European, it could have contributed to the increase in storm damage.

### *Fire*

The estimated area affected by fire also shows an increase over time, especially in the 1970s and 1980s. In the 1990s it seems to have decreased slightly, but there is much variation between years. The estimates of forest fire area in the 1960s are based on only a small sample (coverage 36%), so here the uncertainty is larger, which weakens conclusions on trends. The increase in forest fire area can mainly be found in the eastern Mediterranean region, which is confirmed by several national sources, such as for Greece, Albania and Bulgaria (Xanthopoulos 2000, Meta 2003, Konstantinov 2003). The same trend is observed in Spain. For the period 1968-1994 CREAM (1999) reports an increase in the number of wildfires and an increase in the fire hazard ratings in Catalonia, Spain. The increase of the hazard ratings correlates with an increase of temperature and a decrease in minimum relative humidity. However, their conclusion is that the increase in wildfires cannot be attributed wholly to this effect, since other relevant variables such as land use and human activity also changed over the same period. Their findings probably also apply to the European scale.

Forest management plays a role in fires, too. Generally, conifers are more prone to fires than broadleaves, so an increase in the area of conifers could contribute to an increase in forest fires. However, the increase in forest fire area is more than the increase in coniferous forest area. Not only the area of forest, but also its distribution over the landscape is very important. The abandoning of marginal agricultural land has allowed the area of continuous forest to increase, reducing the possibilities to stop fires. Forest management is also important in its influence on the structure and density of the forest and the amount of available fuel. It is difficult to assess the exact influence of forest management on the occurrence of forest fires, since this will vary very much per country.

In response to the increase in forest fires, in various countries more effort is being put into fire fighting and fire prevention (e.g. Dimitrakopoulos, 1990, Teusan, 1995). Furthermore, new technologies are becoming available for detecting and fighting fires. The possible decrease in fire area over the last years could reflect the increased alertness and improved fire-fighting techniques. Another indication of a more efficient fire-fighting system is that the average area burned per fire has decreased (data not shown).

Figure 3 shows a fast increase in the number of forest fires over time. This apparent trend is most likely also influenced by the afore-mentioned increase in forest fire detection and alertness. Improved fire-fighting techniques will prevent small fires to get out of hand. However, since the fuel load is not diminished, new fires will easily start. This leads to a change in fire size distribution, i.e. less large fires and more small ones. On the other hand, better detection systems will detect more very small fires, also the ones that would otherwise have remained undetected. Very few countries report size distributions. From the few regional data sources available, it indeed seems that the number of fires larger than 1 ha slightly decreases, against an increase in fires

smaller than 1 ha (data not shown). However, it is impossible to separate this trend into a real increase and an apparent increase due to better detection methods.

### *Biotic causes*

The reported biotic damage also seems to have increased, but not as dramatically as the abiotic damage. Here, too, the records are probably more complete for recent decades than earlier, which could explain at least part of the increase. The changes that have increased the abiotic damage also affect the biotic damage: increasing the growing stock puts larger areas of forest at risk, and monocultures are generally more susceptible to pests than mixed stands. Moreover, biotic damage is to a certain extent correlated with abiotic damage, since outbreaks of bark beetles are usually triggered by events such as storms that result in a large mass of dead or dying wood. Warm and dry summers are favourable for insect outbreaks, so the observed increase in average temperature could have had an influence on the amount of biotic damage. Holmsgaard (1986) and Eriksson (1986) also mention an observed increase in fungal infections of roots.

### *Implications*

It seems likely that changes in the state and management of forest have increased the amount of damage due to disturbances. At the European level, growing stock and the average age of forest are projected to continue to increase; this might lead to an increase in damage. A case study on Switzerland by Schelhaas et al. (2002) with a large-scale scenario model showed that due to the continuing increase in average growing stock and shifts in the age class distribution, the simulated amount of damage caused by abiotic factors increases. By 2048 the amount of damaged wood volume had increased by 40% as compared to 2004.

For this reason, forest managers in Europe should be prepared for an increase in the risk of the occurrence of disturbances. However, it is impossible to predict when and where these will occur. Given that forest management has probably had a large impact on the increase in damage, it seems logical that forest management could also be important in reducing these risks. Some of the current trends towards more nature-oriented forest management seem to have potential, such as the trend to replace conifers by broadleaved species. The greater interest in natural processes does not directly decrease the impact of disturbances, but makes them easier to accept and might even result in disturbances being integrated into the management to some extent.

In terms of their impact on the total European harvest and growing stock, disturbances are not very important. However, on a more local or regional scale their impact can be disastrous (Wiebecke, 1973): they can destroy a considerable proportion of the volume of growing stock (see for example Kühnel, 1994), up to virtually the whole forest. Putting the timber salvaged from the damaged forest up for sale floods the market and causes the price for wood to fall (UN-ECE/FAO, 2000b). So the

forest owner suffers a triple blow: higher harvesting costs (the timber from damaged forests is more difficult to extricate), losses of unrecoverable wood, and a drop in the revenue from the wood. Furthermore, the existing management plans will have to be modified and the damaged areas will have to be regenerated. Given the likelihood of increasing risk of disturbance damage, it would be wise to incorporate such risks into the management plans so that in the event of a disturbance event there will be more room for manoeuvre. Regional and national authorities should also be prepared for the occurrence of such an event and have contingency plans in ready.

Although disturbances are usually judged by their negative impacts, they can also provide opportunities. Research in areas regenerating after the storms of 1990 in Switzerland has shown that the mosaic of cleared and uncleared patches after a windthrow contributes to an increased biodiversity (Lässig, 2000). This is a bonus, given that biodiversity is an increasingly important goal in European forestry. The investigations in Switzerland provided insights into the behaviour of insect pest populations. The findings were used to draw up guidelines for the forest managers after the 1999 storm, on which areas had priority in clearing and where the wood could be left in the forest without too much risk. This enhanced the effectiveness of the labour and led to more wood being left in the forest to benefit biodiversity, with relatively little risk of outbreaks of insect pests. The evaluation of the 1999 storms in France was also not entirely negative. The reaction of the French State Forest Service (ONF) was that French forest policy had been too centralised, but that this storm had provided a unique opportunity to remodel the French forests, which had become too conifer-dominated and uniform (Rietbergen, 2000).

## Conclusions

From the data collected in the literature review we estimate that on average over the period 1960-2000 213,000 ha of forest was burned annually in Europe, which is about 0.15% of the total forest area. More than 90% of the area of forest burned per annum is in Portugal, Spain, France, Italy, Greece and Turkey. In these countries the area burned per annum is on average about 0.3% of the total forest area.

We estimate that about 0.15% of the standing volume is damaged annually (average 1950-2000), which is equivalent to 35 million m<sup>3</sup>. This is 8.1% of the total annual felling of the countries examined. Abiotic causes account for 77% of the total damage, with storm the most important factor, accounting for 53% of the total damage. Biotic causes account for 16%, and for the remaining 7% the cause is unknown or a combination of causes. These figures must be seen as rough estimates, taking into account the variety of information sources, the rough way of upscaling and the sometimes small information base.

Despite the uncertainties, an increasing trend in the amount of damage due to disturbances is discernable. The two most probable reasons for the increase in storm damage have to do with changes in the forest structure: the increase in the area of

coniferous forest and the dramatic increase in the total amount of coniferous growing stock. The occurrence of forest fires is probably influenced by forest management, but climate might also be influential. However, most important are the changes in the socio-economic situation. It is less easy to discern a trend for the biotic damage. Such damage is correlated with the occurrence of abiotic damage but is also dependent on the forest management. Given the increases in volume of growing stock, forest area and average stand age in Europe, it seems likely that the amount of damage due to disturbances will continue to increase. Given its importance in the observed increasing trend, forest management could also be crucial in reducing the risks.

To better understand the frequency and extent of damage due to natural disturbances than is possible now, a pan-European standardised monitoring system should be set up. Standardising the reporting, as has already been done for the statistics on forest fires, could be a first step towards such a monitoring system.

## References

- Annala, E and Petäistö, R-L (1978) Insect attack of windthrown trees after the December 1975 storm in Western Finland. Seloste: Hyönteisten lisääntyminen tuulen kaatamissa puissa Länsi-Suomessa vuoden 1975 joulukuun myrskyn jälkeen. *Communications Instituti Forestalis Fenniae*, **94**, 24.
- Avram, C (1983) Die Sturmschäden in den Wäldern Rumäniens und Europas. *Allgemeine Forstzeitschrift*, **38**, 301-302.
- Böhm, P (1981) Sturmschäden in Schwaben von 1950 bis 1980. *Allgemeine Forstzeitschrift*, **36**, 1380.
- CREAF (1999) *CREAF 1988-1998*, Center for Ecological Research and Forestry Applications - CREAF, Barcelona.
- Dimitrakopoulos, AP (1990) A Synopsis of the Greek Wildland Fire Problem. *International Forest Fire News*, **4**.
- Doll, D and Riou-Nivert, P (1991) Les chablis en Europe : Courons-nous au désastre? *Forêt-entreprise*, **77**, 1-16.
- Dorland, C, Tol, RSJ and Palutikof, JP (1999) Vulnerability of the Netherlands and Northwest Europe to storm damage under climate change: A model approach based on storm damage in the Netherlands. *Climatic Change*, **43**, 513-535.
- Emmer, IM, Fanta, J, Kobus, AT, Kooijman, A and Sevink, J (1998) Reversing borealization as a means to restore biodiversity in Central-European mountain forests - an example from the Krkonoše Mountains, Czech Republic. *Biodiversity and Conservation*, 229-247.
- Eriksson, H (1986) Windthrow damage in forests in relation to stand treatment - present state of knowledge in Sweden In *Minimizing wind damage to coniferous stands*(Ed, Communities, CotE) Løvenholm Castle, Denmark, pp. 36-39.
- FAO (1948) Forest Resources of the World. *Unasylva*, **2**, 161-182.



- FAO (1955) *World Forest Resources - Results of the Inventory undertaken in 1953 by the Forestry Division of FAO*, FAO, Rome, Italy. 120 pp.
- FAO (1960) *World Forest Inventory 1958*, FAO, Rome, Italy. 136 pp.
- FAO (1963) *World Forest Inventory 1963*. 113 pp.
- FAO (1976) *European Timber Trends and Prospects 1950 to 2000*. 308 pp.
- FAO (1982) *Forest Fire Statistics*, UN-ECE, FAO, Geneva. 23 pp.
- FAO (1984) *Forest Fire Statistics*, UN-ECE, FAO, Geneva. 32 pp.
- Hendrick, E (1986) Appropriate cultivation and drainage techniques for sites liable to windthrow In *Minimizing wind damage to coniferous stands*(Ed, Communities, CotE) Løvenholm Castle, Denmark, pp. 30-32.
- Holmsgaard, E (1982) Stormfald. *Tidsskr. For Landøkonomi*, **169**, 133-138.
- Holmsgaard, E (1986) Historical development of wind damage in conifers in Denmark In *Minimizing wind damage to coniferous stands*(Ed, Communities, CotE) Løvenholm Castle, Denmark, pp. 2-4.
- Jalkanen, A and Mattila, U (2000) Logistic regression models for wind and snow damage in northern Finland based on the National Forest Inventory data. *Forest Ecology and Management*, **135**, 315-330.
- Können, GPr (1999) *De toestand van het klimaat in Nederland 1999*, Koninklijk Nederlands Meteorologisch Instituut, De Bilt. 64 pp.
- Konstantinov, V. (2003). "Bulgaria - Analysis of the Problems Connected to Forest Fires in Bulgaria." *International Forest Fire News*, **28**.
- Kühnel, I (1994) *Dokumentation der Sturmschäden 1990*, Schriftenreihe der Landesforstverwaltung Baden-Württemberg 75. 190 pp.
- Kuusela, K (1994) *Forest resources in Europe*, European Forest Institute Research Report 1 Cambridge University Press. 154 pp.
- Laiho, O (1987) *Metsiköiden alttius tuulitubolle Etelä-Suomessa. Susceptibility of forest stands to windthrow in southern Finland*, Folia Forestalia 706. 24 pp.
- Lässig, R (2000) Windwürfe - Chancen für artenreiche Wälder nutzen. *Wald und Holz*, **81**, 56-60.
- Lässig, R and Mocalov, SA (2000) Frequency and characteristics of severe storms in the Urals and their influence on the development, structure and management of the boreal forests. *Forest Ecology and Management*, **135**, 179-194.
- Lässig, R and Schönenberger, W (2000) Nach 'Lothar' und 'Vivian'-Erfahrungen profitieren. *Wald und Holz*, **81**, 31-35.
- Lekes, V and Dandul (2000) Using airflow modelling and spatial analysis for defining wind damage risk classification (WINDARC). *Forest Ecology and Management*, **135**, 331-344.
- Lohmander, P and Helles, F (1987) Windthrow probability as a function of stand characteristics and shelter. *Scandinavian Journal of Forest Research*, **2**, 227-238.
- Maurer, E (1982) 25 Jahre Sturm- und Schneeschaden in der Bundesrepublik Deutschland (1953/54 bis 1977/78). *Allgemeine Forstzeitschrift*, 395-397.
- Maxwell Macdonald, J (1952) Wind damage in middle aged crops of Sitka spruce. *Scottish Forestry*, **6**, 82-85.

- McCarthy, J. J., O. F. Canziani, et al., Eds. (2001). *Climate Change 2001: Impacts, Adaptation, and Vulnerability*. Contribution of Working Group II to the Third Assessment Report of the Intergovernmental Panel on Climate Change (IPCC). Cambridge, Cambridge University Press.
- Meta, M. (2003). "Albania - Update on the Forest Fire Situation." *International Forest Fire News*, **28**.
- Miller, KF (1985) *Windthrow Hazard Classification*, Forestry Commission Leaflet 85 HMSO, Londonpp.
- Møller, CM (1957) Stormfahdelts betydning for dansk skovbrug. [The importance of wind damage in Danish forestry]. *Dansk Skovforenings Tidsskrift*, **42**, 526-543.
- Mosandl, R and Felbermeier, B (1999) Auf dem Weg zum naturnahen Wald. *AFZ/Der Wald*, 910-914.
- Nykänen, M-L, Peltola, H, Quine, CP, Kellomäki, S and Broadgate, M (1997) Factors affecting snow damage of trees with particular reference to European conditions. *Silva Fennica*, **31**, 193-213.
- Offergeld, JP (1986) The consequences of the tempest of November 1984 on the forest In *Minimizing wind damage to coniferous stands*(Ed, Communities, CotE) Lövenholm Castle, Denmark, pp. 21-26.
- Pfeffer, A and Skuhavy, V (1995) Der Buchdrucker *Ips typographus* L. (Col., Scolytidae) und seine Problematik in der Tschechischen Republik. *Anzeiger für Schadlingskunde, Pflanzenschutz, Umweltschutz*, **68**, 151-152.
- Plochmann, R and Hieke, C (1986) *Schadereignisse in den Wäldern Bayerns. Eine Zusammenstellung der forstlichen Literatur seit Beginn des 18. Jahrhunderts*, Forstliche Forschungsberichte München 71. 151 pp.
- Poepfel, S (1994) Der Windwurf am 12. Februar 1894. *Forst und Holz*, **49**, 190.
- Rhein-Zeitung (1999) Orkan gilt als Zeichen des Klimawandels In *Rhein-Zeitung*.
- Rietbergen, S (2000) The tempest - A tale of two countries. *Arbor Vitae - The IUCN/WWF Forest Conservation Newsletter*, 11.
- Rottmann, M (1986) *Wind- und Sturmschäden im Wald*, J.D. Sauerländer's Verlag, Frankfurt am Main. 128 pp.
- Schelhaas, MJ, Nabuurs, GJ, Sonntag, M and Pussinen, A (2002) Adding natural disturbances to a large-scale forest scenario model and a case study for Switzerland. *Forest Ecology and Management*, **167**, 13-26.
- Schelhaas, MJ, Varis, S and Schuck, A (2001) *Database on Forest Disturbances in Europe (DFDE)*. <http://www.efi.fi/projects/dfde/> European Forest Institute, Joensuu, Finland
- Schelhaas, MJ, Varis, S, Schuck, A and Nabuurs, GJ (1999) *EFISCEN's European Forest Resource Database (EEFR)*. <http://www.efi.fi/projects/eefr/> European Forest Institute, Joensuu, Finland
- Schiesser (1997) Winter storms in Switzerland. *Theoretical and applied climatology*, **58**, 1-19.
- Schmid-Haas, P and Bachofen, H (1991) Die Sturmgefährdung von Einzelbäumen und Beständen. *Schweizerische Zeitschrift für Forstwesen*, 477-504.

- Schwerdtfeger, F (1955) *Pathogenese der Borkenkäfer - Epidemie 1946 - 1950 in Nordwestdeutschland*, Schriftenreihe der forstlichen Fakultät der Universität Göttingen und Mitteilungen der niedersächsischen forstlichen Versuchsanstalt 13-14 Sauerländer, Frankfurt a.M. 135 pp.
- SFSO and FOEFL (1996) *Forestry and Timber Industry in Switzerland, Yearbook*, Swiss Federal Statistical Office, Bernpp.
- Teusan, S (1995) France: Forest Fire Profile. *International Forest Fire News*, **13**.
- UN-ECE and FAO (1985) *The forest resources of the ECE region (Europe, the USSR and North America)*, United Nations Economic Commission for Europe, Food and Agriculture Organization. Geneva. ECE/TIM 27.pp.
- UN-ECE and FAO (1992) The Forest Resources of the Temperate Zones, The UN-ECE/FAO 1990 Forest Resource Assessment. Vol 1. General Forest Resource Information. *United Nations Economic Commission for Europe. Food and Agriculture Organization. ECE/TIM*, 27.
- UN-ECE and FAO (2000) Forest resources of Europe, CIS, North America, Australia, Japan and New Zealand In *Geneva Timber and Forest Study papers No*, Vol. 17.
- UN-ECE/FAO (1996) Forest fire statistics 1993-1995 In *Timber Bulletin*, Vol. 49 United Nations Economic Commission for Europe (Geneva), Food and Agriculture Organization of the United Nations (Roma), pp. 1-19.
- UN-ECE/FAO (1998) Forest fire statistics 1995-1997 In *Timber Bulletin*, Vol. 51 United Nations Economic Commission for Europe (Geneva), Food and Agriculture Organization of the United Nations (Roma), pp. 1-19.
- UN-ECE/FAO (1999) Forest fire statistics 1996-1998 In *Timber Bulletin* United Nations Economic Commission for Europe (Geneva), Food and Agriculture Organization of the United Nations (Roma).
- UN-ECE/FAO (2000a) Forest fire statistics 1997-1999 In *Timber Bulletin*, Vol. 53 United Nations Economic Commission for Europe (Geneva), Food and Agriculture Organization of the United Nations (Roma).
- UN-ECE/FAO (2000b) *Forest Products Annual Market Review*, United Nations Economic Commission for Europe (Geneva), Food and Agriculture Organization of the United Nations (Roma)pp.
- UN-ECE/FAO (2001) Forest fire statistics 1998-2000 In *Timber Bulletin*, Vol. 54 United Nations Economic Commission for Europe (Geneva), Food and Agriculture Organization of the United Nations (Roma).
- van Boxel, J and Cammeraat, E (1999) Wordt Nederland steeds natter? Een analyse van de neerslag in deze eeuw. *Meteorologica*, **99**, 1-15.
- Waldschutzsituation (1999) Waldschutzsituation im deutschsprachigen Raum. *Allgemeine Forstzeitschrift*, **54**, 324-329.
- Waldschutzsituation (2000) Waldschutzsituation im deutschsprachigen Raum. *Allgemeine Forstzeitschrift*, **55**, 332-337.
- Wiebecke, C (1973) Volkswirtschaftliche Bedeutung der Sturmkatastrophe 1972 und forst- und holzwirtschaftspolitische Folgerungen. *Forstarchiv*, **44**, 140-143.

- Xanthopoulos, G. (2000). "The 1999 Forest Fire Season." *International Forest Fire News*, **22**, p. 76-84.
- Zari (1981) Studie o výskytu kalamit na území CSR od roku 1900 In, Lesprojekt, Brandýs nad Labem, pp. 107.
- Zatloukal, V (1998) Historické a současné příčiny kurovcové kalamity v Národním parku Šumava. [Historical and current factors of the bark beetle calamity in the Šumava National Park]. *Silva Gabreta*, **2**, 327-357.
- Zimmermann, H (1985) Zur Begründung von Mischbeständen mit Fichte und Buche auf Sturmwurfflächen im öffentlichen Wald Hessens. *Allgemeine Forstzeitung München*, **49**, 1326-1330.

## II Adding natural disturbances to a large scale forest scenario model and a case study for Switzerland

*Forest Ecology and Management 167: 13-26*

*M.J. Schelbaas<sup>1,2</sup>, Nabuurs<sup>1,2</sup>, M. Sonntag<sup>3</sup>, A. Pussinen<sup>2</sup>*

<sup>1</sup>) Alterra, Green World Research, PO Box 47, NL-6700 AA Wageningen, the Netherlands

<sup>2</sup>) European Forest Institute, Torikatu 34, FIN-80100, Joensuu, Finland

<sup>3</sup>) Center for Environmental Systems Research, University Gh Kassel, Kurt-Wolters-Str.3, D-34109 Kassel, Germany

### Abstract

In this study we assessed the impact of climate change and the role of natural disturbances on the development of forest resources. A module dealing with natural disturbances was developed for the European Forest Information Scenario Model (EFISCEN), based on the observed distribution of damages in the past and the dependency of disturbances on forest characteristics. To put the model to the proof, the development of Swiss forest resources was projected until 2048 under three alternative scenarios. The first scenario consisted of a run without the new module (the base run), in the second scenario the model was run with the new module under current climate (natural disturbance run) and the third scenario was a run with the new module under a changed climate (natural disturbance and climate change run), where outcomes of the process based model TREEDYN3 were used to simulate the influence of a changing climate on the volume increment and where the frequency of disturbances was increased to simulate the effect of climate change on natural disturbances.

Incorporating natural disturbance dynamics in the EFISCEN model resulted in a more realistic simulation of the total fellings and natural mortality, due to the fact that killed, but unrecovered, timber is also taken into account, resulting in a better simulation of volume and increment development. When the natural disturbances module was used in the simulation, the growing stock increased from 366 m<sup>3</sup> ha<sup>-1</sup> in 1984 to 460 m<sup>3</sup> ha<sup>-1</sup> in 2048, while without disturbances it increased up to 592 m<sup>3</sup> ha<sup>-1</sup>. The simulation under current climate showed an increase in damage due to natural disturbances of 40% over the period 2004-2048, due to an increase of growing stock and a higher proportion of older stands. Under a simulated climate change scenario the frequency of disturbances was assumed to increase, which resulted in 25% higher damages. However, the increment increased more than the damage done by disturbances, which resulted in a simulated growing stock volume of 530 m<sup>3</sup> ha<sup>-1</sup> in

2048. The increase in damage must be attributed to an increasing average standing volume. Because of uncertainties caused by the assumptions made in the model and the stochastic character of the disturbances, the results of this study must be seen as merely indicative.

Keywords: Switzerland, natural disturbances, large-scale scenario modelling, EFISCEN, climate change

## Introduction

The European forests<sup>2</sup> cover 168.5 million hectares, mostly regularly managed (UN-ECE/FAO 2000). These are the most intensively managed forests in the world, accounting for only 4% of the world's forest area, but producing 24% of the global industrial roundwood. At the same time, they have to fulfil many other demands. European-scale projections are therefore essential to foresee the long term impacts of alternatives in forest policy and management under environmental changes (Nabuurs and Päivinen 1996, Nabuurs 2001).

The European Forest Information Scenario Model (EFISCEN) is used to make these long-term projections of the European forest resources (Nabuurs 2001), but the analyses have never incorporated the combined impact of natural disturbances and climate change. The impacts of these natural disturbances seem to be increasing. For example, in 1990 and 1999, Western and Central Europe suffered from severe storms that caused unprecedented damage to the forests. In 1990, 120 million m<sup>3</sup> timber was windthrown and in 1999 another 190 million m<sup>3</sup> was damaged (Schelhaas and Nabuurs In prep.); in comparison to the usual annual fellings of around 400 million m<sup>3</sup> (UN-ECE/FAO 2000), these are large amounts. After the 1990 storms, bark beetles developed fast and caused additional damage of about 30 million m<sup>3</sup> in the affected countries (Schelhaas and Nabuurs In prep.).

In the Mediterranean area, forest fire is the main disturbance factor. Over the period 1970-1997 on average 336,000 ha of forest burned per annum (Schelhaas and Nabuurs In prep.).

Hence, disturbances are already having a marked impact on the forests and forest management in Europe. Several simulation studies investigated the influence of a changing climate on natural disturbances. Some studies found an increase in storminess (Zwiers and Kharin 1998, Lunkeit et al 1996), while others reckon with a decrease (Lässig and Schönenberger 2000). Changing precipitation patterns and an

---

<sup>2</sup> European excludes the forests of the European part of the Russian Federation and Newly Independent States.

increase in temperature could lead to a higher forest fire risk (Kurz et al 1995, Gerstengarbe et al 1999), although the opposite is found as well (Flannigan et al 1998). Also the biotic disturbances will be influenced by climate change, but due to the many possible interactions and feedbacks, it is very difficult to predict how this influence will develop (Fleming 1996).

Therefore, insight in the future role of natural disturbances in European forests is essential because 1) growing stocks in Europe are increasing, which increases the potential amount of damage, 2) it is unsure what will happen with the frequency and intensity of disturbances under climate change, 3) forest managers are increasingly interested in mimicing natural disturbance patterns in the forest under a more close to nature forest management.

The first objective of our study was to extend the large scale scenario model EFISCEN with a module that takes into account the influence of natural disturbances, depending on the actual state of the forest and which should be able to handle hypothetical changes in disturbance regimes due to climate change. The second objective was to parametrise and apply the newly developed module to a test area in order to study its behaviour and impact. For this case study Switzerland was chosen, because natural disturbances play an important role in the forest management in Switzerland and because detailed and reliable datasets were available.

## Methods and Models

### *The EFISCEN model*

The core dynamics of the European Forest Information SCENario model (EFISCEN) are based on a model developed by Sallnäs (1990). EFISCEN is an area-based matrix model that is especially suitable for application on a regional or a country level. National forest inventory data are used as input for the EFISCEN model, including area, volume and net annual increment by age classes. Per country, forest types can be distinguished by region, owner class, site class and tree species, depending on how detailed the input data are. The following summary is based largely on the description by Nilsson et al. (1992) and Nabuurs et al. (2000).

The state of the forest is depicted as an area distribution over age and volume classes in a matrix. A separate matrix is set up for each forest type that can be distinguished. Growth is simulated by moving a certain part of the area in a cell to a higher volume class. The growth models are depicted by the following function:

$$I_{vf} = a_0 + \frac{a_1}{T} + \frac{a_2}{(T)^2} \quad (1)$$

where

$I_{vf}$  is the five-year volume increment in percent of the standing volume,  $T$  is the stand age in years, and  $a_0$ ,  $a_1$ ,  $a_2$  are coefficients. This function is used to calculate the

fraction of the area per cell that will be moved to a higher volume class per time step of five years. These growth functions can be modified in order to simulate growth changes, for instance due to climate change.

Management is controlled at two levels in the model. First a basic management regime per forest type, such as thinning and final felling regime, is incorporated. These regimes define per forest type the age and volume classes in which thinning and final felling can be carried out, thus constraining the maximum amount of wood that can be felled. Second, in the scenario, total required volume of harvest from thinnings and final felling are specified for the country as a whole per species group for each time step. Taking into account the actual state of the forest and the constraints set by the basic management regime, the model tries to find the specified felling levels.

The fraction of a cell that is actually growing (i.e. changing volume class) can be subjected to thinning. In this case, the area stays in the same volume class instead of moving up one volume class. The volume that would otherwise have been gained is regarded as having been removed by thinning. The area that has recently been thinned will react to the thinning by an increased volume increment. When a certain area is clear cut, the area is transferred from the matrix to the “bare forest land” class, from where it can enter the matrix again.

In a scenario, a felling level is defined as what should be achieved given the restrictions set by the management regimes. Further, in the scenarios afforestation can be added and growth changes can be defined. The output consists of the state of the forest after each time step, given in terms of e.g. increment, fellings, growing stock and age class distribution.

### ***The TREEDYN3 model***

The TREEDYN3 forest simulation model (Bossel 1996, Sonntag 1998) is a process-based model of tree growth, carbon and nitrogen dynamics for even-aged, monospecies forest stands. The model includes explicit formulations of ecophysiological processes, such as: computation of solar radiation as a function of seasonal time, daytime and cloudiness, light attenuation in the canopy, and canopy photosynthesis as a function of latitude, seasonal time and daytime, respiration of all tree parts, assimilate allocation, increment formation, nitrogen fixation, mineralization, humification, and leaching, forest management (thinning, felling, litter removal, fertilization etc.), temperature effects on respiration and decomposition, and environmental effects (pollution damage to photosynthesis, leaves, and fine roots). Only ecophysiological parameters that can be either directly measured or estimated with reasonable certainty are used. Consequences of increased greenhouse gas concentrations affect increment via temperature and precipitation changes as well as via CO<sub>2</sub> fertilisation.



## *Incorporating natural disturbances into EFISCEN*

In this study, only those disturbances that cause tree mortality are considered, divided into fire, storm/snow, and insects, as well as natural mortality due to self-thinning or ageing. The disturbances are subdivided into those ones that destroy the whole stand (further referred to as stand-replacing disturbances) and those ones that only kill part of the trees (non-stand-replacing disturbances). In the model, stand-replacing disturbances are dealt with in the same way as a final felling: an area that undergoes a stand-replacing disturbance is transferred from the matrix to the bare forest land class and will be able to move back into the matrix again the next time step. The total standing volume of that area before the disturbance is regarded as the disturbed volume.

An area that is subjected to a non-stand-replacing disturbance is moved one volume class down in the matrix. The difference between the average volume of the volume classes is the disturbed volume. These areas do not show an increased increment, like after thinning in the normal management. This represents the difference between a thinning through management, with the aim of better growth afterwards, and a disturbance that does not necessarily kill the trees that would have been removed in a thinning.

### *Storm and fire*

Disturbances have a stochastic character, depending on the weather and other circumstances. Mandallaz and Ye (1997) demonstrated that the annual number of forest fires in South Switzerland follows a lognormal distribution. We assumed here that the area burned per annum and the annual storm damaged area are lognormally distributed. A lognormal distribution is characterised by an average and a standard deviation, where the expected value is calculated with formula 1.

$$E = e^{\left( \bar{x} + s^2 / 2 \right)} \quad (2)$$

- E = expected value, which is the annual percentage of area that is disturbed  
 x = average of the lognormal distribution  
 s = standard deviation

A forest's susceptibility to a certain disturbance depends on varying characteristics, such as tree species composition (Bosshard 1967, Conedera et al. 1996, Netherer and Führer 1998), thinning history (Miller 1985), altitude (Bosshard 1967, Winterhoff et al. 1995), development stage (Laiho 1987) and age (Putz 1968, Grayson 1989, Becker and Schröter 2000). In order to express the susceptibility of the forest in relation to its

characteristics, a susceptibility matrix is set up for each forest type. In this matrix, the disturbance susceptibility for each disturbance type is defined per cell, depending on forest type, age and volume, relative to other cells.

$S_{i,j,k,l}$  : susceptibility of cell (i,j,k) to disturbance, relative to other cells,

where:

- i : forest type descriptor
- j : age class
- k : volume class
- l=1 : stand replacing disturbance
- l=2 : non-stand replacing disturbance

Using the initial areas per cell, the weighted average of the susceptibilities is calculated, using formula 3.

$$W = \frac{\left( E * \sum_i \sum_j \sum_k X_{i,j,k} \right)}{\sum_i \sum_j \sum_k \sum_l (X_{i,j,k} * S_{i,j,k,l})} \tag{3}$$

- W = weighed average of the susceptibilities
- $X_{i,j,k}$  : initial area in cell i,j,k

Every time step, five random draws are made from the lognormal fire and storm distributions. Per disturbance type an actual "hazard" ratio is calculated, by comparing the sum of the random draws with the expected value (formula 3).

$$A = \frac{\sum^5 RND}{E * 5} \tag{4}$$

- A = actual hazard ratio
- RND = random draw from the lognormal disturbance distribution

The actual disturbed fraction per time step (5 years) of the area per cell is then calculated using formula 5.

$$F_{i,j,k,l} = S_{i,j,k,l} * 5 * W * A \tag{5}$$

$F$  = fraction of the area in the cell that will be subject to disturbance

The wind damage in the recently thinned area is doubled, to reflect the greater susceptibility of recently thinned stands to storm damage (Miller 1985, Laiho 1987).

### ***Insect***

In our study, insect damage was directly linked to the amount of storm damage to reflect the dependence of insects on the amount of breeding material (Forster 1993, Weslien and Schröter 1996). Insect development also depends on weather conditions (Wermelinger et al. 1999). Generally, warm and dry weather stimulates rapid development, but cold and wet summers hamper this. The risk of forest fire is related to weather conditions in the same way. Therefore, insect damage is increased in years with a high occurrence of forest fires, relative to the fire hazard ratio. In the model, insect outbreaks as a consequence of a severe storm occur in the same time step as the storm itself, so in the simulations insect outbreaks last on average 2.5 years, while in reality they can last up to 8 years (Wermelinger et al. 1999).

### ***Natural mortality***

The factor natural mortality is not incorporated stochastically. Instead, in every cell, a percentage of the area is moved one volume class down, to simulate the death of a few trees. This percentage can vary, according to forest type, volume and age.

In the simulation, disturbances take place before the normal fellings are applied. First, the stand-replacing disturbances take place, and then the non-stand-replacing disturbances are applied to the resulting area. In reality, after a large disturbance occurs, the planned fellings are usually stopped or decreased (see e.g. Rohmeder 1933, Poecke 1984). However, it was not possible to quantify this effect of disturbances on the planned harvest. In the model, therefore, feedback is not applied to the planned harvest level.

Because of the stochastic character of the approach, Monte Carlo simulation was used: in our study, the average of 300 runs was calculated, as well as the standard deviation. The extra output from the natural disturbances module consists of disturbed volume and area, separated per disturbance type and tree species.

All volume values in this study are overbark, unless mentioned otherwise. Total drain is defined as the difference in growing stock between two periods. It includes fellings, both planned and sanitation fellings, as well as trees that have died for other reasons.

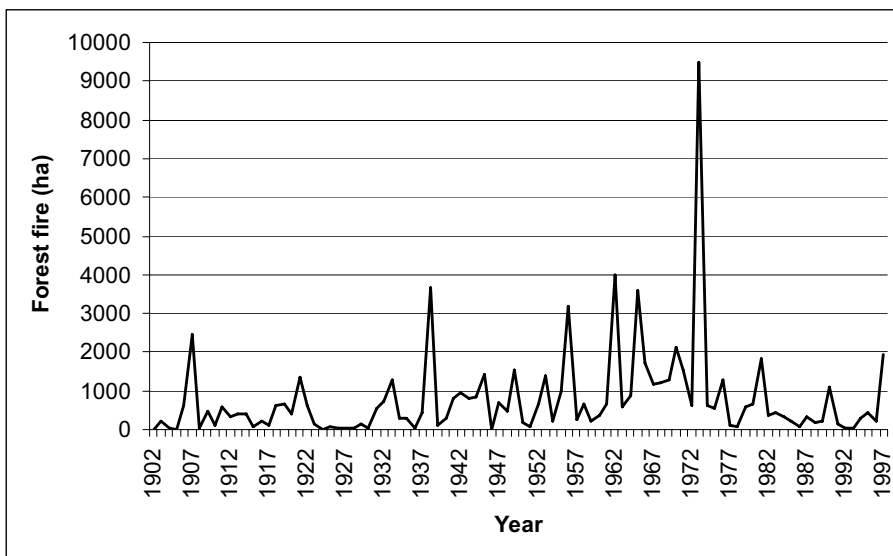
## **Case study: Switzerland**

A case study was carried out for Switzerland in order to study the performance of the disturbance module and its impact. Detailed and reliable data sets were available for Switzerland, both on current forest state and on disturbances in the past. Natural disturbances play a significant role in the management of the forest in Switzerland,

where about 30% of the fellings between 1985 and 1995 were sanitation fellings (Brassel and Brändli 1999).

The results of the first nation-wide forest inventory (NFI 1, Mahrer 1988), describing the state of the forest, were used as input for the model. The input data distinguish 5 regions, 6 site classes and 2 tree species groups, namely coniferous species and deciduous species. The results of NFI 2 (Brassel and Brändli 1999) were used to parameterize the disturbance module and serve as an intermediate check for the simulations.

Fire does not play a big role in Switzerland; the average area destroyed by fire per annum is only 0.075% of the total forest area, or almost 800 ha (Figure 1). The share of sanitation fellings necessitated by fire in the 10-year period between the two forest inventories was only 1.6% of the total volume of sanitation fellings (Brassel and Brändli 1999). Fire area data for the whole of Switzerland were available for the period 1977-1997 only. For the region Alpensüdseite, data were available for the period 1903-1997. Over the period 1977-1997, Alpensüdseite accounted on average for 76.3% of the total fire area. This figure was used to extrapolate the total fire area for the period 1903-1976.



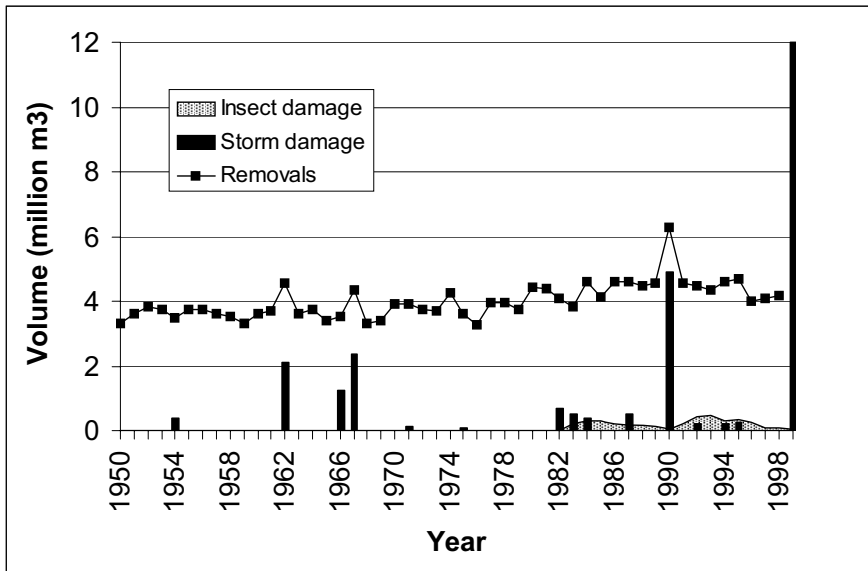
**Figure 1.** Burned forest area per annum in Switzerland 1903-1997 (Conedera et al. 1996, SFSO and FOEFL 1996, UN-ECE/FAO 1984, 1996, 1998). For the period 1903-1976, the area burned per annum was extrapolated from data for the Alpensüdseite region.

According to Conedera et al. (1996), most fires occur in broadleaved forest, thus we assumed that 75% of the total forest fire area is broadleaved. According to Brassel and Brändli (1999), sanitation fellings in response to fire were carried out on an area of 600 ha in the 10-year period between NFI 1 and 2. We regarded this as being the total amount of stand-replacing fire. According to Figure 1, total forest fire area in the same

period was 2632 ha. We concluded that 22.6% of the annual burned area is stand-replacing fire. This agrees with Conedera et al. (1996), who state that most forest fires are ground fires only. These figures were used to parametrise the susceptibility matrix for fire.

In Switzerland's forests, storms account for the largest proportion of damage due to natural disturbances. In the 10-year period between NFI 1 and NFI 2, 66.4% of the sanitation fellings were necessitated by storms and 1.6% by snow. The storms of 1990 and 1999 were especially severe, damaging 5 and 12 million m<sup>3</sup> of timber respectively (Figure 2). This is 1.1 and 2.6 times the average annual removal respectively.

We derived the statistical distribution of storm damage over the period 1950-1999 using Figure 2 for the more extreme damage in combination with an assessment of "background damage", derived from the difference in total storm damage in the period between NFI 1 and NFI 2 (Brassel and Brändli 1999) and the sum of damage as given in Figure 2 for the same period. This led to a background storm damage of 1.16 million m<sup>3</sup> per year. We assumed that this background damage has remained constant since 1950. The data in Figure 2 were then converted to area, based on the ratio of stand-replacing to non-stand-replacing volume and the affected volume per hectare for stand- and non-stand-replacing disturbances (Lässig and Schönenberger 1997). This yielded an average of 11,100 ha for the area that is disturbed by storms per annum.



**Figure 2.** Major storm damage 1950-1999 (SFSO and FOEFL 1996), insect damage 1983-1998 (Meier et al.1999, Wermelinger et al. 1999) and removals 1950-1999 (Schlaepfer and Haemmerli 1990, SFSO and FOEFL 1996, UN-ECE 1999). The damage due to the 1999 storm is a preliminary

According to Bosshard (1967), the risk of storm damage is lower in regions at altitudes above 900 m, because storms usually have constant high wind speeds at higher levels and a gusting character on lower altitudes. Based on this, the proportion of forests above 1000 m was assessed for each region and the relative risk was set accordingly. The damage distribution in Norway spruce (*Picea abies* (L.) Karst.) stands per age class after a storm given in Putz (1968) was used to express the coniferous forests' increasing susceptibility to storm with increasing age. Damage in spruce stands starts to occur at ages above 20 years and in broadleaved species after 80 years (Winterhoff et al. 1995). In general, broadleaved species are less susceptible to storm damage (Bosshard 1967, Leitner 1968).

The total amount of sanitation fellings necessitated because of insect attack during the period between the two forest inventories was 2.598 million m<sup>3</sup>, which is 13.3% of the total sanitation fellings (Brassel and Brändli 1999). As already explained, insect damage in the model depends on the risk of windthrow (which supplies dead stems suitable to breed in) and the fire risk (warm and dry summers). Even when no large storms occur, there will be some insect damage. According to Wermelinger et al. (1999), the level of insect damage before the storm of 1990 was "normal". Figure 1 shows that this is about 0.1 million m<sup>3</sup> per year (1989). The model was parameterized in such a way that in average fire and storm years the insect damage was about 0.1 million m<sup>3</sup> per year.

Coniferous species suffer most from insect attacks (Forster 1993, Frey et al. 1995, Wermelinger et al. 1999). We assumed that 80% of insect damage occurs in conifers and 20% in broadleaves. Further we assumed that insect attacks occur only in stands older than 40 years (Becker and Schröter 2000) and that stand replacing disturbance is 33% of the total amount of insect damage.

In the 10-year period between the two inventories, 11.6% of the sanitation fellings were necessitated by loss of vitality and 5.6% by other reasons (Brassel and Brändli 1999). In the model, natural mortality is solely dependent on the state of the forest. For the natural mortality we assumed that in forests younger than 80 years per 5 years 1% of the area will move down one volume class. For older forests, the percentage increases by 0.25 percent point per 5-year age class.

## Scenarios

In this study, we ran three different scenarios, all based on the "business as usual scenario" of the European base line study (Nabuurs 2001). In the "business as usual scenario", the current felling level is also applied in future. This scenario is realistic, since fellings have not changed greatly in recent decades (Figure 2). All scenarios were run for the period 1984-2048.

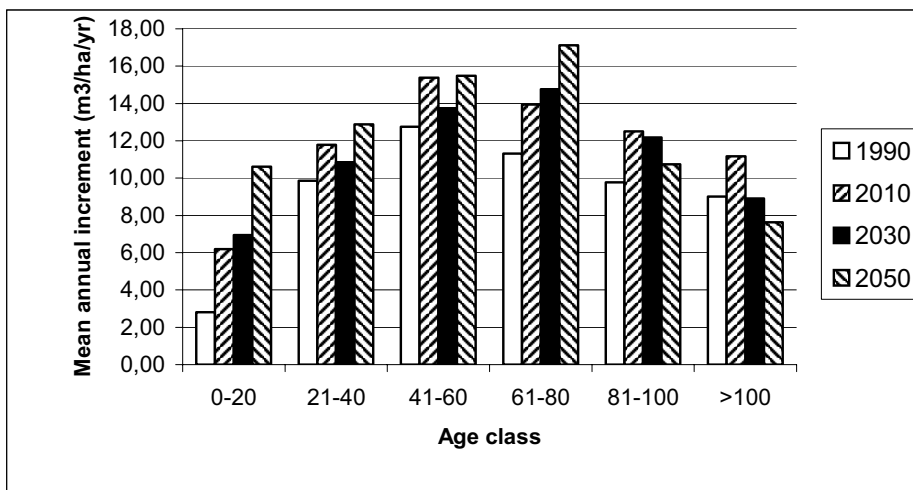
In the first scenario, the model was run without the new dynamic disturbance module. Disturbances were only taken into account via the sanitation fellings that are included in the historical felling level. In the second scenario, natural disturbances

were included using the newly developed disturbance module. The third scenario was a climate change scenario, with an increase in both the natural disturbance level and the increment level.

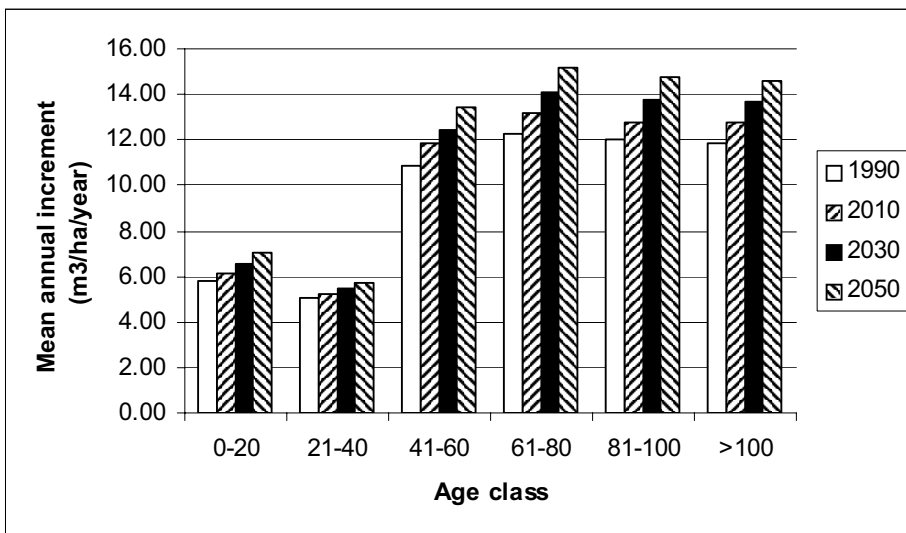
According to Brassel and Brändli (1999), total fellings in the 10-year period between NFI 1 and NFI 2 amounted to 62.3 million m<sup>3</sup> (overbark), of which 15.91 million m<sup>3</sup> were sanitation fellings. In scenario 1 an annual felling level of 6.23 million m<sup>3</sup> was applied, except for the period 1999-2003. Here annual felling level was set at 7.58, to simulate the increase in fellings necessitated by the 1999 storms. In scenarios 2 and 3, with dynamic disturbances, an annual felling level of 4.62 million m<sup>3</sup> was applied, which is total annual fellings minus the average annual sanitation fellings. In scenarios 2 and 3, for the period 1984-1999 the observed annual areas of storm damage and fire in those years (Figure 1 and 2) are used for parameterisation of the model runs, instead of generating them randomly from their distributions.

The input data covered 1,044 million ha, which is the exploitable, stocked forest, without scrubland in NFI 1 (Mahrer 1988). According to Brassel and Brändli (1999), the area of exploitable, productive forest in 1995 was 1,109 million ha, 65,000 ha more than our input. Part of this difference is the temporarily unstocked area (regeneration area) that is not included in the input data. The rest is afforestation, mainly on abandoned agricultural lands. In all scenarios these 65,000 ha was added as afforestation, distributed evenly over the 10 years between NFI 1 and NFI 2.

In scenario 3, possible effects of climate change are taken into account. Results from a climate change impact assessment study with TREEDYN3 were used to simulate the effects of elevated CO<sub>2</sub> and temperature on the growth of forests (Kramer and Mohren 2001). In that study, TREEDYN3 was run for a Norway spruce stand in Switzerland and a beech (*Fagus sylvatica* L.) stand in Austria. In this climate change assessment, weather data were generated (see Karjalainen et al. In press) under the HadCM2 scenario (Mitchell et al. 1995), which assumes that CO<sub>2</sub> concentration doubles in the period 1990 to 2100. On average the climate scenario predicted an increase in mean temperature of 1.5 °C between 1990 and 2050, and an increase in annual precipitation of 5 to 15%. The outcome of the TREEDYN3 study was a gradual increase in volume increment for both Norway spruce and beech. Figure 3 and 4 show the mean annual increment per age class for 1990, 2010, 2030 and 2050 as simulated by TREEDYN3. The growth curves in the EFISCEN model for these years were adapted relative to these figures, where the results for Norway spruce are used for the coniferous species and the results of beech for the broadleaved species. In the periods 1990-2010, 2010-2030 and 2030-2050 the growth changes were linearly interpolated.



**Figure 3.** Mean annual increment per age class in 1990, 2010, 2030 and 2050 as simulated by TREEDYN3 under the HadCM2 scenario, for Norway spruce.



**Figure 4.** Mean annual increment per age class in 1990, 2010, 2030 and 2050 as simulated by TREEDYN3 under the HadCM2 scenario, for beech.

Climate change affects not only tree growth, but possibly also the rate of natural disturbances. Warmer and drier summers can stimulate the fast development of insect populations (see e.g. Cannon 1998) and increase the risk of forest fire (see e.g. Kurz et al. 1995). Zwiers and Kharin (1998) have found indications of an increase in wind speed extremes for Northern Europe. In order to simulate an increase in disturbance risk, we changed the distributions of the fire and wind regime, by deriving the

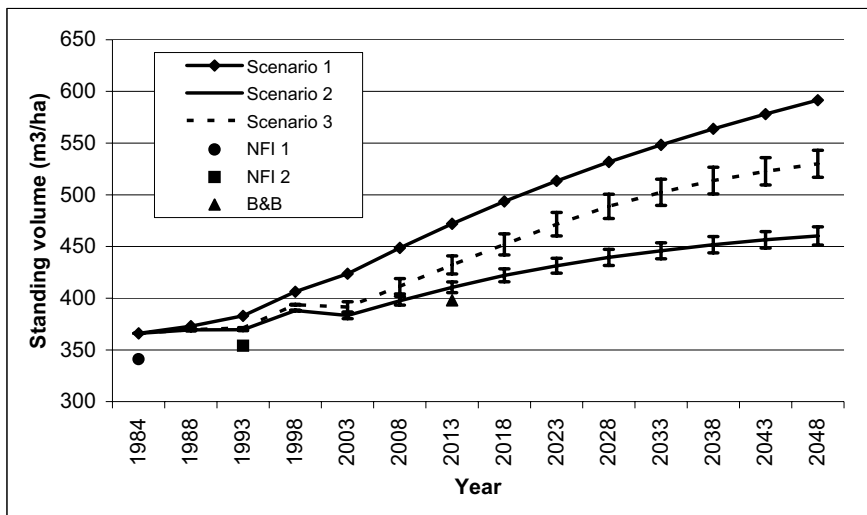


statistical distributions from periods with higher disturbance damages. The distribution for the fire regime was derived using the fire data from the period 1950-1980. Over this period, the average area destroyed by fire per annum was 65% higher than over the period 1903-1997. The wind damage distribution was derived using data from the period 1980-1999, which resulted in an increase of average annual damage by about 22%.

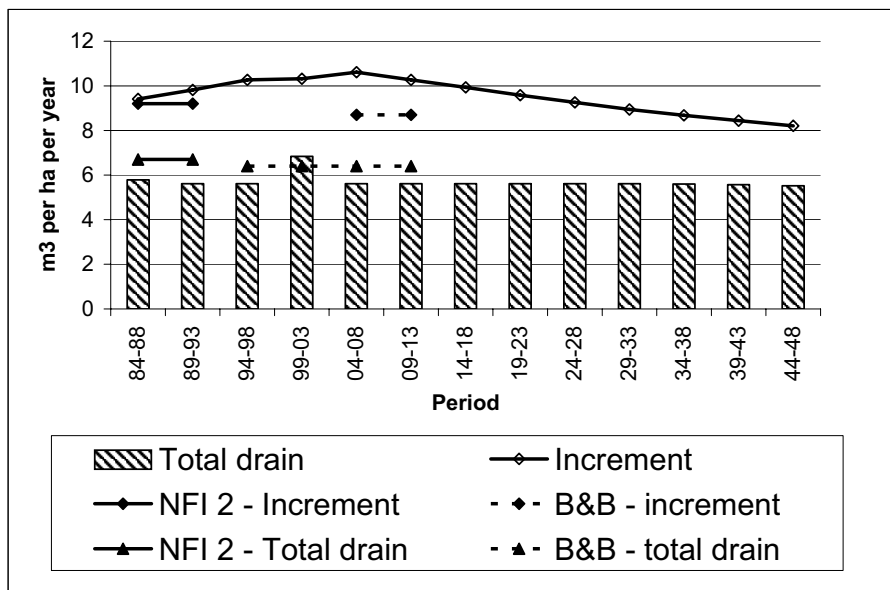
## Results

### *Scenario 1, base run*

Without the new disturbances module, the projected average standing stock in 2048 increased up to 592 m<sup>3</sup> ha<sup>-1</sup> (figure 5). Increment increased from 9.4 m<sup>3</sup> ha<sup>-1</sup> year<sup>-1</sup> in 1984 to 10.6 m<sup>3</sup> ha<sup>-1</sup> in 2004-2008, but then it declined to 8.2 m<sup>3</sup> ha<sup>-1</sup> year<sup>-1</sup> in 2048 (figure 6). Ageing of the forest and the fast increase of standing volume caused the decline in increment. The total drain, in this case equal to the asked felling level, remained at 5.6 m<sup>3</sup> ha<sup>-1</sup> year<sup>-1</sup>.



**Figure 5.** Development of standing volume under the three different scenarios, including the standard deviation for scenarios 2 and 3, standing volume as given in National Forest Inventory 1 (NFI 1) (Mahrer 1988), NFI 2 (Brassel and Brändli 1999) and according to the projection by Brassel and Brändli (1999).



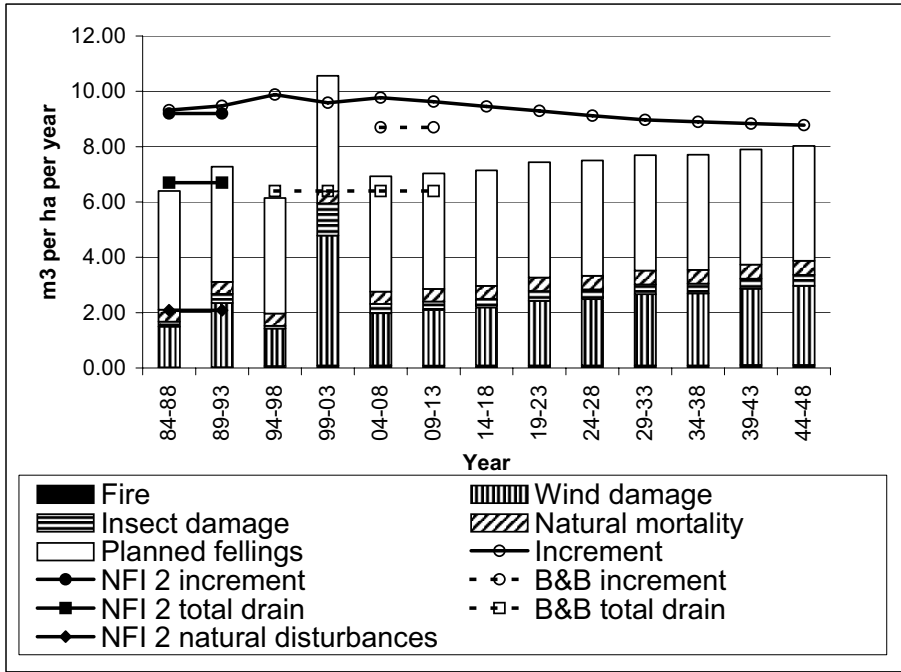
**Figure 6.** Increment and drain under scenario 1 from 1984 till 2048 (the base run), according to the Second National Forest Inventory (NFI 2) and according to the projection by Brassel and Brändli (1999).

*Scenario 2, natural disturbance run*

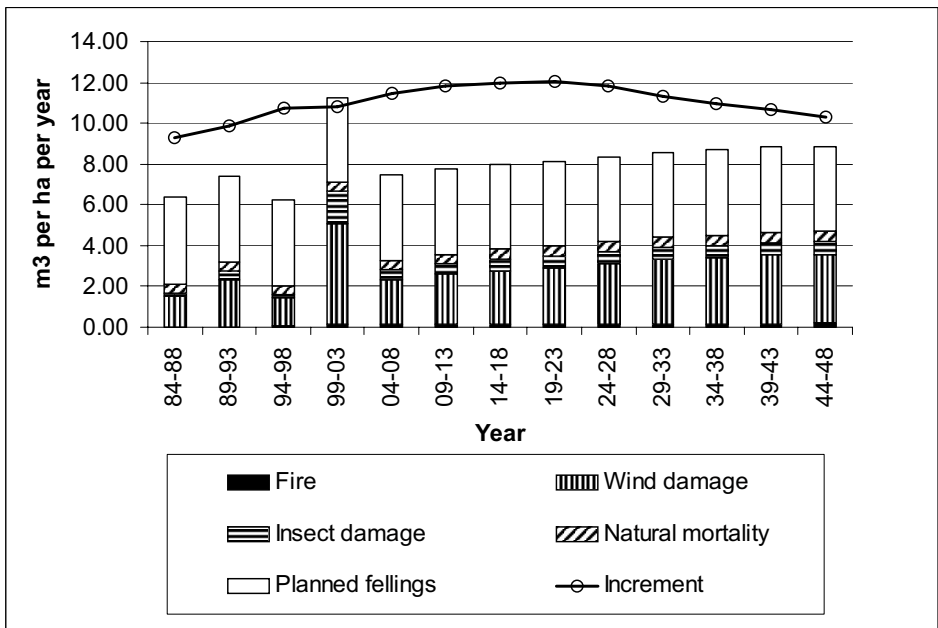
Under scenario 2, the average standing stock increased to 460 m<sup>3</sup> ha<sup>-1</sup> in 2048 (figure 5), with a standard deviation of 8.86. The increment increased to almost 10 m<sup>3</sup> ha<sup>-1</sup> year<sup>-1</sup> in 1994-2008 and then decreased to 8.8 m<sup>3</sup> ha<sup>-1</sup> year<sup>-1</sup> in 2048 (figure 7). This decrease can be attributed to a changing age class structure and an increasing standing volume. In the period 1999-2003, the total drain exceeded the increment, because of the 1999 storms and the foreseen insect damage. After that, the projected amount of damage increased gradually from 2.8 in 2003 to 3.9 m<sup>3</sup> ha<sup>-1</sup> in 2048. This can be attributed to an increased susceptibility of the forest to disturbances, due to an increase of the average standing stock and an increase of the proportion of very old stands.

*Scenario 3, climate change run*

The average standing stock under the climate change scenario increased up to 530 m<sup>3</sup> ha<sup>-1</sup> (figure 5) in 2048, with a standard deviation of 13.04. Increment increased up until 12 m<sup>3</sup> ha<sup>-1</sup> year<sup>-1</sup> in the period 2009-2023 and then declined to 10.3 m<sup>3</sup> ha<sup>-1</sup> year<sup>-1</sup> at the end of the simulation (figure 8). The total drain peaked in 1999-2003 due to the 1999 storm. Thereafter, total disturbance damage increased from 3.3 in 2003 to 4.7 m<sup>3</sup> ha<sup>-1</sup> in 2048.



**Figure 7.** Increment and drain under scenario 2 from 1984 till 2048 (natural disturbance run), according to the Second National Forest Inventory (NFI 2) and according to the projection by Brassel and Brändli (1999).



**Figure 8.** Increment and drain under scenario 3 from 1984 till 2048 (natural disturbance and climate change run).

## Discussion

### *Comparison of the scenarios*

The outcomes of scenarios 1 and 2 differ considerably, especially regarding total drain and average standing stock. The total drain was higher under scenario 2, since not only sanitation fellings were included, but also the unrecovered wood that is felled in response to disturbances. Because of the higher drain, the increase in standing stock was slower. Due to the increase in average standing volume and a greater share of old stands, the forest's susceptibility to disturbances increased, which can be seen in figure 7 by the increase in disturbance damage. The increment declined gradually for the same reasons. In the long run, drain and increment will reach equilibrium, resulting in a constant average standing volume.

Under the climate change scenario, both disturbances and increment increased compared to scenario 2. The increase in increment was greater than the increase in disturbance damage, which resulted in larger average standing stocks (figure 5).

### *Comparison with other sources*

As can be seen in figure 5, in the simulations the volume for 1984 was higher than the results of NFI 1. According to Mahrer (1988), the average volume should be  $341 \text{ m}^3 \text{ ha}^{-1}$ , but our simulations yielded  $366 \text{ m}^3 \text{ ha}^{-1}$ . The difference is caused by the fact that the temporarily unstocked area was not included in our input data.

Brassel and Brändli (1999) describe a projection of the Swiss forest resources until 2015 that is comparable to scenario 2. In their projection the average standing stock in 2015 is  $398 \text{ m}^3 \text{ ha}^{-1}$ ; we found  $411 \text{ m}^3 \text{ ha}^{-1}$  (standard deviation 5.12) under scenario 2 in 2013 (figure 5). Taking into account the initial overestimation of our projection of  $25 \text{ m}^3 \text{ ha}^{-1}$ , the results seem to be fairly consistent. The projected increment under scenario 2 in the first two simulation periods agrees very well with the observed increment in NFI 2 (figure 7). The increment in the period 2004-2013 is about  $1 \text{ m}^3 \text{ ha}^{-1}$  more under scenario 2 than in Brassel and Brändli. The large amount of damage in the period 1999-2004 has a stimulating effect on the increment, because old forests are replaced by young, well growing forests and because competition has decreased. If this damage had not occurred, the difference would probably have been smaller.

The projected total drain in the period 1984-1993 is comparable to the actual total drain in that period. Because of the huge storm damage in 1999, the drain over the period 1999-2003 is very large compared to the projected drain by Brassel and Brändli. For the periods 1994-1998 and 2004-2013, the drain of both projections is comparable.

In the present study, various assumptions had to be made, because data were lacking or insufficient. For example, assumptions were made about the nature of the

distribution of the disturbances, the dependence of disturbances on the state of the forest, the ratios of occurrence of stand-replacing and non-stand-replacing disturbances and the extrapolation of the forest fire area for the whole of Switzerland from one region. Furthermore, Switzerland has a very complicated relief, which has a great influence on the actual impact of a storm. The EFISCEN model is designed to work on a large-scale basis and should be able to work throughout whole Europe. Due to the desired general applicability of the model, these local circumstances can often not be taken into account in a very detailed way. Despite the sometimes rather coarse assumptions and the limited capability of the model to take the topography into account, the outcome of the model projections of scenario 2 correlates well with the projection made by Brassel and Brändli (1999). Nevertheless, the quantification of the dependency of disturbances on the forest characteristics and the description of the disturbance distributions deserve attention in future applications of the model.

Another important assumption was that the state of the forest had not changed in the period from which the statistical distribution had been derived. But in reality, total forest area and total growing stock increased considerably during the 20<sup>th</sup> century (Kuusela 1994). The effects of this assumption can be minimized by reducing the observation period, but this will reduce the accuracy of the statistical distribution. For disturbance types that are sensitive to the state of the forest, like storms in this study, the period should be shortened, while for types that are less sensitive, such as forest fire, the period can be lengthened. On the other hand, storm damage is a highly stochastic event, and the shorter the observation period, the less accurate the distribution will be. As is demonstrated in the difference in damage between scenarios 2 and 3, the choice of the reference period is very relevant.

In order to avoid these problems, the observed climate during the 20<sup>th</sup> century, not the observed damage, could serve as a basis for deriving the statistical distribution of the disturbances. A problem, however, is that the occurrence of storm damage is not linearly related to the occurrence of storms. If other circumstances are unfavourable (e.g. soils are waterlogged), less severe storms may have a larger impact.

Forest fire indices that describe the relation between climate parameters and the forest fire risk have been constructed in many countries. Gerstengarbe et al. (1999) used such an index, combined with a weather generator, to predict the occurrence of forest fires in Brandenburg under a climate change scenario. A comparable approach may be followed for the EFISCEN model. An advantage of this approach is that the effect of weather on insect damage can also be modelled more accurately.

In scenario 3, the effect of climate change on the volume increment is incorporated by extrapolating the simulation results from two forest stands to the whole of Switzerland. A wide variety in growing conditions occurs, which can not be modelled accurately by only two sites. Therefore, the approach chosen here must just be seen as a what-if scenario. The basis for the simulation could be made more robust by running more sites under climate change, but this would require also significantly more data.

A very important aspect of a scenario study is the assumption that all other factors and influences stay the same. The outcomes of this study must therefore be seen in the light of this assumption. Given the uncertainties caused by the assumptions made and the stochastic character of the disturbances, the results of this study must not be regarded as predictions what will happen, but as projections what can happen under the assumptions that have been done.

The results presented are averages for the whole of Switzerland. Mostly, disturbances affect only part of the country. Therefore, local damage can far exceed the values presented here. With an increase in disturbance intensity due to climate change, the effect on local forests may be even more pronounced than nowadays.

## Conclusions

The new natural disturbance module in the EFISCEN model simulated the natural disturbances in a satisfactory way. In the old situation, only the removed wood was counted as fellings, while with the new module also unrecovered wood is deducted from the standing stock volume, which seems to be more realistic. This resulted in a 30% higher total drain, which in turn influences the growing stock and increment development. When the natural disturbances module was used in the simulation, the growing stock increased from 366 m<sup>3</sup> ha<sup>-1</sup> in 1984 to 460 m<sup>3</sup> ha<sup>-1</sup> in 2048, while without disturbances it increased up to 592 m<sup>3</sup> ha<sup>-1</sup>.

Changes in the state of the forest influence the susceptibility of the forest to disturbances, which has been taken into account with the new module, resulting in more credible simulations. The simulation under current climate showed an increase in damage due to natural disturbances of 40% over the period 2004-2048, due to changes in the state of the forest. Both effects will be especially pronounced in long-term simulations.

Given the assumptions of the model, we can expect that under the current climate there will be an increase in damage by disturbance in Switzerland, because of an increase in average standing volume and a higher proportion of old stands. The simulations indicate that under a climate change scenario with increased frequency of disturbances, disturbance damage may increase by almost 25% compared to the current climate. Under climate change the increment rate may increase by about 2 m<sup>3</sup> ha<sup>-1</sup> year<sup>-1</sup>, counteracting the effect of increasing disturbances and resulting in larger average standing stocks.

## Acknowledgement

We are greatly indebted to Prof. Ola Sallnäs and Jocke Fredericksson from the Swedish University of Agricultural Sciences for providing the EFISCEN model to the European Forest Institute. The research in this study has been made possible through co-operation between Alterra and the European Forest Institute and was part of the

EU funded project Long term regional effects of climate change on European forests: impact assessment and consequences of carbon budgets (LTEEF II, ENV4-CT97-0577). It received counter-funding from the Dutch Ministry of Agriculture, Nature Management and Fisheries. Further we would like to thank Dr. Urs-Beat Brändli from the Swiss Federal Research Institute WSL for providing the detailed results of the First Swiss National Forest Inventory and to Edgar Kaufmann from the same institute for his valuable comments on the manuscript. Also we thank two anonymous reviewers for commenting the manuscript.

## References

- Becker, T., Schröter, H., 2000. Ausbreitung von rindenbrütenden Borkenkäfer nach Sturmschäden. *Allgemeine Forstzeitschrift* 55, 280-282.
- Bossel, H., 1996. TREEDYN3 forest simulation model, *Ecol. Modelling* 90, 187-227.
- Bosshard, W., 1967. Erhebungen über die Schäden der Winterstürme 1967. *Schweiz. Z. Forstw.* 118, 806-820.
- Brassel, P., Brändli, U.-B.(Eds.), 1999. Schweizerisches Landesforstinventar. Ergebnisse der Zweitaufnahme 1993-1995. Birmensdorf, Eidgenössische Forschungsanstalt für Wald, Schnee und Landschaft. Bern, Bundesamt für Umwelt, Wald und Landschaft. Bern, Stuttgart, Wien, Haupt.
- Cannon, R.J.C., 1998. The implications of predicted climate change for insect pests in the UK, with emphasis on non-indigenous species. *Global Change Biology* 4, 785-796.
- Conedera, M., Marxer, P., Hofmann, C., Tinner, W., Ammann, B., 1996. Forest fire research in Switzerland. Part 1: Fire Ecology and History Research in the Southern Part of Switzerland. *International Forest Fire News*, No. 15, 13-21.
- Flannigan, M.D., Bergeron, Y., Engelmark, O., Wotton, B.M., 1998. Future wildfire in circumboreal forests in relation to global warming. *J. Veg. Sci.* 9, p. 469-476.
- Fleming, R.A. 1996. A Mechanistic Perspective of Possible Influences of Climate Change on Defoliating Insects in North America's Boreal Forests. In: Korpilahti, E., Kellomäki, S., Karjalainen, T. (eds), 1996. *Climate Change, Biodiversity and Boreal Forest Ecosystems*. Peer-reviewed papers selected from the International Boreal Forest Research Association Conference held in Joensuu, Finland, 30 July-5 August, 1995. Reprint from *Silva Fennica* 30(2-3), 1996. The Finnish Forest research Institute, The Finnish Society of Forest Science, The Finnish Research Programme on Climate Change – SILMU. 315 p.
- Forster, B., 1993. Entwicklung der Borkenkäfersituation in den Schweizer Sturmschadengebieten. *Schweizerische Zeitschrift für Forstwesen* 144, 767-776.
- Frey, W., Forster, B., Gerber, W., Graf, F., Heiniger, U., Kuhn, N., Thee, P., 1995. Risiken und Naturgefahren auf Windwurfflächen. *Schweizerische Zeitschrift für Forstwesen* 146, 863-872.

- Gerstengarbe, F.-W., Werner, P.C., Lindner, M., Bruscek, G., 1999. Estimation of Future Forest Fire Development in the State of Brandenburg. *International Forest Fire News*, No. 21, , 91-93.
- Grayson, A.J. (ed), 1989. The 1987 Storm: Impacts and Responses. *Forestry Commission Bulletin* 87.
- Karjalainen, T., G.J. Nabuurs, A. Pussinen, J. Liski, M. Erhard, and G.M.J. Mohren In press. Impacts of forest management and climate on the forest sector carbon budget in the Nordic countries. In Apps, M. et al. (Eds.), *The role of boreal forest in the global carbon cycle. Proceedings of the meeting held in Edmonton. May 2000*
- Kramer, K., Mohren, G.M.J.(eds), 2001. Long term effects of climate change on carbon budgets of forests in Europe. *Alerra Report* 194. Alterra, Wageningen, the Netherlands. 209 p.
- Kurz, W.A., Apps, M.J., Stocks, B.J., Volney, W.J.A., 1995. Global Climate Change: Disturbance Regimes and Biospheric Feedbacks of Temperate and Boreal Forests. In: Woodwell, G.M., Mackenzie, F.T. (Eds), *Biotic Feedbacks in the Global Climatic System*, pp. 119-133.
- Kuusela, K., 1994. Forest resources in Europe. *European Forest Institute, Research Report* 1. Cambridge University Press.
- Laiho, O., 1987. Metsiköiden alttius tuulituholle Etelä-Suomessa. Susceptibility of forest stands to windthrow in southern Finland. *Folia Forestalia* 706.
- Lässig, R., Schönenberger, W., 1997. Was passiert, wenn man die Natur sich selber überlässt? Ergebnisse der Sukzessionsforschung auf Windwurfflächen. *Laufener Seminarbeiträge* 1/1997., pp. 67-74.
- Leitner, K., 1968. Rückblick auf die Sturmholzaufarbeitung bei den Österreichischen Bundesforsten. *Allgemeine Forstzeitung* 79, 142-144.
- Lunkeit F., Ponater M., Sausen R., Sogalla M., Ulbrich U., Windelband M. 1996: Cyclonic Activity in a Warmer Climate. *Contributions to Amospheric Physics/Beiträge zur Physik der Atmosphäre* 69, 393-407.
- Mahrer, F. (ed.), 1988. Schweizerische Landesforstinventar. Ergebnisse der Erstaufnahme 1982-1986. *Swiss forest land inventory. Results of the first period 1982-1986. Eidgenössische Anstalt für das Forstliche Versuchswesen, Birmensdorf, Switzerland. Berichte* no. 305.
- Mandallaz, D., Ye, R., 1997: Prediction of forest fires with Poisson models. *Can J. For. Res.* 27, p. 1685-1694.
- Meier, F., Engesser, R., Forster, B., Odermatt, O., 1999. *Forstschutz-Überblick 1998. Eidgenössische Forschungsanstalt für Wald, Schnee und Landschaft, Birmensdorf, Switzerland.*
- Miller, K.F., 1985. *Windthrow Hazard Classification. Forestry Commission Leaflet* 85. HMSO, London
- Mitchell, J.F.B., Johns, T.C., Gregory, J.M., Tett, S.F.B., 1995. Climate response to increasing levels of greenhouse gases and sulphate aerosols. *Nature* 376, 501-504.



- Nabuurs, G.J., 2001. European forests in the 21<sup>st</sup> century: impacts of nature-oriented forest management assessed with a large-scale scenario model. Alterra Scientific Contributions 2, Alterra, Wageningen.
- Nabuurs, G.J., Schelhaas, M.J., Pussinen, A., 2000. Validation of the European Forest Information Scenario Model (EFISCEN) and a projection of Finnish forests. *Silva Fennica* 34, 167-179.
- Nabuurs, G.J., Päivinen, R. 1996. Large scale forestry scenario models : a compilation and review. European Forest Institute, Working Paper 10, Joensuu, Finland.
- Netherer, S., Führer, E., 1998. Konzipierung eines Risikoschätzmodells zur Bestimmung der Prädisposition von Standorten und Beständen gegenüber Gradationen von *Pristiphora abietina* (Christ)(Hym. Tenthr.). *Allgemeine Forst- und Jagdzeitung*, 170, 53-60.
- Nilsson, S., Sallnäs, O. and Duinker, P., 1992. Future Forest Resources of Western and Eastern Europe, International Institute for Applied Systems Analysis. The Parthenon Publishing Group, UK.
- Poecke, M.Q.H. van, 1984. Een Deense bosbouwpraktijk; praktijkverslag van de periode augustus t/m november 1983. Vakgroep Bosteelt, Landbouwhogeschool Wageningen.
- Putz, W., 1968. Die Windwurfkatastrophe im Bereich der Forstverwaltung Gusswerk. *Allgemeine Forstzeitung* 79, 147-148.
- Rohmeder, E., 1933. Der Nonnenraupenfrass 1889 - 91 und der Wirbelsturm von 14. Juli 1894 im Ebersbergerforst: geschichtlicher Verlauf, Folgerungen, Wiedergutmachung und Wirkung auf Waldbau und Forsteinrichtung zweier Grossschäden. S.l.
- Sallnäs, O. 1990. A matrix growth model of the Swedish forest. *Studia Forestalia Suecica* No 183. Swedish University of Agricultural Sciences, Faculty of Forestry, Uppsala.
- Schelhaas, M.J. Nabuurs, G.J., In prep. Impact of natural disturbances on European forests in the 20<sup>th</sup> century. *Forest Ecology and Management*.
- Schlaepfer, R., Haemmerli, F., 1990. Das "Waldsterben" in der Schweiz aus heutiger Sicht. *Schweizerische Zeitschrift für Forstwesen* 141, 163-188.
- SFSO and FOEFL, 1996. Forestry and Timber Industry in Switzerland, Yearbook. Swiss Federal Statistical Office, Bern.
- Sonntag, M., 1998. Klimaveränderungen und Waldwachstum: TREEDYN3-Simulationen mit einer Analyse modellstruktureller Unsicherheiten. Verlag Mainz, Aachen. Dissertation, University of Kassel, Germany.
- UN-ECE, 1999. Timber Bulletin, Forest Products Statistics 1994-1998, ECE/TIM/BULL/52/2. UN-ECE, Geneva.
- UN-ECE/FAO, 1984. Forest Fire Statistics. Timber Bulletin for Europe, Volume XXXVI, Supplement 7. United Nations Economic Commission for Europe (Geneva), Food and Agriculture Organization of the United Nations (Roma).

- UN-ECE/FAO, 1996. Forest fire statistics 1993-1995. Timber Bulletin, Volume XLIX , No.4. United Nations Economic Commission for Europe (Geneva), Food and Agriculture Organization of the United Nations (Roma).
- UN-ECE/FAO, 1998. Forest fire statistics 1995-1997. Timber Bulletin, Volume LI, No. 4. United Nations Economic Commission for Europe (Geneva), Food and Agriculture Organization of the United Nations (Roma).
- UN-ECE/FAO, 2000 Forest Resources of Europe, CIS, North America, Australia, Japan and New Zealand. Main Report. Geneva Timber and Forest Study Papers, No. 17. United Nations Economic Commission for Europe (Geneva), Food and Agriculture Organization of the United Nations (Roma).
- Wermelinger, B., Obrist, M.K., Duelli, P., Forster, B., 1999. Development of the bark beetle (*Scolytidae*) fauna in windthrow areas in Switzerland. Mitteilungen der Schweizerischen Entomologischen Gesellschaft 72, 209-220.
- Weslien, J., Schröter, H., 1996. Natürliche Dynamik des Borkenkäferbefalls nach Windwurf. AFZ Der Wald, Allgemeine Forst Zeitschrift für Waldwirtschaft und Umweltvorsorge 51, 1052-1055.
- Winterhoff, B., Schönfelder, E., Heiligmann-Brauer, G., 1995. Sturmschäden des Frühjahrs 1990 in Hessen – Analyse nach Standorts- Bestandes- und Behandlungsmerkmalen. Forch. Ber. Hess. Landesanst. Forsteinrichtung, Waldforch. Und Waldökologie, Hannover Münden 20.
- Zwiers, F.W., Kharin, V.V. 1998. Changes in the extremes of the climate by CCC GCM2 under CO2 doubling. J. Climate 11, 2200-2222.

### III Modelling bark beetle disturbances in a large scale forest scenario model to assess climate change impacts and evaluate adaptive management strategies

Rupert Seidl<sup>a,b</sup>, Mart-Jan Schelbaas<sup>c</sup>, Marcus Lindner<sup>a</sup>, Manfred J. Lexer<sup>b</sup>

a: European Forest Institute, Torikatu 34, 80100 Joensuu, Finland

b: Institute of Silviculture, Department of Forest- and Soil Sciences, University of Natural Resources and Applied Life Sciences (BOKU) Vienna, Peter Jordan Strasse 82, 1190 Wien, Austria

c: Alterra, Centre for Ecosystem Studies, Wageningen, P.O. Box 47, 6700 AA Wageningen, The Netherlands

#### Abstract

Climate change has the potential to significantly alter natural disturbance regimes in forest ecosystems. Particularly affected could be biotic disturbances, where climate change could amplify the disturbance frequency and severity via (i) increased annual life cycles and reproduction capacity of insect herbivores; (ii) increased area where the insect may potentially occur, and (iii) decreased host resilience due to additional pressures as for instance drought.

To simultaneously study these effects a simulation approach is presented that incorporates a climate-sensitive model of damages by the European spruce bark beetle *Ips typographus* (L. Scol. Col.) within the framework of the large-scale forest scenario model EFISCEN. Within EFISCEN a two-stage multivariate statistical meta-model was used to upscale stand level damages by bark beetles as simulated in the hybrid forest patch model PICUS v1.41. Comparing EFISCEN simulations including the new bark beetle disturbance module against 15 years of observed bark beetle damage for Austria showed good agreement between observed and predicted data at province level.

Subsequently a scenario analysis focusing on climate change impacts on bark beetle induced damages for Austrias Norway spruce forests was conducted. Results showed a dramatic increase in damages from 1.33 Mm<sup>3</sup> a<sup>-1</sup> (106 m<sup>3</sup> a<sup>-1</sup> total stem volume over bark, evaluation period 1990 to 2004) to 4.46 Mm<sup>3</sup> a<sup>-1</sup> (period 2095 to 2099) as a consequence of a climate change scenario, with peak damage levels of 6.09 Mm<sup>3</sup> a<sup>-1</sup>. Average damage levels per hectare were found to be highest in pre-alpine Norway spruce stands, but simulated increases in accumulated damage under climate change were strongest in the inner Alps (+166%). Studying two adaptive management strategies (species change) revealed a considerable time-lag between the start of

adaptation measures and a decrease in simulated damages by bark beetles. The chosen simulation approach as well as implications of an altered disturbance regime on future forest management are discussed.

## Introduction

A key driver of forest ecosystem dynamics are natural disturbances, playing an important role in natural forest development. Consequently disturbances are also highly relevant factors in the sustainable management of forest ecosystems. Climate change has the potential to distinctly alter disturbance regimes and in doing so imposing adverse feedbacks on the sustainable provision of important forest services and functions (e.g. Ayres and Lombardero 2000, Dale et al. 2000). According to Schelhaas et al. (2003) windthrow was the most important abiotic natural disturbance in European forests over the last century whereas bark beetles were the most important biotic disturbance agent. Since poikilothermal organisms are directly dependent on the climate regime several authors expect increasing damages from insects as for instance bark beetles under warmer climatic conditions (e.g., Harrington et al. 2001, Bale et al. 2002). The increasing risk from biotic disturbances results from positive feedbacks of changes in climate on essential elements of the herbivore-host system.

The temperature regime directly affects life-cycles and winter survival of insects which are generally limited by the thermal environment (e.g., Wermelinger and Seifert 1999, Netherer and Pennerstorfer 2001, Wermelinger 2004). Thus, an increase in temperature is expected to increase the reproductive capacity and number of completed life cycles per year of important biotic disturbance agents such as bark beetles (Bale et al. 2002). Moreover, climate change is likely to cause shifts in outbreak ranges of insect species (e.g., Parmesan et al. 1999, Williams and Liebhold 2002). Due to the wide latitudinal and altitudinal distribution of the main European tree species the spatial distribution of important insect herbivores is in many cases limited by harsh environmental conditions rather than host availability. For instance, for the Norway spruce (*Picea abies* (L.) Karst.) bark beetle *Ips typographus* (Scol. Col.) the spatial distribution of the host species strongly exceeds the thermally feasible area of insect development. A shift in climatic conditions could trigger a dramatic expansion of herbivores to host areas currently not susceptible to these disturbance agents. Additionally, an increase in environmental stress factors for the host species may reduce host resilience to infestation by insects (e.g., Rouault et al. 2006). Drought stress, for instance, which is likely to increase under warmer climate conditions, has been identified as a major factor increasing predisposition of Norway spruce to attacks of *Ips typographus* (e.g., Wermelinger 2004, Rouault et al. 2006).

Addressing these manifold interactions, considerable efforts have been made recently to investigate prominent biotic disturbance agents in North America (e.g., Williams and Liebhold 2002, Sturtevant et al. 2004, Candau and Fleming 2005). For

the most important European bark beetle, *Ips typographus*, a recent focus on bark beetle ecology (cf. Wermelinger 2004) resulted in increased understanding of the herbivore-host complex at individual tree to stand level. Increased understanding inter alia facilitated the design of models to assess climate change impacts on *Ips typographus* disturbances on stand and forest management unit level (e.g., Seidl et al. 2007a). However, at the landscape level currently no tools are available to assess the effects of biotic disturbances as *Ips typographus* on forest development and the provision of forest functions and services. Nonetheless, analysis beyond plot to stand level are necessary in order to capture not only herbivore – host interactions with regard to climate change but to simultaneously address potential climate change induced shifts in outbreak ranges of herbivores in relation to host distribution. Moreover, in order to support policy makers with regard to decisions under changing environmental conditions analysis need to be conducted on the respective scales (province-, country-, continental level). At the latter scales the model EFISCEN was widely applied over the last years with regard to questions of forest resource development (e.g., Nabuurs et al. 2003, Schröter et al. 2004, Pussinen et al. 2005, EEA 2006, Schelhaas et al. 2006, Nabuurs et al. 2007). Although EFISCEN contains a module of natural disturbances (Schelhaas et al. 2002) the representation of biotic disturbances as bark beetles therein is currently not satisfactorily particularly with respect to studying climate change induced feedbacks on the disturbance regime.

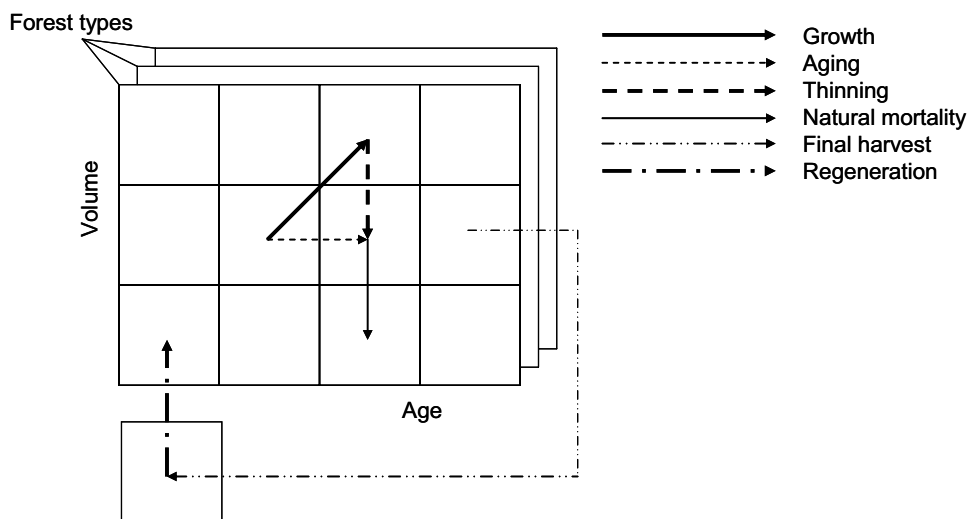
Therefore our objectives were (i) to develop and integrate a climate sensitive approach of simulating bark beetle damages in the large-scale forest model EFISCEN; (ii) to evaluate the approach against a 15 year timeseries of independent data on bark beetle damage; (iii) to study potential climate change impacts on the disturbance regime, and (iv) tentatively investigate the effects of adaptive management strategies on bark beetle damages. The development of a bark beetle disturbance module for integration into the EFISCEN framework built on the stand level hybrid model PICUS v1.41 (Lexer and Hönninger 2001, Seidl et al. 2005) which includes a detailed sub-model of bark beetle induced mortality of Norway spruce (Seidl et al. 2007c). A two-stage meta modelling approach was used to extract process behaviour from PICUS and integrate it into EFISCEN. Austrias Norway spruce forests (1.99 million hectares (Mha)) were chosen as study object since they represent a wide ecological gradient over strongly heterogeneous landscape from the colline to the subalpine vegetation belt. The extended EFISCEN model was evaluated by comparing independent province level bark beetle damage data from Austria with EFISCEN simulation results. Finally, the extended EFISCEN model was applied to estimate the impacts of climate change on future damages from bark beetle disturbances, and to tentatively study the mitigating effect of alternative forest management strategies.

## Methods and Material

### Basic model concepts

#### *EFISCEN v3.2*

EFISCEN is an area-based matrix model that is especially suitable for scenario analysis at the regional or country level. The core of the EFISCEN model is developed in the late 1980s at the Swedish Agricultural University (Sallnäs 1990). National forest inventory data are used as input for the EFISCEN model, including forest area, volume and net annual increment by age classes. Per country, forest types can be distinguished by region, owner class, site class and tree species, depending on the available input data. The state of the forest is depicted as an area distribution over age- and volume classes in a matrix. For each forest type a separate matrix is set up. Forest development is simulated by moving forest area between cells in the matrix (Figure 1). Aging is simulated by moving area to a higher age class, while volume growth is simulated as moving area to a higher volume class. Thinning and natural mortality is simulated by moving area one volume class down. In case of a final felling, area is moved to a bare forest land class outside the matrix. Regeneration is simulated by moving area into the first age-volume class of the matrix again.



**Figure 1.** Structure and simplified processes of the matrix model EFISCEN v3.2.

Management is controlled at two levels in the model. First a basic thinning and final felling regime is incorporated. This regime defines per forest type the probability that a thinning or final felling is carried out dependent on stand age. Second, total required

harvest volume from thinnings and final felling is specified for the country, region and/or species per time step. Taking into account the actual state of the forest and the constraints set by the basic management regime, the model then tries to fulfil the specified felling levels. Other variables that can be included in scenario analysis are changes in forest area, changes in species composition by regenerating with a different species, and growth changes due to for instance changed environmental conditions. The default timestep of the simulations is five years. EFISCEN has been applied in a number of continental scale studies recently, assessing the future development of European forest resources (e.g., Nabuurs et al. 2003, Schelhaas et al. 2006, Nabuurs et al. 2007). Moreover, the model has been adapted for the application under climate change by means of inter alia accounting for changed growing conditions in implementing growth responses from process models (e.g., Schröter et al. 2004, Pussinen et al. 2005). I.e. climate drivers are not explicitly considered as input in EFISCEN but model parameters are updated according to meta information on climate change effects. A more detailed model description can be found in Pussinen et al. (2001) and Schelhaas et al., 2007.

Schelhaas et al. (2002) extended EFSICEN with a module to simulate the impact of natural disturbances on forest resources. Three types of disturbances are considered: storm, fire and insect damage. Disturbance impacts in the model framework can either affect a part of the stand (non-stand replacing disturbance) or the total stand (stand-replacing disturbance). Non-stand replacing disturbances are simulated by moving area one volume class down, while stand-replacing disturbances are simulated by moving area to the bare forest land class. Each cell in all matrices is assigned a relative susceptibility, according to variables such as species, age, and thinning status. For storm and fire, each year a severity index is drawn randomly from observed severity distributions. The combination of severity, susceptibility and the actual distribution of area over the matrix then yields the total damage level. Insect damage is considered to stay at a constant “background” level, but can be increased by large storm events and high fire severity mediating warm and dry years. In an indicative study for Switzerland the EFISCEN disturbance module was parameterised with statistical disturbance data on the study region to investigate the influence of natural disturbances on forest resource development (see Schelhaas et al. 2002). Whereas the integration of the disturbance module was found to improve EFISCEN predictions of forest development (e.g. standing stock) compared to an “undisturbed” model variant (cf. Schelhaas et al. 2002) the investigative power of the approach with regard to climate change was strongly limited by the required empirical parameterisation (distribution of annually damaged area) and the lack of direct climate dependency of for instance insect damages.

### ***PICUS***

PICUS v1.41 is a modular modeling framework centred around a hybrid forest patch model and incorporates a number of flexible sub-models for scenario analysis (e.g.,

forest management, bark beetle damage, rockfall protection; Seidl et al. 2005, Seidl et al. 2007a, Woltjer et al. 2007). Here, besides a brief overview of the general model logic the bark beetle disturbance sub-model will be described in more detail.

The hybrid model approach adopted in PICUS aims at combining the strengths of patch models and process based production models while circumventing the limitations of the individual approaches (see Mäkelä et al. 2000). Spatial core structure of PICUS is an array of 10 m × 10 m patches with crown cells of 5 m in depth. In extension to classical patch models (compare Shugart 1984, Botkin 1993) spatial interactions between the patches are taken into account with regard to inter alia a detailed three-dimensional light regime and spatially explicit seed dispersal. Inter- and intra-species competition, seed dispersal and mortality are modelled based on the approach presented by Lexer and Hönninger (2001) whereas stand level net primary production is modelled according to the simplified physiological principles of radiation use efficiency of the 3-PG model (Landsberg and Waring 1997). The hybridisation of both concepts is described in Seidl et al. (2005). The model has been successfully evaluated with regard to the simulation of equilibrium species composition over broad environmental gradients in the Eastern Alps as well as against long-term growth and yield data of uneven-aged, multi-species stands (Seidl et al. 2005). The latest version PICUS v1.41 includes a process-based soil model of dynamic C and N cycling (Currie et al. 1999) addressing interactions between aboveground production processes and belowground C and N dynamics (cf. Seidl et al. 2007b).

PICUS v1.41 contains a bark beetle disturbance sub-model which includes (i) the stochastic computation of the infestation risk for a simulated forest stand; (ii) the estimation of damage intensity if an infestation occurs; and (iii) the spatial distribution of bark beetle induced Norway spruce mortality within the stand (Seidl et al. 2007c). Probability of infestation (pBB) is calculated combining an estimate of potential annual bark beetle generations with a predisposition assessment index including site and stand related parameters. Annual potential beetle generations are calculated according to thermal requirements for beetle development represented by a sum of degree-days above a threshold temperature for beetle development (8.3 °C). Bark temperatures above the development optimum (30.4°C) lead to a slowing and finally complete halt (>38.9°C) of beetle development in the model. Swarming requirements in terms of a combined day length and air temperature threshold are considered for the start of a new insect generation. The approach of calculating potential annual bark beetle generations adopted within the PICUS bark beetle disturbance sub-model is described in detail in Baier et al. (2007). In the current version, the simulation of potential generations does not account for a perennial bark beetle gradation – i.e. beetle development starts from zero every year. Annual potential generations are taken as a proxy of thermal environmental conditions for beetle development and are transformed into a stand level hazard score (Netherer and Nopp-Mayr 2005). A stand predisposition index is adopted from Lexer (1995) and Netherer and Nopp-Mayr (2005), including four stand level predisposition indicators (share of host trees in a stand, stand density, stand age, and Norway spruce drought index, SMI). Stand



predisposition and the potential insect generation score are combined to derive an annual probability of bark beetle damage at stand level using the bark beetle infestation data set of Lexer (1995) for calibration (cf. Seidl et al. 2007c). Damage intensity (*intBB*), i.e. the amount of damaged Norway spruce trees per infestation, is calculated according to the empirical findings of Lexer (1995) using a stand hazard index combining south- and east-exposed stand edges, drought stress, and the proportion of Norway spruce host trees in the stand as predictors (see Lexer 1995) (Eq. 1-3).

$$\text{int } BB = \frac{1}{1 + e^{3.9725 - 2.9673 \cdot SHI}} \quad (1)$$

$$SHI = SEE \cdot SMI \cdot HTS \quad (2)$$

$$SMI = 1 - \frac{\sum_{bgs}^{egs} AET}{\sum_{bgs}^{egs} PET} \quad (3)$$

*intBB* = annually damaged relative stem number [0-1]

*SHI* = stand hazard index

*SEE* = stand edge index (cf. Lexer 1995) [dimensionless]; currently set to 1

*SMI* = soil moisture index over the growing season [0-1]

*HTS* = host tree share [0-1]

*SMI* = soil moisture index [0-1]

*AET* = actual evapotranspiration [mm]

*PET* = potential evapotranspiration [mm]

*bgs* = begin of growing season

*egs* = end of growing season

The spatial allocation of damages to patches and finally the selection of infested trees within a patch is based on a predisposition ranking of all patches within the simulated stand. A more detailed description of the bark beetle disturbance sub-model alongside a thorough sensitivity analysis can be found in Seidl et al. (2007c). Recently, the sub-model was successfully employed to assess the impacts of bark beetle disturbances on timber production and carbon sequestration under climate change at the forest management unit level (Seidl et al. 2007a).

## Model development

### *A statistical meta-model of bark beetle damage*

In generalizing the PICUS stand level bark beetle disturbance model logic for the application in EFISCEN the two-stage modelling approach (i.e., separate estimation of pBB and intBB) was retained. In order to derive multivariate regression models for pBB and intBB PICUS simulations were conducted over an array of environmental and stand conditions. Simulations covered all combinations of a mean annual temperature gradient from 2°C to 15°C (1°C interval) and a mean annual precipitation gradient spanning from 500 mm to 2000 mm (100 mm intervals). All other site parameters (e.g., soil conditions) were kept constant. For every combination of temperature and precipitation a variety of stand conditions was assessed within PICUS. Stand characteristics were generically initialised and held constant in order to control potential dynamic feedbacks between stand development and bark beetle damages (i.e., independent estimates). Stand structure (i.e., stem number, diameter, height of trees) was taken from yield tables (Marschall 1975) for an average site index for Austria (cf. Schadauer 1999). The studied array of stand ages ranged from 40 years (assumed minimum age for *Ips typographus* damage) to 120 years (10 year interval), stand density was studied from 60% to 100% of yield table stocking density (20% intervals). Moreover, the influence of mixed species on simulated bark beetle damage in PICUS was accounted for by simulating three levels of non-host species share in a stand (corresponding to 100%, 50% and 10% Norway spruce share respectively). Target variables of the PICUS simulations were the annual probability of bark beetle damage (pBB) and the soil moisture index (SMI), the latter a main predictor of annual damage intensity (intBB, see Eq. 1). In total, the combination of climate and stand conditions resulted in 18,144 PICUS estimates of pBB and SMI as basis for the development of the statistical meta-models.

The simulated probability of bark beetle damage in the PICUS experiments ranged from 0.0 to 0.719 with a mean value over all combinations of climate and stand conditions of 0.185. A logit model (function call `glm()`, error distribution: binomial, link: logit, R Development Core Team 2006) containing mean annual temperature and precipitation, stand age, relative stocking density and host tree share as explanatory variables was fitted to the PICUS results (Eq. 4). Host tree share was included as categorical predictor with three levels using dummy coding. Only model parameters which had been found influential on pBB in a multiple sensitivity analysis of the PICUS model (Seidl et al. 2007c) were included in the meta model. First order interactions between climatic predictors and stand structure predictors were allowed in line with the PICUS model logic on combining site- and stand-level predisposition factors in the computation of bark beetle damage. Logarithmic transformation of both climate variables was necessary in order to achieve a satisfactory residual distribution. Model parameters and graphical residual analysis can be found in the Appendix (Table A1, Figure A1).

$$pBB = \frac{e^{z_{ijklm}}}{1 + e^{z_{ijklm}}} \quad (4)$$

$$z_{ijklm} = \mu + a_i + b_j + c_k + d_l + e_m + (a \cdot b)_{ij} + (a \cdot c)_{ik} + (a \cdot d)_{il} + (a \cdot e)_{im} + (b \cdot c)_{jk} + (b \cdot d)_{jl} + (b \cdot e)_{jm} + \varepsilon_{ijklm}$$

- pBB = probability of bark beetle damage  
 z<sub>ijklm</sub> = linear combination of predictor variables  
 μ = intercept  
 a<sub>i</sub> = logarithmic mean annual temperature (i=2°C-15°C)  
 b<sub>j</sub> = logarithmic mean annual precipitation (j=500mm-2000mm)  
 c<sub>k</sub> = stand age (k=40yrs-120yrs)  
 d<sub>l</sub> = stocking density relative to fully stocked yield table stands (l=0.6-1.0)  
 e<sub>m</sub> = host tree share (categorical, m=10%; 50%; 100%)  
 ε<sub>ijklm</sub> = error term

The statistical pBB-model (Eq. 4) showed sensible model behaviour in line with expectations and PICUS model logic. Simulated pBB increased with increasing temperature and decreasing precipitation. Furthermore, older stands showed higher probability of bark beetle damage than young stands and the susceptibility to bark beetle increased with increasing host tree share. Decreasing stand densities resulted in slightly increasing probabilities for bark beetle damage in the model, related to increasing light levels in the forest and subsequently increasing bark temperatures and more favourable conditions for beetle development in the PICUS environment. The R<sup>2</sup> for the logit-model was calculated as

$$R^2 = \frac{1 - (\hat{L}_0 / L)^{2/n}}{1 - \hat{L}_0^{2/n}} \quad (5)$$

where n is the number of binary observations and  $\hat{L}_0$  is the maximized likelihood under the null (see Faraway 2006). The resulting value of 0.923 showed a good correspondence of the statistical meta-model to the PICUS data. Average model bias (E) was calculated as an average of errors for all predictions by

$$E = \frac{1}{n} \sum (y_i - \hat{y}_i) \quad (6)$$

where y<sub>i</sub> is the observed and  $\hat{y}_i$  is the predicted value. The resulting average model bias for the pBB-model was very small (E=-1.792•10<sup>-15</sup>) and not significantly different from zero (α=0.05). Also the mean absolute error of pBB, |E|,

$$|E| = \frac{1}{n} \sum |y_i - \hat{y}_i| \tag{7}$$

was found to be satisfactorily low with 0.0389. Analysing model residuals over the range of estimation showed no evidence for non-constant residual variation (Figure A1, Appendix).

Damage intensity is simulated in PICUS applying a logistic regression model using host tree share and soil moisture index (SMI) as main predictors (Eq. 3). The same calculation was applied within the EFISCEN environment, approximating SMI by means of a generalized linear model (GLM) of mean annual temperature and precipitation (Eq. 8, see Table A2 and Figure A1, Appendix, for coefficients and graphical residual analysis).

$$SMI_{ij} = \mu + a_i + b_j + (a \cdot b)_{ij} + \varepsilon_{ij} \tag{8}$$

- $\mu$  = intercept
- $a_i$  = mean annual temperature ( $i=2^{\circ}\text{C}-15^{\circ}\text{C}$ )
- $b_j$  = (precipitation)-2; ( $j=500\text{mm}-2000\text{mm}$ )
- $\varepsilon_{ij}$  = error term

Besides temperature and precipitation also soil water holding properties play a role in determining soil moisture conditions (i.e., SMI) in PICUS. However, since soil conditions within the simulation units in EFISCEN are highly variable and detailed information on forest soils at large scales are limited (cf. Lexer and Hönninger 1998) values for SMI were derived utilizing an average soil water holding capacity (156 mm). Over the simulated temperature and precipitation gradients the SMI range predicted with PICUS extended from 0.0018 to 0.528 with an average SMI of 0.0936. GLM behaviour was generally in line with the definition of SMI (see Eq. 3) showing a directly proportional relationship between SMI and temperature and an indirectly proportional relationship between SMI and precipitation. However, SMI response to precipitation was found to be widely insensitive to precipitation levels of more than 1100mm per year in the PICUS simulations with a strong increase in SMI at lower precipitation levels, which made a transformation of the corresponding predictor variable necessary (Eq. 8). Nevertheless, the residual distribution showed a slight trend towards an underestimation of low and an overestimation of high SMI values (Figure A1, Appendix). However, those trends did not result in a significantly different overall residual distribution compared to the normal distribution with only 4.74% exceeding the 95% bounds of the normal distribution. The GLM explained a high proportion of variance in SMI of the PICUS simulations ( $R^2=0.945$ ) and average model bias E (Eq. 6) was as low as  $2.885 \cdot 10^{-19}$  (not significantly different from zero at  $\alpha=0.05$ ). Also the mean absolute error of the GLM was small ( $|E|=0.0234$ ). Eq. 8 was subsequently

applied to derive the SMI required in Eq. 1 to estimate the share of damaged Norway spruce volume in the EFISCEN framework.

### ***Integration into EFISCEN v3.2***

The statistical meta-model of bark beetle damages was integrated into EFISCEN v3.2, henceforward in short referred to as EFISCEN. The integration in general followed Schelhaas et al. (2002), treating stand-replacing disturbances as area transitions to bare forest land and non-stand replacing disturbances as area transitions to a lower volume class (cf. thinnings). However, climatic drivers were explicitly accounted for in the simulation of biotic disturbances in the present approach. In order to preserve the interannual climate variability in the simulation the two-stage regression model of bark beetle damage was applied on annual timestep with a subsequent aggregation of the simulated damages to match the five-year simulation periods in EFISCEN. This aggregation had to inter alia account for the differentiation between non-stand-replacing and stand-replacing disturbances for a five year timestep. As threshold for stand-replacing disturbances we defined either (i) bark beetle damage occurring every year of the five year simulation period or (ii) the accumulated damaged volume exceeding 50% of standing volume within the five year simulation period. To that end the share of every EFISCEN matrix cell being subject to one to five damages per five year period was calculated applying basic probability theory for stochastically independent events (Eq. 9), using pBB as analogue to share of area per matrix cell damaged.

if  $n < t$

$$\begin{aligned}
 pBB_n &= pBB_{1 \cap 2 \cap \dots \cap n} - pBB_{t|n} \\
 pBB_{1 \cap 2 \cap \dots \cap n} &= pBB_1 \cdot pBB_2 \cdot \dots \cdot pBB_n \\
 pBB_{t|n} &= \sum_{i=n+1}^t pBB_{1 \cap 2 \cap \dots \cap i}
 \end{aligned} \tag{9}$$

if  $n = t$

$$pBB_n = pBB_1 \cdot pBB_2 \cdot \dots \cdot pBB_n$$

where n is the number of years damaged out of a t year period (t=5).

This concept accounted for the chance of more than one damage per period affecting a certain forest area per matrix cell. Treating the annual pBB as independent events, however, does not include a potentially increased risk of damage following a prior infestation. Within this framework the total amount of damaged volume per cell was calculated by multiplying the affected area with the respective aggregated estimate of annual damage intensity (Eq. 10). Volume damage percentage was converted to area transitions in the matrix (see Figure 1) accounting for the respective class-width of the volume classes.

$$dV_n = pBB_n \cdot \sum_{i=1}^n \text{int} BB_i \quad (10)$$

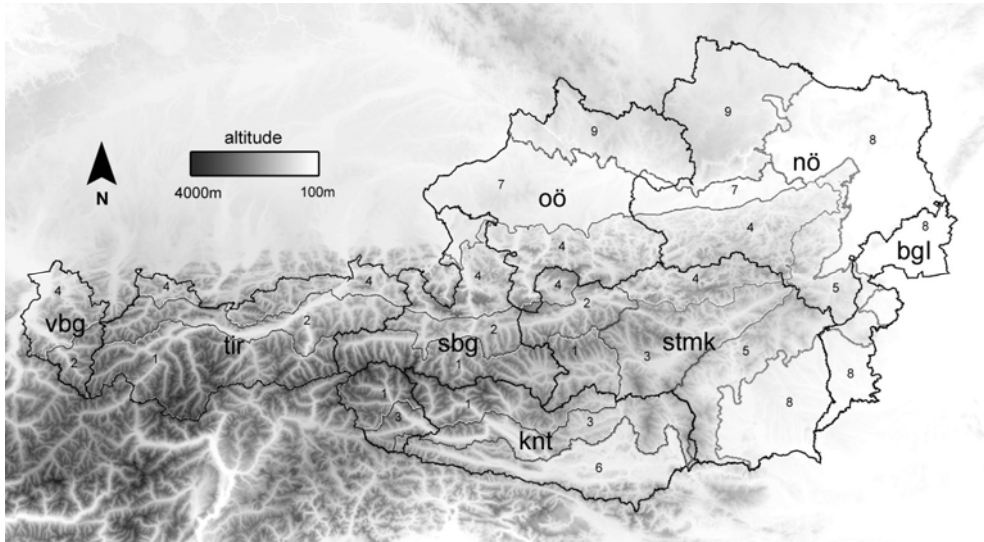
$dV$  = share of volume damaged per EFISCEN matrix cell

### EFISCEN initialisation

The EFISCEN database as used in several European studies (Schröter et al. 2004, Pussinen et al. 2005, EEA 2006, Schelhaas et al. 2006) consists of highly aggregated data from national forest inventories throughout Europe. Initialisation data and model parameters were used from this EFISCEN standard database except where stated otherwise. The spatial scope of the study covered Austrias Norway spruce forests (1.99•Mha), i.e. the full area of potential host trees for *Ips typographus* in the country. Data from the fourth Austrian Forest Inventory period (AFI4, 1986-1990, Anonymous 1997) was utilized for model initialisation, amended with additional information to allow simulations at finer regional scale.

Whereas the standard EFISCEN dataset for Austria distinguishes eight provinces this spatial resolution was found to be too coarse and arbitrary with regard to ecological conditions to reflect the countries heterogeneous environmental conditions. This is particularly the case since several provinces cover a variety of ecological conditions from pre-alpine areas to the high Alps. Thus, for the current study the provinces were subdivided into ecoregions following Kilian et al. (1994). The intersection of the eight provinces with the nine main ecoregions, totalling to 31 simulation entities (see Figure 2), aimed at balancing the need of a more detailed, ecologically meaningful resolution for the simulations and concerns of increasing uncertainties in applying a large scale scenario model to small entities (see Thürig and Schelhaas 2006).

Extending the default EFISCEN province level database on Austrias forests the forest area distribution over age-classes was compiled for the ecoregions in the provinces utilizing Austrian district level information (83 districts, Anonymous 1997). Moreover, since species composition is a highly influential factor with regard to bark beetle susceptibility three mixture classes were distinguished in Norway spruce forests, each being initialised as a separate matrix in EFISCEN. However, the effect of species mixture was confined to the simulation of bark beetle damage in the model. The three classes chosen correspond to the categories reported in Anonymous (1997), representing pure Norway spruce stands (share >80%, Pa.p), mixed Norway spruce stands (share >30% and <80%, Pa.m) and stands with only minor shares of Norway spruce (<30%, Pa.s). District level data (Anonymous 1997) were utilized to derive aggregated values for the three mixture classes on ecoregion level.



**Figure 2.** The spatial resolution of the study consisted of eight federal provinces (bgl: Burgenland, knt: Carinthia, nö: Lower Austria, oö: Upper Austria, sbg: Salzburg, stmk: Styria, tir: Tyrol, vbg: Vorarlberg) subdivided into nine ecoregions (1: inner Alps, 2: northern intermediate Alps, 3: eastern intermediate Alps, 4: northern alpine rim, 5: eastern alpine rim, 6: southern alpine rim, 7: northern pre-alpine area, 8: panonic area, 9: Bohemian massif; cf. Kilian et al. 1994), resulting in 31 simulation entities for Austrias Norway spruce forests.

Furthermore, since the ecoregions represent contrasting growing conditions the province-level Norway spruce increment functions in EFISCEN were substituted. To that end we used a data set of categorized plot level information on Norway spruce site indices (Schadauer 1999) to derive an average site index per ecoregion in the provinces. Subsequently, increment functions were fitted to yield table information (Marschall 1975) and assigned to the simulation entities according to the respective site indices.

## Model evaluation

The amended EFISCEN initialisation procedure as well as the new bark beetle disturbance component were evaluated against independent, observed data. First, province level increment, growing stock and age-class distribution as recorded in the fifth and sixth Austrian Forest Inventory periods (AFI5: 1992-1996; AFI6: 2000-2002) were compared against simulated data. In the simulations forest management in the evaluation period was prescribed at province level applying the harvest data of the Austrian Forest Inventory. Both absolute harvest level and its distribution over thinnings and final fellings were based on inventory information. The observed amount of bark beetle damage (see below) was deducted from the overall inventory harvest level since it was dynamically simulated within EFISCEN in this study.

Simulated bark beetle damages were evaluated against a 15 year time-series of bark beetle damages at province level (Krehan and Steyrer 2005). The main disturbance events in this evaluation period were a major windthrow in 1990 followed by a vast bark beetle outbreak fanned by the warm and dry conditions of the early nineties (Krehan 1993). A second peak in salvage timber from bark beetles was reached towards the end of the evaluation period related to a local foehn-storm and the extremely warm and dry summer of 2003 (Tomiczek et al. 2003, Krehan and Steyrer 2006). Climate data used for the simulation of bark beetle damages in the evaluation period was aggregated for the simulation entities (i.e., ecoregions within provinces) from climate data available for all AFI plots (e.g., Lexer et al. 2002). In the calculation of mean annual temperature and precipitation for the ecoregions within a province the Norway spruce distribution over three elevation zones (Anonymous 1997) was used as weighing factor. This approach was found to give a better climatic representation of the simulated area (i.e., Norway spruce forest area) than an arithmetic mean over all AFI plots. Average climate conditions for the period 1990 to 2004 are given in Table 1 for all combinations of provinces and ecoregions.

**Table 1.** Average temperature and precipitation for the ecoregions in the eight federal provinces for the evaluation period (1990 to 2004) and the last decade of the scenario analysis under the climate change scenario (2090 to 2099). Provinces: bgl: Burgenland, knt: Carinthia, nö: Lower Austria, oö: Upper Austria, sbg: Salzburg, stmk: Styria, tir: Tyrol, vbg: Vorarlberg. Ecoregions (cf. Kilian et al. 1994): 1: inner Alps, 2: northern intermediate Alps, 3: eastern intermediate Alps, 4: northern alpine rim, 5: eastern alpine rim, 6: southern alpine rim, 7: northern pre-alpine area, 8: panonic area, 9: Bohemian massif.

province	ecoregion	temperature [°C]		precipitation [mm]	
		1990-2004	2090-2099	1990-2004	2090-2099
bgl	5	9.3	11.4	684	748
	8	9.5	11.7	683	746
knt	1	4.6	7.0	1099	1135
	3	5.8	8.2	1104	1161
	5	4.5	6.9	1134	1205
	6	6.7	9.1	1169	1216
nö	4	8.4	10.4	1072	1270
	5	8.3	10.4	868	1011
	7	8.5	10.6	753	846
	8	9.7	11.8	624	729
	9	7.9	10.0	765	870
oö	4	8.2	10.0	1310	1453
	7	8.2	10.1	1095	1185
	9	7.0	8.9	1011	1104
sbg	1	5.0	7.1	1201	1171
	2	6.0	8.1	1299	1270



province	ecoregion	temperature [°C]		precipitation [mm]	
		1990-2004	2090-2099	1990-2004	2090-2099
	4	7.5	9.5	1331	1286
	7	8.6	10.8	1257	1207
stmk	1	4.8	6.9	1028	1161
	2	6.3	8.3	1280	1371
	3	5.9	8.0	1149	1264
	4	7.4	9.3	1416	1551
	5	6.9	9.1	1044	1142
	8	9.2	11.3	836	912
tir	1	4.7	7.0	998	911
	2	6.0	8.3	1213	1103
	3	4.4	6.8	1331	1199
	4	6.2	8.5	1285	1170
	6	3.8	6.2	1189	1077
vbg	2	5.6	7.5	1455	1308
	4	6.3	8.1	1795	1645

## Scenario analysis

### *Climate scenarios*

For the period from 2005 to 2099 a scenario analysis was conducted, studying the impacts of a climate change scenario on bark beetle disturbances as well as the effects of adaptive management strategies. To evaluate the impact of climate change two climate datasets were used: First, a baseline climate scenario was constructed by randomly drawing years from the evaluation period climate series (1990 to 2004) for the 95 year scenario period. Second, a transient climate change scenario was simulated, relating to the B2 emission scenario of the IPCC (2000) as predicted with the climate model HadCM3 and downscaled to 10' resolution (see Mitchell et al. 2004). The average temperature change for the last decade of the 21st century relative to the period 1990-2004 at country level was +2.4°C, precipitation showed only a slight increase at country level (+20mm) with limited decreases and increases in the individual provinces (see Table 1). In order to grant consistency with the evaluation period the climate signal of the climate change scenario was imposed on the respective average baseline climate of the ecoregion. Whereas climate drivers were explicitly considered in the approach of simulating bark beetle disturbances the indirect implementation via external process model results was retained for growth response in EFISCEN. Thus in the selection of the climate change scenario we drew on a previous application of EFISCEN, for which the growth response to the selected climate change scenario had been estimated with the process model LPJ (see Schröter et al. 2004). Consequently, in the scenario analysis the climate change effects on forest

growth as well as on bark beetle disturbances (and potential interactions) were considered in this study.

### ***Management strategies***

Three alternative management strategies were studied : Strategy BAU represented business as usual management, implying no species change. The second main strategy (PNV) can be seen as an adaptive management strategy representing a conversion to a species composition close to potential natural vegetation composition (static assessment for current climatic conditions, Kilian et al. 1994). Such strategies are frequently recommended to practitioners particularly for secondary coniferous stands to decrease risks in forest management inter alia with regard to potential changing environmental conditions (cf. Müller 1994, Leitgeb and Englisch 2006). Within strategy PNV, two sub-variants were analysed, one starting adaptation from day one in the simulation (PNV-early), the other studying the effect of a delayed reaction to climate change by starting the conversion efforts not until the year 2020 (PNV-late). In both strategies conversion took place only after final felling or stand-replacing disturbance, i.e. young stands were not subject to conversion treatments. For both PNV-strategies the potential natural species composition for every ecoregion was defined in three elevation belts according to Kilian et al. (1994). Additionally assumptions on the average species composition of the respective forest types were made (see Table A3, Appendix). Aggregated with the respective share of the elevation belts the desired species composition was derived for every ecoregion and implemented in the conversion rules (see Table A4, Appendix). In both PNV strategies Norway spruce forests with a share of less than 30% (category Pa.s) were only subject to conversion in areas where Norway spruce is not a dominant species of the potential natural tree species composition (Kilian et al. 1994). If two main forest types had been classified for one elevation belt and ecozone in Kilian et al. (1994) we assumed an equal share of both forest types. Overall the Norway spruce area was reduced in the PNV strategies to the benefit of beech (*Fagus sylvatica* L.), oak (*Quercus* sp.) and silver fir (*Abies alba* L.), with a focus of conversion activities in Norway spruce stands in pre-alpine regions. Pure Norway spruce forests were only supported in the Inner Alps (ecoregions 1-3), while Norway spruce was fully replaced in ecoregion 8.

Harvest levels built on earlier studies (cf. Pussinen et al. 2005) and were identical in all simulated management strategies, implying a rise in harvest level until the middle of the century (+23.8% from baseline level) and staying constant throughout the second half of the century. Due to the shifts in age-class distribution over the scenario period the share of final fellings was increased relative to the share of thinnings in the second half of the 21st century.

## Study design and analysis

We ran EFISCEN over 110 years or 22 five-year simulation periods where the first 15 years served as evaluation period and the subsequent 95 years were utilized to study the impact of climate change and the effect of adaptive management strategies. In the evaluation period, first the increased spatial resolution in the simulations was evaluated by comparing simulation results with inventory data from AFI5 and AFI6 (Anonymous 1997, 2002). Due to the irregular inventory intervals simulation results for the comparison to AFI6 were linearly interpolated to the year 2001 from the model outputs for 1999 and 2004.

In order to compare simulated tree damage by *Ips typographus* to the data of Krehan and Steyrer (2005) the following assumptions were made. Since Krehan and Steyrer (2005) report total bark beetle damages (i.e. including all bark beetle species) the share of *Ips typographus* on total bark beetle damages had to be derived. An estimate of *Ips typographus* share was available for four recent years (Krehan and Steyrer 2004, 2005, 2006) which was averaged and applied for the evaluation period (average *Ips typographus* share on total observed bark beetle damages: 81.7%). Furthermore, data in Krehan and Steyrer (2005) are related to damage amounts reported by local forest authorities, i.e. the data correspond to salvaged wood from bark beetle damage. Since the salvage of trees damaged by bark beetle is legally binding in Austria we assumed a salvage level of 95% in the comparison of simulations to observed data.

The scenario analysis conducted from 2005 to 2099 focused mainly on two aspects: The impact of climate change with regard to bark beetle disturbances was quantified by relating the simulated damages in the baseline climate scenario to damages under the climate change scenario. For this comparison both scenarios were simulated under business as usual management. Temporal development and cumulative damage were reviewed at country level as well as at the scales of individual ecoregions. The second main objective in the scenario analysis was to assess the effects of adaptive management strategies implying species change. This analysis applied a comparison of the different management strategies under the climate change scenario, focusing on the potential to confine bark beetle damages by species change and assessing the effect of an early or late starting point for adaptation (comparison of the strategies PNV-early and PNV-late). Reported timber volumes are given in total stem volume over bark except where indicated (e.g., u.b.m.= merchantable timber volume under bark).

## Results

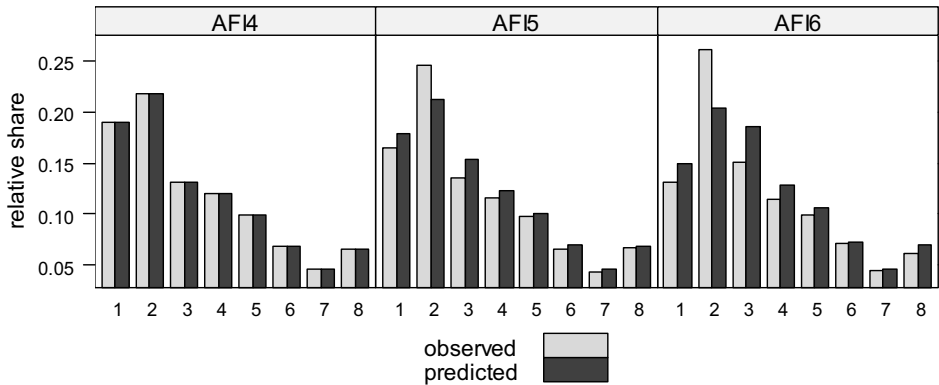
### Evaluation

#### *Simulated forest development*

Between AFI4 (the initial condition for this study) and AFI6 the observed standing stock in Austrias Norway spruce forests increased by 16.6% from 289.1 m<sup>3</sup> ha<sup>-1</sup> to 337.0 m<sup>3</sup> ha<sup>-1</sup> (Anonymous 2002). The corresponding EFISCEN simulations captured this trend, simulating a slightly lower increase in standing stock of 12.4% from 1990 to 2001 (Table 2). Largest deviations between observed and predicted values were found for the province Vorarlberg, which can be related to a considerably different management regime in this province, consisting mainly of continuous cover systems rather than age-class forestry.

The age-class distribution of Austrias spruce forests was found to be strongly skewed towards young age-classes with a distinct peak in the second age-class (20-40 years) in 1990. Whereas this peak even increased in the observations over the available inventory periods EFISCEN simulated a partly shift to the next age-class (Figure 3). Possible reasons for this deviation might be a strongly skewed age-distribution within the individual age-classes of the observed forest data, whereas in EFISCEN an even distribution of area within one age-class is assumed. However, the general pattern of the age-class distribution (peak in young forest stands, low share of stands older than 100 years) was consistently retained in the simulation over the evaluation period.

For long-term applications of the model the evaluation of annual increment is of particular relevance since increment is a main driving factor of forest development. EFISCEN showed a slight overestimation of mean annual increment compared to AFI5 (+0.66 m<sup>3</sup> ha<sup>-1</sup> a<sup>-1</sup>) and underestimated increment relative to the data of AFI6 (-0.52 m<sup>3</sup> ha<sup>-1</sup> a<sup>-1</sup>). Considering the fact that the EFISCEN increment estimations only account for the prevailing forest structure but not for particular environmental conditions in the respective periods the correspondence can be considered satisfactorily. Moreover, the simulated province-level pattern agreed well to the observations, finding increments to be highest in Upper Austria and lowest in Tyrol (Table 2).



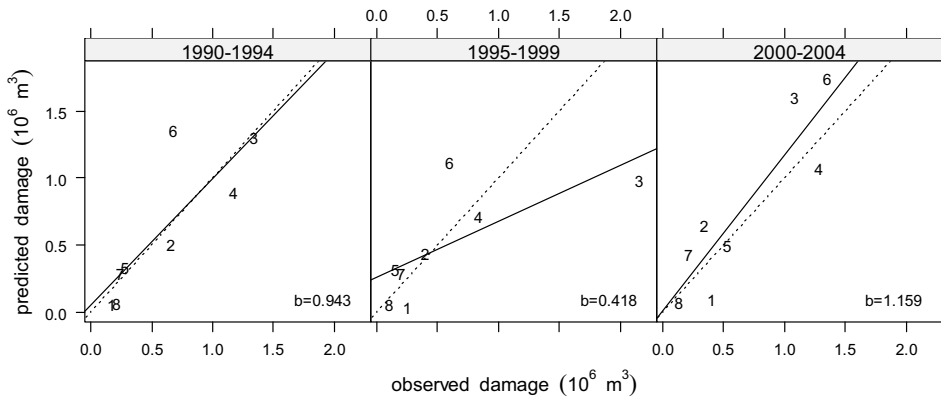
**Figure 3.** Comparison of inventory and EFISCEN age-class distribution (20-year age-classes; 1 = 0-20 years, 7 = 121-140 years, 8=>140 years) for the initialization (AFI4) as well as for two subsequent forest inventory periods (AFI5: 1992-1996; AFI6: 2000-2002).

**Table 2.** Comparison of EFISCEN simulations to forest inventory data (AFI5: 1992-1996; AFI6: 2000-2002) on province level. Provinces: bgl: Burgenland, knt: Carinthia, nö: Lower Austria, oö: Upper Austria, sbg: Salzburg, stmk: Styria, tir: Tyrol, vbg: Vorarlberg.

inventory- period	province	average standing volume stock			average increment		
		observed	predicted	$\Delta\%$	observed	predicted	$\Delta\%$
AFI5	bgl	219.8	208.1	-5.3	10.3	9.5	-7.9
	knt	284.7	292.8	+2.8	8.8	9.4	+7.7
	nö	275.8	289.9	+5.1	8.5	9.4	+10.3
	oö	325.5	337.3	+3.6	10.5	11.3	+7.4
	sbg	325.0	325.1	+0.0	8.5	8.7	+2.7
	stmk	291.1	301.6	+3.6	8.8	9.6	+9.3
	tir	286.5	290.1	+1.3	6.5	6.8	+5.5
	vbg	382.0	342.4	-10.4	7.2	7.5	+3.7
AFI6	bgl	254.7	249.0	-2.2	11.5	10.3	-13.2
	knt	324.3	313.6	-3.3	10.9	10.0	-8.0
	nö	315.6	315.7	+0.0	10.6	9.9	-7.1
	oö	376.3	373.8	-0.7	12.3	11.8	-4.0
	sbg	369.6	343.5	-7.1	9.7	9.2	-4.3
	stmk	331.3	322.5	-2.7	10.4	10.3	-1.0
	tir	326.7	296.5	-9.2	7.7	7.1	-8.2
	vbg	387.2	326.5	-15.7	9.3	7.8	-15.8

### ***Evaluation of simulated bark beetle damage***

Over the 15-year evaluation period the overall simulated damage with EFISCEN (15.11•million cubic meters (Mm<sup>3</sup>), salvaged volume u.b.m.) agreed well with the observation (14.71•Mm<sup>3</sup> , salvaged u.b.m.). In the individual five-year simulation periods, observation and prediction matched well in the period 1990 to 1994 (difference +3.2%), were slightly underestimated in the second and overestimated in the third five-year period (-14.2% and +17.1% respectively). The simulations matched well with the pattern of damage in the 15 year period, simulating a decrease in damage from the first to the second simulation period and a significant increase with the highest damages in the period 2000 to 2004. At province level, damages simulated with EFISCEN corresponded well to the observations (Figure 4). In a regression of observed against predicted data the highest coefficient of determination was achieved in the period 2000 to 2004 with  $R^2=0.802$ , values for periods 1990 to 1994 and 1995 to 1999 were 0.683 and 0.496 respectively. In the first and the last five-year period the slopes of the regression analysis were not significantly different from one whereas in period 1995 to 1999 the slope was significantly lower than one ( $\alpha=0.05$ ). In the second period, a dramatic increase in damages in the province Lower Austria was observed. Based on an already high beetle population levels after the windthrow of the early nineties this rise was triggered by a local snow breakage event in 1995 in combination with non-efficient forest protection measures (Donaubauer et al. 1995, 1996). This local outbreak could not be reproduced in the model, resulting in a considerable underestimation of damages in Lower Austria in the period 1995 to 1999, which exerted a strong leverage on the overall regression results. Re-analysing the intermediate five-year period omitting the value for Lower Austria resulted in a regression slope of 1.12 (not significantly different from one at  $\alpha=0.05$ ) and a percentage of variance explained comparable to the other two evaluation periods ( $R^2=0.624$ ).

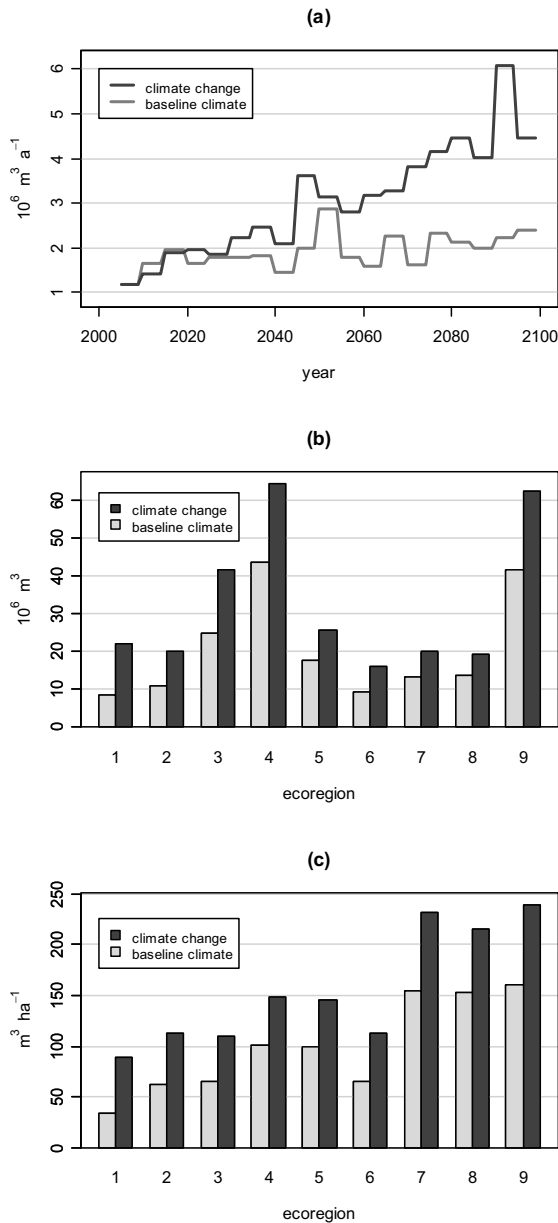


**Figure 4.** Observed versus predicted five-year *Ips typographus* damages for Austria's federal provinces (1: Burgenland, 2: Carinthia, 3: Lower Austria, 4: Upper Austria, 5: Salzburg, 6: Styria, 7: Tyrol, 8: Vorarlberg) in three evaluation periods ( $b$  denotes the slope coefficient of the regression model, solid line: regression, dashed line: 1:1 line). Values are given in salvaged merchantable stemwood under bark.

## Analysis of climate change impacts and adaptation strategies

### *Impact of climate change on bark beetle damage*

In analysing the impact of climate change on bark beetle disturbances in Austria's Norway spruce forests results for the baseline climate and the climate change scenario were compared under business as usual management. Reviewing the temporal pattern of simulated damages over the scenario period from 2005 to 2099 (Figure 5a) it can be seen that already under baseline climate damages increased significantly to more than  $2.38 \text{ Mm}^3 \text{ a}^{-1}$  in period 2095 to 2099 (+79.7% relative to the average damage of the evaluation period 1990-2004). This increase was a result of a considerable change in age-class structure over the 21st century, from the current large areas of young Norway spruce stands to higher shares of mature forests highly susceptible to damage by *Ips typographus*. Under the climate change scenario damages showed a dramatic increase over the 21st century with simulated damage levels being 3.4 times higher in 2099 than in the evaluation period and reaching at peak  $6.09 \text{ Mm}^3 \text{ a}^{-1}$  (period 2090 to 2094; 4.6-fold increase relative to 1990-2004).



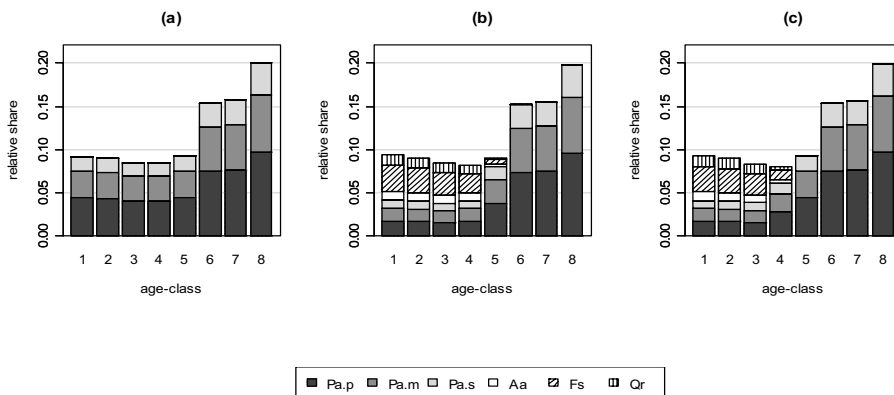
**Figure 5.** Temporal development of annual bark beetle damage (panel a) and spatial distribution of total accumulated bark beetle damages (2005 to 2099, panels b and c) under baseline climate and a climate change scenario. Ecoregions range from the Inner Alps (1) to pre-alpine areas (7,8) and the Bohemian massif (9), see Figure 2.



Accumulated damage over the 95-year scenario period under climate change was +159% higher than under the baseline climate. Particularly the pre-alpine ecoregions (7 to 9) showed very high average damage levels under climate change (Figure 5c). However, relative increases in total accumulated damage compared to the baseline climate were highest in the alpine ecoregions with the strongest increase in ecoregion 1 (inner Alps, +166%). In total, ecoregion 4 had the highest contribution to accumulated bark beetle damage over the 95 year period (22.4% under the climate change scenario, see Figure 5b). However, the spatial distribution of total damages changed considerably over the simulation period: In period 2005 to 2009 absolute damages in the alpine ecoregions 1 and 2 were distinctly lower than those of the pre-alpine ecoregions 7 and 8 despite much larger Norway spruce areas in the Alps (ecoregions 1 and 2: 0.55 Mm<sup>3</sup>; ecoregions 7 and 8: 0.99 Mm<sup>3</sup>). This relation reversed in the climate change scenario resulting in a considerably higher damage from the two inner alpine ecoregions compared to the two pre-alpine ecoregions in the period 2095 to 2099 (3.59 Mm<sup>3</sup> and 2.63 Mm<sup>3</sup> respectively).

### *Effect of adaptive management strategies*

In order to tentatively assess potential adaptive management strategies to cope with a drastic increase in bark beetle damages two adaptation strategies differing in the onset of adaptation measures were compared to business as usual management under the climate change scenario. The stand conversion activities in the PNV-strategies resulted in a considerable reduction in Norway spruce area (Figure 6), however, affecting only young to intermediate age-classes in the simulation period since conversion activities were restricted to areas subject to final felling.



**Figure 6.** Tree species composition over age-classes (20-year class width, 1 = 0-20 years, 7 = 121-140 years, 8=>140 years) for the year 2099 under the climate change scenario and business as usual management strategy (a) as well as under the species change strategies PNV-early (b) and PNV-late (c). Pa.p = pure Norway spruce; Pa.m = mixed Norway spruce; Pa.s = only minor Norway spruce share in stand; Aa = Silver fir; Fs = beech; Or = oak.

Consequently, since young forests were not susceptible to *Ips typographus* and old forests were only marginally affected by the conversion management, simulated bark beetle disturbances in the PNV-strategies showed only minor effects in the first part of the scenario period. Overall, the accumulated bark beetle damage over the 95 year scenario period was 7.81 Mm<sup>3</sup> and 2.92 Mm<sup>3</sup> lower than under business as usual management in PNV-early and PNV-late respectively. In line with the design of the conversion strategies (i.e., target species mixture based on the current potential natural vegetation) little to no changes were simulated for the alpine ecoregions, where Norway spruce is dominant or co-dominant species in the current natural vegetation composition. The highest overall reduction was simulated in ecoregion 8, where Norway spruce was fully replaced by deciduous species in the conversion strategies. Over the simulation period the effect of delaying the onset of the adaptive management for 15 years (PNV-late) was considerably reducing the overall conversion effect on bark beetle damages from -2.69% in PNV-early to -1.01%.

Since long lead times for the effects of the conversion strategies were found in addition to the total accumulated damages over the 95-year scenario period, the period 2095 to 2099 is reviewed in detail (Table 3). In the last five-year simulation period the effect of adaptive management was much more pronounced due to species changes already affecting age-classes of increased susceptibility to *Ips typographus*. From 2095 to 2099, the management strategy PNV-early reduced damages for 1.85•Mm<sup>3</sup> relative to baseline management (-8.3%). PNV-late resulted only in a reduction in damages of -4.1%.

**Table 3.** Simulated average damage for the business as usual management (BAU) in the 95 year scenario period and in the last five-year simulation period. Relative changes under the two adaptive management strategies PNV-early and PNV-late are indicated.

ecoregion	average damage period 2005-2099			average damage period 2095-2099		
	BAU [m <sup>3</sup> ha <sup>-1</sup> ]	ΔPNV-early	ΔPNV-late	BAU [m <sup>3</sup> ha <sup>-1</sup> ]	ΔPNV-early	ΔPNV-late
1	89.19	±0.0%	±0.0%	7.96	±0.0%	±0.0%
2	112.45	-0.8%	-0.3%	9.15	-2.6%	-1.2%
3	109.13	-1.2%	-0.5%	9.84	-3.7%	-1.8%
4	148.19	-3.0%	-1.1%	11.13	-9.7%	-4.6%
5	144.87	-3.9%	-1.5%	11.36	-12.4%	-6.2%
6	113.47	-4.0%	-1.4%	9.95	-11.8%	-5.5%
7	231.07	-4.1%	-1.5%	15.24	-13.5%	-6.8%
8	215.21	-6.3%	-2.3%	14.81	-20.5%	-10.3%
9	239.60	-2.5%	-1.0%	15.78	-8.0%	-4.2%

---

## Discussion and conclusion

### Simulating bark beetle damages in a large scale scenario model

We presented a climate sensitive approach to simulate damage by *Ips typographus* in the large scale scenario model EFISCEN by means of a two-stage multivariate meta model based on extensive stand level model simulations. The particular aim of the approach was to combine information on climate response of beetle development and host resilience with forest structure and composition at the landscape level to cover a broad range of potential interrelationships between climate change and biotic disturbances. The chosen intermediate spatial resolution of the simulation approach (federal provinces forest ecoregions) was found to balance the needs of detailed information on environmental and stand conditions in heterogeneous landscapes as well as sufficiently large simulation entities for robust predictions of forest development within the EFISCEN matrix framework (cf. Thürig and Schelhaas 2006). The evaluation against forest inventory data showed good agreement between the model and the observations, with EFISCEN being able to simulate the observed increase in standing stock and also capturing observed periodical increment levels. The latter result particularly supported the application of EFISCEN over the extended time horizon of this study (cf. Nabuurs et al. 2000). However, deviances as found for the province Vorarlberg point towards the general limitations of the EFISCEN approach, which is restricted to even-aged forests managed under an age-class system. The simulation of uneven-aged forestry as practiced in large parts of Vorarlbergs forests and as generally promoted as a promising alternative management option is strongly limited within the current EFISCEN framework.

Furthermore, good correspondence between observed and predicted bark beetle damages at province level was found. Using averaged climate information for ecoregions (two to six entities per province) EFISCEN including the bark beetle disturbance module was able to accurately reproduce the observed damage levels per province in three consecutive five-year periods. This result is particularly satisfactorily since no overall calibration of the simulation approach was performed to large-scale data sets. Model results were a combination of simulated forest structures in EFISCEN and an upscaling of PICUS model logic on bark beetle damages. The current implementation of bark beetle damages in PICUS was found to suit the requirements for such an upscaling exercise well. Whereas the absence of interannual bark beetle gradations has to be seen as major limitation of the current approach in PICUS (cf. Seidl et al. 2007c), the annually independent calculation fits well for an upscaled application in the EFISCEN environment. Since EFISCEN is a non-spatial model with regard to the distribution of individual forest areas within a simulation entity the simulation of a spatial spread of bark beetle outbreaks as a result of an interannual gradation is generally confined by the modelling framework. Such

limitations could be overcome by bolstering increased spatial detail in the design of an advanced large scale scenario model.

Furthermore, the approach of utilizing annual climate information but aggregate estimated damages to five-year time steps in EFISCEN was found reasonable in the climate-dependent simulation of bark beetle damages at landscape scale. Observations show that a time-lag between climate conditions particularly favourable for the beetle and an actual increase in bark beetle damages exist (e.g., Krehan 1993, Tomiczek et al. 2005), which is not accounted for in the annual calculations of damage probability and damage intensity. Thus, in a comparison to observed data a periodical approach is meaningful. However, it has to be noted that in the evaluation experiment the arbitrary choice of periods can have an influence on the results of the comparison. With that regard the over-estimation in the period 2000 to 2004 is somewhat put into perspective by increases in observed bark beetle damages in Austria in 2005, although climate conditions in this year were not particularly favourable for bark beetle development (Krehan and Steyrer 2006). The overall simulated damage over the 15 year evaluation period corresponded very well to the accumulated observed damages with a difference of only +2.7%.

A major limitation of the presented approach is the implicit assumption of an average level of additional breeding material (e.g., wounded trees, fresh standing and downed deadwood) for bark beetles. Several studies find a considerable impulse to bark beetle gradations from the increasing availability of such highly susceptible breeding material through for instance windthrow or snow breakage events (e.g., Schroeder 2001, Eriksson et al. 2005). This limitation is clearly demonstrated for the province of Lower Austria in the evaluation period 1995 to 1999, where the model was not able to simulate a dramatic increase in observed damages as a result of a local snow breakage event in 1995 (Donaubauer et al. 1996). Improvements could be made by a direct inclusion of such breeding material which, however, would *inter alia* require temporally accurate prediction of abiotic disturbance events as well as detailed knowledge on deadwood pools suitable for breeding (cf. Göthlin et al. 2000). With regard to this limitation future predictions have to be seen as indicative investigations of the climate-dependencies of the herbivore-host relationship. Since several studies point at the possibility for increases in extreme weather events as storms under climate change (e.g., Leckebusch and Ulbrich 2004) the results presented in the scenario analysis have to be seen as conservative estimates.

## **Climate change impacts and effects of adaptive management**

Simulations of a climate change scenario for the 21st century resulted in a dramatic increase in bark beetle damages in Austrias Norway spruce forests. In low-lying areas corresponding damage levels would render controlled forest management in Norway spruce dominated forests unfeasible. Several of these areas (e.g., ecoregions 7 and 8) are already currently under high risk (cf. Krehan 1993, Donaubauer et al. 1996, Tomiczek et al. 1997, Spiecker et al. 2004) and will face increasing pressure under

climate change. However, simulation results also showed a particular increase in damages in alpine areas as a result of a strongly increased Norway spruce area susceptible to bark beetle damage. Such trends have already been observed in recent years (Krehan and Steyrer 2006) and adumbrate the problems of increased bark beetle damages in alpine terrain (i.e., difficult recovery of damaged timber, limitations to the application of conventional forest protection routines in steep terrain). Additional adverse effects associated with such an expansion of biotic disturbances into alpine space are to be expected: For instance, the protective function of forests is of high importance for alpine areas. In Austria the main forest function of almost one fifth of the forest area (Anonymous 2002) is protection against inter alia soil erosion, avalanches, mudflows or rockfall. Distinct increases in damage under climate change have the potential to hamper a sustainable provision of these forest functions, and thus require a particular focus in the development and implementation of adaptive management strategies for vulnerable areas. Moreover, forests are currently the largest terrestrial C storage in Austria (Weiss et al. 2000). An increase in disturbances as found in this study could impose a negative impact on forest C storages and counteract efforts to climate change mitigation in forestry (cf. Seidl et al. 2007a).

The preliminary results of this study should be expanded in future works taking the considerable variability in climate predictions into account and investigating additional adaptive management alternatives with focus also on alpine areas. This need is emphasized by the investigated adaptive management strategies in this study, which resulted in very limited reduction in bark beetle damage especially in alpine areas. In this regard the widespread recommendation among practitioners (e.g., Müller 1994, Leitgeb and Englisch 2006) to aim at a species composition close to a static potential natural vegetation related to current environmental conditions (e.g., PNV strategies based on Kilian et al. 1994) proved inefficient considering shifting disturbance regimes. Notwithstanding the relevance of broad forest type classifications on ecoregion level for practitioners, forest management planning needs to account for changing environmental conditions. Considering the long-term effect of forest management decisions simulation tools can help to evaluate decisions under changing environmental conditions.

The importance of thorough management planning explicitly addressing climate change is underlined by the fact that considerable lead-times in effects of adaptive management were found in this study. Strategy PNV-early, starting the conversion immediately, resulted in reduced bark beetle damages not before the second half of the century. Moreover, a delay in the onset of adaptive management of 15 years diminished the total effect of PNV-early by  $-62.4\%$  over the course of one approximate rotation period. Although both adaptive management strategies showed only limited overall reduction potential with regard to bark beetle damage, the effect in the last simulation period indicated the high long-term potential of conversion management. However, additional strategies should be scrutinized in the future, for instance increasing harvest levels and not limiting conversion to stands subject to final felling in highly vulnerable areas in order to achieve higher reduction potentials.

Nonetheless, scenarios also have to address operationality and practical feasibility in order to be potentially implementable. With this regard efforts to adapt forest ecosystems to climate change are to be balanced with economical and ecological sustainability under the framework of sustainable forest management.

## Acknowledgements

This work was partly funded by a scholarship grant of the European Forest Institute. Additionally funds came from the EU project EFORWOOD (Contract Nr FP6-518128-2).

## References

- Anonymous (1997) *Ergebnisse der Österreichischen Waldinventur 1992 bis 1996. Mit Vergleichsdaten 1986 bis 1990*. Forstliche Bundesversuchsanstalt – Waldforschungszentrum Wien. CD-ROM.
- Anonymous (2002) *Ergebnisse der Österreichischen Waldinventur 2000 - 2002*. Bundesforschungs- und Ausbildungszentrum für Wald, Naturgefahren und Landschaft (BFW) Wien. <http://web.bfw.ac.at/i7/oewi.oewi0002> (2006-12-07)
- Ayres MP, Lombardero MJ (2000) Assessing the consequences of global change for forest disturbance from herbivores and pathogens. *The Science of the Total Environment*, 262, 263-286.
- Baier P, Pennerstorfer J, Schopf A (2007) PHENIPS - a comprehensive phenology model for risk assessment of outbreaks of the European spruce bark beetle, *Ips typographus* (L.) (Col., Scolytidae). *Forest Ecology and Management*, in revision.
- Bale JS, Masters GJ, Hodkinson ID et al. (2002) Herbivory in global climate change research: direct effects of rising temperature on insect herbivores. *Global Change Biology*, 8, 1-16.
- Botkin DB (1993) *Forest dynamics: an ecological model*. Oxford University Press, Oxford. 309 pp.
- Candau JN, Fleming RA (2005) Landscape-scale spatial distribution of spruce budworm defoliation in relation to bioclimatic conditions. *Canadian Journal of Forest Research*, 35, 2218-2232.
- Currie WS, Nadelhoffer JK (1999) Dynamic redistribution of isotopically labeled cohorts of nitrogen inputs in two temperate forests. *Ecosystems*, 2, 4-18.
- Currie WS, Nadelhoffer JK, Aber JD (1999) Soil detrital processes controlling the movement of <sup>15</sup>N tracers to forest vegetation. *Ecological Applications*, 9, 87-102.
- Currie WS, Nadelhoffer KJ, Aber JD (2004) Redistribution of <sup>15</sup>N highlight turnover and replenishment of mineral soil organic N as a long-term control on forest C balance. *Forest Ecology and Management*, 196, 109-127.

- Dale VH, Joyce LA, McNulty S, Neilson RP (2000) The interplay between climate change, forests, and disturbances. *The Science of the Total Environment*, 262, 201-204.
- Donaubauer E, Krehan H, Tomiczek C (1995) Forstschadenssituation in Österreich. *Allgemeine Forstzeitung*, 07/95, 376-379.
- Donaubauer E, Krehan H, Tomiczek C (1996) Forstschadenssituation in Österreich. *Allgemeine Forstzeitung*, 07/96, 386-388.
- EEA (2006) *How much bioenergy can Europe produce without harming the environment?* European Environment Agency. EEA report 07/2006, Copenhagen. ISSN 1725-9177. 67 pp.
- Eriksson M, Pouttu A, Roininen H (2005) The influence of windthrow area and timber characteristics on colonization of wind-felled spruces by *Ips typographus* (L.). *Forest Ecology and Management*, 216, 105-116.
- Faraway JJ (2006) *Extending the linear model with R: generalized linear, mixed effects and nonparametric regression models*. Chapman and Hall, Boca Raton. 301 pp.
- Göthlin E, Schroeder LM, Lindelöw A (2000). Attacks by *Ips typographus* and *Pityogenes chalcographus* on windthrown spruces (*Picea abies*) during the two years following a storm felling. *Scandinavian Journal of Forest Research*, 15, 542-549.
- Harrington R, Fleming RA, Woiwood IP (2001) Climate change impacts on insect management and conservation in temperate regions: can they be predicted? *Agricultural and Forest Entomology*, 3, 233-240.
- IPCC (2000) *Emissions Scenarios 2000*. Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge. 570 pp.
- Killian W, Müller F, Starlinger F (1994) *Die forstlichen Wuchsgebiete Österreichs. Eine Naturraumgliederung nach walddökologischen Gesichtspunkten*. Bericht 82, Forstliche Bundesversuchsanstalt, Wien. ISSN 0374-9037. 60 pp.
- Krehan H (1993) Forstschadenssituation in Österreich. *Allgemeine Forstzeitung*, 07/93, 352-354.
- Krehan H, Perny B, Preier P, Steyrer G (2006) *Österreichisches Borkenkäfer-Monitoring 2005*. <http://bfw.ac.at/400/2168.html> (2006-12-07)
- Krehan H, Steyrer G (2004) Borkenkäferkalamität 2003. *Forstschutz Aktuell*, 31, Bundesforschungs- und Ausbildungszentrum für Wald, Naturgefahren und Landschaft (BFW) Wien, ISSN 1815-5103. 10-12.
- Krehan H, Steyrer G (2005) Borkenkäfer-Monitoring und Borkenkäfer-Kalamität 2004. *Forstschutz Aktuell*, 33, Bundesforschungs- und Ausbildungszentrum für Wald, Naturgefahren und Landschaft (BFW) Wien, ISSN 1815-5103. 12-14.
- Krehan H, Steyrer G (2006) Borkenkäfersituation und Borkenkäfer-Monitoring 2005. *Forstschutz aktuell*, 35, Bundesforschungs- und Ausbildungszentrum für Wald, Naturgefahren und Landschaft (BFW), Wien. pp.10-14. ISSN 1815-5103
- Landsberg JJ, Waring RH (1997) A generalized model of forest productivity using simplified concepts of radiation-use efficiency, carbon balance and partitioning. *Forest Ecology and Management*, 95, 209–228.

- Leckebusch GC, Ulbrich U (2004) On the relationship between cyclones and extreme windstorm events over Europe under climate change. *Global and Planetary Change*, 44, 181-193.
- Leitgeb E, Englisch M (2006) Klimawandel- Satandörtliche Rahmenbedingungen für die Forstwirtschaft. *BFW Praxisinformation*, 10, ISSN 1815-3895. pp. 9-11.
- Lexer MJ (1995) *Beziehungen zwischen der Anfälligkeit von Fichtenbeständen (Picea abies (L.) Karst.) für Borkenkäferschäden und Standorts- und Bestandesmerkmalen unter besonderer Berücksichtigung der Wasserversorgung*. Dissertation, University of Natural Resource Management and Applied Life Sciences (BOKU) Vienna. 210 pp.
- Lexer MJ, Hönninger K (2001) A modified 3D-patch model for spatially explicit simulation of vegetation composition in heterogeneous landscapes. *Forest Ecology and Management*, 144, 43–65.
- Lexer MJ, Hönninger K, Scheifinger H et al. (2002) The sensitivity of Austrian forests to scenarios of climatic change: a large scale risk assessment based on a modified gap model and forest inventory data. *Forest Ecology and Management*, 162, 53-72.
- Mäkelä A, Landsberg J, Ek AR et al. (2000) Process-based models for forest ecosystem management: current state of the art and challenges for practical implementation. *Tree Physiology*, 20, 289-298.
- Marschall J (1975) *Hilfstafeln für die Forsteinrichtung*. Österreichischer Agrarverlag, Wien. 199 pp.
- Mitchell TD, Carter TR, Jones PD, Hulme M, New M (2004) A comprehensive set of high-resolution grids of monthly climate for Europe and the globe: the observed record (1901-2000) and 16 scenarios (2001-2100). *Tyndall Centre Working Paper*, 55. 25 pp.
- Moorhead DL, Currie WS, Rastetter EB, Parton WJ, Harmon ME (1999) Climate and litter quality controls on decomposition: an analysis of modeling approaches. *Global Biogeochemical Cycles*, 13, 575–589.
- Müller F (1994) Müssen wir waldbauliche Konzepte ändern? In: *Klimaänderung in Österreich*. Herausforderung an Forstgenetik und Waldbau (eds Geburek T, Müller F, Schultze U), pp.67-75. Forstliche Bundesversuchsanstalt Wien, Berichte 81. ISSN 1013-0713.
- Nabuurs GJ, Paivinen R, Pussinen A, Schelhaas MJ (2003) *Development of European Forests until 2050*. European Forest Institute Research Report 15. Brill, Leiden - Boston. 242 pp.
- Nabuurs GJ, Pussinen A, van Brusselen J, Schelhaas MJ (2007) Future harvesting pressure on European forests. *European Journal of Forest Research*, 126, 391-400
- Nabuurs GJ, Schelhaas MJ, Pussinen A (2000) Validation of the European Forest Information Scenario Model (EFISCEN) and a Projection of Finish forests. *Silva Fennica*, 34, 167-179.
- Netherer S, Nopp-Mayr U (2005) Predisposition assessment systems (PAS) as supportive tools in forest management – rating of site and stand-related hazards



- of bark beetle infestation in the High Tatra Mountains as an example for system application and verification. *Forest Ecology and Management*, 207, 99–107.
- Netherer S, Pennerstorfer J (2001) Parameters relevant for modelling the potential development of *Ips typographus* (Coleoptera: Scolytidae). *Integrated Pest Management Reviews*, 6, 177-184.
- Parmesan C, Ryrholm N, Stefanescu C et al. (1999) Poleward shifts in geographical ranges of butterfly species associated with regional warming. *Nature*, 399, 579-583.
- Pussinen A, Meyer J, Zudin S, Lindner M (2005) European mitigation potential. In: *Management of European forests under changing climatic conditions* (eds Kellomäki S, Leinonen S), pp. 383-400. Research Note 163, University of Joensuu. ISBN 952-458-652-5.
- Pussinen A, Schelhaas MJ, Verkaik E, Heikkinen E, Paivinen R, Nabuurs GJ (2001) *Manual for the European Forest Information Scenario Model (EFISCEN); version 2.0*. EFI Internal report 5. European Forest Institute. Joensuu, Finland. 49 pp.
- Rouault G, Candau JN, Lieutier F, Nageleisen LM, Martin JC, Warzée N (2006) Effects of drought and heat on forest insect populations in relation to the 2003 drought in Western Europe. *Annales of Forest Science*, 63, 613-624.
- Sallnäs, O, 1990. *A matrix growth model of the Swedish forest*. Studia Forestalia Suecica 183. 23 p.
- Schadauer K (1999) *Oberhöhenbonität und Standort der Fichte nach Daten der Österreichischen Forstinventur*. Mitteilungen der Forstliche Bundesversuchsanstalt Wien 171. ISBN:3-901347-18-6. 135 pp.
- Schelhaas MJ, Eggers J, Lindner M, Nabuurs GJ, Pussinen A, Päivinen R, Schuck A, Verkerk PJ, van der Werf DC, Zudin S (2007). *Model documentation for the European Forest Information Scenario model (EFISCEN 3.1.3)*. Wageningen, Alterra, Alterra report 1559, EFI Technical Report 26, Joensuu, Finland. 118 p.
- Schelhaas MJ, Nabuurs GJ, Schuck A (2003) Natural disturbances in the European forests in the 19th and 20th centuries. *Global Change Biology*, 9, 1620-1633.
- Schelhaas MJ, Nabuurs GJ, Sonntag M, Pussinen A (2002) Adding natural disturbances to a large-scale forest scenario model and a case study for Switzerland. *Forest Ecology and Management*, 167, 13-26.
- Schelhaas MJ, van Brusselen J, Pussinen A, Pesonen E, Schuck A, Nabuurs GJ, Sasse V (2006) *Outlook for the development of European forest resources*. Geneva timber and forest discussion papers 41, United Nations Economic Commission for Europe, Food and Agriculture Organization of the United Nations. ISSN 1020-7228 118 pp.
- Schroeder LM (2001) Tree mortality by the bark beetle *Ips typographus* (L.) in storm-disturbed stands. *Integrated Pest Management Reviews*, 6, 169-175.
- Schröter D, Acosta-Michlik L, Arnell AW et al. (2004) *ATEAM final report 2004. Advanced Terrestrial Ecosystem Analysis and Modelling*. Potsdam Institute for Climate Impact Research (PIK). 139 pp.

- Seidl R, Lexer MJ, Jäger D, Hönninger K (2005) Evaluating the accuracy and generality of a hybrid forest patch model. *Tree Physiology*, 25, 939-951.
- Seidl R, Rammer W, Baier P, Schopf A, Lexer MJ (2007c) Modelling tree mortality from bark beetle infestation in Norway spruce forests. *Ecological Modelling*, accepted.
- Seidl R, Rammer W, Jäger D, Currie WS, Lexer MJ (2007b) Assessing trade-offs between carbon sequestration and timber production within a framework of multi-purpose forestry in Austria. *Forest Ecology and Management*, 248, 64-79.
- Seidl R, Rammer W, Jäger D, Lexer MJ (2007a) Impact of bark beetle (*Ips typographus* L.) disturbance on timber production and carbon sequestration in different management strategies under climate change. *Forest Ecology and Management*, accepted.
- Shugart HH (1984) *A theory of forest dynamics: the ecological implications of forest succession models*. Springer, New York. 278 pp.
- Spiecker H, Hansen J, Kilmo E, Skovsgaard JP, Sterba H, von Teuffel K (2004) *Norway spruce conversion – options and consequences*. EFI Research Report 18. Brill, Leiden. 320 p.
- Sturtevant BR, Gustafson EJ, Li W, He HS (2004) Modelling biological disturbances in LANDIS: a module description and demonstration using spruce budworm. *Ecological Modelling*, 180, 153-174.
- Thürig E, Schelhaas MJ (2006) Evaluation of a large-scale forest scenario model in heterogeneous forests: a case study for Switzerland. *Canadian Journal of Forest Research*, 36, 671-683.
- Tomiczek C, Cech T, Krehan H, Perny B (2003) Forstschutzsituation 2002 in Österreich. *Allgemeine Forstzeitung*, 07/03, 367-368.
- Tomiczek C, Cech T, Krehan H, Perny B, Steyrer G (2005) Forstschutzsituation 2004 in Österreich. *Allgemeine Forstzeitung*, 07/05, 377-379.
- Tomiczek C, Krehan H, Cech T, Donaubaauer E (1997) Forstschadenssituation in Österreich. *Allgemeine Forstzeitung*, 07/97, 387-389.
- Wermelinger B (2004) Ecology and management of the spruce bark beetle *Ips typographus* – a review of recent research. *Forest Ecology and Management*, 202, 67-82.
- Wermelinger B, Seifert M (1999) Temperature-dependent reproduction of the spruce bark beetle *Ips typographus*, and analysis of the potential population growth. *Ecological Entomology*, 24, 103-110.
- Williams DW, Liebhold AM (2002) Climate change and the outbreak ranges of two North American bark beetles. *Agricultural and Forest Entomology*, 4, 87-99.
- Woltjer M, Rammer W, Brauner M, Seidl R, Mohren GMJ, Lexer MJ (2007) Coupling a 3D patch model and a rockfall module to assess rockfall protection in mountain forests. *Journal of Environmental Management*, in press. doi:10.1016/j.jenvman.2007.01.031.

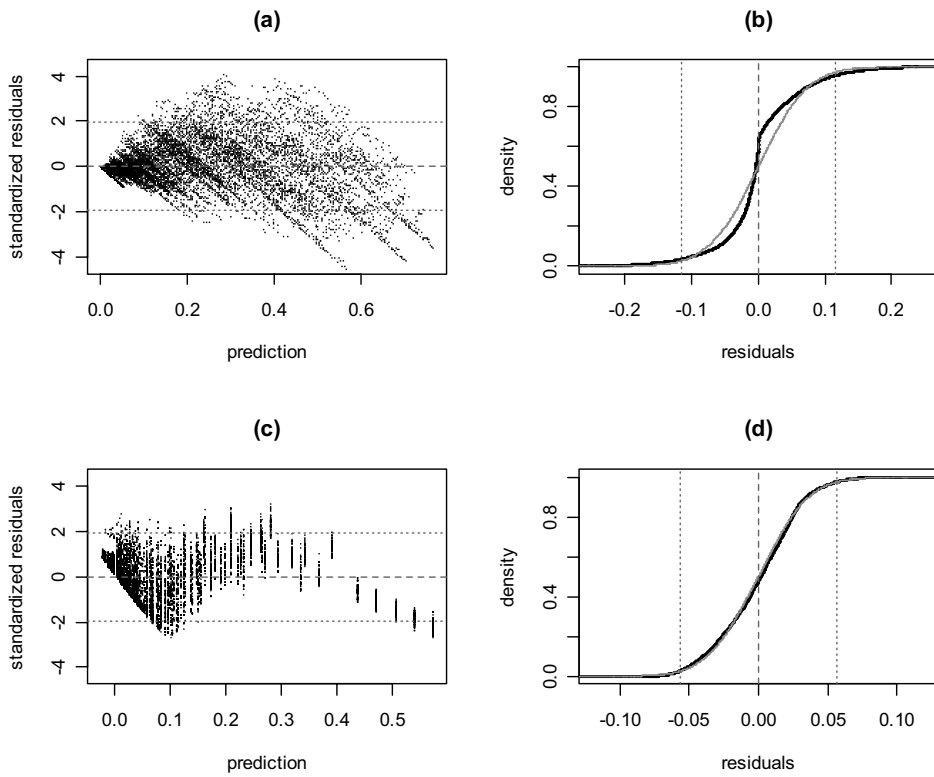
## Appendix

**Table A1.** Parameters for the multivariate logit model of bark beetle probability (Eq. 4)

predictor	factor description	parameter	standard error
a	mean annual temperature (logarithmic)	4.234	0.0123
b	mean annual precipitation (logarithmic)	-1.196	0.00531
c	stand age	-0.0433	0.000142
d	stocking density	0.658	0.0249
e (dummy 50%)	host tree share	-1.425	0.0107
e (dummy 100%)	host tree share	-2.606	0.0104
a · b	interaction temperature x precipitation	-0.214	0.00171
a · c	interaction temperature x stand age	-0.000383	0.0000239
a · d	interaction temperature x density	-0.0527	0.00417
a · e (dummy 50%)	interaction temperature x host tree share	0.0993	0.00183
a · e (dummy 100%)	interaction temperature x host tree share	0.245	0.00178
b · c	interaction precipitation x stand age	-0.0527	0.0000195
b · d	interaction precipitation x density	-0.104	0.00341
b · e (dummy 50%)	interaction precipitation x host tree share	0.278	0.00146
b · e (dummy 100%)	interaction precipitation x host tree share	0.457	0.00143
$\mu$	intercept	0.301	0.0371

**Table A2.** Parameters for the multivariate regression model of soil moisture index (Eq. 8)

predictor	factor description	parameter	standard error
a	mean annual temperature	0.00107	0.0000967
b	(mean annual precipitation) <sup>-2</sup>	25490	574.4
a · b	interaction temperature × precipitation	8282	58.93
$\mu$	intercept	-0.0413	0.000943



**Figure A1.** Graphical residual analysis for the pBB (Eq. 4, panels a and b) and SMI-models (Eq. 8, panels c and d). Panels a and c show standardized residuals over the prediction, panels b and d give the cumulative density function (cdf) of residuals (black line) and the corresponding cdf of the normal distribution (grey line). The 95% bounds of the normal distribution are indicated with dotted lines.

**Table A3.** Distribution of the simulated forest area (i.e. current Norway spruce forest area) to elevation belts and the associated potential natural vegetation for the ecoregions as applied for the calculation of conversion area in the PNV scenarios. Pa = Norway spruce; Aa = Silver fir; Ps = Scots pine; Ld = European larch; Qr = oak; Fs = beech. Assumed species composition per forest type: Pa-Aa-forest: 50% Pa, 50% Aa; Pa-Aa-Fs-forest: 40% Pa, 30% Aa, 30% Fs; (Pa)-Aa-Fs-forest: 20% Pa, 40% Aa, 40% Fs; Fs-(Aa)-forest: 80% Fs, 20% Aa; Pa-(Aa-Fs)-forest: 70% Pa, 15% Aa, 15% Fs

ecoregion	elevation belt [m asl]	% of Norway spruce area	potential natural vegetation composition
1	<900	8.1	Pa, Pa-Ps
	900-1200	20.0	Pa
	>1200	71.9	Pa, Pa-Ld, Ld-Pc
2	<900	12.9	Pa-Aa
	900-1200	29.4	Pa-Aa
	>1200	57.7	Pa
3	<900	20.2	Pa-Aa, Pa-Aa-Fs
	900-1200	31.5	Pa-Aa
	>1200	48.3	Pa
4	<900	45.3	Fs
	900-1200	31.6	Pa-Aa-Fs
	>1200	23.1	Pa-Aa-Fs
5	<900	52.9	Qr, Fs
	900-1200	27.5	Pa-Aa-Fs, Fs
	>1200	19.6	Pa-Aa-Fs, Pa-Aa
6	<900	39.9	Fs, Qr
	900-1200	25.9	Pa-Aa-Fs
	>1200	34.2	Pa-Aa, Fs, Pa
7	<900	93.8	Qr, Fs
	900-1200	5.4	(Pa)-Aa-Fs
	>1200	0.8	(Pa)-Aa-Fs
8	<900	77.8	Qr, Fs
	900-1200	13.3	Fs-(Aa)
	>1200	8.9	Fs-(Aa)
9	<900	89.6	Qr, Fs
	900-1200	10.1	Pa-Aa-Fs
	>1200	0.3	Pa-(Aa-Fs)

**Table A4.** Species change in the PNV conversion strategies. The rows gives the source of the area converted (Pa.p = pure Norway spruce; Pa.m = mixed Norway spruce; Pa.s = only minor Norway spruce share in stand) for the ecoregions, columns gives the species to which the area was converted. Values are given in percent of the area subject to final felling and stand-replacing disturbance in the ecoregion per simulation period.

ecoregion	source	converted to					
		Pa.p	Pa.m	Pa.s	Aa	Fs	Qr
1	Pa.p	100	0.0	0.0	0.0	0.0	0.0
	Pa.m	0.0	100	0.0	0.0	0.0	0.0
	Pa.s	0.0	0.0	100	0.0	0.0	0.0
2	Pa.p	57.6	21.2	0.0	21.2	0.0	0.0
	Pa.m	57.6	21.2	0.0	21.2	0.0	0.0
	Pa.s	0.0	0.0	100	0.0	0.0	0.0
3	Pa.p	48.3	24.8	0.0	23.8	3.1	0.0
	Pa.m	48.3	24.8	0.0	23.8	3.1	0.0
	Pa.s	0.0	0.0	100	0.0	0.0	0.0
4	Pa.p	0.0	21.9	0.0	16.5	61.6	0.0
	Pa.m	0.0	21.9	0.0	16.5	61.6	0.0
	Pa.s	0.0	0.0	54.6	0.0	45.4	0.0
5	Pa.p	0.0	14.3	0.0	11.9	47.3	26.5
	Pa.m	0.0	14.3	0.0	11.9	47.3	26.5
	Pa.s	0.0	0.0	47.0	0.0	26.5	26.5
6	Pa.p	0.0	27.4	0.0	13.5	39.1	20.0
	Pa.m	0.0	27.4	0.0	13.5	39.1	20.0
	Pa.s	0.0	0.0	60.0	0.0	20.0	20.0
7	Pa.p	0.0	1.2	0.0	2.5	49.4	46.9
	Pa.m	0.0	1.2	0.0	2.5	49.4	46.9
	Pa.s	0.0	0.0	6.2	0.0	46.9	46.9
8	Pa.p	0.0	0.0	0.0	4.5	56.6	38.9
	Pa.m	0.0	0.0	0.0	4.5	56.6	38.9
	Pa.s	0.0	0.0	0.0	4.5	56.6	38.9
9	Pa.p	0.0	4.2	0.0	3.1	47.9	44.8
	Pa.m	0.0	4.2	0.0	3.1	47.9	44.8
	Pa.s	0.0	0.0	10.4	0.0	44.8	44.8

## IV Introducing tree interactions in wind damage simulation

*Ecological Modelling 207: 197-209*

*M.J. Schelbaas<sup>1,2,\*</sup>, K. Kramer<sup>1</sup>, H. Peltola<sup>3</sup>, D.C. van der Werf<sup>1</sup>, S.M.J. Wijdeven<sup>1</sup>*

1) Alterra, Green World Research, PO Box 47, NL-6700 AA Wageningen, the Netherlands

2) European Forest Institute, Torikatu 34, FIN-80100, Joensuu, Finland

3) University of Joensuu, Joensuu, Finland

\*) Corresponding author:

Alterra, Green World Research

PO Box 47, NL-6700 AA Wageningen, the Netherlands

E-mail: [martjan.schelhaas@wur.nl](mailto:martjan.schelhaas@wur.nl)

Fax: +31.317.419000

### Abstract

Wind throw is an important risk factor in forest management in North-western Europe. In recent years, mechanistic models have been developed to estimate critical wind speeds needed to break or uproot the average tree of a forest stand. Based on these models, we developed a wind damage module for the individual tree model ForGEM (Forest Genetics, Ecology and Management). For a given wind speed this module assesses the forces on each individual tree, based on the tree dimensions, and support and sheltering provided by other trees. Due to this individual approach, irregular stands can also be assessed. The module is demonstrated on Douglas fir stands (*Pseudotsuga menziesii* (Mirb.) Franco) of different densities in the Netherlands. Patterns of damage are explained, both in freshly exposed stands as well as in sheltered stands. Wind speeds needed to cause damage approximated those of known wind throw events. The wind damage module proved to be very sensitive to simulated tree heights and diameters. Furthermore, the newly introduced support mechanism played an important role in the stability of trees and stands. Lower individual tree stability in dense stands was clearly compensated for by the support of other trees.

### Introduction

Wind throw is an important risk factor in forest management, especially in the temperate and boreal zone. In North-western Europe, the importance of wind throw is illustrated by a series of severe storm events during the last decades, interspersed with numerous smaller events (Schelhaas et al., 2003). In countries where high winds

are frequent, such as Ireland and the United Kingdom, forest management strategies have been adapted to this risk. Even with a low storm frequency, it can be expected that a forest stand will experience some high wind speed events. However, due to the long return intervals of such events, it is difficult for forest managers to learn from past events and to adapt their management. This is supported by a recent study by Blennow and Sallnäs (2002) in Southern Sweden. Although forest owners rank storm highly as a potential risk, they generally do not know how to change their forest management to reduce the risk of damage. There is a clear need for decision support systems for forest managers, but currently there are only a few such systems in place (Peltola et al., 2000b). Mathematical models provide an opportunity for objectively calculating these risks and can show forest managers the long-term implications of their actions (Gardiner and Quine, 2000).

In recent years, substantial progress has been made in the modelling of wind throw. Peltola et al. (1999) and Gardiner et al. (2000) have developed mechanistic models to estimate, for a given tree, the critical wind speed needed for breakage or uprooting. Both models have been used extensively in different types of studies. Talkkari et al. (2000) combined the HWIND model of Peltola et al. (1999) with an airflow model and spatially explicit forest inventory data to identify stands at risk in Finland. Blennow and Sallnäs (2004) did the same for stands in Southern Sweden. The ForestGALES model of Gardiner et al. (2000) was linked to plantation yield models to provide estimates of future damage risks (Suárez et al., 2002). Cucchi et al. (2005) linked ForestGALES to a stand simulator for maritime pine in France. In all these applications, the stand under study is represented by one “average” tree in the wind throw models; assuming even-aged, mono-species stands with little variation in height or diameter. In stands with more variation among trees, such as unmanaged or uneven-aged stands, a different approach is needed. Ancelin et al. (2004) started the work in this direction by using a mechanistic model to evaluate all trees in a stand. However, all these models only evaluate the stability of individual trees and largely ignore interactions among trees. An important factor in stand stability is crown contact between trees. Not only can trees dissipate absorbed wind energy through crown contact (Milne, 1991), they can also physically support each other (Quine et al., 1995). One of the reasons for increased wind damage risk after thinning is that trees receive less support by their neighbours (Ruel, 1995).

In the present study, we made the step from modelling individual tree stability to modelling stand stability. To do this, we developed a wind damage module for the individual tree growth model ForGEM (henceforth ForGEM-W). The use of an individual tree growth model instead of yield tables (as used by Suárez et al., 2002 and Cucchi et al., 2005) allows a much greater range of management options to be investigated, including the management of uneven aged, mixed forests. However, in the current study we only focussed on the development and sensitivity of the model. As a first test and example, we parameterised the model for Douglas fir (*Pseudotsuga menziesii* (Mirb.) Franco) in the Netherlands. Douglas fir has proved to be sensitive to wind throw during storms in the past (Faber, 1975; Sissingh, 1975). Furthermore, a



tree pulling study is available (Schooten, 1985) for this species, as well as measurements in forest reserves and a set of yield tables with different stand densities.

## Material and Methods

### The ForGEM model

ForGEM (Forest Genetics, Ecology and Management) is a forest model that simulates the growth and development of individual trees, on a scale up to several hectares. Several processes can be simulated with different levels of detail according to the needs of the user. In the current application, we used the simplest simulation methods. This means simulation of light interception based on the gap-type approach and no simulation of water and nitrogen balances. Here we will explain only the basics of the model; a more detailed description can be found in Kramer (2004) and Kramer et al. (2007). Most approaches are essentially similar to those of SORTIE (Pacala et al., 1993; Pacala et al. 1996), except in this case the gap-type approach for light interception.

ForGEM simulates individual trees that have fixed coordinates. For each tree relevant characteristics are tracked over time. These are among others age, height, stem diameter at breast height (dbh), crown length and radius, and mass of foliage, branches, heartwood, sapwood, coarse roots and fine roots. Tree growth is driven by light interception. In the gap type approach, light is assumed to come straight from above. The light interception in each 20x20m grid cell is divided over the trees in the cell, according to their foliage mass and its vertical distribution (see Bugmann, 2001). The relative position of the trees is not taken into account. Intercepted radiation is converted into photosynthates via the radiation use efficiency (RUE) parameter (Landsberg and Waring, 1997). The RUE must be calibrated against independent data, such as growth and yield tables. Photosynthates are allocated in such a way that specific ratios between different tree compartments are maintained (see Kramer et al., 2007). These ratios depend on the total aboveground biomass. Photosynthates allocated to the stem are converted into height and dbh increment. Maximum height increment follows a Richards' growth curve (Richards, 1959; Jansen et al., 1996). Actual height increment can be lower due to insufficient resources. The relationship between tree volume, height and dbh is described by species-specific allometric functions, derived from yield table data (Jansen et al., 1996). Resource related mortality appears when all reserves of a tree are depleted and there is no foliage left. Effects of competition are assumed to translate into reduced dbh growth, which determines the probability that a particular tree will die (Kobe et al., 1995). Age related mortality is assumed to follow a 2-parameter Weibull distribution. Recruitment is not used in the present application. Weather variables such as daily radiation, temperature and precipitation are generated by a weather generator.

## Adjustments to ForGEM

A precise simulation of crown radius and dbh is important for the wind damage module. The crown radius influences the sail area (i.e. the area that captures wind) of the tree, and determines how much stress can be transferred to other trees. Both anchorage strength and stem resistance to breakage are derived from dbh, which makes dbh a critical variable. To increase the accuracy of the simulation of these two variables, we introduced competition for space between crowns and an age effect on dbh increment.

### *Crown competition*

In reality, a crown can have almost any shape due to its ability to adapt to local light conditions. For light interception and crown expansion, the crown is always treated as a cylinder. To correct for the irregular shape of a real crown, we represent the crown as two cylinders. Both have the same height, the crown length (CL). The outer cylinder represents the potential influence sphere of the crown, and can be seen as having a radius equal to the longest branch. This radius is further referred to as crown radius (CR). In case of competition not all of the volume within this cylinder will contain the tree's own foliage. Competition is measured as the overlap in volume of these outer cylinders between trees. The shared volume is distributed between the trees according to their actual Leaf Area Index (LAI). Species with a higher LAI will thus be better competitors. We can now calculate a virtual crown volume (VCV) for each tree  $i$ :

$$VCV_i = CV_i - \sum_j CVO_{ij} * \frac{LAI_j}{(LAI_i + LAI_j)} \tag{1}$$

with

$CV_i$  is the crown volume of tree  $i$  and

$CVO_{ij}$  the overlap in crown volume between tree  $i$  and  $j$ .

The virtual crown volume is also assumed to be a cylinder, with a virtual crown radius (VCR <sub>$i$</sub> ):

$$VCR_i = \sqrt{\frac{VCV_i / CL_i}{\pi}} \tag{2}$$

The development of the virtual crown radius only depends on the competition with other trees. Under heavy competition pressure the virtual radius can decrease. If the virtual crown radius reaches zero, we assume the tree is dead. This replaces the original increment-dependent mortality. The maximum increase of the projected

crown area (Maximum Percentage Crown Area Increment, %CAI<sub>MAX</sub>, % per year) of the outer cylinder is expressed as a relative logarithmic function:

$$\%CAI_{MAX} = 1.25 * (b_1 * \ln(CA) + b_2) \quad (3)$$

where

$$b_1 = \frac{-b_2}{\ln((CR_{MAX})^2 * \pi)} \quad (4)$$

with  $b_2$  and  $CR_{MAX}$  site-dependent parameters.

Parameter  $b_2$  can be fitted to local growth and yield tables. The factor 1.25 is introduced in the model to compensate for the effect of crown competition that is already present in stands of normal density in yield tables. This factor is derived from the difference between stands with normal and low density for Douglas fir in Jansen et al. (1996). The maximum increase is then modified according to the ratio between virtual crown volume and crown volume, which is an indication of the competition status of the tree:

$$\%CAI = \%CAI_{MAX} * \frac{VCV}{CV} \quad (5)$$

Crown length is fixed in the model, but will in reality also be influenced by competition with other trees. However, according to Peltola et al. (1999) the HWIND model is much more sensitive to crown width than crown length.

#### *Dbh increment in older trees*

The original model overestimated dbh increment at higher ages, as compared to growth and yield tables. A reduction in dbh increment at higher age is generally observed in trees, but the underlying mechanisms are still a topic of debate (Binkley et al., 2002). In the ForGEM model, no processes are included that cause a reduction of dbh increment at higher ages. Dbh is an important variable for the wind damage module, and thus we included a simple mechanism that decreases the RUE, depending on age:

$$RUE = RUE * AgeEffect \quad (6)$$

$$AgeEffect = 1 \quad \text{if } Age \leq T$$

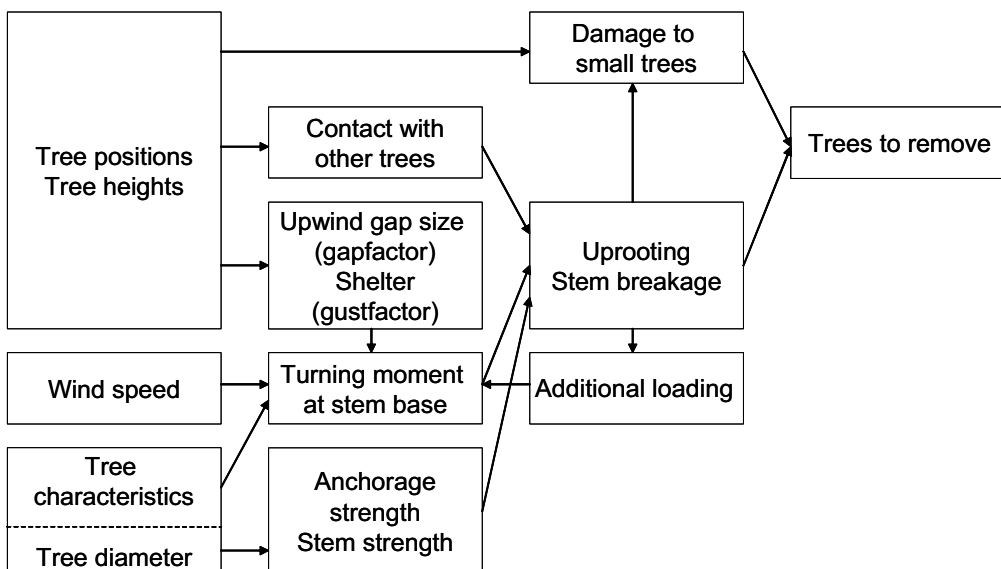
$$AgeEffect = 1 - c^{(T/(Age-T))} \quad \text{otherwise} \quad (7)$$

T determines the age after which the decrease in RUE should start, and c determines the speed of decrease.

### Modelling the mechanism of wind damage

#### *The wind damage module*

The wind damage module in ForGEM-W is largely based on the principles of the HWIND model as developed by Peltola et al. (Peltola and Kellomäki, 1993, Peltola et al., 1999 and Gardiner et al., 2000). A general outline of the wind damage module is shown in Figure 1.



**Figure 1.** Outline of wind damage module in ForGEM-W.

All trees in the simulated object are individually subjected to the forces of the wind. The wind is characterised by a constant mean speed and direction. Forces on the tree are initially calculated as if it was located at the stand edge without support from other trees, with an infinite upwind gap size. Corrections are then made for the real size of the gap (gapfactor) and the sheltering a tree receives from upwind trees (gustfactor). Both factors are derived from the actual configuration of the stand in front of the tree. Three forces contribute to the to the bending moment of the stem: (1) a horizontal force due to the impact of the wind; (2) a vertical force due to gravity, acting on the displaced stem and crown; and (3) an additional vertical force if the tree is hit by other trees that fall down. Through crown contact with other trees, the bending moment acting on the stem can be reduced. If the total bending moment exceeds the maximum resistive moment provided by the root system the tree is

assumed to uproot. Similarly, a tree is assumed to break if the stress acting on the stem exceeds its maximum resistance. Young trees are very flexible and will usually not break or uproot (Mencuccini et al., 1997; Stokes, 1999). We therefore assumed that trees smaller than 5m cannot break or uproot (Holtam, 1971). However, such trees can be killed when they are hit by other trees falling on them.

### *Wind*

Input in the wind module is the wind direction and maximum mean hourly wind speed per day at a certain reference height (here 10m). We will further refer to this as “wind speed”, unless otherwise stated. While running a long-term simulation this input is provided by a random generator which can be calibrated to measured wind climate. Alternatively, it is possible to subject a certain stand or initial situation to a user-specified wind speed and direction. The latter option is used in this study to investigate the sensitivity of the model. The input wind speed is used to construct a logarithmic wind profile at the stand edge (Monteith, 1975):

$$u(z) = \left( \frac{u_*}{k} \right) \ln \left( \frac{z-d}{z_0} \right) \quad (8)$$

with

- $u(z)$  wind speed ( $\text{m s}^{-1}$ ) at height  $z$ ,
- $u_*$  friction velocity ( $\text{m s}^{-1}$ ),
- $k$  von Karman’s constant (dimensionless),
- $d$  zero plane displacement (m) and
- $z_0$  roughness length (m).

### *Vertical tree profile*

To calculate the wind loading, for each individual tree the projected vertical area of the stem is derived from dbh, tree height and the taper functions. The projected vertical area of the crown is calculated from tree height, crown base and virtual crown radius, assuming a diamond shaped crown (Figure 3). The relative crown height where the crown reaches its maximum width is species-specific. This is further referred to as the crown centre. The tree is assumed to be symmetric in all directions. During wind loading, the crown will reduce its sail area against the wind, called streamlining. This reduction is calculated in the same way as described by Peltola and Kellomäki (1993). If there is no foliage present (broadleaves in winter), the projected area is assumed to be 20% (Peltola et al., 1999) of the projected crown area, and no further streamlining of the crown is applied.

*Wind loading*

For the calculations of forces on the tree, the tree is divided in segments of 1m. The wind-induced force ( $F_w(z)$  (N)) acting at the midpoint of each segment is calculated as (Monteith, 1975):

$$F_w(z) = \frac{C_d \times \rho \times u(z)^2 \times A(z)}{2} \quad (9)$$

with

- $u(z)$  mean wind speed ( $\text{m s}^{-1}$ ) at height  $z$  (m),
- $A(z)$  projected area ( $\text{m}^2$ ) of the tree against the wind,
- $C_d$  drag coefficient (dimensionless) and
- $\rho$  air density ( $\text{kg m}^{-3}$ ).

The bending of the stem is assumed to be proportional to the wind force and inversely proportional to the stem's stiffness, according to the formulas given by Pennala (1980). Stem stiffness is related to the modulus of elasticity (MOE) of the tree and its dbh.

*Gravity loading*

The displacement of the tree will invoke a force ( $F_g(z)$  (N)) due to gravity (Peltola and Kellomäki, 1993):

$$F_g(z) = M(z) * g \quad (10)$$

with

- $M(z)$  green mass of the stem and crown and
- $g$  gravitational constant

Each stem segment is assumed to be a cylinder. Its diameter is estimated by the taper functions halfway along the segment. The green mass is calculated from the segment volume and the green density of the wood. Total dry mass of branches and foliage as available in ForGEM are converted to their green mass using the respective ratios between green density and dry density. For branches, the same densities as for the stem are assumed. Branch and foliage mass are assumed to be distributed over the crown in the same way as the vertical area projection, analogous to Peltola et al. (1999).

The combined forces of wind and gravity can be resolved in a mean turning moment ( $T_{\text{mean}}(z)$  (Nm)) at any height in the tree, at stand edge:

$$T_{\text{mean}}(z) = F_w(z) * z + F_g(z) * x(z) + F_a(z) * x(z) \quad (11)$$

With  $x(z)$  horizontal stem displacement of the stem at height  $z$ .

The last term (Fa) represents an additional gravity load that could arise if a falling tree gets entangled in the subject tree. To estimate the maximum turning moment of each segment ( $T_{\max}(z)$ , Nm) at the actual location of the tree, we need to correct for the real upwind gap size, the sheltering and support by other trees and the gustiness of the wind:

$$T_{\max}(z) = T_{\text{mean}}(z) * \text{Gust}_{\text{factor}} * \text{Gap}_{\text{factor}} * (1-\text{Sup}) \quad (12)$$

with

$\text{Gust}_{\text{factor}}$  integrating the sheltering of the tree and the gustiness of the wind,  
 $\text{Gap}_{\text{factor}}$  reflecting the influence of the actual size of the upwind gap and  
 $\text{Sup}$  reflecting the support provided by other trees.

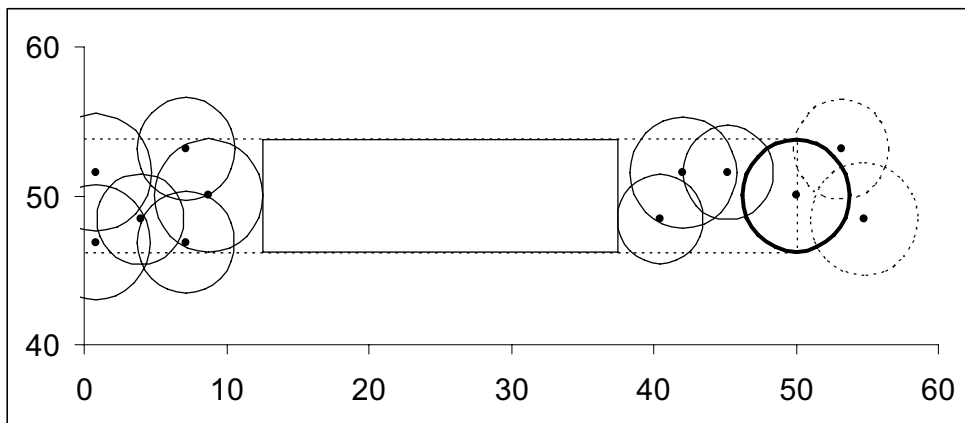
The maximum turning moment at stem base ( $T_{\max}$ ) can then be calculated as (Peltola et al., 1999):

$$T_{\max} = \sum_{z=0}^h T_{\max}(z) \quad (13)$$

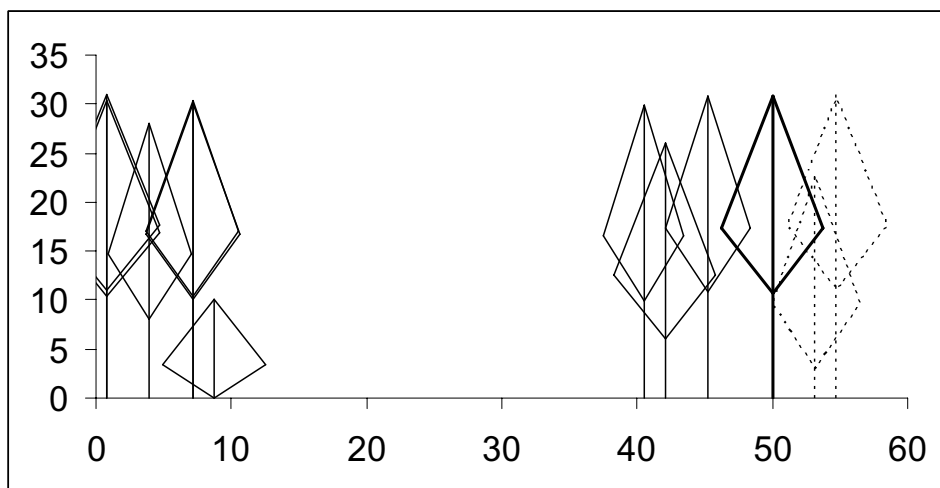
### *Upwind gap size*

The wind speed profile at the stand edge is calculated assuming an infinite upwind gap. To take into account the real gap size, the correction factor  $\text{Gap}_{\text{factor}}$  is used. The calculation of this factor is described in Peltola et al. (1999). To determine if any gaps are present upwind of the subject tree, we draw a transect from the upwind forest edge to the subject tree (dotted rectangle in Figure 2). The transect width equals the crown width of the subject tree.

According to wind tunnel tests by Gardiner et al. (2005), the presence of small trees in front of large ones did not affect the wind loading of the larger ones. We therefore exclude trees that are smaller than the crown base of the subject tree. Only trees with their stem base inside this transect are taken into account. If the distance between two neighbouring crowns exceeds half the tree height of the subject tree, we define this area as a gap (solid rectangle in Figure 2). If one or more gaps are present, the size of the first gap going upwind is used in the calculation of  $\text{Gap}_{\text{factor}}$ . Small trees present in this gap will have an effect on the zero plane displacement (d) and thus on the wind profile. However, to simplify the calculations we ignored this effect. If no gap is present inside the transect, the user defined gap size outside the stand is used.



**Figure 2.** Determining distance to gap (soild rectangle) or forest edge for the subject tree (thick circle). Wind direction is from the west (from the left). Only the three trees directly in front of the subject tree contribute to shelter. The dotted circles are two trees that support the subject tree.



**Figure 3.** Profile view of forest transect shown in Figure 2.

*Gustfactor*

The mean turning moment as calculated above applies only under static conditions. Actual forces on the tree can be much higher as a consequence of wind gustiness. The correction factor  $Gust_{factor}$  is introduced to take this into account (Peltola et al., 1999).  $Gust_{factor}$  is a function of the mean tree height, mean tree spacing, and distance from the forest edge. For its calculation, see Peltola et al. (1999). In the HWIND version, mean tree height and mean tree density were taken from the stand as a whole. In contrast, we consider only those trees that are between the subject tree and the



upwind edge (Figure 2), and that are taller than the crown base of the subject tree. Moreover, we use an adapted tree density ( $TD_a$ ), taking into account trees without foliage and height differences between trees:

$$TD_a = \frac{\sum_{i=1}^n SW_i}{d \times 2 \times CR} \quad (14)$$

with

$d$  the distance to the upwind edge (Figure 2),

$CR$  the crown radius of the subject tree,

$SW_i$  the relative shelter weight of tree  $i$  and

$N$  the number of trees contributing to the shelter of the subject tree.

$SW_i$  is calculated as:

$$SW_i = \frac{h_i}{h_s} \text{ if foliage is present on tree } i \quad (15)$$

$$SW_i = \frac{h_i}{h_s} \times 0.2 \text{ if tree } i \text{ is leafless} \quad (16)$$

with  $h_i$  the height of tree  $i$  and  $h_s$  the height of the subject tree.

In case a tree is leafless, the sheltering effect is assumed to be only 20% of that of a coniferous tree, analogous to the streamlining effect (Peltola et al., 1999). Furthermore, the sheltering effect is assumed to be linearly related to the ratio between the height of the sheltering tree and the subject tree. A tree in front of the subject tree of only half its height will thus contribute only 50% to the shelter effect, while a tree twice the height will contribute 200% to the shelter effect. A wind tunnel study by Gardiner et al. (2005) confirms that intermediate trees in irregular stands receive more shelter than trees of the same height in a regular stand.

*Support by other trees*

An important aspect of tree stability is the distance to neighbouring trees. Trees can be physically supported by neighbours (Quine et al., 1995) and roots can interlock (Savill, 1983; Smith et al., 1987). As an estimate of the physical support of neighbouring trees (Sup), we determine the overlap between the crown of the subject tree and the crowns of its downwind neighbours:

$$Sup_i = \frac{\sum_j CAO_{ij}}{CA_i} \tag{17}$$

with  $CAO_{ij}$  the crown area overlap between subject tree  $i$  and its downwind neighbour  $j$ , at the height of the crown centre. The maximum turning moment acting on the tree is decreased proportionally by the ratio of the crown that overlaps with other crowns (Equation 12).

*Resistance to uprooting and stem breakage*

The resistance to uprooting ( $ROOT_{RES}$  (Nm)) is included as a function of dbh:

$$ROOT_{RES} = a_1 \times DBH^{a_2} \tag{18}$$

With  $a_1$  and  $a_2$  parameters. The resistance to stem breakage ( $STEM_{res}$  (Nm)) is calculated from the dbh and the modulus of rupture ( $MOR$  (Pa)) of green wood (Peltola et al., 1999):

$$STEM_{RES} = \frac{\pi}{32} \times MOR \times DBH^3 \tag{19}$$

When the maximum turning moment exceeds the resistance to uprooting or the stem resistance to breakage, the tree is considered to be respectively uprooted or broken.

*Damage by falling trees*

Falling trees can hit others and will thus exert an additional force. We assume that the crown above the crown centre has small branches and is so flexible that it will not contribute to entanglement with its neighbours. A falling tree will only get stuck in another tree if the lower part of the crown or the stem of the falling tree hits the stem of the standing tree, below the crown centre. The falling tree is assumed to add a static load at the height where the trees get entangled. This load is added to the corresponding segment's weight and further treated as gravity load. The additional load ( $Fa$  (N)) is calculated according to its leverage in relation to the gravity point of the falling tree:

$$Fa = Fg \times \frac{h_g}{l} \quad (20)$$

with

$Fg$  is the total gravity load of the fallen tree,

$h_g$  is the length from stem base to the gravity centre of the fallen tree and

$l$  is the length from stem base of the fallen tree to the point where the trees are entangled.

Trees smaller than 5m are assumed to die if hit by the stem or lower part of the crown of a falling tree. No partial damage to crowns is incorporated. Trees that are broken, uprooted or killed by other trees are assumed to die at the end of the time step and are further treated in the model as lying deadwood.

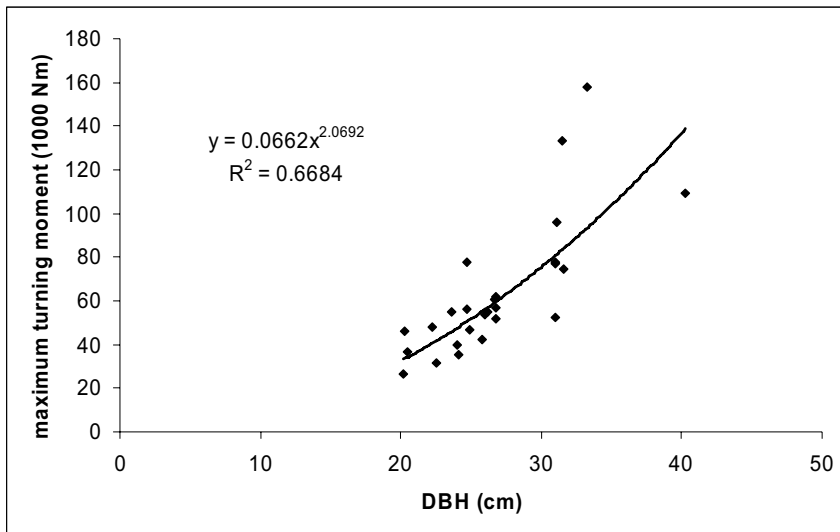
## Parameters and initial situation

### *Parameters*

In order to explore the behaviour of the wind damage module and to test its sensitivity, we parameterised and calibrated the model for Douglas fir in the Netherlands (Table 1). For the calibrations we used the growth and yield tables of Jansen et al. (1996), site class 16, normal density. Maximum crown radius and height of crown centre were estimated from the Dutch Forest Reserves Database (Bijlsma, Alterra, unpublished data), which contains measurements of 920 Douglas firs in the Netherlands. Anchorage parameters were estimated from tree pull tests by Schooten (1985) (Figure 4). Other parameters were taken from literature. Table 1 shows only those parameters that are relevant for the processes as described in this article. Other parameters are unchanged in comparison to Kramer et al. (2007).

**Table 1.** Parameters used in ForGEM-W.

Parameter	Value	Unit	Reference
Dry wood density	530	kg m <sup>-3</sup>	Wiselius, 1990
Green wood density	670	kg m <sup>-3</sup>	Datenbank Holzeigenschaften, 2005
Foliage green density	2*foliage dry density		assumption
MOR	31.96	MPa	Schooten, 1985
MOE	5500	MPa	Schooten, 1985
Cd (drag coefficient)	0.22	-	Mayhead, 1973
Height of crown centre, relative to crown length	0.33	-	Dutch Forest Reserves Database
CR <sub>MAX</sub>	7.7	m	Dutch Forest Reserves Database
RUE	3.07E-09	kg DM MJ <sup>-1</sup>	calibrated
T (AgeEffect)	35	year	calibrated
c (AgeEffect)	0.35	-	calibrated
a <sub>1</sub> (root resistance)	66.217	-	Estimated from Schooten 1985
a <sub>2</sub> (root resistance)	2.0692	-	Estimated from Schooten, 1985
b <sub>2</sub> (crown expansion)	0.1137	-	Estimated from Jansen et al., 1996
k (von Karman's constant)	0.41	-	Peltola et al., 1999
d (zero plane displacement)	0.0	m	Peltola et al., 1999
z <sub>0</sub> (roughness length)	6% of the mean tree height	m	Peltola et al., 1999
ρ (air density)	1.226	kg m <sup>-3</sup>	Peltola et al., 1999



**Figure 4.** Regression of maximum turning moment needed to uproot a tree against DBH (data from Schooten, 1985).

### *Test stands*

We simulated a 10-year-old forest stand (200 x 100m) without the wind damage module for 50 years under three different management regimes: (1) low density regime; (2) normal density regime; and (3) no management. The respective situations at age 60 (Table 2) are used to explore the behaviour of the wind damage module. They are further referred to as L60 (low density), N60 (normal density) and U60 (unmanaged). Initial conditions were taken from Jansen et al. (1996) for a stand with normal density (4000 stems/ha) at age 10, assuming a regular spacing. Management consisted of a thinning from below every 5 years to the density as specified in the respective yield table. The gap size in front of the forest was assumed to be 100m in all directions.

**Table 2.** Characteristics of simulated test stands, L60=low density, N60=normal density, U60=unmanaged. Numbers in brackets are standard deviations.

	L60	N60	U60
Age (year)	60	60	60
Density (N ha <sup>-1</sup> )	152	322	656
Basal area (m <sup>2</sup> ha <sup>-1</sup> )	34.6	40.8	42.2
Growing stock (m <sup>3</sup> ha <sup>-1</sup> )	434	541	550
Average dbh (cm)	53.7 (2.62)	40.1 (2.95)	26.2 (11.6)
Average height (m)	31 (0.31)	31 (0.37)	24.4 (8.77)
Dominant height (m)	31.2	31.3	31
Average crown radius (m)	4 (0.05)	3.8 (0.24)	3.5 (0.28)
Average crown virtual radius (m)	3.8 (0.14)	3.1 (0.37)	2.7 (0.44)
Average HD ratio (cm/cm)	58 (2.5)	78 (5.2)	98 (22)

## Testing of validity and performance

### *The wind damage module*

The three test stands were subjected to a western wind, with wind speeds ranging from 10 to 40 m s<sup>-1</sup>, with steps of 1 m s<sup>-1</sup>. Damage was expressed as the percentage of the initial standing volume that was damaged. Analyses were done separately per 100x100m block, to distinguish between exposed stands and sheltered stands. The damage pattern at wind speeds 25 and 32 m s<sup>-1</sup> was explored in more detail. A wind speed of 25 m s<sup>-1</sup> represents conditions where damage has been reported in the past. Additionally a wind speed of 32 m s<sup>-1</sup> was selected for analysis, because at this speed all stands had enough damage to perform a statistical analysis. Tree coordinates of damaged and undamaged were plotted. A range of tree characteristics was tested to explain the damage pattern. Student t-tests were used to test for significant differences between damaged and undamaged trees.

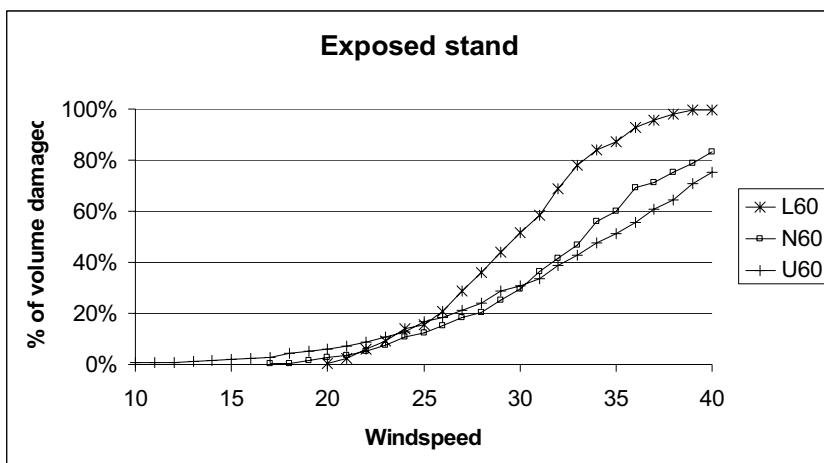
*Sensitivity*

We conducted a sensitivity analysis for the wind damage module, analogous to an analysis carried out for HWIND (Peltola et al., 1999). Important input variables and parameters were changed by plus or minus 20%. The effect on the percentage of volume that was damaged was assessed for all stands at wind speeds of 25 m s<sup>-1</sup> and 32 m s<sup>-1</sup>. Furthermore, we corrected simulated tree heights and dbh of stands L60 and N60 to match exactly the yield table values and assessed the effects. Additionally, we evaluated the effects of disabling the support mechanism.

**Results**

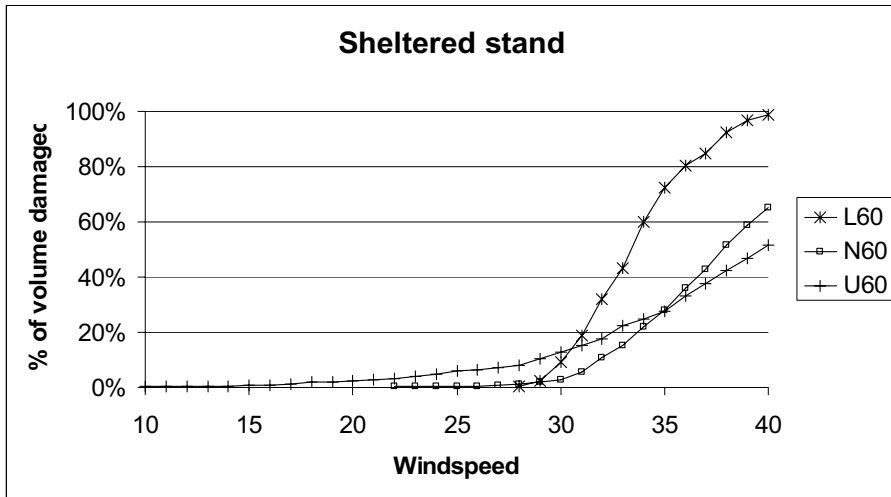
*The wind damage module*

The damage pattern for varying wind speed differs between the three stands in exposed conditions (Figure 5). Damage in the unmanaged stand (U60) starts already at 10 m s<sup>-1</sup> and increases gradually to 75% at a wind speed of 40 m s<sup>-1</sup>. In the normal density stand (N60), damage starts to occur at 17 m s<sup>-1</sup> and increases to 83% at 40 m s<sup>-1</sup>. The onset of damage in the low density stand (L60) is at 20 m s<sup>-1</sup>. The damage level increases quickly with wind speed and approaches 100% at 40 m s<sup>-1</sup>.



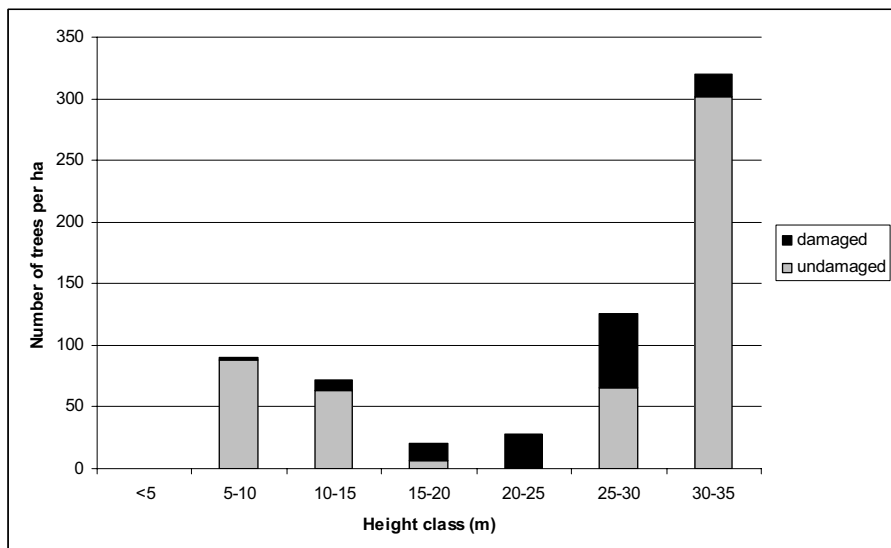
**Figure 5.** Percentage of standing volume damaged at different wind speeds for exposed stands (L=low density, N=normal density, U=unmanaged).

The pattern in sheltered stands shows similar behaviour (Figure 6). At low wind speeds, there is already some damage in U60, but it increases only gradually with wind speed. At 40 m s<sup>-1</sup>, the damage is 52%. The onset of damage is more or less the same for the managed stands, with about 2% damage at 29 m s<sup>-1</sup>. However, the slope for L60 is much higher than for N60. At 40 m s<sup>-1</sup>, N60 has 65% damage, but L60 has 99% damage.



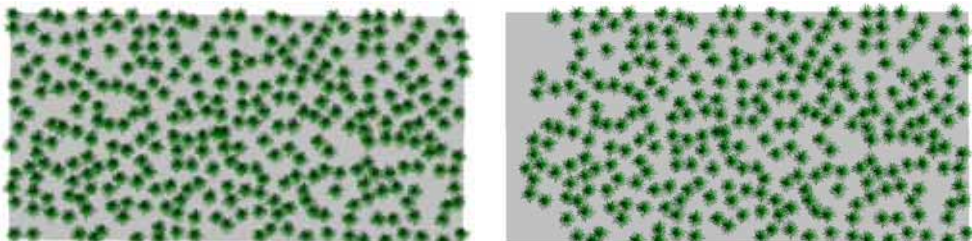
**Figure 6.** Percentage of standing volume damaged at different wind speeds for sheltered stands (L=low density, N=normal density, U=unmanaged).

In order to see what determines if trees are damaged or not, we compared some tree characteristics in all stands at wind speeds of 25 and 32  $\text{m s}^{-1}$ . The unmanaged stand had different height layers with clearly different damage patterns (Figure 7). Nearly all trees with a height between 15 and 25 m were damaged. In the managed stands no sub layers were present. For a better comparison we therefore only considered trees taller than 25 m in the unmanaged stand.

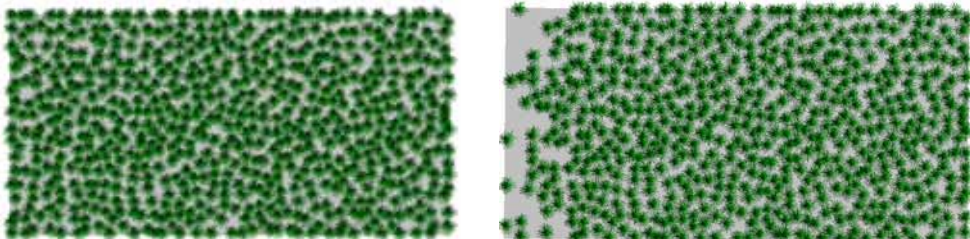


**Figure 7.** Number of trees damaged per height class in U60 after a storm with a wind speed of 25  $\text{m s}^{-1}$  (exposed and sheltered combined).

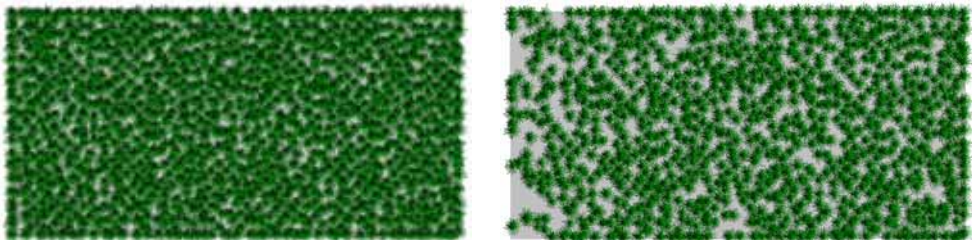
In the managed stands trees were only uprooted, while in the unmanaged stand both uprooting and stem breakage played a role. Since the difference between uprooting and stem breakage is solely determined by dbh (Equations 18 and 19), we compared damaged to undamaged trees. Table 3 shows characteristics of damaged and undamaged trees for the two selected wind speeds, under exposed and sheltered conditions. At  $25 \text{ m s}^{-1}$ , there was hardly any damage in sheltered managed stands. At  $32 \text{ m s}^{-1}$ , all stands showed considerable damage in both exposed and sheltered conditions. At both wind speeds, damaged trees in exposed stands are located significantly closer to the upwind edge than undamaged trees. This is also clearly visible in Figures 8-10. These figures show the damage distribution as overhead views for a wind speed of  $25 \text{ m s}^{-1}$ .



**Figure 8.** Overhead view of stand L60 before (left) and after occurrence of a western storm of  $25 \text{ m s}^{-1}$  (right). Visualised with Stand Visualisation Software (SVS, McGaughey, 1999).



**Figure 9.** Overhead view of stand N60 before (left) and after occurrence of a western storm of  $25 \text{ m s}^{-1}$  (right). Visualised with SVS (McGaughey, 1999).



**Figure 10.** Overhead view of stand U60 before (left) and after occurrence of a western storm of  $25 \text{ m s}^{-1}$  (right). Visualised with SVS (McGaughey, 1999).



In all stands, damaged trees had less support by other trees than undamaged trees (Table 3). Only for L60 in exposed and sheltered conditions at  $25 \text{ m s}^{-1}$  and N60 in sheltered condition at  $25 \text{ m s}^{-1}$  the difference was not significant. Damaged trees in the main canopy of sheltered stands all had smaller diameters and thus higher height/diameter (H/D) ratios than undamaged trees. This effect occurred at both wind speeds, but at  $25 \text{ m s}^{-1}$  in L60 and N60 this difference was not significant due to the low number of damaged trees. Differences between damaged and undamaged trees in the managed stands are small: about 2 cm in dbh and 2-4 in H/D ratio. The differences in the unmanaged stands are much more pronounced and are larger at  $25 \text{ m s}^{-1}$  than at  $32 \text{ m s}^{-1}$ . The difference is 10-13 cm in dbh and 29-40 in H/D ratio, despite the fact that damaged trees are on average 2-3 m lower. The same pattern between damaged and undamaged trees is visible in the unmanaged stand under exposed conditions. The differences in tree characteristics are less pronounced than in the sheltered situation, but are again larger at lower wind speed. The difference is 6-8 cm in dbh, 1-2 m in height and 17-24 in H/D ratio. Damaged trees shorter than 25 m (only in U60) are also characterised by a high H/D ratio, around 150 for trees between 15 and 25 m (not shown in table). In all managed stands at high wind speed, damaged trees have a significantly larger virtual crown radius. In L60 the difference between damaged and undamaged trees is about 10 cm and in N60 about 25 cm. At a wind speed of  $25 \text{ m s}^{-1}$ , only in N60 under exposed conditions a significantly larger virtual crown radius could be detected of about 20 cm.

Not many trees are damaged due to additional loading by other trees. In L60, 0% and 1.9% of the trees are loaded by other trees at  $25$  and  $32 \text{ m s}^{-1}$  respectively. For N60, the respective figures are 9.3% and 5.2% and for U60 6.9% and 6%. No trees smaller than 5 m were present, so no trees were killed by toppling neighbours.

**Table 3.** Characteristics of damaged and undamaged trees in all stands after storms with wind speeds of 25 and 32 m s<sup>-1</sup>. Bold numbers indicate statistically significant ( $\alpha < 0.01$ ) differences between damaged and undamaged groups. In the unmanaged stand (U60) results are only shown for trees larger than 25m to keep the results comparable to the other stands.

	exposed at 25 m s <sup>-1</sup>		sheltered at 25 m s <sup>-1</sup>		exposed at 32 m s <sup>-1</sup>		sheltered at 32 m s <sup>-1</sup>	
	damaged	undamaged	uprooted	undamaged	damaged	undamaged	damaged	undamaged
number	96	56	1	151	106	46	51	101
dbh	53.8	53.1	51.9	53.9	53.2	54.3	<b>52.4</b>	<b>54.6</b>
height	31.1	31.0	31.2	31.0	31.0	31.1	31.0	31.1
H/D ratio	57.8	58.6	60.1	57.7	58.4	57.4	<b>59.2</b>	<b>57.0</b>
edge distance	<b>43.2</b>	<b>60.8</b>	177.0	149.9	<b>40.9</b>	<b>70.0</b>	154.4	147.9
virtual crown radius	3.84	3.80	3.96	3.82	<b>3.86</b>	<b>3.75</b>	<b>3.88</b>	<b>3.79</b>
crown radius	3.97	3.97	3.96	3.98	3.97	3.97	3.97	3.98
support	0.06	0.09	0.00	0.07	<b>0.05</b>	<b>0.13</b>	<b>0.03</b>	<b>0.09</b>
number	41	281	2	319	136	186	38	283
dbh	39.7	40.1	36.6	40.2	39.8	40.2	<b>38.1</b>	<b>40.4</b>
height	31.0	31.0	30.7	31.0	31.0	31.0	30.8	31.0
H/D ratio	78.5	77.6	83.9	77.5	78.2	77.4	<b>81.4</b>	<b>77.0</b>
edge distance	<b>9.0</b>	<b>55.6</b>	135.7	149.9	<b>29.1</b>	<b>64.6</b>	149.2	149.9
virtual crown radius	<b>3.31</b>	<b>3.12</b>	3.02	3.15	<b>3.27</b>	<b>3.05</b>	<b>3.39</b>	<b>3.11</b>
crown radius	3.83	3.78	3.86	3.78	<b>3.84</b>	<b>3.74</b>	3.84	3.77
support	<b>0.22</b>	<b>0.32</b>	0.15	0.29	<b>0.24</b>	<b>0.36</b>	<b>0.16</b>	<b>0.31</b>

Continued Table 3

	exposed at 25 m s-1		sheltered at 25 m s-1		exposed at 32 m s-1		sheltered at 32 m s-1	
	damaged	undamaged	uprooted	undamaged	damaged	undamaged	damaged	undamaged
number	104	342	52	393	209	237	122	323
dbh	<b>26.7</b>	<b>35.0</b>	<b>21.5</b>	<b>34.7</b>	<b>29.7</b>	<b>36.0</b>	<b>25.7</b>	<b>36.0</b>
height	<b>28.7</b>	<b>30.5</b>	<b>27.4</b>	<b>30.4</b>	<b>29.4</b>	<b>30.6</b>	<b>28.6</b>	<b>30.6</b>
H/D ratio	<b>113.0</b>	<b>88.9</b>	<b>129.9</b>	<b>89.5</b>	<b>103.5</b>	<b>86.6</b>	<b>114.9</b>	<b>86.4</b>
edge distance	<b>31.2</b>	<b>54.7</b>	148.6	151.7	<b>34.9</b>	<b>61.8</b>	150.7	151.6
virtual crown radius	2.65	2.59	2.49	2.61	2.64	2.58	2.56	2.61
crown radius	3.48	3.51	<b>3.36</b>	<b>3.52</b>	3.50	3.52	<b>3.41</b>	<b>3.53</b>
support	<b>0.21</b>	<b>0.38</b>	<b>0.12</b>	<b>0.35</b>	<b>0.26</b>	<b>0.41</b>	<b>0.18</b>	<b>0.37</b>
U60 trees>25m								

*Sensitivity analysis*

The most sensitive parameters are those that determine the resistance to uprooting (Table 4). Especially parameter  $a_2$  has a decisive influence, but at higher wind speeds  $a_1$  also becomes more influential. The influence of the drag coefficient is very similar to that of  $a_1$ . Changing the drag coefficient linearly affects the turning moment, while changing  $a_1$  linearly affects the uprooting resistance. Among the tree characteristics, the wind damage module is very sensitive to tree height and dbh. At higher wind speeds, crown radius becomes more important as well. At higher wind speeds the wind damage module is also sensitive to turning off the support mechanism. In general, the sensitivity to parameters and variables clearly changes with wind speed. Furthermore, the sensitivity seems to decrease with increasing stand density, except for those parameters and variables that affect the support mechanism. Average tree height was underestimated by ForGEM in L60 by 4.9% and in N60 by 1.9%. At the same time, average dbh was overestimated in L60 by 6.8% and in N60 by 1.3%. When we corrected the values in the initial stands accordingly, damage increased in L60 from 7.7% to 21.1% at a wind speed of 25 m s<sup>-1</sup>, and from 50.3% to 88% at a wind speed of 32 m s<sup>-1</sup>. In N60, damage increased from 6.4% to 8.1% at a wind speed of 25 m s<sup>-1</sup>, and from 26.3% to 34% at a wind speed of 32 m s<sup>-1</sup>.

**Table 4.** Difference between lowest and highest volume percentage damaged when varying parameters and variables by  $\pm 20\%$  at wind speed 25 and 32 m s<sup>-1</sup>.

Wind speed (m s <sup>-1</sup> )	25			32		
	L60	N60	U60	L60	N60	U60
MOR	0%	0%	3%	0%	0%	3%
MOE	0%	0%	2%	2%	3%	4%
Cd	12%	7%	8%	59%	32%	21%
$a_1$	14%	7%	6%	60%	37%	22%
$a_2$	100%	98%	87%	95%	91%	79%
Height of crown centre	2%	1%	2%	15%	7%	6%
$z_0$	3%	2%	3%	26%	11%	6%
Gap in front	1%	1%	2%	12%	6%	4%
Crown length	0%	1%	2%	5%	5%	2%
Crown radius	4%	5%	5%	26%	16%	17%
Tree height	37%	27%	26%	93%	77%	57%
Dbh	48%	29%	31%	87%	75%	62%
without support	3%	9%	11%	10%	41%	34%

## Discussion

### *General model performance*

The H/D ratio is generally seen as an important variable in predicting or explaining wind damage. It essentially expresses the stability of individual trees and is mainly influenced by competition during the life of a tree. The average H/D ratio clearly increases from L60 to U60 (Table 2). Also in our results this ratio proved to be important in explaining the damage pattern. In sheltered stands, regardless of management, the average H/D ratio of damaged trees was higher than the H/D ratio of undamaged trees (Table 3). However, there seems to be no absolute limit where trees are safe. At  $32 \text{ m s}^{-1}$ , the average ratio of undamaged trees in U60 was 86, whereas the ratio of damaged trees in N60 was 81. Similarly, the ratio of undamaged trees in N60 of 77 is much higher than the 59 of damaged trees in L60. So trees that survived in one stand would have been damaged in another stand. An important difference between the stands is the tree density. A higher density results in more shelter and more support. The sensitivity to the inclusion of the support mechanism increases clearly from L60 to U60 (Table 4) and the support values found for damaged and undamaged trees are higher in N60 and U60 than L60 (Table 3). Another factor in the susceptibility of a tree is the size of the crown and thus the sail area of the tree. This becomes more important at higher wind speeds. At  $32 \text{ m s}^{-1}$  the virtual crown radius was significantly larger for damaged trees in all managed stands. However, differences were not very large.

In the management gradient L60-N60-U60, individual tree stability (H/D ratio) decreases and stand stability (support) increases. With lower individual tree stability, a lower wind speed is needed to initiate damage (Figure 5). However, more support and variability in tree sizes lead to a much slower increase of damage with wind speed. A similar pattern has been described by Ancelin et al. (2004).

Within the sheltered stands, trees with a relatively high H/D ratio and large virtual crown radius (only in managed stands and at higher wind speed) were most vulnerable. In the exposed stands, these principles were mostly overruled by the exposure factor: distance to edge was significantly smaller for damaged trees in all exposed stands. Crown size was still significant in managed conditions (except L60 at  $25 \text{ m s}^{-1}$ ), but only in U60 could a difference in H/D ratio still be proven. This is probably because of the very large difference between damaged and undamaged trees under unmanaged conditions.

Only in U60 more than one tree layer was present. Nearly all trees between about 15 and 25m height were damaged, at both wind speeds. These trees were characterised by a very high H/D ratio, about 150 on average. The allocation mechanism of ForGEM favours height growth over dbh growth. Apparently these trees received enough light for height growth but not enough for dbh growth. In reality these trees would have already collapsed earlier, since wind speeds required to

damage them are very low and occur annually. The group of trees lower than 15m received even less light and did not become very tall. Consequently, their H/D ratio is lower and many even survived the wind speed of 32 m s<sup>-1</sup>.

Our findings agree well with the recommendations given by the Forestry Commission in the UK (Quine et al., 1995). Thinnings should be light, so as not to open up the canopy too much. However, in high risk areas they recommend a strategy of planting at low density and no thinnings. We only tested a high density unmanaged stand. The effect of planting at lower density would be less competition, so individual trees would be more stable. A higher initial wind speed would therefore be required to initiate damage than our unmanaged stand. Support would still be very high, leading to a only a small increase in damage with increasing wind speed.

All exposed stands show that damage is concentrated on the upwind edge. If this edge had been present during the whole life of the stand, trees would have been subjected to wind loading earlier. Trees can react to mechanical stress by changing their allocation pattern (Stokes, 1999). This can lead to increased diameters (Telewski and Jaffe, 1981; Telewski, 1995) and more stable root systems (Cucchi et al., 2004). Cucchi et al. (2004) for example showed that the turning moment needed to uproot edge trees was 20% higher than for inner trees, after correcting for stem mass. Such mechanisms are not present in ForGEM, and thus we should regard the simulations as being done on newly created stand edges. Such a mechanism could be easily included by changing the anchorage parameters when trees experience wind loading. However, parameterisation will require dose-response curves which are currently not available (Gardiner et al., 2005).

There are a few storms in Dutch history that are known to have caused relatively large damage to the forest. These storms were in 1972, 1973, 1976 and 1990 (Schelhaas et al., 2001). Douglas fir is explicitly mentioned to have experienced damage in all storms in the 1970s. Maximum mean hourly wind speeds from weather stations in the most affected regions ranged from 25 to 28 m s<sup>-1</sup> (Royal Netherlands Meteorological Institute, 2006). Gust speeds to over 40 m s<sup>-1</sup> were measured. According to our findings, such wind speeds would cause considerable damage to freshly exposed edges, and only minor damage in sheltered stands. Indeed, damage has often been reported from such edges, but whole stands have also been thrown (Sissingh, 1975). The latter might be caused by differences in soil conditions and actual stand management. Anyhow, a common difficulty when comparing real wind speeds causing damage to modelled wind speeds is that wind speeds are rarely measured at the exact location of damage (Ancelin et al., 2004; Cucchi et al., 2005). Wind can be very variable, even at relatively short distances. Uncertainty in actual wind speed, combined with a lack of detailed inventory data, currently prohibits a validation of the observed spatial damage pattern.

*Sensitivity to parameters and input variables*

The wind throw module was very sensitive to some parameters and tree characteristics. The parameters and characteristics can be divided into two groups: (1) those affecting the wind force acting on a tree (drag coefficient, tree height); and (2) those that help to resist damage (anchorage parameters and dbh). The results of the sensitivity analysis are comparable to the conclusions by Peltola et al. (1999) concerning the HWIND model. Modulus of rupture (MOR) is less important under our circumstances, since the majority of trees passed the dbh where stem breakage is the limiting factor (26cm).

The wind force acting on a tree is determined by wind speed, frontal area of the tree and the drag coefficient. These three factors are interrelated. At higher wind speeds, the frontal area of a tree will decline through streamlining (Nobel, 1981; Vollsinger et al. 2005), and consequently the drag coefficient decreases with increasing wind speed (Mayhead, 1973; Rudnicki et al., 2004). In wind throw modelling, one factor is usually assumed to be constant (here drag coefficient), while the other (crown frontal area) is assumed to vary with wind speed (through streamlining). However, Gaffrey and Kniemeyer (2002) showed that decreasing sail area and keeping drag force constant or keeping sail area constant and decreasing drag force did not lead to the same results. Moreover, streamlining is shown to be species-specific (Rudnicki et al., 2004; Vollsinger et al., 2005); in this study we simply assumed that Douglas fir would respond similarly to other species. Further, Rudnicki et al. (2004) concluded that incorrect estimation of drag could be a significant source of error in mechanistic wind throw models, due to the often small underlying sample sizes.

The high sensitivity to the anchorage parameters shows a need to test the model with more experimental data. The sample size on which our model is based was rather small and we had to extrapolate in some cases. Further, the small sample size meant that our model can only be applied to soil conditions similar to those where the tree pull tests were carried out. Another method of determining root support is by taking into account the root system characteristics explicitly. HWIND for example includes a simple estimation of root system width and depth and the contribution of the root soil plate to the total anchorage. However, we found this approach led to unacceptable large sensitivities to parameters (such as ratio of root system width to crown radius) and tree characteristics (such as crown radius) that were difficult to determine or to simulate. Also with such an approach additional experimental data would be needed. Furthermore, independently modelling the rooting system will be a major challenge, since rooting systems are highly flexible and root research is complicated.

The sensitivity of the wind damage module makes an accurate simulation of tree characteristics by ForGEM crucial. ForGEM was able to simulate the most important variables, tree height and dbh, within 5% of the values in the yield tables for low and normal density at age 60. However, even such small differences can have rather large effects. Continued attention to processes influencing tree height and dbh

is therefore needed. Especially the rather crude mechanism to reduce dbh increment at higher ages should be improved. The sensitivity to tree characteristics proves the strength of the combination of an individual tree model with a wind damage model. Since individual tree models are able to simulate a wide range of stand conditions and management practices, analysis of wind damage is not confined to yield table situations.

#### *Other improvements*

The wind damage module as presented here can still be refined in many ways. Wind may only damage parts of the crown and wind snap can occur at other heights than just breast height. Root rot is also known to be an important factor affecting susceptibility to wind damage (Schelhaas et al., 2003). Furthermore, various relationships with the weather exist. Wet snow can add a considerable weight, up to twice the weight of the canopy itself (Quine et al., 1995). This is especially important in northern European countries and is included in the original HWIND model (i.e. Peltola et al., 1999). Waterlogged soils cause trees to uproot more easily (Schelhaas et al., 2003), while a frozen soil effectively prevents uprooting (Peltola et al., 2000a). Temperatures below zero also have a large influence on wood characteristics (Peltola et al., 2000a; Silins et al., 2000). Silins et al. (2000) found values for MOR and MOE to be about 50% higher in winter. Another important aspect is the duration of a storm. Currently, the model simulates a storm as a short event where trees can become damaged or not. In reality, storms can last for hours or even days, causing the wind to penetrate further into the stand as trees start to collapse. Furthermore, wind speed is very much influenced by the surrounding landscape. If a certain real stand or forest situation is to be simulated, this must be taken into account. In studies with HWIND at a larger scale, such effects were included via the use of airflow models (Talkkari et al., 2000; Blennow and Sallnäs, 2004).

## **Acknowledgement**

Part of the work for this article has been financed by the EU projects CAMELS (EVK2-CT-2002-00151) and EFORWOOD (contract nr FP6-518128-2). It has also benefited from a grant from the Dutch Ministry of Agriculture, Nature management and Food quality (DWK Programme 381: Zelfregulatie 11358). We would like to thank the Forest Reserves Program (Dr. Rienk-Jan Bijlsma) for using their data. Further we would like to thank Dr. Gert-Jan Nabuurs and Prof. Timo Pukkala for their helpful comments on the manuscript. We thank Tim Green for correcting the English language.



## References

- Ancelin, P., Courbaud, B., Fourcaud, T., 2004. Development of an individual tree-based mechanical model to predict wind damage within forest stands. *Forest Ecology and Management* 203:101-121.
- Binkley, D., Stape, J.L., Ryan, M.G., Barnard, H.R., Fownes, J., 2002. Age-related decline in forest ecosystem growth: An individual-tree, stand-structure hypothesis. *Ecosystems* 5:58-67.
- Blennow, K., Sallnäs, O., 2002. Risk perception among non-industrial private forest owners. *Scandinavian Journal of Forest Research* 17:472-479.
- Blennow, K., Sallnäs, O., 2004. WINDA - a system of models for assessing the probability of wind damage to forest stands within a landscape. *Ecological Modelling* 175:87-99.
- Bugmann, H., 2001. A review of forest gap models. *Climatic Change* 51:259-305.
- Cucchi, V., Meredieu, C., Stokes, A., Berthier, S., Bert, D., Najar, M., Denis, A., Lastennet, R., 2004. Root anchorage of inner and edge trees in stands of Maritime pine (*Pinus pinaster* Ait.) growing in different podzolic soil conditions. *Trees – Structure and Function*, 18:460-466.
- Cucchi, V., Meredieu, C., Stokes, A., Coligny, F. de, Suarez, J., Gardiner, B.A., 2005. Modelling the windthrow risk for simulated forest stands of Maritime pine (*Pinus pinaster* Ait.). *Forest Ecology and Management* 213:184-196.
- Datenbank Holzeigenschaften, 2005. [www.holzdatenbank.de](http://www.holzdatenbank.de) Technische Universität Dresden, Interessengemeinschaft Holz. Last visited 9 May 2005.
- Faber, P., 1975. Stabiliteit van bos ten opzichte van wind. Een theoretisch gezichtspunt. *Nederlands Bosbouw Tijdschrift* 47:179-187.
- Gaffrey, D., Kniemeyer, O., 2002. The elasto-mechanical behaviour of Douglas fir, its sensitivity to tree-specific properties, wind and snow loads, and implications for stability – a simulation study. *Journal of Forest Science* 48:49–69.
- Gardiner, B.A., Quine, C.P., 2000. Management of forests to reduce the risk of abiotic damage - a review with particular reference to the effects of strong winds. *Forest Ecology and Management* 135:261-277.
- Gardiner, B., Peltola, H., Kellomäki, S., 2000. Comparison of two models for predicting the critical wind speeds required to damage coniferous trees. *Ecological Modelling* 129:1-23.
- Gardiner, B., Marshall, B., Achim, A., Belcher, R., Wood, C., 2005. The stability of different silvicultural systems: a wind-tunnel investigation. *Forestry* 78:471-484.
- Holtam, B.W. (1971). Windblow of Scottish forests in January 1968. Forestry Commission Bulletin 45. HMSO, London.
- Jansen, J.J., Sevenster, J., Faber, P.J. (Eds.), 1996. Opbrengsttabellen voor belangrijke boomsoorten in Nederland. IBN Rapport 221, Hinkeloord Report No 17. 240 pp.
- Kobe, R.K., Pacala, S.W., Silander, J.A Jr., Canham, C.D., 1995. Juvenile tree survivorship as a component of shade tolerance. *Ecological Applications* 5:517-532.

- Kramer, K. (Ed.), 2004. Effects of silvicultural regimes on dynamics of genetic and ecological diversity of European beech forests; impact assessment and recommendations for sustainable forestry. Alterra, Wageningen.
- Kramer, K., Buiteveld, J., Forstreuter, M., Geburek, T., Leonardi, S., Menozzi, P., Povillon, F., Schelhaas, M.J., Teissier du Cros, E., Vendramin, G.G., van der Werf, D.C., 2007. Bridging ecophysiological and genetic knowledge to assess the adaptive potential of European beech. Submitted to Ecological Monographs.
- Landsberg, J.J., Waring, R.H., 1997. A generalised model of forest productivity using simplified concepts of radiation-use efficiency, carbon balance and partitioning. *Forest Ecology and Management* 95:209-228.
- Mayhead, G.J., 1973. Some drag coefficients for British forest trees derived from wind tunnel studies. *Agricultural Meteorology* 12:123-130.
- McGaughey, R.J., 1999: SVS - Stand Visualization System. A product of the USDA Forest Service, Pacific Northwest Research Station.  
<http://faculty.washington.edu/mcgoy/svs.html>
- Mencuccini, M., Grace, J., Fioravanti, M., 1997. Biomechanical and hydraulic determinants of tree structure in Scots pine: anatomical characteristics. *Tree Physiology* 17:105-113.
- Milne, R., 1991. Dynamics of swaying of *Picea sitchensis*. *Tree Physiology* 9:383-399.
- Monteith, J.H., 1975. Principles of environmental physics. Edward Arnold, London.
- Nobel, P.S., 1981. Wind as an ecological factor. In: O.L. Lange, P.S. Nobel, C.B Osmund, H. Ziegler (Editors), Responses to the physical environment. Springer-Verlag GmbH & Co, Berlin, pp. 475–500.
- Pacala, S.W., Canham, C.D., Silander, J.A. Jr., 1993. Forest models defined by field measurements: 1. The design of a northeastern forest simulator. *Canadian Journal of Forest Research* 23: 1980-1988.
- Pacala, S.W., Canham, C.D., Saponara, J., J Silander, J.A. Jr., Kobe, R.K., Ribbens, E., 1996. Forest models defined by field measurements: II. Estimation, error analysis and dynamics. *Ecological Monographs* 66:1-43.
- Peltola, H., Kellomäki, S., 1993. A mechanistic model for calculating windthrow and stem breakage of Scots pines at stand edge. *Silva Fennica* 27:99-111.
- Peltola, H., Kellomaki, S., Vaisanen, H., Ikonen, V.-P., 1999. A mechanistic model for assessing the risk of wind and snow damage to single trees and stands of Scots pine, Norway spruce, and birch. *Canadian Journal for Forest Research* 29:647-661.
- Peltola, H., Kellomäki, S., Hassinen, A., Granander, M., 2000a. Mechanical stability of Scots pine, Norway spruce and birch: an analysis of tree-pulling experiments in Finland. *Forest Ecology and Management* 135:143-153.
- Peltola, H., Gardiner, B., Kellomäki, S., Kolstrom., T, Lässig, R., Moore, J., Quine, C., Ruel, J., 2000b. Introduction: Wind and other abiotic risks to forests. *Forest Ecology and Management* 135:1-2.
- Pennala, E., 1980. Lujuusopin perusteet, No 407. Otakustantamo, Espoo, Finland.
- Quine C., Coutts, M., Gardiner, B., Pyatt, G., 1995. Forests and wind: Management to minimize damage, Forestry Commission Bulletin 114. HMSO, London.

- Richards, F.J., 1959. A flexible growth function for empirical use. *Journal of Experimental Botany* 10:290-300
- Royal Netherlands Meteorological Institute, 2006.  
<http://www.knmi.nl/samenw/hydra/> Last visited 1 August 2006.
- Rudnicki, M., Mitchell, S.J., Novak, M.D., 2004. Wind tunnel measurements of crown streamlining and drag relationships for three conifer species. *Canadian Journal of Forest Research* 34:666-676.
- Ruel, J-C. 1995. Understanding windthrow: silvicultural implications. *Forestry Chronicles* 71:435-445.
- Savill, P.S., 1983. Silviculture in windy climates. *Forestry Abstracts* 44:473-488.
- Schelhaas, M.J., Varis, S., Schuck, A., 2001. Database on Forest Disturbances in Europe (DFDE). European Forest Institute, Joensuu, Finland.
- Schelhaas, M.J., Nabuurs, G.J., Schuck, A., 2003. Natural disturbances in the European forests in the 19th and 20th centuries. *Global Change Biology* 9:1620-1633.
- Schooten, J.P.v., 1985. Fysische aspecten van windworp bij de douglasspar, Vakgroep Bosteelt. Landbouwhogeschool, Wageningen, 200 pp.
- Silins, U., Lieffers, V.J., Bach, L., 2000. The effect of temperature on mechanical properties of standing lodgepole pine trees. *Trees - Structure and Function* 14:424-428.
- Sissingh, G., 1975. Stabiliteit van bos ten opzichte van wind en storm gezien vanuit de praktijk. *Nederlands Bosbouw Tijdschrift* 47:188-193.
- Smith, V.G., Watts, M., James, D.F., 1987. Mechanical stability of black spruce in the Clay Belt region of northern Ontario. *Canadian Journal of Forest Research* 17:1080-1091.
- Stokes, A., 1999. Strain distribution during anchorage failure of *Pinus pinaster* Ait. at different ages and tree growth response to wind-induced root movement. *Plant and Soil* 217:17-27.
- Suárez, J.C., Gardiner, B.A., Quine, C.P., 2002. Windthrow risk management in Scottish forests. A computer-based model to calculate wind risk for forestry. *Metsätieteen päivä*, Metla, Helsinki.  
<http://www.metla.fi/org/sms/paiva/esit/02-suarez.htm>
- Talkkari, A., Peltola, H., Kellomäki, S., Strandman, H., 2000. Integration of component models from the tree, stand and regional levels to assess the risk of wind damage at forest margins. *Forest Ecology and Management* 135:303-313.
- Telewski, F.W., 1995. Wind-induced physiological and developmental responses in trees. In M.P. Coutts and J. Grace (Editors), 1995. *Wind and Trees*. Cambridge University Press, 237-263.
- Telewski, F.W., Jaffe, M.J., 1981. Thigmomorphogenesis: changes in the morphology and chemical composition induced by mechanical perturbation of 6-month-old *Pinus taeda* seedlings. *Canadian Journal of Forest Research* 11:380-387.
- Vollsinger, S., Mitchell, S.J., Byrne, K.E., Novak, M.D., Rudnicki, M., 2005. Wind tunnel measurements of crown streamlining and drag relationships for several hardwood species. *Canadian Journal of Forest Research* 35:1238-1249.
- Wiselius, S.I., 1990. *Houtvademecum*, 6th edition. Kluwer, Deventer. 375 pp.

## V The wind stability of different silvicultural systems for Douglas fir in the Netherlands: a model based approach

Mart-Jan Schelhaas, Centre for Ecosystem Studies, Alterra, Wageningen UR

### Summary

The aim was to evaluate different silvicultural systems for Douglas fir (*Pseudotsuga menziesii* (Mirb.) Franco) in the Netherlands in terms of timber production and wind stability over a full rotation. This was done using the ForGEM-W model, which combines a distant dependent tree growth simulator with a mechanical-empirical wind damage module. Six different silvicultural systems were evaluated: normal yield table management, free thinning from above in a mono-species and a mixed stand (50% mixture of beech (*Fagus sylvatica* L.)), uneven-aged system, no thinning regime with low initial stand density of Douglas fir with and without admixture of beech. Silvicultural systems leading to low height-diameter (h/d) ratios were most successful in avoiding damage. Low h/d ratios were obtained in the system with low stand density and no thinning, and in the uneven-aged system by systematically removing trees with the highest ratios during thinning. Especially the uneven-aged system combined a high timber production with low risk. The use of Douglas-fir-beech mixtures changed the competition pressure on Douglas fir, and thus the h/d ratio and the wind risk. Results from this study indicate that the current trend towards more nature-oriented management could lead to lower wind risks and even to an increase in overall productivity.

### Introduction

Wind damage is an important risk factor in forest management, especially in the temperate and boreal zone. In North-western Europe, the importance of wind is illustrated by a series of severe wind events during the last decades, interspersed with numerous smaller events (Schelhaas et al., 2003). In addition to wind speed, the vulnerability of trees and stands to wind damage are determined by many different factors such as tree and stand characteristics. The height and crown size of a tree largely determine how much force the wind exerts on the tree: the larger the tree and the crown, the higher the wind drag will be. Wind force will cause the stem to bend and exerts a force on the root-soil system. The resistance to stem breakage can be determined by the strength properties of the wood and by the diameter of the stem (Peltola et al., 1999). Correspondingly, the resistance to overturning depends on the

shape and size of the root system, the soil type and on soil conditions (e.g. Coutts 1986). Stem diameter or stem mass has been found to correlate well with root mass and anchorage (Nicoll et al., 2006; Schelhaas et al., 2007).

Trees having high height to stem diameter ratio at breast height (h/d ratio) for a fixed tree height require less wind speed to be damaged (Peltola et al., 1999; Wilson and Oliver, 2000; Cameron, 2002). With increasing height, h/d ratios should be kept lower to maintain the same risk level (Cremer et al., 1982; Becquey and Riou-Nivert, 1987; Ruel, 1995). However, forest stands with relatively high h/d values can still be stable if stand density is high enough, because of mutual support and sheltering amongst the individual trees (Schelhaas et al., 2007).

Management decisions and silvicultural treatments influence the state of the forest and thus its susceptibility to windthrow. For example, clear fellings suddenly expose adjacent stands to the wind (Zeng, 2006). Thinnings are known to reduce the stand stability for several years (Cremer et al., 1982; Savill, 1983; Ruel, 1995; Cameron, 2002), but can lead in the longer term to lower average h/d ratios. In highly exposed sites it is recommended to use a relatively sparse initial stand density and not to thin (Quine et al., 1995; Ruel, 1995). As low initial stand densities will generally negatively affect the wood quality (Deans and Milne, 1999; Fahlvik et al., 2005), the use of a self-thinning mixture is proposed as alternative (Quine et al., 1995; Ruel, 1995).

Tree species choice is also an important management decision with regard to vulnerability to wind damage. Species differ in wind vulnerability among others due to differences in wood properties, rooting strategy and crown shape. Broadleaves are generally considered windfirm (Holmsgaard, 1986; Quine et al., 1995) whereas among the conifers especially Norway spruce (*Picea abies* (L.) Karst.) and Sitka spruce (*Picea sitchensis* (Bong.) Carr.) are seen as unstable species (Slodicák, 1995). Several authors (Eriksson, 1986; Slodicák, 1995; Schütz et al., 2006) advocated that introducing more windfirm species into a stand of less windfirm species would yield a higher stability of the total stand (see Lüpke and Spellmann (1999) for a literature overview), although others suggest that this will result only in the loss of the unstable component (Lüpke and Spellmann, 1997) or that the combination could become even more unstable (Savill, 1983). As stabilising species, usually broadleaves are recommended (Eriksson, 1986).

Despite a vast body of literature on the relation between vulnerability of trees to wind, forest characteristics and management, forest managers still express the need for more support. Blennow and Sallnäs (2002) showed that forest owners in Southern Sweden rank the risk of wind damage highly, but they generally do not know how to change their forest management to reduce the risk of damage. Mathematical models provide an opportunity for objectively calculating these risks and can show forest managers the long-term implications of their actions (Gardiner and Quine, 2000).

In recent years, substantial progress has been made in the modelling of windthrow. Peltola et al. (1999) and Gardiner et al. (2000) have developed mechanistic models to estimate, for a given tree, the critical wind speed needed for breakage or uprooting. In these models, the stand under consideration is assumed to be even-aged

and comprised of a single species. Furthermore, it is assumed to consist of an evenly-spaced array of trees with the same height and diameter (the so-called “average tree”). Ancelin et al. (2004) extended this approach and evaluated all trees in a particular stand. Schelhaas et al. (2007) introduced a similar mechanistic module into a single tree model and improved shelter and support mechanisms between trees in Douglas fir (*Pseudotsuga menziesii* (Mirb.) Franco). This integrated model, ForGEM-W, is able to simulate a wide variety of silvicultural practices under a specified wind climate.

One of the countries in North-western Europe that is affected regularly by storms is the Netherlands. Major wind damage to the forest occurred in 1972/73 and 1990 (Schelhaas et al., 2001), with wind-felled timber between 0.6 and 1.5 million m<sup>3</sup>. Lesser damages occurred in 1976 and 2007 with respectively 0.15 and 0.25 million m<sup>3</sup> (Schelhaas et al., 2001). Douglas fir was one of the species reported to be vulnerable in these and other events (Van Soest, 1954; Faber, 1975; Sissingh, 1975).

Although the share of Douglas fir in the total forest area is only about 6% (Dirkse et al., 2007), it is considered an important species due to its high productivity and good wood quality. Moreover, it regenerates well naturally and is expected to increase its share in future. Traditionally it was managed in monocultures with thinning from below. Nowadays the future tree method and thinning from above are common ways of managing Douglas fir. In the future tree method, about 60-100 trees are selected that represent the most valuable trees and that are expected to form the final crop. Selection usually takes place at an age of about 40. During the further life of the stand, these trees are consistently favoured in thinnings. As Douglas fir is a rather shade-tolerant species, it can also be grown in uneven-aged and selection systems. In regeneration groups it can be mixed with fast growing species as larch (*Larix* sp.) and birch (*Betula* sp.), but eventually these species will be overgrown by the Douglas fir. Similarly, Douglas fir is known to out compete also beech (*Fagus sylvatica* L.) in individual mixtures, and therefore, Hekhuis and van Nierop (1988) recommended mixing both species in strips of 10-15 m or in groups. Despite the importance of Douglas fir and its reported wind vulnerability, not much is known yet about the consequences of different silvicultural systems on its vulnerability to wind damage, for example, in the Netherlands.

The aim of this study was to evaluate the effectiveness of a set of silvicultural systems with regard to timber production and wind stability by means of ForGEM-W simulations. Silvicultural systems were selected to represent the whole range of possible systems that would likely be applied to Douglas-fir stands in the Netherlands. They were evaluated over a full rotation period to enable a good comparison. The focus was on sheltered stands. However, although trees are known to adapt to more exposed conditions in permanent stand edges (Cucchi et al., 2004; Gardiner et al., 2005), underlying mechanisms are still insufficiently understood and not included in the ForGEM-W model.

## Methods

### Outline of ForGEM-W

#### *ForGEM*

ForGEM (Forest Genetics, Ecology and Management) is a physiologically-based model that simulates the growth and development of individual trees, on a scale up to several hectares (Kramer, 2004). Several processes can be simulated with different levels of detail according to the needs of the user. In the current application, the simplest simulation methods were used. This means simulation of light interception based on the gap-type approach and no simulation of water and nitrogen balances. Here only the basics of the model are explained, a more detailed description can be found in Kramer (2004) and Kramer et al. (2007). Most approaches are essentially similar to those used in SORTIE (Pacala et al., 1993; Pacala et al. 1996), except in this case the gap-type approach for light interception.

ForGEM simulates individual trees that have known coordinates. Tree growth is driven by light interception. In the gap-type approach, light is assumed to come straight from above. The light interception in each 20x20 m grid cell is divided amongst the trees in the cell, according to their foliage mass and its vertical distribution (see Bugmann, 2001). Intercepted radiation is converted into photosynthates via the radiation use efficiency (RUE) parameter (Landsberg and Waring, 1997). The RUE must be calibrated against independent data, such as growth and yield tables.

Photosynthates are allocated in such a way that specific ratios between different tree components (stem mass, branches, root mass, etcetera) are maintained (see Kramer et al., 2007). These ratios depend on the total aboveground biomass. Photosynthates allocated to the stem are converted into height and stem diameter increment. Maximum height increment follows a Richards' growth curve (Richards, 1959; Jansen et al., 1996). Actual height increment can be lower due to insufficient resources. The relationship between tree volume, height and stem diameter at breast height (DBH) is described by species-specific allometric functions, derived from yield table data (Jansen et al., 1996). Crown expansion is influenced by overlap with other tree crowns (Schelhaas et al., 2007).

Competition-induced mortality is assumed to occur if the crown of a tree is completely suppressed by those of its neighbours (see Schelhaas et al. (2007) for details). Age-related mortality is assumed to follow a 2-parameter Weibull distribution. Mortality is calculated at the end of each month. Dead trees are removed from the tree list and are assumed to be left in the forest. Regeneration is explicitly simulated by seed production, dispersal and germination. Seedlings are treated as cohorts in 5x5 m grid cells. Self thinning is simulated in these cohorts using the  $-3/2$  power law (e.g.

Reineke, 1933; Drew and Flewelling, 1979). Upon reaching 2 m height they are further treated as individuals and receive x,y-coordinates. Several processes have a stochastic character (mortality, weather), so simulations should be repeated several times.

A wide range of management options is available in ForGEM. These include tree planting, tending, thinning, cutting gaps, clear-cut, shelterwood, a future tree selection system, and thinning aimed at a target diameter distribution. Thinnings can be specified according to age-density schemes, removal of a certain fraction of the basal area increment or according to the available crown space. Thinnings can be done from below, from above or randomly.

### ***Wind damage module***

The wind damage module uses a static mechanistic approach, largely following the principles of the HWIND model (Peltola et al., 1999; Gardiner et al., 2000). Based on characteristics of the tree (height, DBH, crown size), but also its surroundings, it calculates which trees will be broken or uprooted for a given mean hourly wind speed. Wind drag on a tree is calculated according to Monteith (1975), where the crown and stem shape determine the effective sail area of a tree, adjusted for streamlining of the crown (see Peltola and Kellomäki, 1993). The weight of the displaced stem and crown add to the total turning moment at stem base.

However, unlike in HWIND, this turning moment is modified according to the shelter and support received by surrounding trees. Support is assumed to be linearly related to the crown overlap of trees. For each tree a shelter zone is defined, consisting of a strip equal to the crown width and reaching until the first upwind edge. This can be either the forest edge or a gap inside the forest. A gap is defined as an area free of trees higher than the crown base of the subject tree, with a minimum gap length of half the tree height of the subject tree. Sheltering effects depend on the stand density in this zone, modified for relative heights of trees and the presence of foliage. The relationship between stand density and sheltering was adapted from wind tunnel studies in even-aged stands (Gardiner et al., 1997), as described in Peltola and Kellomäki (1999) and Gardiner et al. (2000).

Trees are assumed to break or uproot if respectively the maximum stem resistance or the maximum anchorage resistance is exceeded. Stem resistance is a function of MOR and DBH to the power 3 (see Peltola et al. 1999).

Resistance to uprooting depends also on DBH according to:

$$ROOT_{RES} = a_1 \times DBH^{a_2}$$

where  $a_1$  and  $a_2$  are species specific parameters, estimated from tree pulling tests (Schooten, 1985). Trees can experience additional loading if hit by fallen trees (the domino effect). Trees smaller than 5 m cannot be uprooted or broken, but they can be destroyed by falling trees. Trees are removed from the tree list at the end of each



monthly time step. More details on the wind damage module can be found in Schelhaas et al. (2007).

## Model computations

### *Selected silvicultural systems*

The following six silvicultural systems were selected to be tested, providing a broad range of likely systems:

1. Normal yield table system. Every five years, the number of stems was reduced to the values indicated in the yield table for normal stand density from Jansen et al. (1996). A thinning from below was applied, i.e. trees to be removed were those with the lowest stem diameters, compatible with Jansen et al. (1996). In figures, this system is further referred to as “yield table”.
2. Free thinning from above. The implementation of this silvicultural system was based on the management guidelines for Douglas fir forests from the Dutch State Forest Service. Every five years the best trees were selected and liberated from their immediate competitors. Competitors were defined as trees that have an overlapping crown projection and that reached up to at least half the crown of the tree of interest. The number of trees to be liberated decreased stepwise from 200 at age 20 to 60 at age 80. Preferred maximum h/d ratios for these trees decrease from 100 at age 20 to 70 at ages above 40. This system is further referred to as “free thin mono”.
3. Free thinning from above with mixture of more windfirm beech. The same silvicultural system as described for the “high mono” was used, but 50% of the selected trees to be liberated had to be beech. This system is further referred to as “free thin mix”.
4. Uneven-aged system. This system aims to maintain an exponentially decreasing diameter distribution:

$$N_j = N_{j+1}^q$$

where  $N_j$  is the number of trees in diameter class  $j$  and  $q$  a constant.  $Q$  was 1.3 and the target diameter 60 cm (De Klein and Jansen, 1992). Thinning starts from the highest diameter class. The required number of trees was selected with preferably low h/d ratios and liberated as described before. Excessive trees within the diameter class were removed. Subsequently the lower diameter classes were evaluated for thinning, taking into account the already removed trees. This system is further referred to as “uneven”.

5. No thinning. A no-thin system with an initial stand density of 200 trees per hectare was selected, which is the final density reached by the normal yield table management. This system is further referred to as “no thin mono”.
6. Self-thinning mixture. A mixture of 200 Douglas firs and 3800 beech trees per hectare were used, where the beech was expected to be outcompeted by the Douglas fir. This system is further referred to as “self-thin mix”.

## Simulations and comparisons of silvicultural systems

All simulated systems were studied on a square forest plot of 1 ha. On the north, west and south side a buffer strip of 50 m was assumed to provide shelter to the experimental plot, since only sheltered stands were dealt with. This buffer strip consisted of exactly the same forest as the simulated stand and received the same treatment. However, no wind damage was allowed in the buffer. A buffer strip to the east was deemed unnecessary because high wind speeds are always connected to south to north-westerly directions. Still, in the simulations, 5-10% of the volume damaged was due to easterly winds. However, omitting the easterly buffer strip considerably reduced the computing time needed. The whole forest plot was assumed to be surrounded by an open strip of 100 m, needed to calculate the wind profile outside the forest. The site class was assumed to be 16 for Douglas fir and 8 for beech (Jansen et al. (1996), referring to the maximum mean annual volume increment over a rotation that can be reached for that species on that site).

A young 3 ha stand was generated as initial situation for all even-aged mono-species systems. Initial conditions were taken from Jansen et al. (1996) for a stand with normal density (4000 stems/ha) at age 10, assuming a regular spacing. For the free thinning from above with mixture of beech (system 3) the same initial situation was used, but assuming that all trees in every second 12.5 m strip were beech. Strips were located north-south, providing most shelter to the predominant westerly winds. For the self thinning mixture (system 6) all trees were assumed to be beech, except 200 trees per ha located in a regular grid. To create an uneven-aged initial stand, gaps were cut randomly in the regular stand every 10 years for a period of 100 years. Thereafter uneven-aged management was applied until the diameter distribution was more or less in balance. For the even-aged simulations, no regeneration was allowed to occur in order to save computing time.

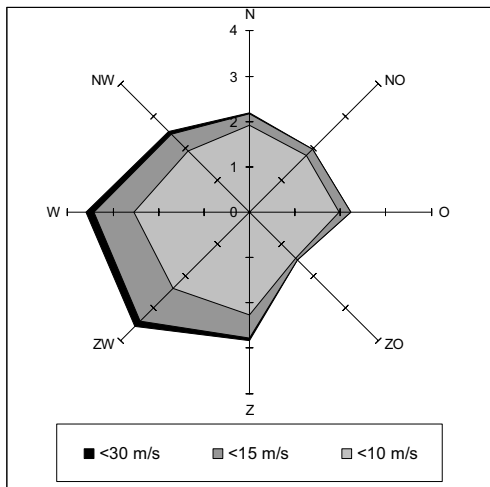
Model parameters for Douglas fir were the same as those used by Schelhaas et al. (2007), except that the maximum crown radius was decreased from 7.7 to 6.5 m. This was done because of excessive mortality at older ages compared with yield table values (Jansen et al., 1996). Consequently the RUE had to be re-calibrated, as well as the parameters that govern the decrease of RUE over age ( $T$  and  $c$ , Table 1, see Schelhaas et al., 2007). Beech was chosen as an example of wind stable species for admixture. For purposes of demonstration and not to complicate the analysis further, this species was assumed not to be damaged by wind at all, so not all parameters were needed for this species. Relevant parameters are shown in Table 1, others are equal to those in Kramer et al. (2007). Furthermore, a 10% variation around the mean was introduced in the height growth parameter for individual trees. This was done to ensure that trees emerging from the same cohort would develop in slightly different ways, allowing for processes as competition and selection to work well.

**Table 1.** Relevant parameters used in ForGEM-W.

Parameter	Douglas fir	Beech	Unit	Reference
Height of crown centre, relative to crown length	0.33	0.41	-	Dutch Forest Reserves Database
CR <sub>MAX</sub> (maximum crown radius)	6.5	14	m	Dutch Forest Reserves Database
RUE	2.56E-09	1.78E-09	kg DM MJ <sup>-1</sup>	calibrated
T (onset of age effect on RUE)	30	30	year	calibrated
c (impact of age effect on RUE)	0.2	0	-	calibrated
b <sub>2</sub> (speed of crown expansion)	0.1137	0.0722	-	Estimated from Jansen et al., 1996

Meteorological variables such as daily radiation, temperature and precipitation are generated by a weather generator for the growth simulations. A separate generator was added to provide daily maximum mean hourly wind speed data (further referred to as wind speed). This generator is based on the Rijkooort-Weibull model for mean daily wind speed (Rijkooort, 1983) and can be parameterised using measured wind speed series. For a simulation, a year was divided in six seasons of two months duration, starting with January/February. Each season has its own wind direction frequencies. Each of the combinations of the eight wind directions and six seasons has its own Weibull distribution. In the simulation, a wind direction is randomly selected each day according to the frequency distribution for that season, and a wind speed is also randomly selected based on the corresponding Weibull distribution.

Wind speed data for the period April 1961 - December 2005 were obtained from the Dutch Royal Meteorological Institute (KNMI, 2007) for the measurement station Leeuwarden (53°13' N, 5°46' E). Measurements were already corrected for roughness of the surrounding terrain. Figure 1 gives an impression of the wind climate with regard to wind direction and wind speed. The highest wind speed measured at Leeuwarden was 28.1 m/s. Based on this data, the daily maximum mean hourly wind speed and the corresponding direction were extracted. Data were grouped per two-month season and eight wind directions (north between 337.5 and 22.5 degrees, etc). For each of the 48 combinations of season and wind direction a 2-parameter Weibull distribution was fitted, using the maximum likelihood method. Furthermore, seasonal direction frequencies were derived. Using the weather generator, 10 time series of daily weather were generated, each covering a period of 100 years. All silvicultural systems were tested for these 10 weather scenarios and all results presented are the means of these 10 replicates.



**Figure 1.** Relative number of observed daily maximum hourly mean wind speeds per wind speed class and wind direction at measurement station Leeuwarden (in % of total number of observations).

Each of the silvicultural systems had its own dynamics with respect to the development of average diameter, standing volume, harvest and other stand characteristics. Considering the differences in diameter development, it was not realistic to compare the systems over the same time span of 100 years. Therefore in all cases a target diameter of 60 cm was assumed, equal to the maximum diameter in the uneven-aged system (De Klein and Jansen, 1992). In the even-aged cases, the analysis was limited to the period needed to obtain an average (quadratic) diameter of 60 cm of the standing Douglas fir trees. In case this diameter was not reached during the simulation period, the full 100 year period was analysed. In the uneven-aged case a 100 years period was analysed, since no information was available on the average time required for a single tree to reach the target diameter.

All systems were compared visually for the development of h/d ratio against height, where for the uneven-aged system the average height and h/d ratio per 2.5 m height class for a randomly selected year was used as comparison. Even-aged systems were also visually compared with regard to development of stand density, DBH, height and h/d ratio over time.

For all silvicultural systems, the gross annual volume increment (GAI) over the period under consideration was assessed, calculated as following:

$$GAI_i = \left( \sum_{t=0}^{T_i} W_{it} + \sum_{t=0}^{T_i} M_{it} + \sum_{t=0}^{T_i} H_{it} + (S_{T_i} - S_0) \right) / T_i$$

where  $GAI_i$  is the gross annual increment ( $m^3 ha^{-1} yr^{-1}$ ) for system  $i$ ,  $W_{it}$ ,  $M_{it}$ ,  $H_{it}$  respectively the wind damage, the other mortality and the harvest in system  $i$  in year  $t$  (all in  $m^3 ha^{-1}$ ),  $T_i$  the evaluated timespan for system  $i$  and  $S_{T_i}$  and  $S_0$  the standing wood volume at the end and the start of the simulation. It was assumed that all dead

timber is left in the forest, both due to mortality and wind damage. Total timber production is then calculated as the components harvest and standing wood volume increase in above formula. All silvicultural systems were also visually compared in terms of gross annual increment and the distribution of its components.

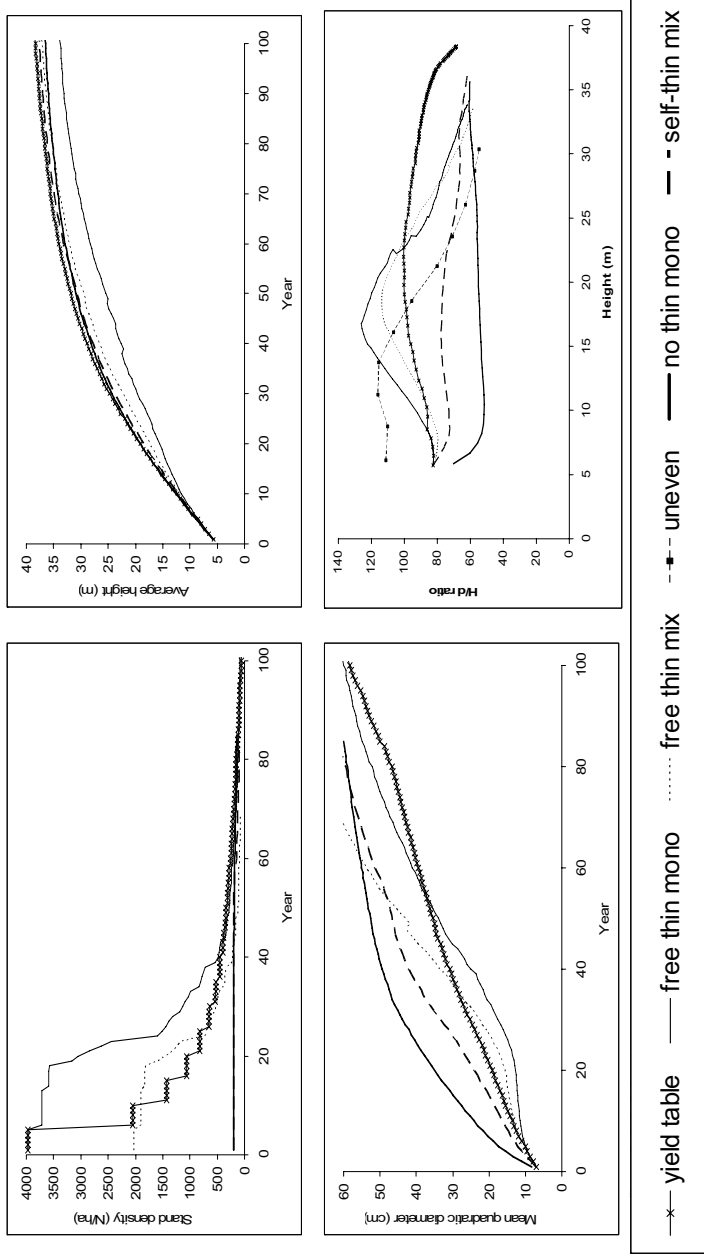
## Results

### Stand dynamics development

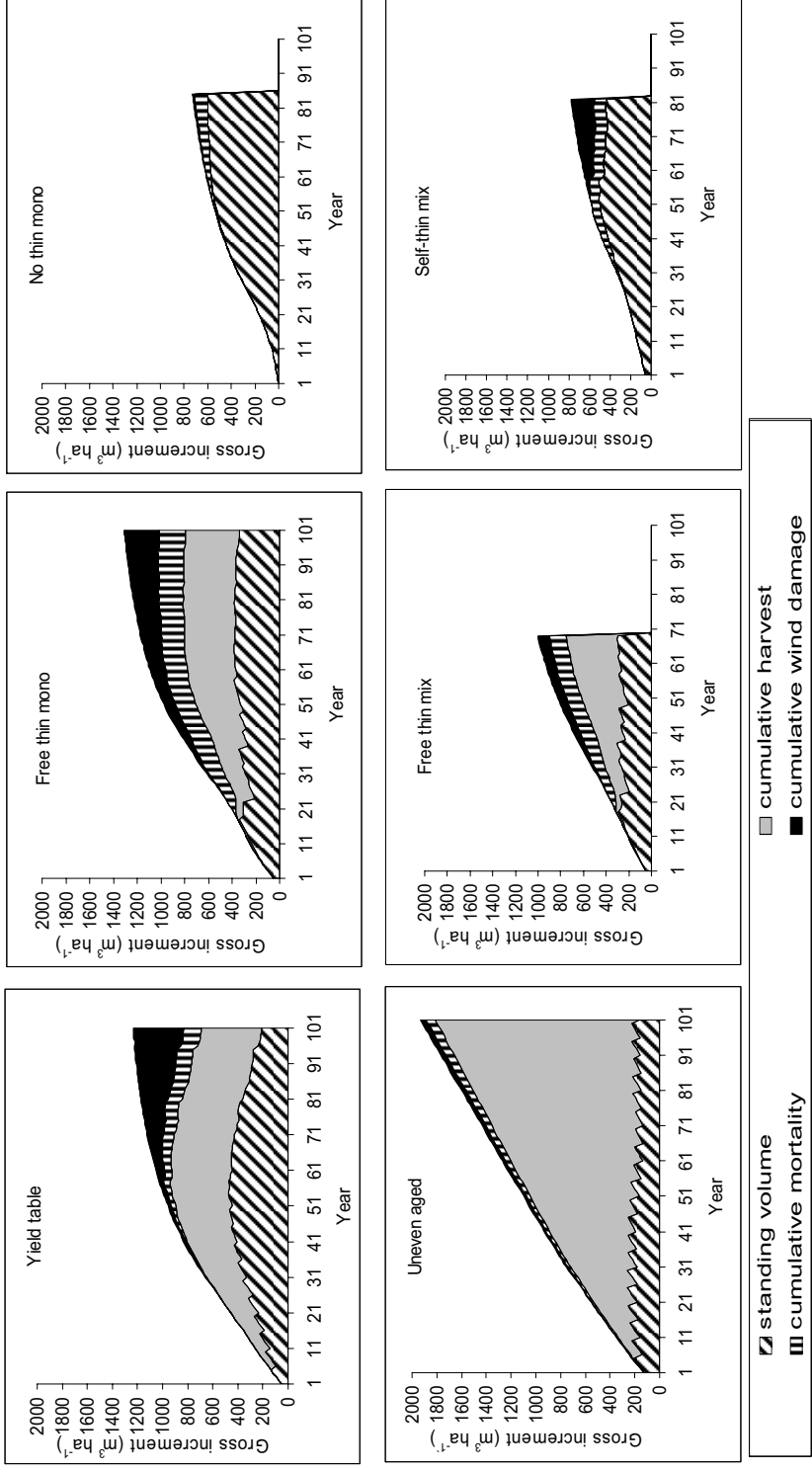
The development of the mean diameter over time differed between systems (Figure 2) and was related to the differences in stand density development. The normal yield table system showed a very regular development of diameter, caused by a very regular decrease in stand density over time. After 100 years of simulation a diameter of 58.7 cm was reached. The system free thinning from above mono maintained a higher stand density in the first half of the simulation, leading to lower mean diameters. Later stand densities were lower, leading to a more rapid increase in diameter. After 100 years a diameter of 59.9 cm was reached.

In the system free thinning from above mixed with beech, total stand density equalled that of the free thinning from above mono system and was higher later on in the simulation. However, the share of Douglas fir in the stand density decreased quickly to around 20%. The mean diameter of the Douglas fir increased rapidly, and reached the 60 cm target already after 69 years. Diameter growth was the fastest in the system no thinning mono, but in the second half of the simulation the speed of growth decreased. On the contrary, the system self thinning mixture showed a slower diameter growth to start with, but a higher growth rate in the second half of the simulation. As a result, both systems reached the 60 cm diameter at about the same time: after 85 years for the no thinning mono and 82 years for the self thinning mixture system. Development of average height was almost equal in all systems, except for the system free thinning from above mono where average height was 4-6 m lower during most of the simulation.

Also the development of the h/d ratio showed large differences between systems. In the normal yield table system the h/d ratio varied between 80 and 90 most of the time and dropped only below 80 when trees were over 35 m high. In the system free thinning from above mono, the h/d ratio increased to over 120 at a tree height of 15 m, but decreased steadily to 60. The system free thinning from above mixed followed a similar pattern, but reached at maximum only a ratio of 114. In the free thinning from above mono system, the h/d ratio varied mostly between 50 and 60, whereas the system self thinning mixture showed a gradual decline from 80 to 60. In the uneven-aged system, average h/d ratios of small trees were high, with values close to 120 for trees until 15 m. However, h/d ratios decreased rapidly for higher trees, with a value of 55 for the tallest trees in the stand



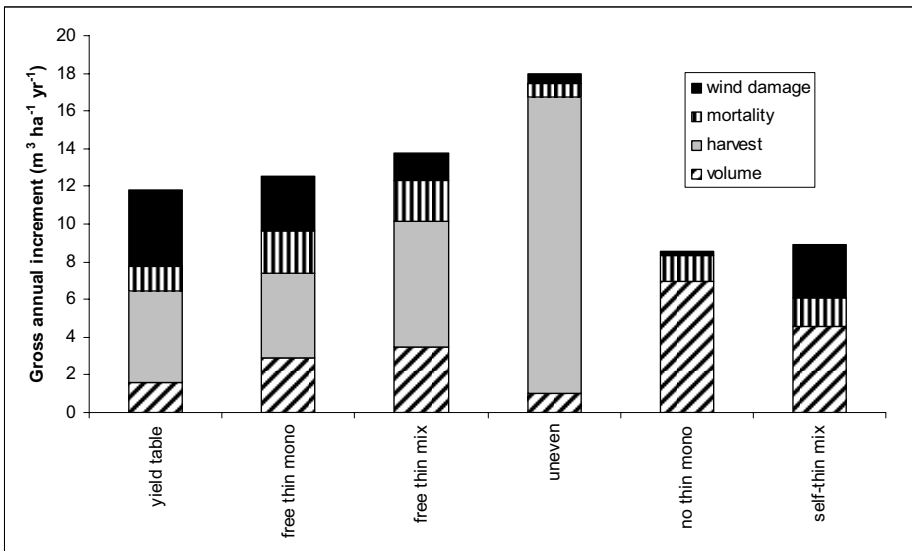
**Figure 2.** Development of a) mean (quadratic) diameter, b) stand density, c) tree height of Douglas fir in the even-aged silvicultural systems and d) development of h/d ratio in relation to average height for all systems. All figures show the Douglas fir part only and are averages over 10 replicates.



**Figure 3.** Development of standing volume, cumulative mortality, cumulative harvest, cumulative wind damage during the simulations, averaged over 10 replicates

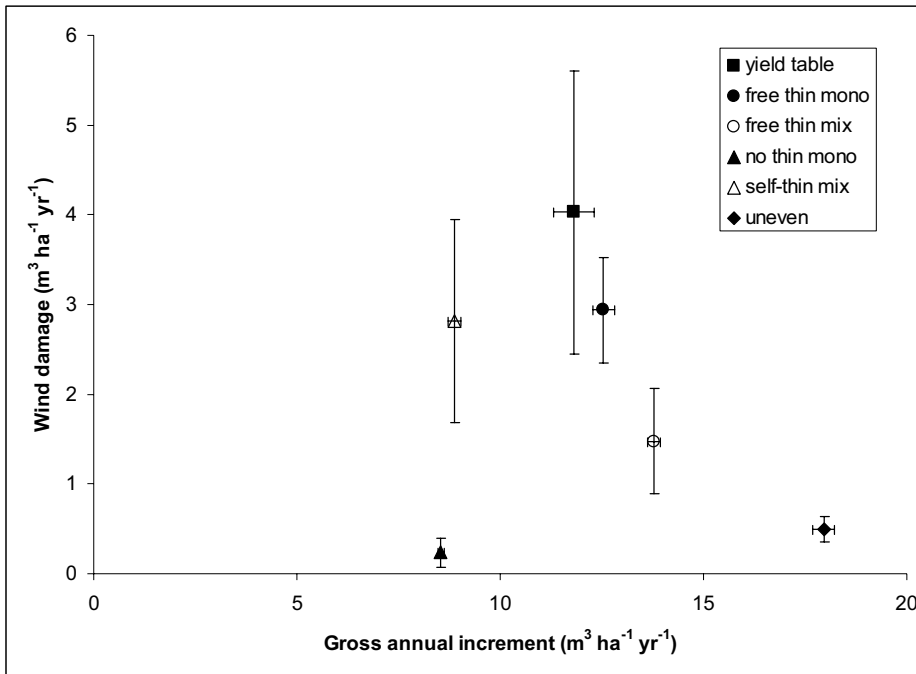
### Timber production and intensity of wind damage

Systems differed substantially in development of growing stock, harvest, mortality, wind damage and gross annual increment (Figure 3). In the normal yield table system, wind damage started to be substantial after about 60 years, leading to a decrease in growing stock level from 470 m<sup>3</sup> ha<sup>-1</sup> to 200 m<sup>3</sup> ha<sup>-1</sup> in the end. Also after 50 years some mortality started to occur. The total gross annual increment was 11.8 m<sup>3</sup> ha<sup>-1</sup> yr<sup>-1</sup>, with standing volume and harvest accounting for just over half of it (Figure 4). In the system free thinning from above mono, wind damage started to occur already after about 40 years, but increased less quickly over time. Mortality occurred especially between 20 and 40 years. The gross annual increment was 12.5 m<sup>3</sup> ha<sup>-1</sup> yr<sup>-1</sup>. Developments in the system free thinning from above mixed were largely comparable, but the target diameter was reached much earlier. The gross annual increment was 13.8 m<sup>3</sup> ha<sup>-1</sup> yr<sup>-1</sup>, with harvest and standing volume accounting for about 75%. The system no thinning mono showed hardly any wind damage and only little mortality. However, gross annual increment was substantially lower with 8.5 m<sup>3</sup> ha<sup>-1</sup> yr<sup>-1</sup>. In the system self thinning mixture the gross annual increment was 8.9 m<sup>3</sup> ha<sup>-1</sup> yr<sup>-1</sup>, with about half of it in harvest and standing volume. Wind damage started to occur after 55 years. The uneven-aged system had the highest gross annual increment with 18.0 m<sup>3</sup> ha<sup>-1</sup> yr<sup>-1</sup>. This system showed hardly any wind damage and mortality.



**Figure 4.** Gross annual increment and its components, averaged over the period needed to get an average diameter of 60 cm or over a period of 100 years. Average of 10 replicates.





**Figure 5.** Wind damage in relation to timber production (volume development plus cumulative harvest), including one standard deviation.

There was no clear relationship between gross annual increment (Figure 5) and the amount of wind damage. Low level of damage was found both in the no thinning mono and uneven-aged system, but gross annual increment was nearly double in the uneven-aged case. Similarly, the systems free thinning from above mono, free thinning from above mixed and normal yield table had comparable gross annual increment, but the amount of wind damage was more than twice as high in the normal yield table system as compared to the system free thinning from above mixed. Within the 10 replicates per each system, gross annual increment showed not much variation, whereas wind damage was much more variable (Figure 5). Systems with higher average wind damage levels showed more variation than systems with low average wind damage levels, too. Furthermore, the systems differed in the lowest wind speed that caused damage. In the uneven-aged system, damage occurred already at wind speeds of 7 m/s. In the system free thinning from above mono, wind damage started at 9 m/s, while in the systems free thinning from above mix and normal yield table this was at approximately 11 m/s. The systems no thinning mono and self-thinning mixture showed the highest wind speed needed to cause damage with 16 m/s. The highest wind speed present in the simulation was 26.9 m/s.

## Discussion and conclusions

### Evaluation of the findings

Differences between systems with regard to stand dynamics can be explained by inter and intra species competition. The system free thinning from above mono had the highest stand density early in the simulation and, thus, the highest competition. This resulted in the lowest diameter growth of all systems, in retarded height growth and in considerable mortality early in the simulation. The normal yield table system had a lower stand density than the system free thinning from above mono until 40 years and thus a higher diameter growth and virtually no mortality. At higher ages the stand density was higher than the system free thinning from above mono, leading to slower diameter growth and more mortality. The system no thinning mono had the lowest stand density, the highest diameter growth and no mortality up to about 50 years. After that, competition increased, causing lower diameter growth and start of mortality. In the system self thinning mixture, 3800 beech trees per hectare were planted in between the sparse (200 stems ha<sup>-1</sup>) Douglas-fir stand. This resulted in increased competition, as evidenced by the lower diameter growth of Douglas fir as compared to the system no thinning mono. However, at a later stage diameter growth recovered as the Douglas fir outcompeted the beech. In the system free thinning from above mixed, half of the Douglas fir were replaced by beech. As Douglas fir is a better competitor, its diameter growth increased as compared to the system free thinning from above mono. Both mixed cases showed that Douglas fir is out competing beech, as expected. Furthermore, reactions of stand characteristics to changes in stand density and tree species composition showed realistic and consistent patterns.

Since height growth was more or less equal among the even-aged systems (except the system free thinning from above mono), the development of the h/d ratio depended on the diameter development and, thus, on the stand density development. Management only had an indirect influence on the h/d ratio through manipulation of stand density. However, in the uneven-aged case management also had a direct influence on the h/d ratio, because h/d ratio was the main selection criterion when the stand density had to be decreased in a certain diameter class. By removing the trees with the highest h/d ratio, the average ratio in a diameter class was decreased after each thinning. Since the number of trees to be retained decreases with increasing diameter class, there was a strong selection on trees with a low h/d ratio. This explains the rapid decrease of h/d ratio with increasing height (Figure 2). Moreover, remaining trees profit from decreased competition and will grow quickly towards the target diameter.

Not all silvicultural systems utilised the growing space equally efficient. Gross annual increment ranged from 8.5 in the system no thinning mono to 18.0 m<sup>3</sup> ha<sup>-1</sup> yr<sup>-1</sup> in the uneven-aged case (Figure 4). Lower increments appeared when stand density deviated from the optimal stand density. Too dense stands, as for example in the first

half of the system free thinning from above mono, causes high competition, with relatively little investment in height and diameter growth of the stem. Too open stands, as for example the start of the system no thinning mono, do not utilise all incoming light. Furthermore, part of the difference between even-aged and uneven-aged stands was caused by the fact that no regeneration was allowed in the even-aged systems. Especially in the systems with low stand densities a second tree layer could considerably increase the overall productivity.

Differences in wind damage level between systems can be linked to differences in development of the  $h/d$  ratio. The longer a tree remains in a situation with relatively high  $h/d$  ratios, the larger the chance that it actually will be damaged. At tree heights above 25 m, the system normal yield table showed the highest  $h/d$  ratios of all silvicultural systems (Figure 2). Moreover, this situation lasted quite some time, increasing the risk that critical wind speeds will be reached. This system also showed the highest damage amount (Figure 4). The system free thinning from above mono showed the highest absolute  $h/d$  ratios reached in any system and had a relatively high damage level as well. In the system free thinning from above mixed,  $h/d$  ratios were somewhat lower, and so was also the damage level. However, this was probably also related to a shorter simulation period, and thus the time trees were exposed to risk of wind damage. The system no thinning mono had hardly any wind damage, connected to very low  $h/d$  ratios throughout the rotation and a short time needed to attain the target diameter. Admixture of beech (the system self thinning mixture) resulted in higher  $h/d$  ratios and a higher damage level in Douglas fir, although the time to reach the target diameter was about the same.

However, the combination of  $h/d$  ratio and height alone are not sufficient to predict the damage level. The systems self thinning mixture and free thinning from above mono had about the same level of damage, but  $h/d$  ratios in the high mono system were much higher. Higher  $h/d$  ratios could, to a certain extent, be compensated for by more shelter and support of surrounding trees as was suggested also previously by Ruel (1995), for example. This was also demonstrated by the uneven-aged case where initial  $h/d$  ratios were rather high, but overall damage was small. This is also caused by the fact that overstory trees will grow quickly to the desired diameter due to the large growing space. Thus, the exposure time is relatively limited.

Several authors state that mixed stands are more stable than mono species stands (e.g. Slodicak, 1995; Schütz et al., 2006). In the case where half of the Douglas fir was replaced by beech (system free thinning from above mixed vs. free thinning from above mono), damage was indeed reduced by 50%, in line with findings of Lüpke and Spellmann (1997). However, this contradicts the study of Schütz et al. (2006) who found that an admixture of 10% wind-firm tree species in Norway spruce stands lead to a decrease in damage by a factor of 3.4. These differences might perhaps be explained by differences in stand history, or preference of mixed stands on certain sites, like sites with lower fertility. Furthermore, use of beech as mixture in the no thinning mono system lead to a considerable increase in damage. This was caused

by the effect beech has on the h/d ratio of Douglas fir through increased competition. This possible effect has also been hypothesised by Cameron (2002). In the current simulations, the effect of mixtures was limited to aboveground competition and sheltering effects. However, belowground competition is an important factor as it can affect the development of deeper or shallower root systems (Hendriks and Bianchi, 1995; Schmid and Kazda, 2001), but may also result in loss of interlocking between trees of the same species (Elie and Ruel, 2005).

The uneven-aged system was found to suffer less damage in comparison with normally practised even-aged systems. This was in line with observations (Dvorak et al., 2001; Schütz et al., 2006). Many authors attribute the greater stability of uneven-aged systems to the fact that overstorey trees will become accustomed to the wind (Cameron, 2002; Mason, 2002). However, such a mechanism was not present in the current model. Instead, the greater stability of the larger trees was caused by high diameter growth of these trees (due to a higher availability of light, nutrients and water) and a strong selection on low h/d ratios. The observed trend for larger trees in this system to have more favourable h/d ratios was confirmed by observations from Kenk and Guehne (2001), but this might at least be partly caused by acclimation to wind.

### **Model aspects**

Especially DBH was strongly influenced in simulations by the silvicultural system. But, still developments of DBH and thus, h/d ratios showed realistic and consistent patterns between the systems. However, it was not possible to validate the absolute values, other than for the normal yield table system. Although the model was calibrated on the normal yield table, differences occurred. At age 100, simulated height and DBH were 38.3 m and 53.0 cm, respectively, compared to yield table values of 37.1 m and 51.4 cm. Differences were most likely caused by the inclusion of wind damage events in the simulations, leading to lower stand densities (i.e. 95 simulated trees per hectare against 206 according to the yield table). However, only the last 10-15 years of the simulation densities were considerably lower than the yield table, so the differences in height and DBH are still rather small.

A larger problem is the sensitivity of wind damage module to anchorage parameters. The underlying sample size was rather small and for trees larger than 40 cm DBH anchorage had to be extrapolated (Schelhaas et al., 2007). Although other tree pulling studies found no changes of relationships between tree size and anchorage at diameters above 40 cm (Papesch et al., 1997; Lundström et al., 2007), more experimental work is needed to confirm this. The shelter function of the wind damage module was also adapted from wind tunnel studies in even-aged stands (Gardiner et al., 1997). In the future, these functions should be refined using wind tunnel studies in irregular stands, as was done for example by Gardiner et al. (2005). Another useful approach would be to replace the shelter functions by models that simulate airflow

through and above the stand directly. However, this would considerably increase the computing time needed.

At the moment, the ForGEM-W model is not able to take into account the acclimation of trees to wind stress, which would be particularly important in edges and stands with sparse canopies. However, due to the simulation set-up the simulation of wind prone permanent edges was avoided. On the other hand, trees in different silvicultural systems were growing under different degrees of shelter, and the taller trees in the no thinning mono and uneven-aged systems, in particular, would be liable to show acclimation to wind loads. However, that would most likely only decrease the already low levels of damage found for these systems and would not influence the outcomes of this study. Currently, the wind damage module also simulates a storm as a short event in which trees are either damaged or not. In reality, storms can last for hours or even days, causing the wind to penetrate further into the stand as trees start to collapse. However, it is not likely that this would change the order of susceptibilities of the silvicultural systems, since all systems were treated in the same way.

## Conclusions

The wind damage module was found to be most sensitive to tree height, DBH and anchorage parameters, and especially to h/d ratio of trees (Schelhaas et al., 2007). As a result, it was found in this work that the silvicultural systems leading to low h/d ratios suffered the least amount of damage. Low h/d ratios were attained through low stand densities. Furthermore, low h/d ratios imply a high diameter growth, and thus a shorter rotation, which in turn lowers the time stands are at risk. Especially the uneven-aged system combined a high timber production with low risks. From the even-aged systems the system free thinning from above mixed had the highest timber production, but included some risk. The no thinning system had the lowest risk, but on account of a considerably lower timber production.

However, low stand densities will also be unfavourable for wood quality, leading to wide growth rings, strongly tapered stems and thick branches. Increasing wood quality through higher stand densities will lead to higher risks of wind damage through increased h/d ratios. Forest owners have to decide individually on this trade-off and how much risk they want to take. This decision will strongly depend on the actual location of a site. The more sheltered a site is, the more focus could be put on aims other than risk consideration. To aid them in this decision, future studies should take into account wood quality and economic parameters, too.

The wind data used in this study are derived from one of the most exposed meteorological stations and risk of wind damage will probably be lower for most of the Douglas fir forests in the Netherlands than as shown here. It is therefore unlikely that foresters will massively adopt a no-thin strategy with low densities, also because of the consequences for wood quality. Currently there is a tendency towards more nature-oriented management. This implies both a move from traditional silvicultural

systems like the normal yield table system towards more free thinning and uneven-aged systems, as well as increased interest in admixture of other species. Results from this study indicate that these trends could lead to lower wind risks and even to an increase in overall productivity.

To conclude, one should still be cautious to transfer the results of this study directly to other wind climates, tree species and soils. More or less severe wind climates might change the specific order of silvicultural systems as obtained in this study because underlying relationships are non-linear (Schelhaas et al., 2007). Other tree species will differ in growth patterns and react differently to inter- and intra-tree species competition, too. However, conclusions on the effects of stand density and h/d ratios on wind stability are likely to be generally valid.

## Acknowledgement

I would like to thank the Forest Reserves Program (Dr. Rienk-Jan Bijlsma) for using their data. Furthermore I would like to thank Dr. Gert-Jan Nabuurs, Prof. Timo Pukkala, Dr. Heli Peltola and two anonymous reviewers for their helpful comments on the manuscript. Funding was provided from European Commission (EFORWOOD, FP6-518128-2)

## References

- Ancelin, P., Courbaud, B., Fourcaud, T., 2004. Development of an individual tree-based mechanical model to predict wind damage within forest stands. *Forest Ecology and Management* 203:101-121.
- Becquey, J., Riou-Nivert, P., 1987. L'existence de zones de stabilité des peuplements. Conséquences sur la gestion. *Forêt-Entreprise* 34, 323-334.
- Blennow, K., Sallnäs, O., 2002. Risk perception among non-industrial private forest owners. *Scandinavian Journal of Forest Research* 17:472-479.
- Bugmann, H., 2001. A review of forest gap models. *Climatic Change* 51:259-305.
- Cameron, A.D., 2002. Importance of early selective thinning in the development of long-term stand stability and improved log quality: a review. *Forestry* 75(1), 25-35.
- Coutts, M.P., 1986. Components of tree stability in Sitka Spruce on peaty gley soil. *Forestry* 59(2):173-197.
- Cremer, K.W., Borough, C.J., McKinnell, F.H., Carter, P.R., 1982: Effects of stocking and thinning on wind damage in plantations. *New Zealand Journal of Forestry Science* 12(2):224-268
- Cucchi, V., Meredieu, C., Stokes, A., Berthier, S., Bert, D., Najjar, M., Denis, A., Lastennet, R., 2004. Root anchorage of inner and edge trees in stands of Maritime pine (*Pinus pinaster* Ait.) growing in different podzolic soil conditions. *Trees – Structure and Function*, 18:460-466.

- Deans, J. D., Milne, R., 1999. Effects of respacing on young Sitka spruce crops. *Forestry* 72:47-57.
- Dirkse, G.M, Daamen, W.P, Schoonderwoerd, H., Japink, M., van Jole, M., van Moorsel, R., Schnitger, P., Stouthamer, W.J., Vocks, M., 2007. Meetnet Functievervulling Bos 2001-2005, Vijfde Nederlandse Bosstatistiek. Directie Kennis, Ministerie van Landbouw, Natuur en Voedselkwaliteit. 95 p.
- Drew, T.J. and Flewelling, J. W., 1979. Stand density management: an alternative approach and its application to Douglas-fir plantations. *Forest Science*, 25:518-532.
- Dvorak, L., Bachman, P., Mandallaz, D., 2001. Sturmschäden in ungleichförmigen Beständen. *Schweiz. Z. Forstwes.* 152(11): 445-452
- Elie, J.G., Ruel, J.C., 2005. Windthrow hazard modelling in boreal forests of black spruce and jack pine. *Canadian Journal of Forest Research* 35(11), 2655-2663.
- Eriksson, H., 1986. Windthrow damage in forests in relation to stand treatment-present state of knowledge in Sweden. In: *Minimizing wind damage to coniferous stands*, Proceedings of the workshop organized jointly by the Danish forest experiment station and the commission of the European Communities at Løvenholm Castle, Denmark, March 3-7, 1986
- Faber, P., 1975. Stabiteit van bos ten opzichte van wind. Een theoretisch gezichtspunt. *Nederlands Bosbouw Tijdschrift* 47:179-187.
- Fahlvik, N., Ekö, P.M., Petterson, N., 2005. Influence of precommercial thinning grade on branch diameter and crown ratio in *Pinus sylvestris* in southern Sweden. *Scandinavian Journal of Forest Research* 20:243-251.
- Gardiner, B., Marshall, B., Achim, A., Belcher, R., Wood, C., 2005. The stability of different silvicultural systems: a wind-tunnel investigation. *Forestry* 78:471-484.
- Gardiner, B., Peltola, H., Kellomäki, S., 2000. Comparison of two models for predicting the critical wind speeds required to damage coniferous trees. *Ecological Modelling* 129:1-23.
- Gardiner, B., Stacey, G.R., Belcher, R.E., Wood, C.J., 1997. Field and wind tunnel assessments of the implications of respacing and thinning for tree stability. *Forestry* 70(3), 233-252.
- Gardiner, B.A. and Quine, C.P., 2000. Management of forests to reduce the risk of abiotic damage – a review with particular reference to the effects of strong winds. *Forest Ecology and Management* 135:261-277.
- Hekhuis, H.J., Nierop, L.J.A.M. van, 1988. De menging douglas - beuk : onderzoek naar de individuele gelijkjarige menging van douglas en beuk, met douglas als hoofdboomsoort, op holtpodzolen in Nederland. *Doktoraalscriptie/Landbouw-universiteit, Vakgroep Bosteelt en Bosoecologie* (88-10).
- Hendriks, C.M.A., Bianchi, F.J.J.A., 1995. Root density and root biomass in pure and mixed forest stands of Douglas-fir and Beech. *Netherlands Journal of Agricultural Science* 43, 321-331.
- Holmsgaard, E. 1986. Historical development of wind damage in conifers in Denmark In: *Minimizing wind damage to coniferous stands*, Proceedings of the workshop organized jointly by the Danish forest experiment station and the commission of

- the European Communities at Løvenholm Castle, Denmark, March 3-7, 1986, pp. 2-4.
- Jansen, J.J., Sevenster, J., Faber, P.J. (Eds.), 1996. Opbrengsttabellen voor belangrijke boomsoorten in Nederland. IBN Rapport 221, Hinkeloord Report No 17. 240 pp.
- Kenk, G. and Guehne, S., 2001. Management of transformation in central Europe. *Forest Ecology and Management* 151(1-3): 107-119.
- Klein, J.P.G. de, Jansen, J.J., 1992. Planning en controle in ongelijkjarig bos. *Nederlands Bosbouw Tijdschrift* 64, p. 313-322.
- KNMI, 2007. Dutch Royal Meteorological Institute [www.knmi.nl](http://www.knmi.nl) Page assessed at 23 July 2007
- Kramer, K. (Ed.), 2004. Effects of silvicultural regimes on dynamics of genetic and ecological diversity of European beech forests; impact assessment and recommendations for sustainable forestry. Alterra, Wageningen.
- Kramer, K., Buiteveld, J., Forstreuter, M., Geburek, T., Leonardi, S., Menozzi, P., Povillon, F., Schelhaas, M.J., Teissier du Cros, E., Vendramin, G.G., van der Werf, D.C., 2007. Bridging ecophysiological and genetic knowledge to assess the adaptive potential of European beech. Submitted to *Ecological Modelling*.
- Landsberg, J.J., Waring, R.H., 1997. A generalised model of forest productivity using simplified concepts of radiation-use efficiency, carbon balance and partitioning. *Forest Ecology and Management* 95:209-228.
- Lundström, T., Jonas, T., Stöckli, V., Amman, W., 2007. Anchorage of mature conifers: resistive turning moment, root-soil plate geometry and root growth orientation. *Tree Physiology* 27:1217-1227.
- Lüpke, B. and Spellmann, H., 1997. Aspekte der Stabilität und des Wachstums von Mischbeständen aus Fichte und Buche als Grundlage für waldbauliche Entscheidungen. *Forstarchiv* 68:167-179.
- Lüpke, B von, Spellmann, H., 1999. Aspects of stability, growth and natural regeneration in mixed Norway spruce-beech stands as a basis of silvicultural decisions. In Olsthoorn, A.F.M., et al (eds). *Management of mixed-species forest: silviculture and economics*. IBN Scientific Contributions 15, IBN-DLO Wageningen. p. 245-267.
- Macdonald, M.J., 1952. Wind damage in middle aged crops of Sitka spruce, and its prevention. *Scottish Forestry* 6:82-85.
- Mackenzie, R.F., 1974. Some factors influencing the stability of Sitka in Northern Ireland. *Irish Forestry* 31:110-129.
- Mason, W. L., 2002. Are irregular stands more windfirm? *Forestry* 75(4):347-355.
- Monteith, J.H., 1975. *Principles of environmental physics*. Edward Arnold, London.
- Nicoll, B.C., Gardiner, B.A., Rayner, B., Peace, A.J., 2006. Anchorage of coniferous trees in relation to species, soil type, and rooting depth. *Canadian Journal of Forest Research* 36: 1871-1883.
- Pacala, S.W., Canham, C.D., Saponara, J., J Silander, J.A. Jr., Kobe, R.K., Ribbens, E., 1996. Forest models defined by field measurements: II. Estimation, error analysis and dynamics. *Ecological Monographs* 66:1-43.



- Pacala, S.W., Canham, C.D., Silander, J.A. Jr., 1993. Forest models defined by field measurements: 1. The design of a northeastern forest simulator. *Canadian Journal of Forest Research* 23:1980-1988.
- Papesch, A.J.G., Moore, J.R., Hawke, A.E., 1997. Mechanical stability of *Pinus radiata* trees at Eyrewell forest investigated using static tests. *New Zealand Journal of Forestry Science* 27:188-204.
- Peltola, H., Gardiner, B., Kellomäki, S., Kolstrom, T., Lässig, R., Moore, J., Quine, C., Ruel, J., 2000. Introduction: Wind and other abiotic risks to forests. *Forest Ecology and Management* 135:1-2.
- Peltola, H., Kellomäki, S., 1993. A mechanistic model for calculating windthrow and stem breakage of Scots pines at stand edge. *Silva Fennica* 27:99-111.
- Peltola, H., Kellomäki, S., Väisänen, H., Ikonen, V.-P., 1999. A mechanistic model for assessing the risk of wind and snow damage to single trees and stands of Scots pine, Norway spruce, and birch. *Canadian Journal for Forest Research* 29:647-661.
- Quine C., Coutts, M., Gardiner, B., Pyatt, G., 1995. Forests and wind: Management to minimize damage, Forestry Commission Bulletin 114. HMSO, London.
- Reineke, L.H., 1933. Perfecting a stand density index for even-aged forests. *Journal of Agricultural Research*, 46:627-638.
- Richards, F.J., 1959. A flexible growth function for empirical use. *Journal of Experimental Botany* 10:290-300
- Rijkoort, P.J., 1983. A compound Weibull model for description of surface wind velocity distributions. Scientific Report, WR 83-13, Royal Dutch Meteorological Institute (KNMI).
- Ruel, J-C., 1995. Understanding windthrow: silvicultural implications. *Forestry Chronicles* 71:435-445.
- Savill, P.S., 1983. Silviculture in windy climates. *Forestry Abstracts* 44:473-488.
- Schelhaas, M.J., Varis, S., Schuck, A., 2001. Database on Forest Disturbances in Europe (DFDE), European Forest Institute, Joensuu, Finland. <http://www.efi.int/projects/dfde/>.
- Schelhaas, M.J., Nabuurs, G.J., Schuck, A., 2003. Natural disturbances in the European forests in the 19th and 20th centuries. *Global Change Biology* 9:1620-1633.
- Schelhaas, M.J., Kramer, K., Peltola, H., van der Werf, D.C., Wijdeven, S.M.J., 2007. Introducing tree interactions in wind damage simulation. *Ecological Modelling* 207:197-209.
- Schmid, I., Kazda, M., 2001. Vertical distribution and radial growth of coarse roots in pure and mixed stands of *Fagus sylvatica* and *Picea abies*. *Canadian Journal of Forest Research* 31: 539-548.
- Schooten, J.P. van, 1985. Fysische aspecten van windworp bij de douglasspar, Vakgroep Bosteelt. Landbouwhogeschool, Wageningen, 200 pp.
- Schütz, J.P., Götz, M., Schmid, W., Mandallaz, D., 2006. Vulnerability of spruce (*Picea abies*) and beech (*Fagus sylvatica*) forest stands to storms and consequences for silviculture. *European Journal of Forest Research* 125, 291-302.

- Sissingh, G., 1975. Stabiliteit van bos ten opzichte van wind en storm gezien vanuit de praktijk. *Nederlands Bosbouw Tijdschrift* 47:188-193.
- Slodiacák, M., 1995. Thinning regime in stands of Norway spruce subjected to snow and wind damage. *Wind and Trees*. M. P. Coutts and J. Grace. Cambridge, Cambridge University Press.
- Soest, J. van, 1954. Stormschade aan Douglas. *Nederlands Bosbouw Tijdschrift* 26(4), 89-99.
- Wilson, J.S., and Oliver, C.D., 2000. Stability and density management in Douglas-fir plantations. *Canadian Journal of Forest Research* 30, p. 910-920.
- Zeng, H. 2006. Influence of clear-cutting on the risk of wind damage at forest edges: A gis-based integrated models approach. University of Joensuu, Faculty of Forestry. *Dissertationes Forestales* 16.

