Fire Ecology of Scots Pine in Northwest Europe

Vuur Ecologie van Grove Den In Noordwest Europa

Promotoren:

Prof. Dr. Ir. G.M.J. Mohren	Hoogleraar Bosecologie en bosbeheer Wageningen Universiteit
Prof. Dr. J.G. Goldammer	Hoogleraar Fire Ecology Freiburg University

Co-promotor:

Dr. Ir. J. den Ouden	Universitair docent, Leerstoelgroep Bosecologie en bosbeheer Wageningen Universiteit

Promotiecommissie:

Prof. Dr. F. Berendse	Wageningen Universiteit
Prof. Dr. S. Stephens	University of California, Berkeley
Prof. Dr. F. Rego	Universidade Técnica de Lisboa

Fire Ecology of Scots Pine in Northwest Europe

Marco Hille

Proefschrift
ter verkrijging van de graad van doctor
op gezag van de rector magnificus
van Wageningen Universiteit,
Prof. Dr. M.J. Kropff
ter beoordeling door een commissie
van drie door het college voor promoties
aangewezen deskundigen

ISBN 90-8504-283-6

Most of the chapters of this thesis have been or will be published as articles in scientific journals. You are kindly requested to refer to these articles rather than to this thesis.

Hille, M. (2006)

Fire Ecology of Scots Pine in Northwest Europe
PhD thesis, Wageningen University, Wageningen

Abstract

In this thesis the ecological consequences of forest fire are studied in North-west European Scots pine (*Pinus sylvestris*) forests. The focus is on post-fire succession, and the factors and mechanisms that influence the successional pathways after fire.
Fuel load and fuel moisture determine the intensity of forest fire and thus the degree of humus consumption. In a controlled laboratory experiment, humus consumption was determined for different moisture levels. Experimental fires showed evidence that variation in precipitation throughfall causes spatial variation in humus consumption in the stand through differences in humus moisture with respect to tree crowns.
Humus consumption influences tree mortality, growth of remaining trees and re-colonization. Surface fires in Scots pine plantations caused a partial reduction of the litter and humus layers and a high mortality in the smaller trees. Reduction in radial growth after surface fire was variable, and was less in large diameter trees and in trees that experienced less humus consumption around their stem bases.
Experimental burning of the humus layer showed that increased removal of organic material by fire resulted in an increase in seedling numbers. Earlier studies have suggested that the charcoal produced by fire improves germination conditions by absorbing phytotoxins produced by ericaceous species. All such studies have used activated carbon as a standardized model for charcoal. Bioassays with pine seeds in aqueous extracts of *Vaccinium myrtillus* and *Calluna vulgaris* showed toxic effects of the two species, but charcoal reduced toxicity less than activated carbon. Therefore, those previous studies have overestimated the effect of charcoal on germination, likely because of the considerably higher active surface area of activated carbon.
The post-fire tree cohort after severe but small-scaled fires in Scots pine stands mainly consisted of Scots pine, but also birch and aspen. Compared to succession after other disturbance types in Scots pine stands, such as windthrow or soil scarification, seedling numbers are higher after small-scale fires by a magnitude of ten.
Based on the good regeneration and for the purpose of fuel load reduction in areas with increased fire hazard, the prescribed burning of Scots pine stands should be reconsidered. Controlled forest fires could be used as an additional silvicultural technique to regenerate and transform single-species pine stands into mixed and more natural forests.

Key words: biodiversity, fire ecology, fuel modelling, succession, tree regeneration

Preface

The author of this thesis, Marco Hille, died in a tragic accident, on December 4, 2004, while cutting firewood near his parents home in Schale, Germany. At the time, his thesis was almost completed, with essentially mainly editorial work remaining.

Based on the draft thesis, the Rector Magnificus of Wageningen University, acting on behalf of the board of the University, decided to award the doctorate degree posthumously, to recognize the work done by Marco Hille and as a tribute to an extraordinary PhD student. This decision was based on an evaluation of the thesis by a reading committee consisting of Prof F. Berendse of Wageningen University, Prof F. Rego of the Technical University of Lisbon, and Prof S. Stephens of the University of California at Berkeley.

Marco Hille's PhD project was funded by a grant of the German Robert Bosch Foundation to the C.T. de Wit Research School in Production Ecology and Resource Conservation. Mrs Ir F. Vodde assisted in the editorial completion of the thesis.

Marco Hille worked with enormous enthusiasm and energy, and was well ahead of his original work plan. At the time of his death he was already preparing for postdoctoral research in Yosemite National Park in his beloved California. Marco had a bright career ahead of him. We miss him dearly.

Wageningen/Freiburg,

Prof Dr Ir G.M.J. Mohren
Prof Dr J.G. Goldammer
Dr Ir J. den Ouden
(Supervisors and Co-Supervisor)

Table of Contents

CHAPTER 1

Introduction

1 Introduction

At first glance, in comparison to other regions of the world, fire does not play a significant role in North-West European forests, and research on forest fires in the area has been largely neglected. The ideology and policy of banning fire from the European landscape, especially from managed forests, lead to an emphasis on research into fire fighting means and tactics (Goldammer 1982, Mißbach 1973). The strong "no fire"-policy was an essential part of European forest management and this ideology was even exported in the 19th century into the "New World" (Pyne 1997). Here, in fire-prone forests, the exclusion of fire caused huge problems of fuel overload and altered forest dynamics. Over the last 30 years, the new field of fire ecology emerged to deal with these issues.

Despite all fire exclusion efforts, between 3,600 and approximately 20,000 ha of forest burned within North-West Europe on an annual average over the last 50 years (Lex and Goldammer 2001, Ubysz and Szczygiel 2002). In this thesis, North-Western Europe is referred to as the countries of Germany, Poland, the Netherlands and Belgium. Data collection for this thesis was done in the Netherlands and Germany, though the forest structures and the fire hazard situations are comparable in Belgium and Poland.
Within all forest types found in North-West Europe (both natural and man-made forest types), pine stands (*Pinus sylvestris* L.) have by far the highest fire risk. This risk is due to the vegetation composition itself (often a dense field layer of grasses and dwarf-shrubs is found beneath pine), the low ignition temperature required to start a fire in pine wood and needles, and the high energy content of it's woody parts (Mißbach 1973).
Especially on dry sandy soil, in regions such as the *Veluwe* in the Netherlands, the *Kempen* region in Belgium or the *Lausitz* and the *Lüneburger Heide* in Germany, fires of varying extent occur annually. Under severe conditions, fires burn large areas and the monotonous forest structure with even-aged stands and no natural fuel brakes makes it hard to control.
To broaden our limited knowledge of the ecological consequences of fires occurring under North-West European conditions and determine how best to deal with them, this thesis focuses on the actual and potential **ecological** role of fire in North-West European *P. sylvestris* forests.

1.1 Fire ecology

In the past centuries, fire has been seen as a big threat to pine forestry and was not tolerated. Looking back at the effectiveness of fire as a human tool to clear "wild" vegetation (forest removal) and it's power in uncontrolled conditions, this view is easily understandable (Pyne 1997).

All ecosystems are exposed to, at the least, gradual changes in climate, nutrient loading, habitat fragmentation, regeneration and other biotic and abiotic factors and ecosystems may respond in a smooth, continuous way to such trends. However, diverse, sudden events, such as fire in a forest, can trigger abrupt changes and switch the ecosystem to a contrasting state (Scheffer *et al.* 2001). The magnitude of change depends on a) the resilience of the ecosystem and b) the severity of the disturbance.

Fire ecology deals with the variation of forest fire impacts and factors that cause this variation. Since fire ecology is quite a young field of ecological research, the following paragraphs aim to give the reader a short introduction to general mechanisms and effects of forest fire.

1.1.1 Fire as disturbance factor

Traditionally, forest landscapes have been regarded as increasingly stable during succession over time, until eventually reaching a "climax state" of complete stability. Currently, such ideas are being challenged by a new understanding of the importance and inevitability of large-scale disturbances such as storms and fires that cause ecosystems to be in a state of constant change. The recognition of fire as a complex agent of change in ecosystems is not new, but it's incorporation into the succession theory is more recent. Once, fire was considered an exogenous, catastrophic event that affected ecosystems. It is now seen as a natural phenomenon linked to the dynamics of many plant communities and animal populations. However, the effects of fires vary widely and the outcome of post-fire ecosystem development is a multivariate process (Table 1.1, Biswell 1989, Johnson 1992).

Fire ecology has emerged as a central feature of this new understanding and aims to explain the variation of fire effects on ecosystems. Currently, results from fire ecology research are used to redefine traditional assumptions about the growth and development of forest ecosystems (Agee 1993).

1.1.2 Fire regimes

The main ignition source under a natural fire regime is lightning. Depending on weather and forest conditions before and during the occurrence of lightning, a fire can develop varying characteristics and consequently, varying ecological impacts.
In general, fires are divided into two types: those which kill all trees on site are referred to as "stand-replacing disturbances" (most often found in ecosystems where high severity fires occur), and those which only kill some trees but leave the overstory intact are referred to as "non-stand-replacing" (low-severity fire regime; Oliver 1981). Additionally, forest fires are classified as *surface fires* and *crown fires*. In general, surface fires are less severe and are most often not stand-replacing, whereas crown fires

are mortal for trees affected and therefore stand-replacing. However, even low-intensity surface fires which do not scorch the canopy can be stand-replacing (e.g. for thin barked trees; Nieuwstadt *et al.* 2001).

High intensity, crown fires cause high mortality and replace the entire stand. Low intensity surface fires accelerate nutrient cycling, causing temporarily increased growth and a change in floral composition (Crutzen and Goldammer 1992). Commonly, the role of fire in an ecosystem is described in terms of disturbance characters, listed in Table 1.1, and summarized in a certain fire regime.

Table 1.1. *Main characteristics of fire regimes (adapted from Agee 1998b).*

Descriptor	Definition
Frequency	Mean number of fire events per time period
Extent	Area burned per time period or fire
Timing	The seasonality of the disturbance
Magnitude	Described either as intensity (physical force) or severity (measure of the effect on the organism or ecosystem)

Although many combinations of the fire characteristics listed above may occur, simplified systems were created to categorize fire effects in different ecosystems. Agee (1993) uses a classification system based on fire severity (the effect of fire on the dominant vegetation). **Low-severity** fire regimes are those where fires have little effect on mortality of the dominating vegetation (e.g. in the Western US, where the natural fire regime of Ponderosa pine stands composes of a fire return interval of 5-10 years with almost no mortality in the overstory; Stephens and Collins 2004). **High-severity** fire regimes include those vegetation types, where the fires kill most of the vegetation (e.g. in the Rocky Mountains, where stand-replacing events occur every 150 to 350 years on entire watersheds; Barret *et al.* 1991). **Moderate-severity** fire regimes vary temporally and spatially between low- and high severity, e.g. in the boreal zone of Northern Europe, where fires occur with a return interval of 50-60 years, and locally kill overstory trees (Lehtonen and Kolström 2000). This classification will be used throughout this report, especially for post-fire succession in section 1.3.

1.1.3 Ecological fire effects

All fire characteristics and effects are driven by fuel-loading (dead and dry biomass on the forest floor), moisture content of the fuel and the structural characteristics of the ecosystem. Additionally, the weather situation before and during the fire plays an

important role. Main factors affecting fire behavior and severity are wind, humidity and temperature (Biswell 1989).
Forest fires are major disturbances in forest ecosystems and effect ecosystem processes in many ways. In general, forest fires affect ecosystem dynamics by:

- initiating mortality, either throughout the whole stand or in patches (Spurr and Barnes 1980),
- creating structural change and unique habitats (Oliver and Larson 1990),
- affecting biogeochemical nutrient cycling (Viro 1969).

A common way of characterizing fire effects is by their order of occurrence. **First-order** fire effects occur at the time of the fire or within a very short time afterwards. They are therefore direct results of the fuel combustion process and are primarily heat induced. First-order effects are directly caused by pre-fire conditions and the fire environment and are the drivers of second-order fire-effects (e.g. vegetation succession; Table 1.2).

Table 1.2. *Examples of First- and Second-Order Fire Effects and their relation.*

First-Order	Second-Order
Plant mortality	Vegetation succession
Fuel consumption	Soil temperature
Smoke production	Erosion
Soil heating	Site productivity
	Charcoal/Phytotoxins

Second-order fire effects occur on a wider time scale; days, months or even decades after the fire event. They are indirect results of fire and other processes such as climate, land use and seed availability. Due to this, they are very variable and hard to predict (Reinhardt *et al.* 2001).

At a very long time scale, some fire effects are evolutionary; certain species developed adaptations to cope with fire occurrence (e.g. thick bark or serotinous cones).

Mineral soil exposure by consumption of the ectorganic soil layers and the alteration by heat input are especially critical for post-fire vegetation composition (Clark 1994). The degree of humus consumption during wildfires or prescribed burnings strongly influences the post-fire succession of the site (Johnson 1992). Prolonged heat release by smoldering combustion kills the fine roots of herbs, shrubs and also overstory trees, which can lead to high mortality (Ryan and Frandsen 1991, Stephens and Finney 2002). As a consequence, species composition can be altered and seed sources for a quick re-colonization will be absent. With a higher consumption of the humus layers, stored seeds and vegetative organs are also destroyed (Schimmel and Granström 1996). In the case of partial consumption of the humus layer, heat tolerance and storage depth of certain species plays an important role in the re-colonization pattern of the site (Granström and Schimmel 1993, Whittle *et al.* 1997). Furthermore, in the case of complete humus consumption, only species with roots and vegetative organs in the mineral soil can survive. Therefore, the amount of humus left on the site also influences species composition, density and early growth in the seedling cohort following the fire (Thomas and Wein 1985, Johnson 1992). In the boreal forest, complete humus consumption was found to favor the establishment and growth of coniferous seedlings (Chrosciewicz 1974, Zasada *et al.* 1983).

Examples of second-order fire effects are a change in the pH of the soil into a more alkaline milieu (Riek *et al.* 2002, Chandler *et al.* 1983, Wright and Bailey 1982), an increase of post-fire temperature in the upper mineral soil (Clark 1994), and the formation of a hydrophobic layer (DeBano 1981, Hetsch 1980).

Another fire effect, which has been discovered just recently, is related to the production of huge amounts of charcoal which remain on the forest floor after the fire. These charcoal particles have the ability to adsorb phytotoxins, which are released by, for example, dwarf-shrubs and usually inhibit seed germination (Zackrisson *et al.* 1996). Trials with activated carbon and charcoal showed that this porous material has very positive effects on germination, seedling survival and growth (e.g. Nilsson 1994, Jäderlund *et al.* 1998). Because woody charcoal has lower active surface area, it can be expected that the adsorption potential is smaller, but charcoal can still be an important factor in the re-colonization of the burned forest floor by reducing phytotoxic impact and therefore improving conditions for germination.

1.1.4 Humus consumption

Humus consumption (humus referred to as the O_F and O_H -layer between the litter (O_L) and the mineral soil, Green *et al.* 1993) has a big influence on fire effects. Prolonged heat released by smoldering combustion of humus kills fine roots of herbs and shrubs as well as overstory trees, which can lead to high mortality (Stephens and Finney 2002). As a consequence, species composition can be altered and seed sources for a quick re-colonization will be absent. With a higher consumption of the humus layers, stored seeds and vegetative organs are also destroyed (Schimmel and Granström 1996). In the case of partial consumption of the humus layer, heat tolerance and storage depth of certain species plays an important role in re-colonization pattern of the site (Granström and Schimmel 1993, Whittle *et al.* 1997). Furthermore, in the case of complete humus consumption, only species with roots and vegetative organs in the mineral soil can survive.

However, the exposed mineral soil is a suitable seedbed for many forest species. In the boreal forest, complete humus consumption was found to favor the establishment and growth of coniferous seedlings (Chrosciewicz 1974, Zasada *et al.* 1983).

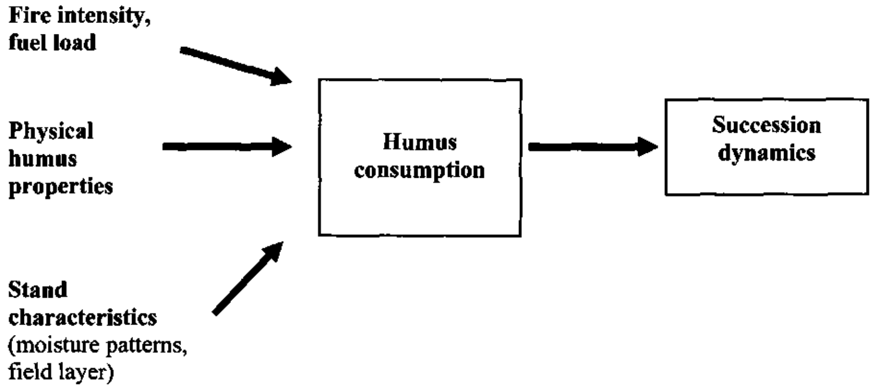


Figure 1.1. *Schematic illustrations of the humus consumption process and factors that influence it's degree (Chapter 2) and spatial pattern (Chapter 4). Humus consumption is a key factor that influences post-fire succession dynamics (see Chapter 5, 8).*

Humus consumption proceeds mostly by slow smoldering combustion, with the amount of consumption mainly depending on humus characteristics. In previous studies it was found that humus moisture has an overriding influence on humus consumption (McArthur and Cheney 1966, Sandberg 1980, Brown *et al.* 1985, Frandsen 1987, Reinhardt *et al.* 1991, Robichaud and Miller 1999). The stored moisture results in a latent heat flux for evaporation of the water, which provides an effective heat sink, stopping smoldering combustion. Other factors, such as packing ratio, humus depth, and amount of surface fuel load also influence humus consumption (Fig. 1.1, Burgan and Rothermel 1984, Miyanishi and Johnson 2002).

1.1.5 Response to the disturbance

Fire disturbance causes the ecosystem to respond and the magnitude and time period of this response varies with the initial conditions, the resilience of the system and the fire severity (Scheffer *et al.* 2001, Oliver and Larson 1996). Possible scenarios of ecosystem development after fire are determined by:

- low magnitude of disturbance and/or high resistance [almost no mortality for dominating species, quick re-growth out of surviving propagules of pre-disturbance plant species],

- medium magnitude of disturbance and/or medium resistance [leading to high mortality rates within species dominating the site, but propagules and seeds of pre-fire species lead to re-colonization, e.g. temporally altered composition of annual species or natural regeneration of species which dominated the site before] or
- high magnitude of disturbance and/or low resistance [leading to a complete change of species composition and vegetation cover, e.g. grass or bracken invasion in previous forest-covered areas].

Depending on fire intensity and ecosystem stability, several different pathways of development can follow a fire disturbance. After high severity fires, a change in species and gradual recovery over time through various plant communities have been observed (Tonn *et al.* 2000), but also an immediate post-fire re-colonization by the same species which were present before the fire (Weber and Stocks 1998). An establishment of natural tree regeneration (especially in pine stands) has been observed (Kuuluvainen and Rouvinen 2000, Engelmark 1993, Engelmark *et al.* 1998), as well as an invasion of grasses or bracken in the burned areas, leading to a monotonous and less diverse ecosystem (Gliessmann 1978, Oinonen 1967, Jahn 1980).

The response of plants to fire can vary significantly, due to both variability in the heat regime of the fire and differences in plant species' ability to survive a fire and to recover (Miller and Findley 1994). As a result, highly diverse structures are created in the fire-adapted landscape (Weber and Stocks 1998). Fire in boreal forests is important for maintaining the diversity of rare herbs (Esseen *et al.* 1997). Fires also create habitats for specialized plants and animals (Punttila and Haila 1996, Kaila *et al.* 1996). Some detailed studies have been carried out on the impact of fire on soil fertility (Vega *et al.* 1983, Viro 1969), soil microbial activity (Fritze *et al.* 1993), and the development of ground vegetation (Trabaud 1983, Lindholm and Vasander 1987).

1.1.6 Fire and forest management

The importance of natural disturbances in regulating forest ecosystem properties, including biodiversity was shown in several studies in the past (Franklin *et al.* 1987, Busing and White 1997), and recent ecological research offers innovative silvicultural approaches designed to manage forests similar to their natural disturbance regime (Granström 2001, Angelstam 1998, Bergeron and Harvey 1997).
Today's forest management practices throughout Europe put an emphasis on sustainability, biodiversity and the use of silvicultural systems which are close to natural disturbance regimes. It is therefore important to understand e.g. the role of forest fires as a natural disturbance event in forest ecosystem dynamics, especially in *P. sylvestris* stands. In recent years, the important role of fire in determining ecological functioning of forest ecosystems and the need to consider natural disturbances in forest management has become increasingly clear (e.g. Agee 1998b).

However, only sporadically has research been published which deals with fire effects under North-Western European conditions (Goldammer 1979, Jahn 1980), which offers the ability to give predictions about fire impact, fire effects and post-fire succession.

In world-wide forestry, prescribed fire uses traditionally include:

a) Reducing fuel hazards. Prescribed fires are often used to reduce the amount of dead-and-down fuels on the forest floor. Larger fuels and parts of the humus layer remain, and overstory trees are not damaged. As a consequence of the reduced fuel load, the impact of subsequent wildfires is reduced (Mutch 1994, Deeming 1990, Wagle and Eakle 1979).

b) Pre-commercial thinnings. Fire with a prescribed intensity girdles small trees which bark is not thick enough to protect the cambium from the heat input. Effects on the stand structures are similar to that of a thinning from below - but in contrast to mechanical pre-commercial treatments, fire is more cost effective (Graham *et al.* 1999, Wade 1993).

c) Improving ecosystem diversity and wildlife habitat (Van Lear 2000). If used properly, fire is one of the most beneficial and cost-effective wildlife management tools available and is used e.g. in pine stands in the Southern US. Fire creates open areas for foraging and it encourages plants that provide food and shelter for many species (Burger *et al.* 1998).

d) Restoration of entire ecosystems. After decades of fire exclusion in fire-prone regions, it was realized that forest dynamics without fire lead to undesired and unstable conditions. Fire is re-introduced to control stand density, forest structure and ecological processes, in places such as the southwest U.S. (Covington *et al.* 1997) and the Appalachians (Elliott *et al.* 1999).

e) Preparation of seedbeds and sites for forest regeneration. There are ample examples from Northern America and Scandinavia where prescribed fire is used to regenerate conifer forests. Prescribed burnings to introduce natural regeneration are common practice for *P. sylvestris* in Finland, Norway and Sweden (Uggla 1959, Braathe 1974, Sykes and Horrill 1981). Even broad-leaved species are treated with prescribed fires to stimulate natural regeneration (Brose *et al.* 2001).

It is important to note that for all purposes listed above, fire is used in a way similar to the natural fire regime, e.g. at times in the year where it would occur naturally, in similar aerial extent and with a similar return interval. Therefore, it is essential to know the natural fire regime and succession after fire, so that these dynamics can be mimicked. A strong emphasis on fire ecology research is therefore focused on gathering information about natural fire regimes and successional pathways of ecosystem development after fire section (1.2.3).

1.2 Pine stands in North-West Europe - on fire?

P. sylvestris is one of the most important tree species in North-West Europe (the Netherlands, Germany, Belgium and Poland). *P. sylvestris* as a tree species which is not endemic on most North-West European sites (Endtmann *et al.* 1991), but was introduced as a crop species centuries ago. Although this species would naturally stock only on dry, sandy sites or wet, acid bogs, *P. sylvestris* stands can be found on the entire range of moisture and nutrient levels in these countries (Endtmann *et al.* 1991). However, fire has and will occur in all different forest types of *P. sylvestris.*
In addition, future climate change may increase drought over large parts of Europe, possibly increasing the risk of forest fires, and interfering with conditions for sustainable forest management. Due to global climate changes, increased frequency of drought and increased interannual variability of precipitation are expected over Europe (IPCC 2001, Rind *et al.* 1989). These changes in temperature and precipitation will change the fuel characteristics and fire frequency (Badeck *et al.* 2004), as well as the behavior of a potential fire (Kasischke and Stocks 2000, Crutzen and Goldammer 1992).

1.2.1 *P. sylvestris* forests in North-West Europe

P. sylvestris is the most widely distributed conifer in the world and natural forests or plantations are found in all member states of the EU. *P. sylvestris* is of considerable importance as a timber producing species and of high economic value in many rural regions. *P. sylvestris* covers 3,007,000 ha in Germany, 96,000 ha in the Netherlands, 65,000 ha in Belgium (Mason and Alia 2000) and 6,028,200 ha in Poland (Ubysz and Szczygiel 2002).
In Germany, only 3% of the potential natural vegetation would be pine-dominated forest types, most often as mixed stands with *P. sylvestris*, oak and birch. 85% of the existing pine forest stock is found on mesic sites of which 60% are dominated by dwarf-shrubs and *Deschampsia flexuosa* L. in the field layer, another 25% have a dominant *Rubus* spc. field layer (Beck 2000, Endtmann *et al.* 1991). A similar situation can be found in the Netherlands, where the total potential area of *P. sylvestris* dominated forest types is between 2,000 and 4,000 ha, classified into the three forest types of a *Cladonia-Pinetum, Leucobrium-Pinetum* and *Empetrum-Pinetum* (Lust *et al.* 2000).
Most of the existing pine forests are intensively managed as artificial stands which replaced former natural broad-leafed forests (Wiedemann 1948, Zerbe and Brande 2003). These pine stands most often lack the capability of self-organizing and natural regeneration. Typically, the artificial stands show a tendency towards natural forest succession, with invading hardwood saplings of birch, oak and beech, especially on mesic and rich sites. On poor sites, however, a mixed pine/birch/oak-forest with several

cohorts would be the desired forest type, both from an ecological and economic point of view.

Recently, forest management in all North-West European countries places more emphasis on *'close to nature'*-management, and now natural regeneration is the desired way to manage and regenerate pine stands, especially on poor sandy soils (BMF 2001). Silvicultural systems based on heavy shelterwoods and natural regeneration are replacing traditional clearcutting followed by planting.

1.2.2 Fire and *P. sylvestris* forests

Forest management practices, which have introduced and favored single-aged pine stands in North-West Europe over the last two centuries, are facing a high fire hazard in the plantations. Until today, the natural disturbing agent *fire* is seen as the ultimate evil of pine forestry - despite the frequent fires and the tolerance of the species towards fire in its natural habitat.

Although limited scientific information about fire ecology of *P. sylvestris* in North-West Europe exists, the wide variety of responses to fire described below enables this species to cope with the current fire situation here, or even profit from it, compared to endemic tree species. More detailed knowledge on post-fire succession dynamics of pine stands is available for the boreal and hemiboreal forest zone (Viro 1974, Engelmark 1993, Engelmark *et al.* 1998, FIRESCAN 1996, Spanos *et al.* 2000).

Tree species of the genus *Pinus* have developed the most advanced means and methods to survive a forest fire either as an individual or as a species. Serotinous cones, thick buds and insulating bark coverage are the most common properties which reduce fire impacts (Rego and Rigolot 1990). The near to world-wide presence of *Pinus* under different climates and fire regimes lead to a wide variation of fire tolerance within the species. Bark insulation is the primary fire adaptation of *P. sylvestris* - the thicker the bark, the better the protection (Rego and Rigolot 1990) - with all it's consequence of this feature as a "thinning" criteria within a moderate surface fire (Kolström and Kellomäki 1993) to a long-term evolutionary selection mechanism within the species.

P. sylvestris faces mainly moderate-severity fire regimes: the occurrence causes tree mortality in the burned area, but fires are not generally stand-replacing (Agee 1998a). Fire effects and fire impact strongly depend on specific fire characteristics and on-site factors and are therefore hard to predict (Wirth *et al.* 1999). Usually, higher fire intensities and frequencies are found on nutrient richer sites, since here fuels build up faster. In Scandinavian *P. sylvestris* stands it takes at least 20 years after a fire before enough fuel builds up for a new fire to spread (Schimmel 1993, Granström 1996).

Over the wide distribution of *P. sylvestris* all varieties of fire regimes can be found:

- Near the tree line in northern boreal forests, no evidence of fire was found (Kullman 1986), mainly due to a humid climate and extensive mires.
- Near the Finnish-Russian border, the fire frequency was 1.87% (as the proportion of the area burnt per time unit) from 1679 to 1872 and 0.40% from 1873 onwards. The mean fire interval was 62 yrs and the median interval 56 yrs (Lehtonen and Kolström 2000).
- For Scandinavia, a mean fire return interval between 50 and 100 years was estimated for natural pine stands (Tolonen 1983, Agren *et al.* 1983).
- Under dry conditions in northern China and Siberia, an estimated fire-return interval of ~25 years was observed (Goldammer & Di 1990, Wirth *et al.* 1999).

In the natural range of *P. sylvestris*, fire is the most important natural factor, that determines stand dynamics (Wirth *et al.* 1999). Natural successional stages of *P. sylvestris* often include a number of large overstory trees, that survived a forest fire with several younger cohorts of pine in the middle- and understory which regenerates on a plot-scale after fire (Kuuluvainen and Rouvinen 2000, Engelmark *et al.* 1998). As a consequence, multiple-aged stands are found in many areas (Lehtonen and Kolström 2000, Zackrisson 1977, Zackrisson 1980), although even-aged natural stands also exist (Kurbatskii and Ivanova 1980 in Agee 1998a). Even-aged stands result from stand-replacing fires and an extensive regeneration. In southeast Sweden, it was shown that in stands of *P. sylvestris* with frequent fires, an open canopy was maintained in the past, but with fire-exclusion, the canopy is more closed today (Lindbladh *et al.* 2003). For Siberian *P. sylvestris* forests it was shown that fires maintain stand densities and stand biomass below the self-thinning boundary by inducing mortality in the stand. Here, the fire return interval is shorter than the time it takes for subsequent growth by surviving trees (Wirth *et al.* 1999).
Most of the understory species (e.g. *Calluna vulgaris*, *Vaccinium* spc.) are also well-adapted to fire and regenerate by sprouting. Forest floor vegetation changes by fire are minor in 40 to 70 year-old post-fire forests of *P. sylvestris* (Vanha-Majamaa and Lahde 1991).

1.2.3 Fire regimes in North-West European pine stands

Fire regimes for forest ecosystems usually are defined by the usual size of fires, the season they occur, the frequency of fire returning to a given area and their severity on the ecosystem (Tab. 1.1). From studies on natural fire regimes in fire prone areas, such as the Western United States, the successional pathway after fire strongly depends on fire severity and the size of the burn (Agee 1993, Agee 1998b, Turner *et al.* 1998). Regimes with a low severity (usually meaning small patches within the forest landscape burn) lead to a mosaic of trees and groups of trees of different ages. Fire

regimes of moderate to high severity usually create burned areas of a larger extent, which leads to larger areas covered with trees of the same age.
Historically, there was no lack of anthropogenic fire in Europe. Fire was used in pastoralism, farming and forestry intensively, and in each land-use form, a certain fire regime was applied; in some regions until the end of the 19th century (Pyne 1998).
Even today, all forest fires in North-West Europe are strongly influenced by human actions (Pyne 1997, Lex and Goldammer 2001, Ubysz and Szczygiel 2002), both directly by, for instance, arsonists as ignition sources and indirectly by factors such as fire suppression effectiveness, choice of tree species etc. The large degree of fire hazard in pine plantations, which would not occur naturally, is in itself an indirect human impact on the fire regime (Thonicke 2003). Ignition and suppression actions are direct influences of humans on the fire regime. Almost all ignitions are caused by human action (arsonists, negligence or accident); lightning rarely causes forest fire (Lex and Goldammer 2001).
On average over the last 55 years, 1,100 to 9,700 ha of forest burn annually in Germany (Lex and Goldammer 2001), 200 to 2,500 ha in the Netherlands, up to 2,600 ha in Belgium (Schelhaas *et al.* 2001) and 2,500 to 8,600 ha in Poland (Ubysz and Szczygiel 2002). The average size of a burn varies between 0.6 to 4 ha. However, there is a significant variation of area burned for these countries. Severe fire years were 1975, 1976 and 1992, with areas of up to 6,000 ha forest burned.
Two main fire seasons are found under North-West European conditions. The first period of high fire hazard is in early spring, when dead grasses are still present, quickly dry out after a few days without rain and therefore provide fuel for a fast spreading fire. Once fresh green plant material emerges in the field layer, fire hazard usually drops until late summer, when hot days dry out both field layer vegetation, dead woody material and live foliage. Most large scale fires occurred in late summer (Lex and Goldammer 2001).
Lightning is the only event that naturally starts a fire. For North-West Europe, lightning is most often accompanied with rain and therefore only a few fires are caused by lightning. Human action, either through arson or accident, are causing most ignitions.
Since all fires get extinguished by fire fighters, and considering the fact that North-West Europe is densely populated and forests are easily accessible, no natural fire duration or spread is possible as the weather situation or forest conditions would allow. Instead, we have to deal with a fire regime which is more affected by human action than by natural factors. However, ecological factors determine the post-fire successional pathway of the area burned.

1.3 Succession pathways for *P. sylvestris* stands

Two main approaches to describe the structural development of *P. sylvestris* stands have been derived over the last decades. A climax model was developed with a site-specific climax (Fanta 1995, Van Dobben *et al.* 1994), where disturbances lead to a

retrogressive stand dynamic. More recently, a pathway model was created with pathways of structural development depending on site, silvicultural treatment and disturbances (Kint 2003). However, in this model pathways after fire disturbance are not defined yet.
Post-disturbance succession usually depends on both the severity of the disturbance and the life history traits of the available species. Therefore, several interacting effects have to be taken into account to understand mechanisms and patterns of post-fire succession, especially since the effect of these two components varies according to environmental conditions on the burned site (Halpern 1989).

In the boreal and hemi-boreal zone, surface fires that are not stand-replacing occur most often in stands of *P. sylvestris* and cause overstory mortality only in areas with high fire intensity. Thus, wildfires typically create complex stand structures with patchy distribution of surviving and dead trees. The surviving trees provide seed sources for post-fire regeneration, and in these patches, natural pine regeneration creates the post-disturbance forest (Engelmark *et al.* 1998, Agee 1998a, Engelmark 1993).

In North-West Europe, natural regeneration of *P. sylvestris* was a heavily discussed topic in forest management during the 1930's. Often, mechanical site treatments were applied to improve site conditions and to stimulate regeneration (Mallik 2003, Dohrenbusch 1997, Olberg 1957, Wittich 1955). Early works by Recke (1928) and Conrad (1925) observed the beneficial use of surface fire in pine forests in Europe. These authors report about site improvements, at least short-term, that led to increased natural regeneration of pine trees on burned plots. After several unplanned fire events, such as large forest fires around Berlin during World War II, the stimulating effect of fire for the regeneration of *P. sylvestris* was observed (Klein 1964, Hinz 1993).
Main factors found to affect post-fire development of conifer forests are the degree of humus consumption and seed availability (Turner *et al.* 1998). Humus consumption determines overstory mortality, the degree of physiological stress induced to the trees on the burned site (Chapter 6), and prepares suitable seedbed conditions (Olberg 1955, Chapter 8). Seed availability on the burned site might be restricted to certain species after large scale fires, where tree species with short-range seed dispersal ranges can not colonize all parts of the burned site (Gracia *et al.* 2002). *P. sylvestris* has a limited seed dispersal range of max. 120 m from the parent tree (Dohrenbusch 1997), while seeds of *Betula pubescens* can be carried by the wind for more than several kilometers (Leder 1992).

Therefore it is important to differentiate between succession after **high severity** (stand-replacing) and **low severity** (non-stand-replacing) fires, and between high severity fires of small extent (gap-size less than 1 ha) and larger extent (Chapter 9).

However, this strong dichotomy of high and low severity and large- and small-scale fires has to be broadened – after medium severity fires or on medium size burned sites, a mixture of the specific characteristics can be expected (Gracia *et al.* 2002).

1.4 Outline of this thesis

With this thesis I set out to investigate the role of fire in the dynamics of forest succession in North-West Europe, in particular the **ecological consequences** of the disturbance "Fire". The focus of my research is on forests dominated by *P. sylvestris*, both on sites where this species would be part of the natural vegetation type and on sites where it currently replaces broadleaved tree species. My objectives are to determine which factors and mechanisms are responsible for the post-fire development of *P. sylvestris* forests and to show different pathways of secondary succession after fire.

To meet these objectives information about fuel load (Chapter 2) and fire behavior in North-West European pine stands is gathered, which allows simulations about all necessary fire characteristics (modeling of fire behavior; Chapter 3). Factors that influence humus consumption in the field are presented in Chapter 4. With this information, the fire disturbance can be described in more detail and ecological consequences can be correlated to fire intensity and severity.

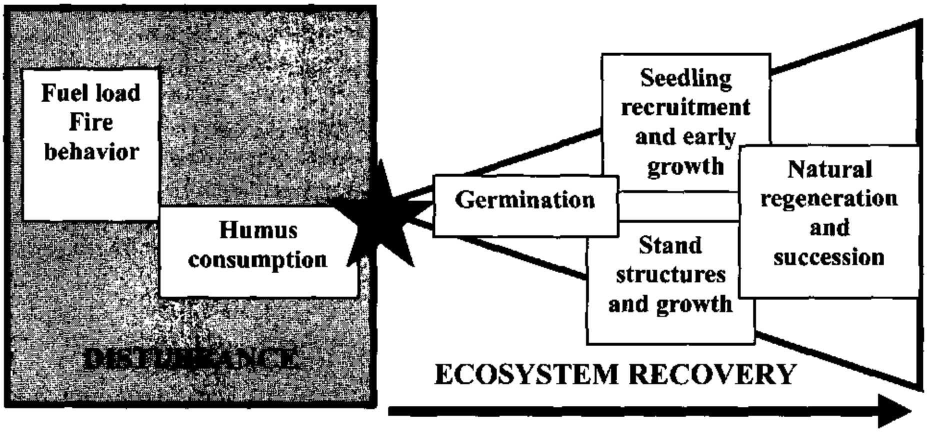


Figure 1.2. *Topics included in this study on the fire ecology of* P. sylvestris *in North-West Europe. The main focus is on the situation right before forest fire (black star) and the ecosystem recovery over time, assessing pre-fire situations, and resulting post-event effects and succession dynamics.*

Secondly, ecological fire effects and recovery processes are assessed. Emphasis is laid on the fire-induced change in stand structures (Chapter 6) and the influence of fire on radial tree growth (Chapter 5). The effect of charcoal on post-fire germination dynamics (Chapter 7) and seedling growth after fire (Chapter 8) are presented to show mechanisms that determine post-fire succession.
From this, forest dynamics after small scale, high severity fires comes into the focus (Chapter 9).

Finally, the potential future role of fire in forest management and nature conservation in North-West Europe is evaluated, taking the results of the previous chapters into consideration. Potential fields in which fire effects cover with forest management objectives are identified and silvicultural models which include fire are developed (Chapter 10).

With this set-up, the pre-fire situation (fuel load) and several ecological fire effects after fires of different intensities are put into one picture, leading to the first approach to describe the fire ecology of *P. sylvestris* in North-West Europe.

CHAPTER 2

Fuel load and humus consumption

Published as: HILLE M, DEN OUDEN J. 2005. FUEL LOAD, HUMUS CONSUMPTION AND HUMUS MOISTURE DYNAMICS IN CENTRAL EUROPEAN SCOTS PINE STANDS. *INT J WILDLAND FIRE* 14: 153-159

Abstract

Samples of Scots pine (*Pinus sylvestris* L.) humus were burned under different moisture and fuel load scenarios to model humus consumption. For moisture levels below 120% on a dry mass basis, a parabolic increase of humus remaining with increasing moisture content was observed, while for higher moisture levels up to 300%, humus was reduced by a constant 10-15% on a dry mass basis. Both fuel load and humus moisture had a highly significant influence on humus consumption.

Humus gross calorific value of Scots pine (19 509 KJ kg^{-1}) is lower than that of other pine species. We found a desorption time-lag for humus moisture of 85 h in this study.

Field data show a steady accumulation of humus in Central European Scots pine stands (up to 45 t ha^{-1} in 120 yr old stands). Amounts of litter remain constant over the different stand ages (~15 t ha^{-1}).

This study provides important information to predict humus consumption in Scots pine stands. The results can be used to build a fire severity and post-fire succession model for Scots pine stands in Central Europe.

2 Fuel load, humus consumption and humus moisture dynamics in Central European Scots pine stands

2.1 Introduction

Fire plays a significant role in western and central European forestry, although it does not receive as much attention from science and the general public as in other parts of the world. Especially in regions where Scots pine (*Pinus sylvestris* L.) grows on poor sandy soils (e.g. the *Veluwe* and *Kempen* regions in The Netherlands and Belgium, or the *Lausitz* region and the *Lüneburger Heide* in Germany) fires of varying size and intensity occur each year (Schelhaas *et al.* 2001). Current scenarios of climate change predict more summer droughts and thus even increased fire danger and higher fire frequency can be expected for the near future (IPCC 2001, Badeck *et al.* 2004).
For Scots pine, a moderate to high severity fire regime is observed in its natural range in the boreal forests of Northern Europe, with severity mainly determined by crown damage and humus consumption (Agee 1998a). In general, the degree of humus consumption during wildfires or prescribed burnings strongly influences the post-fire succession of the site (Johnson 1992). In this paper, humus is defined as the O_F and O_H -layer between the litter (the sum of surface fuels, not decomposed needles and cones; O_L) and the mineral soil. Prolonged heat release by smoldering combustion kills fine roots of herbs, shrubs but also overstory trees, which can lead to high mortality (Ryan and Frandsen 1991, Stephens and Finney 2002). As a consequence, species composition can be altered and seed sources for a quick re-colonization will be absent. With a higher consumption of the humus layers, stored seeds and vegetative organs are also destroyed (Schimmel and Granström 1996). In the case of partial consumption of the humus layer, heat tolerance and storage depth of certain species plays an important role in re-colonization pattern of the site (Granström and Schimmel 1993, Whittle *et al.* 1997). Furthermore, in the case of complete humus consumption, only species with roots and vegetative organs in the mineral soil can survive. In the boreal forest, complete humus consumption was found to favor the establishment and growth of coniferous seedlings (Chrosciewicz 1974, Zasada *et al.* 1983). The amount of humus left on the site also influences species composition, density and early growth in the seedling cohort following the fire (Thomas and Wein 1985, Johnson 1992, Hille and Den Ouden 2004).

Humus consumption proceeds mostly by slow smoldering combustion - portions of the O_F-layer can be consumed by flaming combustion, though - with the amount of consumption mainly depending on humus characteristics. In previous studies it was found, that humus moisture has an overriding influence on humus consumption (McArthur and Cheney 1966, Sandberg 1980, Brown *et al.* 1985, Frandsen 1987, Brown *et al.* 1991, Reinhardt *et al.* 1991, Robichaud and Miller 1999). The stored moisture results in a latent heat flux for evaporation of the water, which provides an effective heat sink, stopping smoldering combustion. Other factors, such as packing ratio, humus depth, and amount of surface fuel load also influence humus consumption (Burgan and Rothermel 1984, Miyanishi 2001, Miyanishi and Johnson 2002).

This study aims to provide information on parameters that influence humus consumption by forest fire in Central European Scots pine stands. The first part assesses current load of litter and humus in pine stands of different age. The second part deals with the consumption of pine humus under several moisture and fuel load scenarios. Finally, we look at moisture dynamics for Scots pine humus and, in a review, at the effect of field layer vegetation on humus moisture and reasons for temporal and spatial variation of humus moisture in pine stands.

2.2 Methods

2.2.1 Fuel load in Central European Scots pine stands

Litter and humus load samples were taken in 18 pure pine stands on poor sandy soils in the Uckermark region, app. 80 km north of Berlin, Germany. These stands are managed for timber production under reference of traditional yield tables (Lembcke *et al.* 1975), which are based on high stocking levels and a rotation period of 120 yrs. The sampled stands were between 20 and 120 yrs old and not thinned for at least five years. Stand parameters such as stem density and basal area deviated less than 10% from these yield tables, with a stem density of 1,340 ha^{-1} and a basal area of 30.4 m^2 ha^{-1} in 50 yr old stands and 435 ha^{-1} and 32.4 m^2 ha^{-1} after 100 yrs, respectively. Fuel load was sampled using the line transect method of Brown (1974), where on a 10 m line all down woody debris is tallied in the standard fire size classes of 1-hour (0-0.64 cm diameter), 10-hour (0.64-2.54 cm), 100-hour (2.54-7.62 cm) fuels (Byram 1963). This classification is based on the time-lag and the corresponding diameter of the fuel. The time-lag is defined as the time needed under specific conditions for a fuel particle to loose about 63.3% of the difference between its initial moisture content and its equilibrium moisture content (NWCG 1996).

Within each stand, ten sample points on a randomly placed 25 by 25 m grid were established. From each sample point, four line transects were laid out in north, east, south and west direction. To be able to calculate litter and humus load on a per hectare basis, samples of 71 cm^2 were taken with a metal cylinder at 0.3 and 1.0 m on the line transect down to the mineral soil, dried at 110° C until they reached constant weight.

2.2.2 Humus consumption by fire

In a mature 80-year old pine stand in Northern Germany humus was collected at ten randomly selected locations, where all organic material between the litter and the mineral layers was collected on an area of 0.7 m^2. The organic material from all ten locations was mixed, then split into amounts of approximately 1,000 g and exposed to 40 different moisture regimes, ranging from oven-drying to storage under wet conditions. Pine cones, big pieces of bark and woody material were removed from the samples.

To simulate field conditions but also to burn the humus under replicable conditions, tin cans (16 cm high and 10 cm in diameter) were filled partly with moist sand, on which smaller samples of the humus were added to fill the top 5 cm of the can. For each moisture regime, the same dry humus volume was used, the actual humus weight varied for the moisture regimes from 25 to 150 g. Bulk density of the created fuel bed for all moisture regimes was within ± 15 % of the bulk density found in the stands which were sampled as described in the previous section.

To simulate different fuel loads and energy inputs, between one and three charcoal lighters (produced by *Landmann GmbH&Co KG, 27711 Osterholz-Scharmbeck, Germany*) were placed on top of the humus and ignited. During the burn, a steady airflow was provided by the exhaust hood in an experimental chamber. The experiment was terminated when neither radiating heat nor smoke was observed. After that, weight loss of the humus was recorded to monitor the degree of consumption.

In addition to the samples that were ignited, a fourth sample of the same material was put into a drying oven at 110°C until it reached a constant weight to determine dry weight and from this the moisture content of all four samples.

In a pre-study calorimetric analysis (according to standardized method *ASTM D5865-04*; http://www.astm.org), a mean heat content (± SD) of 20,200 ± 68 KJ kg^{-1}, 19,509 ± 89 KJ kg^{-1} and 29,305 ± 145 KJ kg^{-1} for pine litter, pine humus and charcoal lighters, respectively, were determined. The used amounts of charcoal lighters can be correlated to the amount of pine litter (as a mixture of needles, twigs, pieces of bark and cones) with the same heat content. For the charcoal lighters of 111 g weight, and thus a heat content of 325 KJ per piece, this corresponds to 161 g pine litter. Extrapolating the area of the tin cans used in this study to a hectare basis, one charcoal lighter used in the humus consumption experiment, is the calorific equivalent to a pine litter fuel load of 2,221 t ha^{-1}.

Humus consumption was modeled with the variables *moisture content, fuel load, bulk density* and their squares, using linear regression procedures with backward elimination of not significant variables using SAS 8.2.

2.2.3 Moisture dynamics

To estimate the drying rate of pine litter and humus under changed environmental conditions, three samples of forest floor from the top of the O_L down to the mineral soil (3 cm litter on top of a 9 cm deep humus layer) were taken from the same 80-year old Scots pine stand as in the humus consumption experiment. We tried not to disrupt the physical structure of the samples by carefully trimming the litter and humus against the edge of 30x30 cm cardboard boxes. The samples were then stored under indoor lab conditions (constant at 20° C, 50% relative humidity). The weight of the samples (litter and humus) was recorded at regular intervals for 200 h. These values were transformed to a non-dimensional variable (*'Normalized moisture', NM*) using Equation 2.1 (Fosberg 1977).

$$NM = \frac{M\% - M_{fin}\%}{M_{init}\% - M_{fin}\%} \qquad \text{Equation 2.1}$$

with M% as the actual humus moisture and $M_{fin}\%$ and $M_{init}\%$ as the final and initial humus moisture, respectively. On a log-linear graph this ratio should approximate a straight line.

2.3 Results

2.3.1 Fuel load

Litter load in Scots pine stands over the entire age range of sampled stands from 20 to 120 yrs (n=19) were constant at around 15 t ha^{-1} (LITTERLOAD=0.0225AGE + 13.95, F=0.45, p=0.51, adj. R^2=0.03; Fig. 2.1), ranging between 8 to 23 t ha^{-1}. Humus loads increased with higher stand age and fitted to a logarithmic function (HUMUSLOAD=9.95 ln(AGE) - 12.98; F=6.86, p=0.017, adj. R^2=0.36*). For stands younger than 30 yrs, the average humus load was lowest with 18 t ha^{-1} on average. For older stands, the humus load reached values of up to 44 t ha^{-1} (Fig. 2.1). Average humus bulk density was 0.154 g cm^{-3} on average with a standard deviation of ± 0.028 g cm^{-3}.

The amount of surface fuels between different stands showed a higher variation than litter and humus loads. Fuel load of fine fuels (1-h and 10-h) is around 1.5 t ha^{-1} on average for all stand ages and was independent from stand age (ANOVA, F=0.38, p=0.88 for 1-h fuels and F=4.32, p=0.36 for 10-h fuels). The load of larger fuels (100-h) increases to up to 8.8 t ha^{-1} for stands around 75 yrs and was influenced by stands age (ANOVA, F=357.58, p=0.04). In stands below an age of 30 yrs, no 100-h fuels were found. The highest load of 100-h fuels was found in stands between 35 and 80 yrs of age (Fig. 2.2).

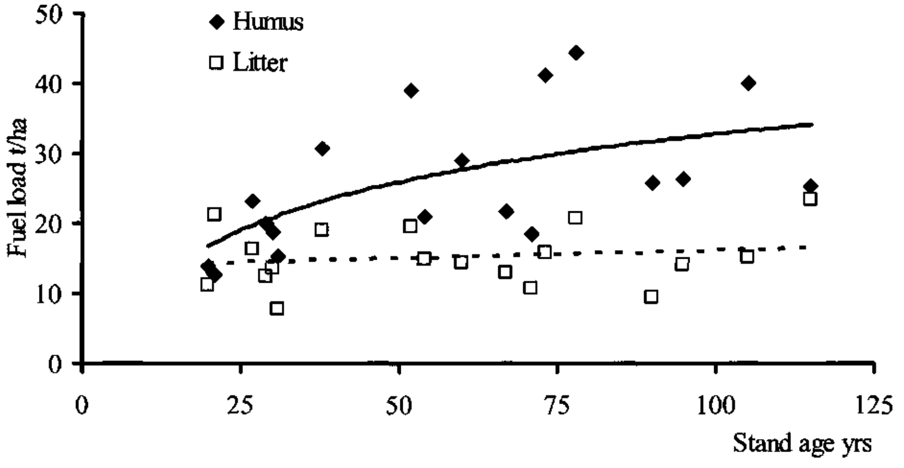


Figure 2.1. *Litter and humus loads for 18 Scots pine stands of different age in the Uckermark region, Germany. All stands are managed according to current yield tables, with thinning operations every 5 to 10 yrs. Regressions for humus (bold line) and litter (dotted line) included.*

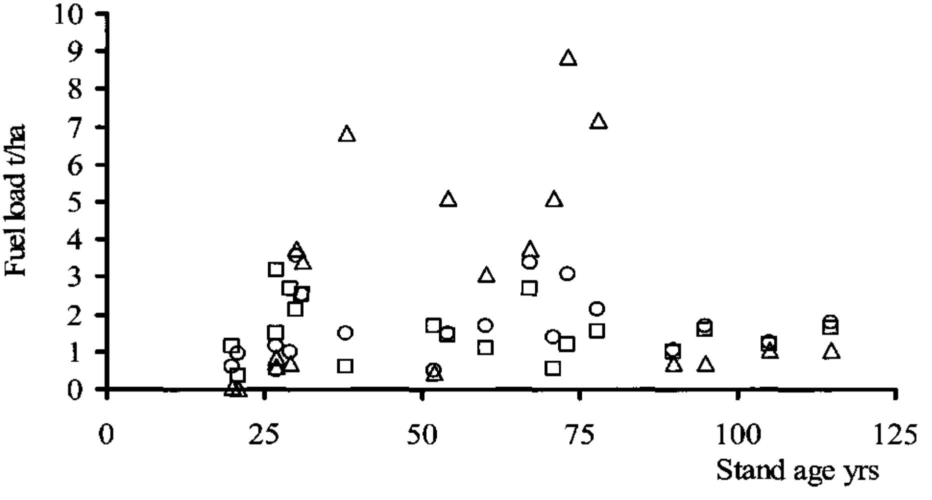


Figure 2.2. *Fuel loads (t ha^{-1}) for 18 Scots pine stands in the Uckermark region, Germany. Fuels were classified by time-lag class (1-h □, 10-h ○, 100-h Δ).*

2.3.2 Humus consumption by fire

Extensive smoldering and humus weight reduction was observed at moisture contents below 120 %. Above that level, the weight loss was marginal (10-15 %).

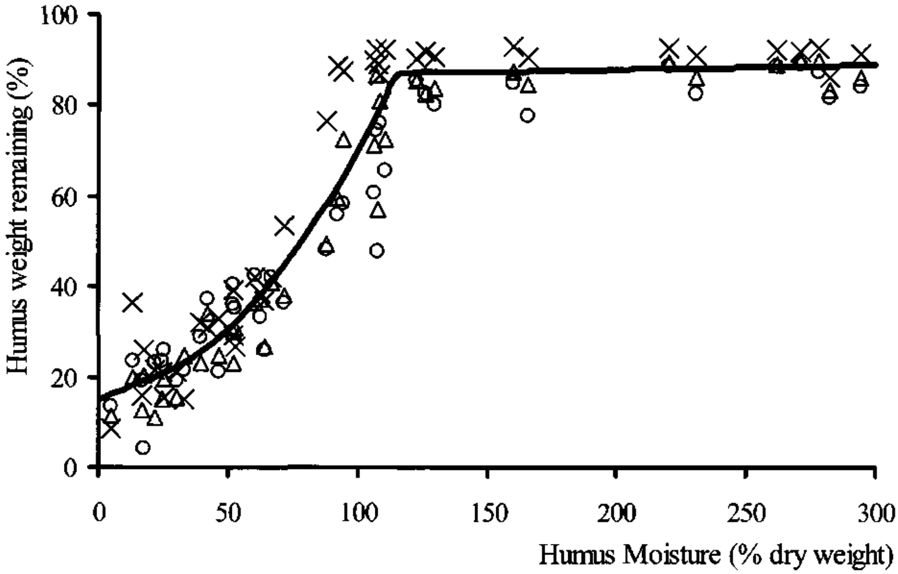


Figure 2.3. *Humus consumption (weight percentage of humus remaining) at different humus moistures. Different fuel loads were simulated with one (×), two (Δ) or three (○) bars of charcoal lighter as an ignition source. The lines show the parabolic relation for humus moisture <120% and an almost constant humus consumption percentage for higher humus moisture.*

For regression of humus consumption data, a combination of a parabolic (for low moisture contents) and a linear function (for high moisture contents) was used (Tab. 2.1). The transition from a parabolic to a linear function was determined by starting with a parabolic regression for data points below 75 % moisture content and adding higher moisture data values until the adj. R^2 of the regression declined. This transition occurred at 120 % humus moisture. From that point on, a linear function was used to fit the data (Fig. 2.3). Regression coefficients were calculated to predict humus consumption from humus moisture (HM), fuel load (FL) and their squares (Tab. 2.1). For the regression, fuel load was transformed from numbers of charcoal lighters to pine litter in t ha^{-1} by their calorific value. Bulk density had no significant influence below ($p=0.689$) or above 120 % ($p=0.965$) and was eliminated in the first regression step.

Table 2.1. *Regression coefficients to predict the weight percentage of humus remaining. The transition zone at 120% determines the break between the parabolic function for low humus moisture content and the linear function for higher moistures. HM=humus moisture, FL=Fuel load (number of fuel sticks).*

Humus moisture	Intercept	HM^2 (%)	HM (%)	FL^2 ($t\ ha^{-1}$)	FL ($t\ ha^{-1}$)	Adj. R^2
<120	40.47***	0.00524***	-	0.888**	-9.92***	0.88
>120	67.10***	-	0.20***	-	-1.59***	0.66

***=significant at the 0.001 level, **= significant at the 0.05 level

2.3.3 Moisture dynamics

The desorption curves for the 9 cm deep Scots pine humus and litter layer are curvilinear rather than strictly linear (Fig. 2.4), similar to the desorption lines of other woody material (Nelson 1969, Mutch and Gastineau 1970). The initial moisture content (M_{init}%) was 291 % on average and a standard deviation of ± 11 % and the final moisture content (M_{fin}%) was 98 % ± 11 %. The time-lag for the combination of Scots pine litter and humus, as the time required to reach 63.3 % of the difference between the initial moisture content and the equilibrium moisture content, was 85 h (Fig. 2.4).

2.4 Discussion

The litter load in managed stands of Scots pine is only slightly influenced by the age of the stand. Similar values for fine surface fuels and litter were found in stands from 20 to 120 yrs (Fig. 2.1; 2.2). This indicates that the production of fine fuels and needles in the crowns is balanced with their decomposition rate. However, the load of larger diameter fuels (100-h) seems to be highest when stem exclusion affects large trees between 35 and 80 yrs. Of course, presence of large diameter dead wood is directly influenced by the harvest of thinned trees. Dwarf-shrubs, mosses and grasses represent additional fuels on the sites. These plants can contribute significantly to the total fuel load (Hille and Goldammer 2002), but are strongly site dependent and have high variation in moisture content, so that they can serve as a heat sink or source during a fire.

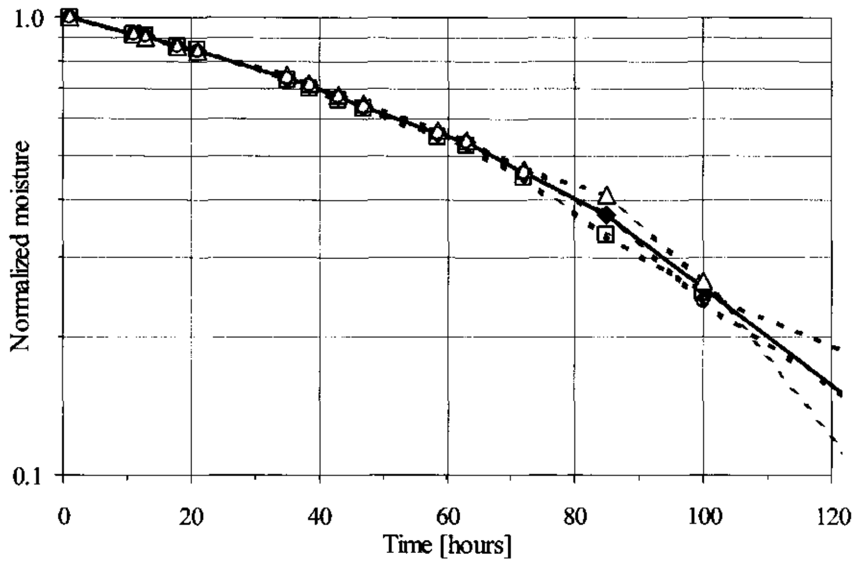


Figure 2.4. *Desorption of the three Scots pine litter and humus samples (dotted lines) on a normalized log-transformed moisture scale, and the average desorption (solid line). Average time-lag by the definition of Byram (1963) is 85 h.*

For the humus load, a logarithmic accumulation over time for stand age up to 120 yrs was observed. However, we expect an asymptotic increase of humus load with higher stand age, but unfortunately we didn't find pure pine stands older 120 yrs in the region we sampled. The O_H and O_F layers added up to 44 t ha^{-1} in 120 yrs old stands.

The use of charcoal lighters to simulate different fuel loads is a new approach for humus consumption studies. A correlation of number of lighters with the litter load in the field by energy content allows a controllable and replicable study. While the litter is a highly variable mixture of needles, twigs and cones, the charcoal lighters are uniform in size and energy content. However, a direct comparison to field situations should be done carefully. Although the calorific value of charcoal lighter and the corresponding amount of pine litter can be calculated - the mean litter load of ~15 t ha^{-1} (Fig. 2.1) would correspond to 6.75 lighters - the litter represent a much larger bulk, and thus affects a much larger area than sticks. Also, the charcoal lighters are more similar to twigs, bark pieces or cones in surface area and burn-out time than fine needle fuels. The longer burn-out time of the lighters may have given the moist humus the opportunity to partially dry and burn - similar to burning woody pieces on the forest floor with contact to the humus layer.

The relationship between percentage humus weight reduction and moisture of pine humus is similar to that of other conifer species (Brown *et al.* 1985). Even under high moisture contents humus is consumed (10 to 15 % of the dry mass), assuming that a spreading fire and therefore the consumption of the litter layer is possible. While for several North American conifers humus consumption remained constant on a low level at moisture contents above 150% (Brown *et al.* 1985), this threshold is reached already at 120% for Scots Pine. This might be due to the lower heat content of Scots pine humus (19,509 KJ kg^{-1}) compared to e.g. Douglas fir (*Pseudotsuga menziesii* Franco; 22,600 KJ kg^{-1}; Frandsen 1987) or Ponderosa pine (*Pinus ponderosa* Laws; 20,370 KJ kg^{-1}; Van Wagtendonk *et al.* 1998*a*). In contrast to the findings of Brown *et al.* (1985), our regression for low humus moisture is not linear, but has a parabolic shape. Bulk density of Scots pine humus (0.154 g cm^{-3}) is similar to that of *Pseudotsuga menziesii* (0.153 g cm^{-3}) and *Pinus ponderosa* (0.155 g cm^{-3}; Van Wagtendonk *et al.* 1998*b*) and did not have a significant influence on humus consumption in this study. Our modeling approach with two distinct regression (parabolic below 120 % humus moisture and linear above) follows the approach of Brown *et al.* (1985), who also used two regression functions. Although S-shaped function might fit the data better and won't have a discontinuity, our approach allows a simple prediction with process-determining parameters instead of dimensionless values.

Even though the fast drying litter layer was included in the weighting of the samples, the calculated average desorption time-lag of 85 hrs is higher than for humus of other pine species. Scots pine humus is therefore less reactive to changes in the milieu than e.g. layers of humus of *Pinus ponderosa* of similar thickness (time-lag of 50 h; Fosberg 1977, Anderson 1990). However, drying rates of the upper organic layer in boreal pine stands are about 120 hrs (15 days with app. 8 hrs drying per day), as modeled in the Canadian Fire Danger Rating System (Lawson *et al.* 1997). With this knowledge about the drying rate of Scots pine humus and litter (Metz 1958, Wittich 1998a) a general model can be built to predict forest floor moisture dynamics under a given weather scenario. However, one should note, that due to the removal of the litter/humus samples from the forest floor no horizontal moisture transport was possible and that our results are valid for one set of atmospheric temperature and relative humidity only.

The regression of humus consumption (Fig. 2.3 and Tab. 2.1) is separated into two distinct zones with strong dependence on humus moisture. Due to the concave shape of the regression curve below 120 % humus moisture, small changes in humus moisture can lead to large changes in the humus consumed, particularly between 50 and 120 % moisture content, which can lead to differences in overstory mortality and impact on seed survival. Spatial and temporal variation in humus moisture can therefore cause strong variation of humus consumed and consequently, with-in site variation in post-fire succession.

2.4.1 Temporal and spatial variation of humus moisture

The temporal variation of humus moisture is strongly correlated to rainfall and throughfall dynamics. Time series from Schaap *et al.* (1997) showed a variation of humus moisture during a summer period in a Douglas fir (*Pseudotsuga menziesii* Mirb. Franco) stand of 282 % on a dry mass basis. Spatial variation of humus moisture is partly caused by crown-structures of overstory trees (Beier *et al.* 1992). Humus is drier in the vicinity of the stem base and becomes increasingly more moist at further distances from it (Zinke 1962, Ziemer 1968, Chrosciewicz 1989, Bruckner *et al.* 1999). The highest moisture contents were found in stand openings, while under crown cover the humus was driest (Möttönen *et al.* 1999). Water drains off the conifer crowns and drops down at their edges (Bouten *et al.* 1992, Otto 1994, Whelan *et al.* 1998). In general, spatial variation of throughfall is reduced with increasing rainfall intensity (Llorens *et al.* 1997). During rainfall events with only little precipitation, the majority (60 %) of the water is intercepted before it reaches the forest floor. With higher rainfall rates, the percentage of water that is intercepted is smaller (15 %; Rutter 1963).
Humus depth is also largely influenced by tree position, with deeper humus layers in the vicinity of the trees (Hokkanen *et al.* 1995) and a strong spatial auto-correlation for humus depth at lag-distances of 0.5 m (Smit 1999). It can therefore be expected, that the highest humus consumption and therefore the strongest fire severity takes place in the vicinity of stem bases, as was observed in mixed-conifer stands in the Sierra Nevada, California (Hille and Stephens 2005), and in the boreal forest (Miyanishi *et al.* 1999).

2.4.2 Field layer vegetation and humus moisture

Humus moisture is highly variable under field conditions, and does not only depend on weather factors. Clerkx and van Hees (1993) showed the influence of different field layer vegetation types on humus moisture in Scots pine stands. In their work, they sampled humus moisture during the vegetative period in 10 to 14-day intervals under different types of vegetation. Humus moisture was lowest at plots without vegetation. For plots with plant cover, their samples beneath grass (*Deschampsia flexuosa* L.) were significantly drier than beneath heather (*Erica tetralix* L.), which was explained by reduced evaporation.
During their entire sampling period, humus moisture varied between 30 and 210 %. Most often the humus moisture was within the range of 70 to 130 %, and thus in the moisture class where humus consumption is strongly influenced by fuel load and moisture content (Fig. 2.3).
Therefore, one could expect a lower humus consumption beneath areas which are covered with dwarf-shrubs or grasses. On the other hand, dense forest floor vegetation contributes to fuel load, especially outside the vegetation period, when tissue moisture content is low (Mißbach 1973).

2.5 Conclusions

The dynamics in humus moisture are crucial in determining and predicting fire severity and post-fire site-developments. In this study we showed that small changes in humus moisture result in large changes in the humus consumption for moisture situations often found in the field. Any factor that determines humus moisture therefore has an impact on humus consumption and, consequently, fire severity and post-fire succession. The use of charcoal lighters as a heat source with known calorific value to start smoldering humus consumption allowed a replicable study. However, a direct comparison to field situations should be done with caution.

Humus moisture is mainly determined by rainfall events, crown coverage and type of field layer vegetation. Climate change (e.g. drier summers in Central Europe) and nitrogen inputs, which cause changes in field layer vegetation (e.g. invasion of grasses in many open forest types) are expected and will have a significant effect on humus moisture.

For future research, a link between humus moisture and current litter moisture models (van Wagner 1987; Wittich 1998a; Ogee and Brunet 2002) could allow the estimation of the humus moisture content and from this, the degree of humus consumption could be modeled. From this, the fuel moisture models, which are used to give a fire hazard rating could be developed to give an ecological hazard rating, taking into account the chances of a fire occurring (calculated from fuel moisture) and the impact of potential fire (humus consumption and fire severity).

As of yet, only few data are available on properties and mechanisms of humus moisture dynamics in Scots pine stands in Europe. Our results can be used to predict humus consumption by fire for different fuel loads and moisture contents in Scots pine stands. This can add to the development of models predicting, overstory mortality, propagule survival and the establishment of tree-regeneration after a forest fire. For the boreal forest, some vegetation models with humus remaining after fire as the key input factor already exist (Thomas and Wein 1985). Our study, for example, indicates that a fire that occurs at average humus moisture levels of 50 % can be expected to reduce humus mass by 70 % (Fig. 2.3). In such a case, only seeds buried in the mineral soil would survive the fire and species which prefer mineral soil for germination would have a good chance to establish. However, further research on the fire ecology of Scots pine under Central European conditions is necessary for this detailed prediction.

CHAPTER 3

Fire modeling

GOLDAMMER JG, HELD AC, HILLE M, WITTICH KP, KUEHRT E, KOUTSIAS N, OERTEL D. 2004. AN INNOVATIVE CONCEPTUAL MODEL OF A FOREST FIRE MANAGEMENT INFORMATION AND DECISION-SUPPORT SYSTEM FOR BRANDENBURG STATE.

Paper presented at the Northeast Asia forest fire International Symposium 5-6 March 2004. Korea Forest Research Institute.

Abstract

Research and development conducted within the Forest Fire Cluster of the German Research Network on Natural Disasters is built on a number of separately evolved concepts that were integrated in a cooperative research project. The Forest Fire Cluster has the responsibility of three major components. The first component consists of an innovative conceptual model for a fire information system and decision-support for early warning, monitoring, information management and simulation of wildfires in pine forests of Brandenburg State, Germany. The second component provides the link between the locally applicable system and a global fire information system provided by the Global Fire Monitoring Center (GFMC). The third component includes modeling of historic occurrence and future trends of fire occurrence due to regional climate change and is implemented by an associated project of the Potsdam Institute for Climate Impact Research (PIK), and it is published separately.

The first component is composed by a number of different modules. Firstly, it includes the adaptation of established fire behavior simulations models (BEHAVE, FARSITE) implemented by the Fire Ecology Research Group. For the first time a fire behavior model has been applied for the specific conditions of pine forests in the eastern, continental part of Germany, including the interspersed heathlands that constitute an important carrier of a wildfire at landscape level. The characteristics of these forests are quite typical for temperate-hemiboreal pine forests of Eurasia. Secondly, it includes a fire detection component (Automated Fire Detection System - AWFS) implemented by the German Aerospace Center (DLR). The development of the AWFS meets the requirements for fast, cost-effective and reliable fire detection system. And thirdly, it includes a fire danger rating and forecast system implemented by the (German Meteorological Service - DWD). The national fire-danger rating system has consolidated during the project lifetime. During the research project the work of the Global Fire Monitoring Center (GFMC) constituted the link from national to international levels.

The value added by the research project is a mutual support of individual research projects and their final merging into a comprehensive decision-support tool for fire management. Insight gained by the research project concerning the operational use of satellite remote sensing information in the management of active wildland fires will be useful for the development of urgently needed operational spaceborne fire systems.

3 An innovative conceptual model of a forest fire management information and decision-support system for Brandenburg State[1]

3.1 Introduction

The current high probability of forest fire occurrence in the region Brandenburg, resulting in part from low precipitation, sandy soil sites with low water-holding capacity, and the fire hazard of the prevailing fire-prone pine forest stands, might further increase due to climatic change.

The cluster "Forest Fire" within the German Natural Disaster Research Network (DFNK) analyses current fire hazards and provides tools required for advanced operational decision support for wildfire response. This cluster research has the responsibility of three major components. The first component consists of an innovative conceptual model for a fire information system and decision-support for early warning, monitoring, information management and simulation of wildfires in pine forests of Brandenburg State, Germany. This component includes the adaptation of established fire behavior simulations models (BEHAVE, FARSITE) implemented by the Fire Ecology Research Group, a fire detection component (Automated Fire Detection System - AWFS) implemented by the German Aerospace Center (DLR) and a fire danger rating and forecast system implemented by the (German Meteorological Service - DWD). The second component provides the link between the locally applicable system and a global fire information system provided by the Global Fire Monitoring Center (GFMC). The third component includes modeling of historic occurrence and future trends of fire occurrence due to regional climate change and is implemented by an associated project of the Potsdam Institute for Climate Impact Research (PIK); the report of the third component is published separately.

Accordingly, the structure of this work follows this general cluster scheme and it is presented in sections each one corresponding to a specific issue raised in each component. To help the readers follow we briefly present this general scheme which it is distinguished into the research components for building a fire information system and the implementation of the fire information system. The former consists of three modules that is fire behavior simulation models, automated fire detection system, and fire danger rating and forecast system.

[1] This paper is based on the results of a multi-disciplinary fire experiment near Cottbus, Germany in 2000, coordinated by JG Goldammer. M. Hille was responsible for the fuel load measurements, the creation of a fire behavior model and the fire modeling approach.

The concepts, methods and results of each one of these modules are presented in details in the following corresponding sections. The implementation of the fire information system operates as an umbrella and intends to put together all modules by providing a common basis for their requirements and needs. Although at this stage this system has not been fully implemented yet, it theoretically describes various issues and intends to provide a common basis for the requirements of each module.

In principle wildland fires as well as their causal factors exist and function inside the spatial and temporal domain. Specific characteristics of space and time define the spatial and temporal scale over which wildland fires are approached and studied. It is very clear that answers at specific questions require appropriate data at appropriate scales to allow their meaningful processing and interpretation.

Time, as one of the scale dimensions, shows how the variation of the information of a process fluctuates throughout the temporal domain. The process itself is static or dynamic. It is clear that topography, when viewed at certain temporal and spatial extents, presents a completely stationary behavior compared to weather which is a very dynamic process. Under this context, the primary data or observations and the tools needed to acquire, maintain and introduce this information to a fire information system are very diverse and must cover different circumstances. One of the basic principles that should be satisfied by a modern and well-organized fire information system is the ability to process and use information corresponding to such dynamic and highly variable processes. We focus especially on dynamic aspects because they are more complicated than the static ones and need special tools for manipulating and introducing the updated information.

The spatial scale is another important dimension that refers to and controls many and different aspects of such processes. By using the term spatial scale we describe specific characteristics of the spatial properties of the data and information. According to Lam and Quattrochi (1992) and Cao and Lam (1997), spatial scale describes the map or cartographic scale, but also the geographic or observational scale (i.e. spatial extent of the study area), the operational scale (i.e. the hierarchical level at which the process operates) and finally the measurement scale (i.e. spatial resolution defined by pixel size in remote sensing images). Under this perspective, spatial scale is a critical aspect of the data that should be well defined and clear, be similar among different geographical data layers to assure the necessary compatibility and be also relevant to the objectives, purposes and operational level that the fire information system aims at.

Besides all these issues, the content of a forest fire management information and decision-support system is another critical aspect; what is aimed at by its implementation. What it is needed is a multi-function based system to cover not only the fire behavior modeling-simulation part but to cover also various research and applied issues in wildland fire management. This fire information system could be also utilized as a warning solution by estimating fire risk potential given that a network of weather stations can be online connected with it. The ability to simulate a real process under hypothetical scenarios can help to acquire a prior knowledge about the effects and outcomes resulting from such processes and contribute for a better fire management and planning. An optimized dispersal of fire fighting forces especially under conditions of limited resources can be achieved by utilizing a prior knowledge of fire behavior acquired by the simulation.

The fire simulation provides a lot of insight of what has to be expected from a particular fire situation, not only in terms of the physical parameter of a fire (rate of spread, intensity). The simulation on a landscape scale allows the prediction of fire direction and its behavior in the field setting. Dispatching of fire fighting resources will be made much easier and effective.

3.2 Forest and heathland characteristics in Brandenburg, Experimental site of 2001

The Forest Fire Experiment 2001 was conducted at various forest stands that have characteristics typical for extended pine forest stands in Brandenburg State, Germany. For the development of a fire behavior model specific data are essential. As data like fuel load, rate of spread, flame length, temperatures and fire weather were not existing, live burning experiments were conducted to collect these input-data.
The experimental sites of 2001 are owned by Vattenfall Mining Europe (former Lausitzer Braunkohle AG) open-cast coal mining enterprise near the city of Cottbus (51° 47′ 03″ N, 14° 24′ 20″ E). The location and characteristics of the experimental plots, each between 0.3 ha and 1 ha surrounded by a clearcut buffer zone, provided suitable conditions in terms of safety for an experimental forest fire. Three of the plots were up to 100 years old low-productivity Scotch Pine (*Pinus sylvestris* L.) stands with minor dimensions, typical for the region. The fourth plot was a 15-years old *P. sylvestris* stand. The fuel bed at all four plots consisted mainly of grass (*Calamagrostis* spp., *Deschampsia* spp.), forest litter and dead downed woody material. The fuel load (available fuel for the experimental fire) varied from 5 to 15 t ha^{-1}.

For the validation of a heathland fire model experimental fires were conducted in continental heathlands (*Calluna vulgaris* (L.) Hull.) in the Federal Forest Service District Lausitz in summer 2002. For these experimental fires the Federal Forest Service provided three plots (0.5 ha each) with homogeneous *C. vulgaris* cover.
The purpose of a heathland model was to include the heathland-forest interface in the decision support system in case of catastrophic wildfires. Open sites covered by heather vegetation located between forest complexes are suitable to rapidly carry a wildfire from a burning forest to the adjoining forest stand. The fuel loads on the heathland plots ranged between 9 and 15 t ha^{-1}.

Both experimental areas are located in the south-eastern part of Brandenburg State, a region with a very low level of precipitation and sandy soils with little water storage capacity. The climatic conditions for both experimental sites are as follows:

Climatic zone:	medium-dry lowland climate
Average temperature (Cottbus):	8.8° C
Average annual temperature scale:	19.3° C
Average annual precipitation (Döbern):	627 mm
Precipitation during the vegetation period:	316 mm

The combination of the site characteristics with the inherent characteristics of the *P. sylvestris* stands and the *C. vulgaris* ecosystems result in a high wildfire hazard. Figure 3.1 provides a scene of the forest fire experiment conducted for the research project in 2001 that shows the spatial arrangement of surface and live crown fuels that lead to high-intensity crowing fire.

3.3 The research components for building a fire information system

3.3.1 Fire behavior simulation models

Since no adequate models exist to describe fire behavior under central European conditions, models developed and successfully applied in other regions had to be used and adapted. The standard software BEHAVE developed at the U.S. Forest Service Intermountain Sciences Laboratory (Rothermel, 1972) provided an appropriate tool, especially since they are representative for homogenous ecosystems and fuel arrangements. Predicted fire behavior parameters from the BEHAVE model were compared with those observed in Brandenburg's pine forests and heathland fires.

In a subsequent step, a fire dispatching and modeling system was created with FARSITE (Finney, 1998) at the forest district level. The FARSITE model contains the same algorithms and formulas as BEHAVE, but can be used to simulate fire on a range of landscape features with different fuel models using a GIS-approach. Thus, the data have to be prepared in raster format. The input data sets contain information about elevation, slope, aspect, fuel type, crown closure, stand height and crown bulk density (Finney, 1998). The fire itself is modeled as a moving elliptical wave, the shape of this ellipse is determined by wind and topography (Huygens's principle, cf. Richards, 1990, 1995).

The work presented in this section describes the construction and testing of new and appropriate fuel models through fuel inventory and comparisons of predicted versus observed fire behavior parameters. Fuels models is one off the basic inputs to fire behavior simulation modeling and they are described by a number of parameters associated to fire propagation dynamics.

Field inventories

Fuel sampling within 35 pine stands in the region was conducted with the transect method by (Brown, 1974). A classification of fuels is possible into one of the four time-lag classes (1, 10, 100 or 1000 hours). The time-lag is defined as the time period required for a fuel particle to reach approximately 63 % of the difference between the initial moisture content and the equilibrium moisture content in a different milieu (see Byram, 1963). This characteristic of the fuel particle is strongly correlated to its diameter, so in fire management one estimates the time-lag period by measuring the particles′ diameter. Dead and downed woody fuels have been grouped into classes that reflect the rate at which they can respond to changes in atmospheric conditions (i.e., 1-hour = <0.6cm, 10-h = 0.6-2.5 cm, 100-h = 2.5-7.6 cm and 1000-h = 7.6-20.3cm diameter). Additionally, grass and duff sampling was done on 0.5 m^2 plots within the stands. The entire above-ground material was sampled to determine the oven-dry weight (load per ha).

Figure 3.1. *View of one of four burning plots of the Brandenburg Fire Experiment, 23 August 2001. The experimental site was structured inhomogeneously, thus allowing to observe a range of different fuel and fire behavior conditions. Source: http://www.forst.uni-freiburg.de/feueroekologie.*

The pine stands were classified using cluster analysis into six groups. Factors determining grouping were stand age and the time since last thinning. In young pine stands (<20yrs.), the litter layer consisted mainly of 1- and 10-h fuels, while grasses were not established yet. Older stands (21-40 yrs.) are structured similarly, but with higher amounts of available fuels. In later stand stages (41-60 yrs.) grasses and shrubs invade due to increased light availability on the forest floor. Old stands are characterized by thick duff layers, a continuous grass layer and less dead and down material. Very high amounts of dead and down material was observed in stands where thinning was conducted before their fifth year. Usually, thinnings take place from stand age 35, so that younger stands are not affected. For detailed information on fuel classification in pine stands and other parameters included in the modeling process see Hille and Goldammer (2002).

Heathlands are a rather homogenous fuels of a single species. The shrub *Calluna vulgaris* is classified as 'live woody fuel', dead parts of the plants and litter beneath them are considered as 1-h fuels.

Two of the created fuel models were actually validated in the field. Fuel model 23 was tested during a forest fire experiment in summer 2001 (Goldammer *et al.*, 2001). The heathland model 26 was validated in summer 2002. Descriptions of all fuel models developed are summarized in Table 3.1.

Table 3.1. *Fuel models for pine stands of different age and continental heathlands. Fuel loads of the different fuel classes were used as the main input parameter in FARSITE. Fuel model 25 corresponds to all stands independently from their age.*

Model	Stand Type	Grass	1-h	10-h	100-h	Live woody	Fuel bed depth
#				t ha^{-1}			m
21	<20 yrs.	0	7.81	7.61	0	0	0.15
22	21-60 yrs.	0	8.06	11.70	4.09	0	0.20
23	61-100 yrs.	0.78	8.13	13.43	2.56	0	0.20
24	>101 yrs.	0.54	11.61	17.84	1.02	0	0.15
25	Thinning<5yrs.	0.42	10.27	20.57	6.47	0	0.30
26	Heathland	0	3.20	0	0	9.60	

Experiment results - Fuel model validation

Figure 3.2 shows the simulation results with the measured fuel and weather data during the fire as input into the BEHAVE-model and the observed fire characteristics. For the conditions measured during the fire, the BEHAVE-model calculates a fast increase of

fire spread for higher wind speeds, the pine model #23 being more influenced by wind speed than the heathland model.
We observed a high variance of observed spread rates, which was caused by fuel inhomogeneity and short-time changes in wind speed. Therefore, the observed values are visualized by ranges (ellipsoids in Fig. 3.2). For the pine model #23, the simulated fire spread for a range of wind speeds (line in Fig. 3.2) goes right through the cloud of observed fire spread (ellipsoids). In heathlands, the predicted rate of spread is below the observed average spread by ~20 %.

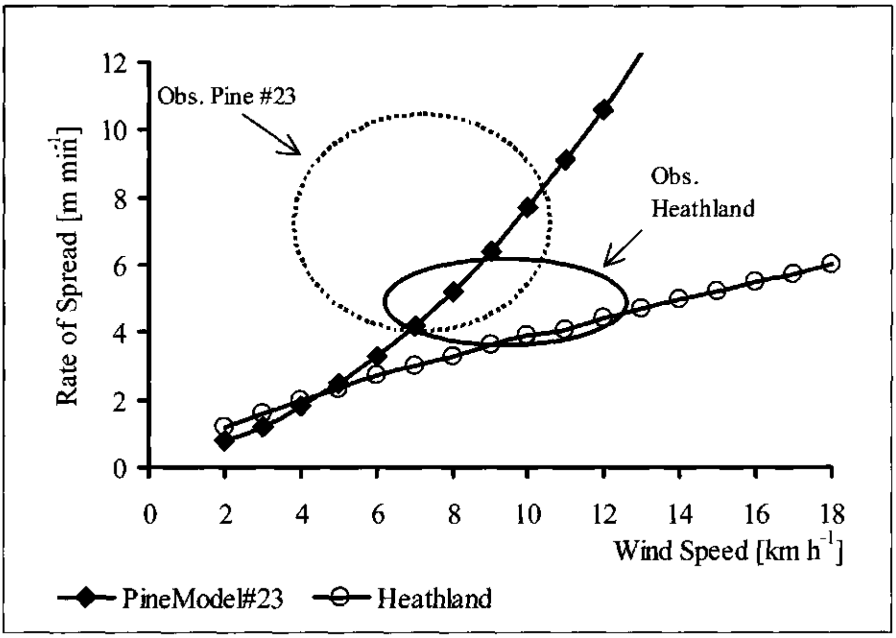


Figure 3.2. Simulated fire spread (lines) for the two tested fuel models and the observed data from the fire experiments (ellipses). Due to a high variation in wind-speed during the experimental burns, it was impossible to measure the exact wind-speed and the time when rate-of-spread measurements were taken. Therefore the range of wind speed during the experiment and the measured spread rates are presented here.

Given the high variability of fuel and wind, the fire behavior is well met with the two models. Especially for the pine model #23, the calculated spread rates and flame lengths (data not shown) are in range of the observed values. We therefore assume that also the other created fuel models for pine stands (Tab. 3.1) will give reasonable results in predicting fire behavior, although they are not validated yet.

Simulation results - Model application

Using the results and the gained experience of the BEHAVE modeling, a FARSITE simulation was created. On a 1000 ha former military bombing range, covered with pine forests and extensive heathland areas (Federal forest in the Lausitz region, Eastern Germany), fuel and stand information was collected to allow a classification by fuel models specified in Table 3.1 (see also Burgan and Rothermel, 1984).

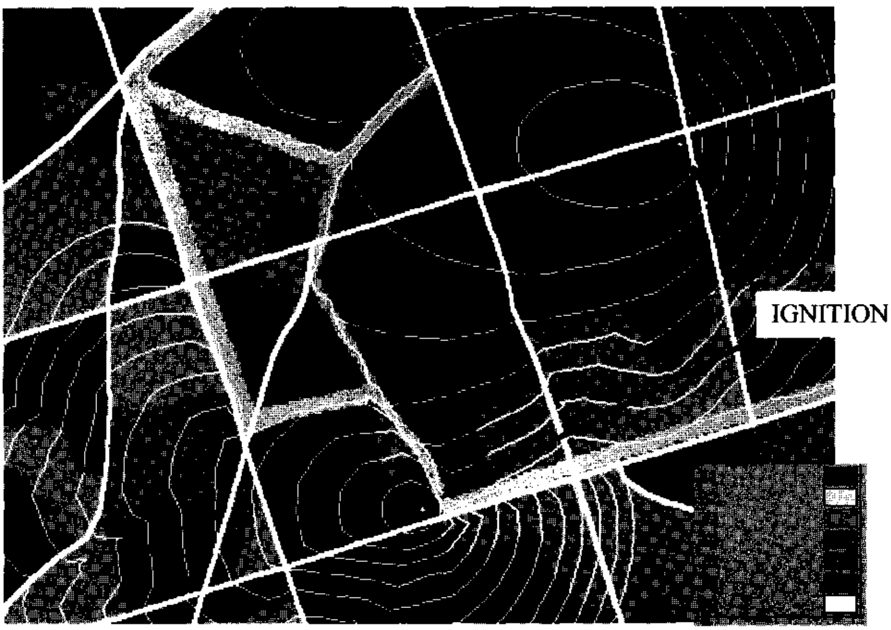


Figure 3.3. *Screenshot of the workable FARSITE 2-D landscape view. Colors represent different fuel models, grey areas (99) are fire barriers. Forest roads are displayed as white lines. Two ignitions are modeled for 6 hours under dry weather conditions with strong winds from the west. The two fires are not stopped by forest roads, only the wide fire barriers of bare mineral soil (30 m wide) are able to stop fire's spread.*

For the fire simulation, a digital landscape was created, using available information such as maps, digital elevation models, stand boundaries, roads etc. A raster grid of 6 x 6 m was chosen to be able to represent even small compartment and fire breaks (which are 30m wide in reality) within the study area. Figure 3.3 shows the workable raster view of parts of the simulation area.

The FARSITE model is very useful in extreme fire weather situations, where several fire suppression resources have to be positioned at places where they can reach high effectiveness. One scenario is given in Figure 3.3: under dry conditions in late summer (Temp. 30° C, RH below 50 % and strong winds from the West [17 km h^{-1}]) two ignitions were observed by the Automated Fire Detection System (cf. section 3.3.2). The coordinates are imported into FARSITE and the simulation runs for six hours.
FARSITE calculates the expected spread of the fire in 30-minute intervals, presented as thin white lines in Figure 3.3. The model outputs reveal that the forest roads are not able to stop or slow down the fire. The fast fire spread in the heathland (Mod. 26) makes suppression very difficult and dangerous. Therefore, fire suppression resources have to be positioned at the wide fire barriers (30 m wide fuel breaks) in the sampling area (grey areas).
The second ignition in the southern part of the test area occurred in a pine forest. Here, the spread is slower, but without suppression activities, the fire would not stop at the forest roads, too. Under a situation, where suppression forces are limited, one would decide to locate all engines around the forest fire and trust on the effectiveness of the fire barriers, which will stop the heathland fire according to the simulation.

3.3.2 Automated Fire Detection System AWFS

The Automated Fire Detection System AWFS (Kührt *et al.*, 2000) provides the fire detection and location component of the Forest Fire Management Decision-Support System (Fig. 3.4). The AWFS was designed to meet the following technical requirements specified by German forest authorities:

- Automatically recognize smoke formation of 10 m expansion within a radius of 10 km and within 10 minutes after becoming visible
- High reliability in respect of fire recognition
- Acceptable rate of false alarms
- Localize the source of the fire
- Easy maintenance
- Automatic transmission of smoke data to a control center
- Full record-keeping of all events
- Data transmission to control center must enable the operator to independently evaluate the potential hazard
- The costs should be lower compared with the conventional method (fire detection towers operated by personnel)

Technical systems for forest fire detection use CCD cameras, infrared sensors, spectrometers for detecting the smoke gases, laser backscattering, or other methods. AWFS tested in Germany is a system based on a high-resolution Frame Transfer CCD camera with special red-free filter which was originally developed for space missions (Michaelis *et al.*, 1999). AWFS detects fire by the trail of smoke within some minutes after its visibility. One system controls an area of about 300 square kilometers. The camera scans the forests from the top of the observation tower. The pictures are resolved with 14 bits and transmitted via optical fibers to the computer unit which is located in the tower. Here they are analyzed by a specially developed software. At any detected smoke formation, compressed pictures and further details (time, position) are reported to the control centre, where they are processed in a PC and displayed on a monitor. With a number of computer-assisted supports the operator is able to make reliable decisions.

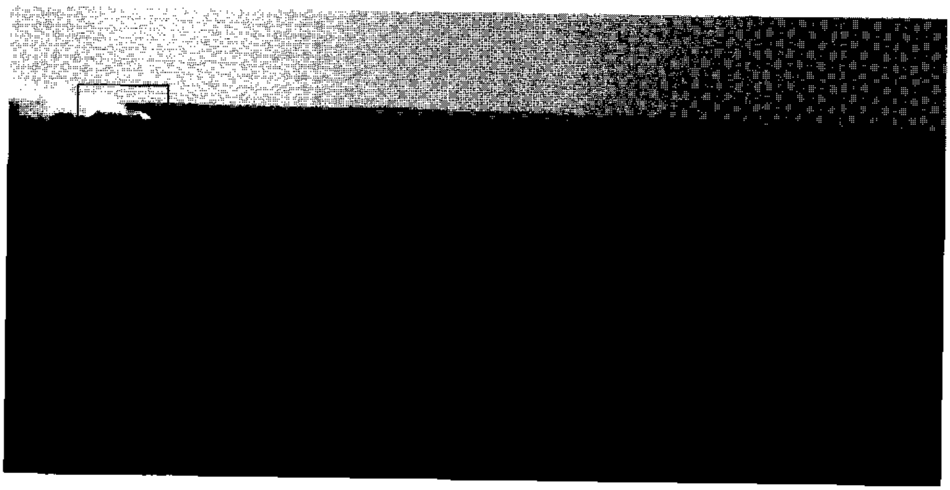


Figure 3.4. *Visual impression of forest fire smoke detection by the AWSF at the start of the 2001 Brandenburg fire experiment. Source: http://www.fire.uni-freiburg.de.*

Tests and Results

AWFS was installed and tested on three observation towers in the State of Brandenburg, Germany, during the four forest fire seasons (1999-2002) and with special test activities. One of these activities was the Brandenburg fire experiment on 23 August 2001.

Each of the more than 120 fires which arose in the observed region of about 1000 km^2 during the test period was recognized within some minutes. The false alarm rate due to special weather conditions and harvest activities (dust clouds) commonly remained below 2 %, which is well acceptable for the operator who evaluates the alarms of several systems and calls the fire brigade.

The absolute bearing exactness of every camera is better than 1°. Therefore, several systems can locate the source of fire with approximately 100 m at a distance of 10 km.

An impressive example for the precision of locating fires was a smoke signal of a structural fire in a small town in Brandenburg State, several kilometers away from the observing towers. With the intersection of bearings from two towers and the use of the digital map it was possible to determine the name of a short street in the town where the fire had started. The fire department was alerted immediately.

In the forest fire experiment in 2001 AWFS detected all four experimental fires within one revolution of the camera, i.e. within seven minutes. In one case the alert was already given about one minute after the smoke came up.

3.3.3 Fire Danger Rating and Forecast System

It is commonly known that some of the facets of weather support the ignition and propagation of forest fires: on the one hand, lightning strikes may directly ignite fires, and on the other hand, precipitation and evaporation affects the water content of dead and living vegetation and therefore indirectly controls the success of anthropogenic ignitions. Additionally, air motion influences the oxygen supply of the source of the fire and the spreading of the fire. Finally, fair weather means that the number of people frequenting the forests increases and permits a broad spectrum of activities of foresters and farmers (on neighboring farmland), so that the number of potential ignition sources (fire risk) increases. In order to prevent fire losses, the objective of the national weather services is to forecast the weather-dependent forest-fire risk and to issue fire-weather warnings to fire-fighting agencies, forest authorities, emergency services and the public when the weather becomes critical.

Implementation of the national to local fire-weather danger forecast

Within the framework of the German Weather Service (Deutscher Wetterdienst – DWD) operational forest-fire danger forecast are currently using domestic and foreign fire-weather ratings, such as the German M-68 index and the Canadian Fire-Weather Index (FWI) (Wittich, 1998b). The indices, together with additional meteorological information, are sent to forest authorities and disaster control centers of the Ministries of Interior of the Federal States of Germany so that they can issue the necessary instructions.

During the fire season the DWD daily issues the M-68 index via the internet under http://www.dwd.de/WALDBRAND. Figure 3.5 shows the danger-rating chart for Germany on 5 June 2002, containing five risk levels (level 1 = low danger, ..., level 5 = extreme danger). Clicking on one of the ~ 200 station circles, one can get a time series over several days, which is composed of the current-day index, the index of two previous days and that of three forecast days, thus illustrating the temporal course of the forest-fire potential.

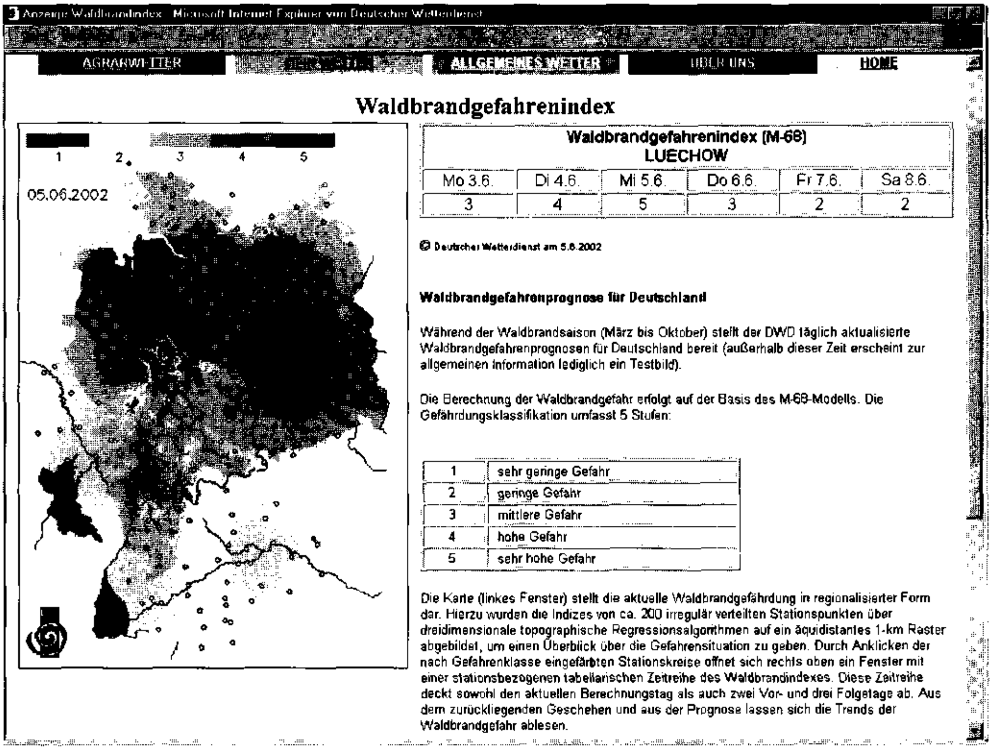


Figure 3.5. *Example of the German Weather Service (DWD) fire-weather / fire-danger forecast via the internet for 5 June 2002. During the fire season a map provides a daily overview for Germany's territory. The system allows to retrieve the fire danger index (M-68) for individual stations to obtain the fire-danger forecast for the current day (right hand example: Luechow, 5 June 2002), for the pas two days and the next two days.*

For the implementation of a local decision-support system based on automatic fire detection and modeling of fire behavior precise on-site real-time fire weather data are required to obtain a realistic model output. In an optimized system weather data would gathered automatically through a dense network of weather stations, transmitted to the data processing centre and integrated into the decision-support system. Alternatively, fire weather data could be obtained at or near the fire site by a mobile weather station or by ground personnel using a mobile fire-weather kit. Taking into account the local variability of fire-weather data the latter alternative will meet the demands of on-site weather information.

An overview of fire danger at regional level, e.g. for assessing fire danger in Europe and the neighboring countries, is provided by the Eurasian Experimental Fire Weather Information System generated of the basis of the Canadian Forest Fire Danger Rating System (CFFDRS) by the Northern Forestry Centre, Canada, for the Global Fire Monitoring Center (GFMC). The system allows downloading a number of Fire Weather Index Components (Fine Fuel Moisture Code – FFMC, Duff Moisture Code – DMC, Drought Code - DC, Initial Spread Index - ISI, Buildup Index - BUI, and the Fire Weather Index - FWI) and Meteorological Data (Fig. 3.6). This regional system is still operating on a provisional basis due to the lack of automated inputs from hourly weather observations, especially in Russia. The Canadian Forest Service is working on a Global Experimental Fire Weather Information System to be displayed at the GFMC in late 2003.

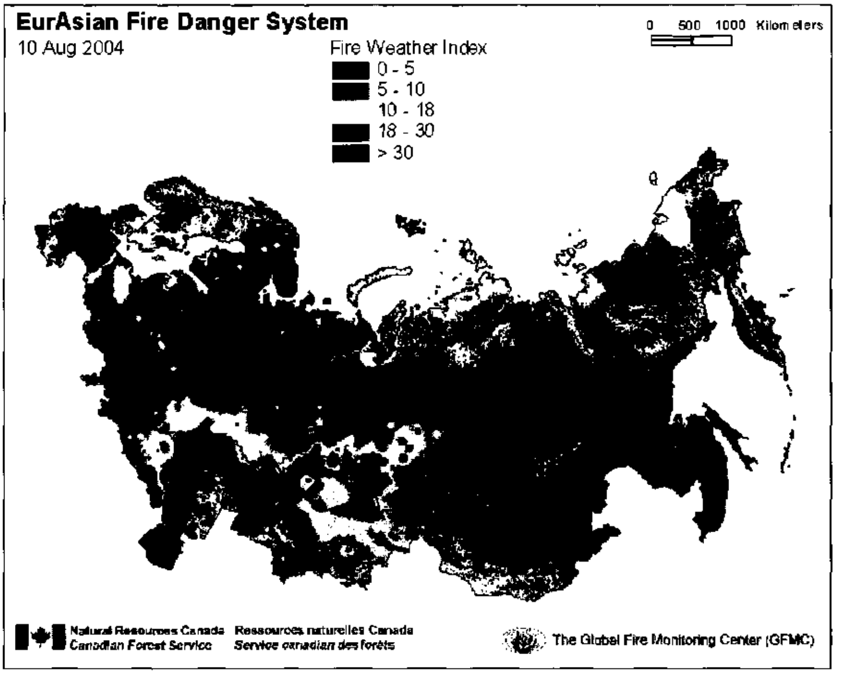


Figure 3.6. Example of a daily fire weather index map of the Eurasian Experimental Fire Weather Information System generated of the basis of the Canadian Forest Fire Danger Rating System (CFFDRS) by the Northern Forestry Centre, Canadian Forest Service, for the Global Fire Monitoring Center (GFMC). Source: http://www.fire.uni-freiburg.de/fwf/eurasia.htm (accessed 8 July 2005).

3.4 Implementation of the Fire Information System

Theoretically, data and information about what we consider as fire structural parameters (i.e. information about fuel, weather, and topography), contain the descriptive (i.e. attributes) as well as the spatial (i.e. coordinates) component. The spatial component sets up the basic requirements to consider it as a geographical information system (GIS); for instance, descriptive information of fuel is spatially distributed within the geographical extent of the study area.
Conceptually, the integration of all the necessary information under a common processing scheme presupposes firstly the necessary compatibility among different data layers. To maintain spatial information of any descriptive parameter in a digital form, a number of different alternatives are available including, among others, the format of the data (i.e. raster vs. vector type), the type of spatial objects (i.e. point, line or polygon), the type of measurements (i.e. nominal, ordinal, interval, or ratio), and the spatial resolution or scale (DeMers, 1997).

3.4.1 Integration of fuel data, fire behavior model, weather and fire detection data in a GIS

Primary observations or data referring to the structural parameters for wildland fire may exist in multiple types and multiple scales that prohibit their integration under a common scheme due to several incompatibilities. For instance, elevation gradient, as well as weather data are better represented by continuously data using the raster data type. However, their primary source data (based on which the final ones are produced) may considerable differ. For instance, fire weather observations are provided at specific points in space that correspond usually to meteorological weather stations. To convert point observations into continuous surfaces by filling the gaps in the between unsampled sites, interpolation procedures have to be applied, like inverse distance weighting, nearest neighbors, splines, or geostatistics (Burrough and McDonnel, 1998). On the other hand, road network and firebreaks, which are depicted as linear or polygon objects depending on the scale level, are introduced into the fire information system as vector or raster type. To allow however their co-processing with other spatial information, as for instance for fire behavior modeling, vector objects should be converted to raster objects by considering during the conversion process how to maintain without misquoting the original information.

In addition to the fire structural parameters information, the fire behavior model and the fire detection and monitoring system are another two critical components of a fire information system. Fire behavior, formally is defined as "the manner in which a fire behaves as a function of the variables of fuel, weather and topography". The fire behavior modeling phase enables us to simulate a real fire event and allows us to test hypothetical scenarios about its propagation, and suppression strategy. A fire can be inserted and simulated into fire behavior modeling system either manually by the system operator or automatically if this system is connected online to an appropriate fire detection system. Apart from the input of the ignition source, real data referring to fire propagation can also be introduced into the system so that the modeling phase of the system be continuously supplied with the updated information for validation and self-correction.

GIS, by providing tools, resources and a proper organizational context to gather, manage and process spatial referenced information (Burrough, 1986), can support the role to integrate fuel data, fire behavior model, weather data, and the fire detection system. The main functional process that has to be resolved is the data management including collection, homogenization, maintenance and future update of the information. Information may come from completely different sources, and be different in scale, content, accuracy, etc. To enable the integration of such different spatial layers of information under a common functional schema, certain procedures have to be implemented and supported.

3.4.2 Design of a prototype of a fire information - decision support system

In principle, a fire information – decision support system should support the requirements of the input, maintenance, update and processing of the appropriate information. Concerning the data management subsystem, the ability to work independently under a semi-automatic or fully automatic mode, when possible, is very important. Furthermore, its ability to receive online information about input (i.e. fire weather data) as well as output data (i.e. fire behavior) is another important aspect. Remote sensing and GIS, being complementary tools for gathering and processing data and information, could constitute the hart of the data management subsystem. Remote sensing can contribute to generation of the information to support the requirements of updated and spatially distributed information. Various remote sensing applications can be found in literature for estimating fuel parameters and fire risk before the fire (Chuvieco and Congalton, 1989; Leblon *et al.*, 2002), for detection and monitoring during a fire (Bourgeau-Chavez *et al.*, 1997; Kasischke *et al.*, 1993), and for burned land mapping and post fire effects assessment after the fire (Jakubuskas *et al.*, 1990; Koutsias and Karteris, 1998).

The research project provided an opportunity to test the advanced spaceborne Bi-Spectral Infrared Detection (BIRD) sensing system for the detection and characterization of high-temperature events (HTE). BIRD is the first space borne sensor that offers the capability to provide daytime detection of small fires with areas exceeding ~15 m^2 and to estimate their radiative energy release. For fires with areas exceeding ~0.15 ha, an estimation of the effective fire temperature and area is also feasible. This capability of BIRD is especially important for the detection of small fires. In addition, the high sensitivity of the BIRD IR sensor system might also allow the characterization of low intensity surface fires in forests (under canopy) which are difficult to be detected by other satellite systems. During the project's scientific forest fire experiment the Advanced BIRD Airborne Simulator (ABAS) was used to test the capabilities of this new spaceborne fire detection and characterization system (Oertel *et al.*, 2002) before BIRD was launched to the orbit in October 2001.

The results of the tests of ABAS and BIRD (Fig. 3.7) confirm the capabilities of the sensor system. An integration of BIRD with the prototype decision-support system would provide an opportunity to generate information of additional value for a fire management decision support system. An operational BIRD satellite system would deliver precise information on the spread and intensity of a fire front, thus allowing to verify and update the outputs of the fire spread model.

Figure 3.7. *Example of a BIRD fire product image fragment showing forest fires in the centre of Portugal on 4 August 2003. The fire radiant power is color coded in Megawatt per pixel and is overlaid on the black and white background showing the dark fire scars.*

3.5 Conclusions and outlook

Research and development conducted within the Forest Fire Cluster of the German Research Network on Natural Disasters is built on a number of separately evolved concepts that were integrated in a cooperative research project. For the first time a fire behavior model has been applied for the specific conditions of pine forests in the eastern, continental part of Germany, including the interspersed heathlands that constitute an important carrier of a wildfire at landscape level. The characteristics of these forests are quite typical for temperate-hemiboreal pine forests of Eurasia.

Thus, the results of this work can be easily adapted to neighboring countries where similar pine forests cover large areas, e.g., Poland, Belarus, and the Russian Federation. The development of the AWFS meets the requirements for fast, cost-effective and reliable fire detection system. The national fire-danger rating system has consolidated during the project lifetime.

During the research project the work of the Global Fire Monitoring Center (GFMC) constituted the link from national to international levels. Besides the function of a support body for the development of national to international policies and fire management strategies the modus operandi of the GFMC provided an opportunity to implement the regional Eurasian Experimental Fire Weather Information System in cooperation with the Canadian Forest Service and to test the BIRD satellite mission in various vegetation types around the world.

The concept of the German Natural Disaster Research Network (DFNK) provided an exemplary opportunity to conduct multi- and interdisciplinary fire research and has contributed to establish a new and unprecedented collaborative culture of wildland fire science in Germany. The value added by the research project is a mutual support of individual research projects and their final merging into a comprehensive decision-support tool for fire management. Insight gained by the research project concerning the operational use of satellite remote sensing information in the management of active wildland fires will be useful for the development of urgently needed operational spaceborne fire systems (Ahern *et al.*, 2001).

CHAPTER 4

Factors that influence humus consumption in the field

Published as: HILLE M, STEPHENS SL. 2005. MIXED CONIFER FOREST DUFF CONSUMPTION DURING PRESCRIBED FIRES: TREE CROWN IMPACTS. *FOREST SCIENCE* 51: 417-424

Abstract

Fire suppression has produced large forest floor fuel loads in many coniferous forests in western North America. This study describes spatial patterns of duff consumption in a mixed-conifer forest in the north-central Sierra Nevada, California. Overstory crown coverage was correlated to spatial patterns of duff depth after prescribed fire. On one site that was burned under dry conditions, almost all duff was consumed, with some remaining in overstory gaps. On a second site that was burned under moist conditions a few days after the first annual precipitation, strong spatial patterns of duff consumption were recorded With increasing distance from the base of the nearest overstory tree, the probability of duff remaining after prescribed fire increased significantly. There is strong evidence that spatial variation of precipitation throughfall resulted in higher duff moisture in gaps, while duff beneath crown cover was drier and, therefore, totally consumed. This study shows, that including a spatial component in a process-based duff consumption model would improve the accuracy of fire-effect predictions.

4 Mixed conifer forest duff consumption during prescribed fires: Tree crown impacts[1]

4.1 Introduction

Consumption of the duff later (O_F and O_H layers) during wildfires or prescribed fires influences post-fire stand development by the destruction of rhizomes and seeds that are stored in the forest floor (Schimmel and Granström 1996), and by overstory-tree mortality caused by the prolonged heat released by smoldering combustion. Additionally, duff consumption is the largest contributor to smoke production and has a large impact on soil nutrient cycling (Neary *et al.* 1999).
In the case of partial consumption of the duff layer, heat tolerance and storage depth of seeds and rhizomes can play an important role in the re-colonization pattern of the site (Granström and Schimmel 1993). Furthermore, the exposure of mineral soil by complete consumption of the duff layer favors the establishment of many forest species (Thomas and Wein 1985; Hille and Den Ouden 2004).

Tree mortality due to heat-induced injury of the cambium or the roots can alter overstory composition and eliminate seed sources (Stephens and Finney 2002). The prolonged heat release by combustion of the accumulated organic material around the stem bases ('tree wells', Johnson *et al.* 2001) can be lethal for the cambium of overstory trees.
Next to safety and smoke production, duff consumption should be of major importance when planning a prescribed fire. Since major fire effects are strongly correlated to duff consumption, a precise estimation of duff consumption is crucial to predict post-fire ecosystem dynamics. Duff consumption mainly proceeds by slow, smoldering combustion. The amount of consumption is dependent on duff characteristics such moisture, inorganic content, layer depth, and density (Frandsen 1987, Stephens *et al.* 2004).

Several empirical models of duff consumption have been developed from data collected from wildfires or prescribed burns (Sandberg 1980, Brown *et al.* 1985, Reinhardt *et al.* 1991). These models have been used to predict duff consumption during prescribed fires. However, these models cannot be generalized beyond the site and the conditions under which the data were collected.
Process-based models of duff consumption are based on a two-step process of smoldering, with an endothermic process of char forming (pyrolysis), followed by an

[1] This study was conducted in mixed-conifer forest in the Sierra Nevada, California. However, the strong spatial variation of humus moisture has also been observed in Scots pine stands (Möttonen *et al.* 1999) and the same effects can be expected here. Duff is an old US-american expression for the O_F and O_H-layer.

exothermic process of oxidation (Miyanishi and Johnson 2002). Therefore, propagation of smoldering combustion is dependent on sufficient heat being transferred from the exothermic oxidation zone to the adjacent duff to cause pyrolysis. Factors that affect this heat transfer therefore strongly influence duff combustion.

Process-based models of duff consumption are based on physical mechanisms of pyrolysis, char oxidation, and heat transfer under all conditions of smoldering combustion, and therefore, generalizable for duff with different bulk density, mineral soil, and water contents (van Wagner 1972, Frandsen 1987). However, these duff consumption models only estimate average duff consumption at the stand level (average duff moisture content is the input variable). Hence, spatial variation of duff properties within the stand, which can cause spatial variation of duff consumption, cannot be predicted.

Duff moisture was found to have the strongest impact on the smoldering combustion process (Frandsen 1987), and is the most important input variable in both empirical and process-based models. Stored moisture results in a latent heat flux for water evaporation, which provides an effective heat sink; thus it can slow down or extinguish smoldering combustion. Other characteristics such as duff depth, bulk density, amount of surface fuels, and inorganic content of the duff layer will influence heat generation and heat transfer, and therefore, smoldering combustion (Burgan and Rothermel 1984, Miyanishi and Johnson 2002).
Duff moisture within a stand is seldom spatially uniform. Factors such as precipitation throughfall, which are mainly influenced by overstory crown structure, and differences in water uptake by ground layer vegetation can cause spatial duff moisture variation. In conifer forests, the highest throughfall rates are found at the periphery of the crown (canopy drip line) and in areas with no crown coverage (Bouten *et al.* 1992, Bruckner *et al.* 1999). At the stand level, Chrosciewicz (1989) and Miyanishi and Johnson (2002) found significantly drier duff beneath trees when compared to duff beyond tree crowns in pine/spruce (*Pinus/Picea*) stands. This spatial variation of duff moisture has not been considered when predicting duff consumption.

As duff moisture strongly influences duff consumption, temporal and spatial variation of duff moisture should become evident in patterns of duff remaining after fire (Robichaud and Miller 1999). A patterning of duff consumption has been observed mainly in boreal forests (Dyrness and Norum 1983, Zasada *et al.* 1983, Miyanishi 2001), but also in dry coniferous stands (Sweeney and Biswell 1961). Miyanishi *et al.* (1999) and Miyanishi and Johnson (2002) reported a significant spatial correlation between burned patches of duff and standing boles of trees killed by the fire.
The objective of this study is to evaluate and explain spatial variation in duff patterns after two prescribed fires in a mixed-conifer forest. Following common practices, only an average duff moisture value was calculated for the first prescribed fire, which was burned 3 days after a significant rainfall event. We therefore cannot present spatial,

pre-fire data for the first prescribed fire, but present the observed strong spatial pattern of duff consumption. For the second prescribed fire, which was burned under dry conditions the following year, spatial pre-fire duff moisture was sampled. The absolute and relative position of a given duff sample within the setting of surrounding trees will be used to explain spatial variation in duff consumption.

Our hypothesis is that spatial variation in duff moisture, as affected by overstory canopy cover, will result in differing patterns of duff consumption. If overall duff moisture is low enough to ensure propagation of smoldering combustion (such as in the prescribed fire under dry conditions), most of the duff on site will be consumed and therefore no spatial pattern can be found. In contrast, if duff moisture varies spatially around marginal conditions for smoldering propagation (such as after the prescribed fire that closely followed the first significant rainfall event, and where tree crowns caused spatial variation in duff moisture), we expect spatial variation in duff consumption. Information from this study could be used to improve predictions of duff consumption by prescribed fire.

4.2 Methods

4.2.1 Study site

The experiment was established in a second-growth mixed-conifer forest (Fuel Model 10; NWCG, 1989) at the University of California Blodgett Forest Research Station, latitude 38° 54′ 45″ N, longitude 120° 39′ 27″ W. The experimental stand (compartment 292 of the research station) is located in the north-central Sierra Nevada at approximately 1400 m above sea level and covers 19 ha of a north facing, gentle slope (slopes <15 %). No harvesting operations have occurred in the stand for more than 25 yr.

The majority of precipitation (total of 1500 to 2000 mm yr^{-1}) in this area falls between November and February, either as snow or rain. During the summer and early fall, almost no rainfall occurs, leading to very dry conditions in late summer. Often summers occur with no significant rainfall for five months.

Fire was once a common ecosystem process in the mixed conifer forests at Blodgett Forest (Stephens and Collins 2004). Mean fire return intervals varied from 5-15 yr from the late 1600's to the beginning of the 20^{th} century. After 1900, fires have been very rare and this coincides with the introduction of fire suppression into this region (Husari and McKelvey 1996).

To characterize existing forest structure, information was collected from 0.04 ha systematic forest inventory plots placed in compartment 292. The circular inventory plots are separated by 120 m. Height and dbh of all trees >2.5 cm dbh were measured. Average basal area, average number of trees per hectare, and average diameter were calculated using the forest inventory plots.

4.2.2 Fuel loads

On the stand level, pre- and post-fire fuel load measurements were performed in the entire 19 ha compartment with the transect method (Brown 1974) the week before and after the fires. Sample points were on a 60 by 60 m grid. From each sample point, two transects were installed in random directions. Litter and duff depth were measured at 0.33, 0.66, and 0.99 m on each transect. Ground and surface fuel loads were calculated by using appropriate equations developed for Sierra Nevada forests (Van Wagtendonk *et al.* 1996, 1998b). Coefficients required to calculate ground and surface fuel loads were arithmetically weighted by the basal area fraction (percent of total basal area by species) to produce accurate estimates of fuel loads (Stephens 2001).
To assess the spatial variation of duff depth and moisture with respect to the position of the canopy, one transect in a random direction was installed from the base of ten overstory trees in the compartment. To be selected, the trees had to have symmetric crowns that were separated by at least 2 m from the crowns of neighboring trees and no understory trees or shrubs present beneath their crowns. In total, ten dominant trees (five sugar pines (*Pinus lambertiana* Dougl.) and five ponderosa pines (*Pinus ponderosa* Laws)) within the compartment were selected for the measurements.

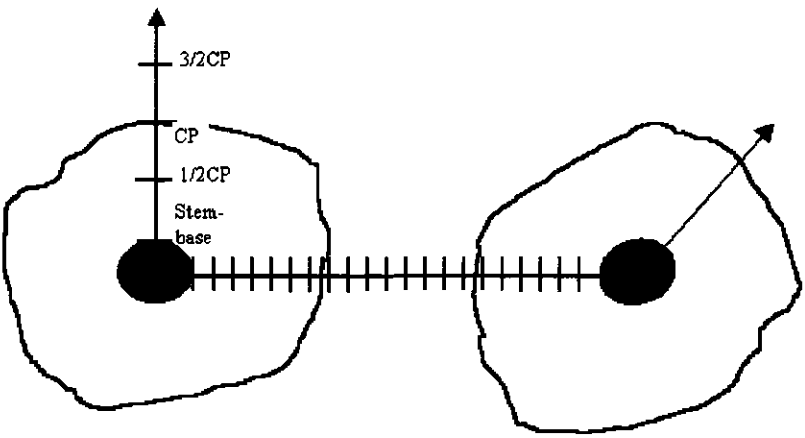


Figure 4.1. *Sample design of duff moisture and depth measurements used in the dry burn, with CP as the distance from the stem base to the crown perimeter. Sampling points were at the stem base, and at 0.5, 1, and 1.5CP. The design of the tree-to-tree lines is also illustrated as a connection between two neighboring dominant trees. From the right tree in the figure, the line was extended into a random direction to the next dominant tree.*

In the four cardinal directions, the distance between the base of the tree and the projected crown perimeter was measured. Duff depth was also measured on each transect: a) directly next to the stem base, b) at half the distance to the crown perimeter, c) directly at the crown perimeter and, d) 1.5 x the distance of the crown perimeter (Fig. 4.1). At each point ($n = 40$), a steel pin was installed flush with the duff surface to allow the measurement of duff depth reduction. Post-fire duff depth was measured the day after the prescribed fire.
An hour before the prescribed fire, duff samples of approximately 250 cm^3 were taken at points a) through d), with a clockwise offset of 15 cm from the original point (Fig. 4.1). These samples were weighed, oven-dried (105° C for 12 hr) and re-weighed to determine duff moisture on a dry weight basis.

4.2.3 Prescribed-fire application

The southwest portion of compartment 292 (2.7 ha) was prescribed burned in November 2001 one week after the first significant precipitation event, which delivered 25 mm of rain (Fig. 4.2). Hereafter this prescribed fire is identified as the 'moist burn'. For this burn only average pre-fire duff moisture data from samples taken at the center of each circular forest inventory plot, are available.

The remaining area of compartment 292 (16 ha) was burned in October 2002 under much drier conditions (also, no rainfall the previous 3 months) (Fig. 4.2). Hereafter this prescribed fire is identified as the 'dry burn'. Previous to the fire, extensive duff moisture samples were collected.

The ignition pattern used in both prescribed fires was a strip head fire (Martin and Dell 1978). Strip width varied from 2 to 3 m. The rate of spread in both burns was approximately 1 m min^{-1}. Flame heights of 0.30 to 1.00 m were observed during the moist burn, flames reached 1.50 m in height in the dry burn.

4.2.4 Post-fire duff sampling

Post-burn measurements were taken after smoldering combustion ended (24 hr after the flaming front passed). Duff consumption was measured using the installed steel pins in the dry burn area; duff consumption was measured as the difference between the head of the pin and the post-fire surface. Duff consumption in the moist burn area was measured at fixed intervals along fuel transects without the use of steel pins. To assess the impact of overstory tree crowns on duff consumption, we sampled post-fire duff patterns in both prescribed fires in compartment 292.

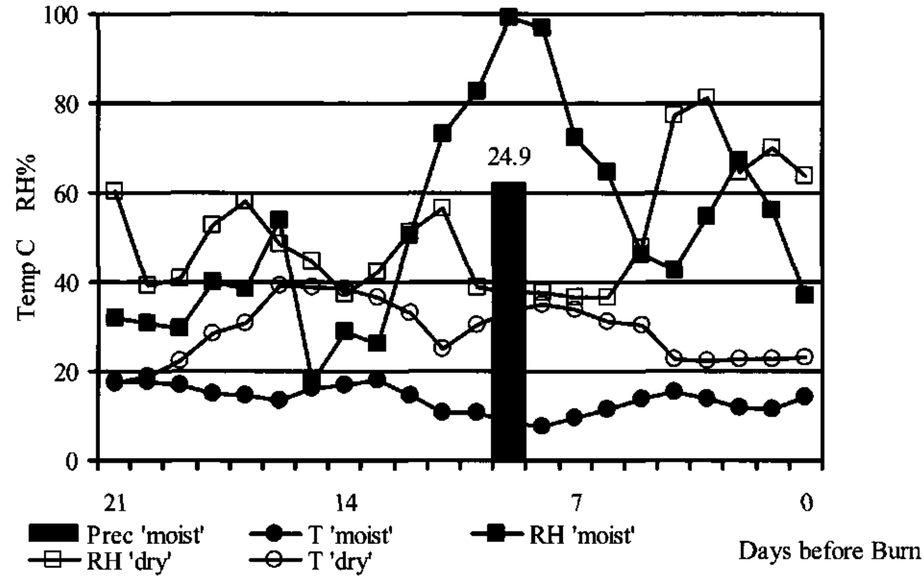


Figure 4.2. Temperature (°C), relative humidity (RH %), and precipitation (mm) 3 weeks before the burn and the day of the prescribed fires. Values are daily averages, generated from 15-minute interval measurements. Note rainfall before the moist burn, but no rain before the dry burn.

From a randomly determined starting point, the nearest overstory fir or pine was selected. A line was extended from this tree to another overstory tree in a random direction (± 20° search angle). If no tree was found within 25 m, another direction was randomly selected. Trees had to be dominant to be selected. On this tree-to-tree line, duff depth was measured every 50 cm and at distances of 0, 10 and 25 cm from each tree base (Fig. 4.1). For the analysis, only lines from trees with a crown-to-crown gap to the neighboring tree of > 2 m were selected (n = 59 for the moist burn, n = 56 for the dry burn).

4.2.5 Statistical Analyses

Pre- and post-fire data were compared with paired t-tests. Moisture content and duff depth from the ten overstory trees in each prescribed fire (dry and moist) were analyzed with one-way ANOVA procedures. The duff depth data from tree-to-tree lines was transformed from absolute distance to relative distance to the stem base by creating classes of 1/10th of the distance from stem base to the projected crown perimeter and assigning the actual values to the nearest relative class. This enabled us to compare duff consumption under trees with different absolute crown widths. The Chapman-Richards equation, as a flexible growth function (Richards 1959), was used to model the probability of duff remaining after the two prescribed fires.

4.3 Results

Overstory tree species in this stand include white fir (*Abies concolor* Gord. & Glend.), Douglas-fir (*Pseudotsuga menziesii* Mirb. Franco), ponderosa pine, and sugar pine. Incense-cedar (*Calocedrus decurrens* Torr.) and California black oak (*Quercus kelloggii* Newb.) comprise a second-canopy strata. White fir seedlings and sapling dominate the understory (Tab. 4.1).

Table 4.1. *Pre-fire stand variables for compartment 292 at Blodgett Research Forest, California.*

Tree species	Stand density N/ha (+- SD)	Average DBH cm	Basal area m²/ha (+- SD)	Height m
Black oak	8 (1)	46.19	1.23 (3.04)	11.45
Douglas- fir	31 (2)	28.11	3.53 (8.72)	26.54
Incense- cedar	252 (13)	19.11	10.02 (25.58)	13.91
Ponderosa pine	6 (1)	74.43	2.95 (7.69)	39.80
Sugar pine	11 (1)	77.85	5.73 (14.42)	39.45
White fir	181 (9)	22.74	15.50 (39.18)	26.40
Σ	497 (15)	23.25	39.53 (100)	20.73

4.3.1 Fuel load

The total pre-fire fuel load in compartment 292 was approximately 155 Mg ha^{-1}, of which approximately 100 Mg ha^{-1} consisted of duff (Tab. 4.2). Litter depth averaged 3 cm across the unit. Duff depth averaged 9 cm throughout the unit, though depths up to 18 cm were sampled on individual transects.

The post-fire fuel load was significantly reduced by both prescribed fires (Tab. 4.2). Total fuel loads were reduced by 51% in the moist burn and 88% in the dry burn. An especially high reduction occurred for the litter and duff layer in the dry burn. Small dead-and-down fuels were also reduced, but at the time of the sampling (1 yr after the burn), new fuels have probably begun to accumulate.

4.3.2 Duff depth and moisture pattern

The dbh, height, and crown radii of the ten trees selected for the duff measurements varied from 75 to 96 cm, 35 and 40 m, and 2.5 and 4 m., respectively. Pre-fire duff depth varied with respect to distance from the stem base. The duff layer around the base of all sampled overstory trees was significantly deeper (depth of up to 20 cm) than at any other location either under the canopy or beyond the canopy (ANOVA, $F_{3,36}$ = 17.51; p = 0.001; Fig. 4.3).

Table 4.2. *Pre-fire fuel load by size classes (average ± SD) for compartment 292 at Blodgett Research Forest, California.*

Fuel Class (size class (cm))	Pre-fire fuel load Mg ha^{-1} (n=25)		Post-fire fuel load moist burn Mg ha^{-1} (n=14)		Post-fire fuel load dry burn Mg ha^{-1} (n=10)	
1-hr (0 to 0.6)	2.0	(0.2)	0.7	(0.2)	0.3	(0.1)
10-hr (0.6 to 2.5)	6.3	(0.7)	2.0	(0.5)	0.4	(0.1)
100-hr (2.5 to 7.6)	5.8	(1.6)	7.0	(1.9)	1.7	(0.2)
1000-hr sound (> 7.6)	6.0	(3.3)	4.6	(3.8)	6.9	(3.5)
1000-hr rotten (> 7.6)	15.8	(4.3)	3.1	(2.4)	0.0	(-)
Litter [O_L]	17.8	(3.6)	19.4	(6.4)	5.2	(1.1)
Duff [O_F + O_H]	101.0	(11.9)	38.4	(15.2)	4.1	(0.4)
Total fuel load	154.5		75.2		18.7	

The average duff depth decreased from 6 to 2 cm with increasing distance from the stem and especially in gaps. However, the decrease in duff depth further away from the tree (under the crown to the gap) was not significant at the 95 % level.

In the dry burn, duff moisture was low with an average of 12 % and a range from 10 to 14 %. There was no significant pattern found for duff moisture and its position to the nearest stem base (Fig. 4.3; ANOVA, $F_{3,36}$ = 1.18, p = 0.33). There was very little variation in duff depth within and between crown position classes (coefficient of variation between 11 and 13 %). In the moist burn, duff moisture varied from 30 to 65 %.

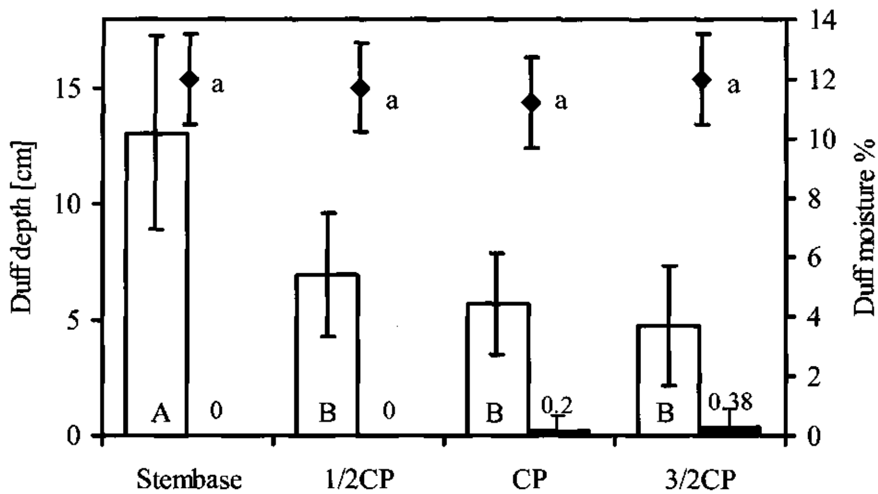


Figure 4.3. *Mean duff moisture and duff depth in the dry burn at different positions related to the nearest overstory tree (mean ± SD, n = 10 for each location). Diamonds display duff moisture before the fire (right y-axis), bars display duff depth before (white) and after the fire (black). Bars/circles with the same letter are not significantly different at the 0.05-level. CP = distance from stem base to crown perimeter.*

4.3.3 Post-fire duff patterns

At all sample-points within the two burned areas, the litter layer was totally consumed and at least the top duff layer was charred. There was little duff remaining after the dry burn. At 97 % of the sample points on the tree-to-tree lines (in the dry burn), there was no organic material on the soil surface. The few remaining areas with duff were less than 3 cm thick. In the moist burn, a higher percentage of duff remained unburned. At 82 % of the sample points, no duff was found after the prescribed fire, but areas with duff remaining were up to 6 cm thick.

In both prescribed fires, a spatial trend of duff consumption with regard to crown coverage was observed. While there was no duff found in the direct vicinity of the overstory trees, the probability to find duff increased to 7 % for the dry burn and 65 % for the moist burn with increasing distance from the tree (Fig. 4.4).

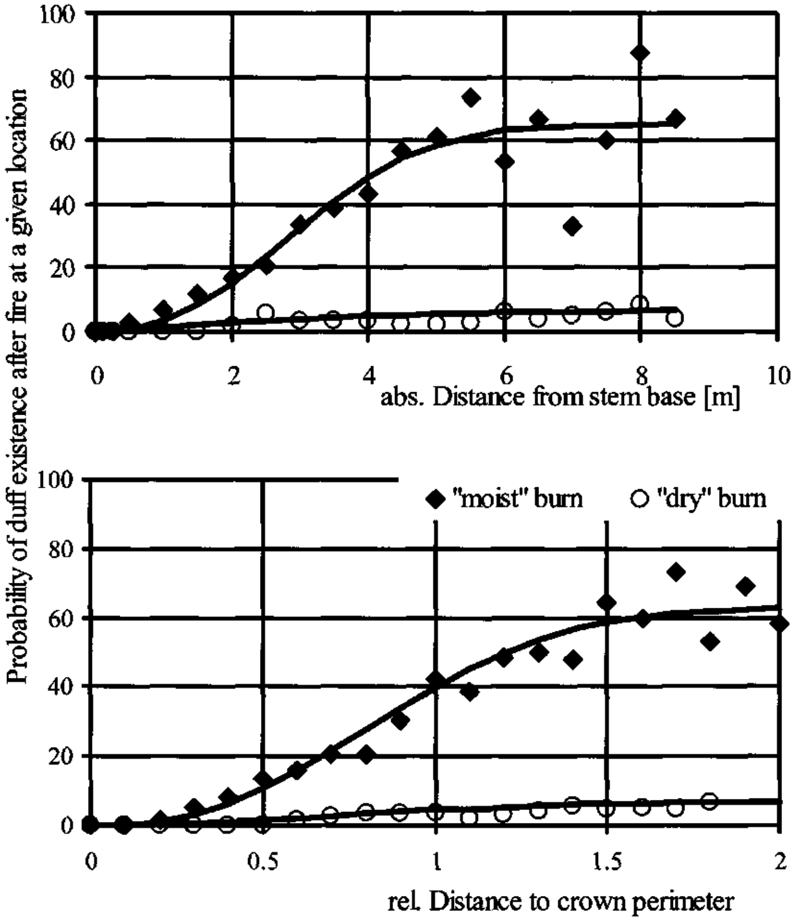


Figure 4.4. A, B. *Spatial variation of duff remaining in the moist (diamonds) and the dry (empty circles) prescribed fires, related to absolute (A) and relative (B) distance from dominant tree. Lines are regressions with the Chapman-Richard equations. The probability of duff remaining (y-axis) is calculated from the percentage of data points where duff survived the fire. The relative distance from the stem base is expressed as ratio to the crown perimeter. A relative distance of "1" marks the edge of the crown.*

In reference to the absolute distance from the tree base, a strong increase in the probability of duff remaining after fire was observed in the moist burn at distances between 2 and 4 m (Fig. 4.4A). In the dry burn, no increase in the probability of duff remaining was found. The same data, transformed to a relative position to the crown structures, shows a very similar trend, with an even stronger increase of probability of duff remaining between 0.5 and 1.5 of the relative distance to the crown perimeter (Fig. 4.4B).

This probability is well described by an S-shaped curve following the Chapman-Richards equation (Richards 1959). The equation to predict the probability to find remaining duff after the fires (P(d)) for the absolute distance is

$$P(d) = A*(1-e^{-((d/3.5)^k)}) \quad (1)$$

and

$$P(d) = A*(1-e^{-(d^k)}) \quad (2)$$

with d = distance from tree, either in meters (1) or as a fraction of crown perimeter of the nearest tree (2). The asymptote A represents the probability of finding duff at locations in gaps (7 % for the dry burn and 65 % for the moist burn) and k determines the shape of the curve. Best-fit parameters are given in Table 4.3. High adjusted R^2-values indicate good model performance.

Table 4.3. *Parameters for the four S-shaped regression curves, following the Chapman-Richards equations in (1) and (2).*

Data	Equation	A	k	adj. R^2
Absolute distance moist burn	(1)	65	2.370	0.937
Absolute distance dry burn	(1)	7	1.235	0.828
Relative distance moist burn	(2)	65	2.430	0.978
Relative distance dry burn	(2)	7	1.879	0.930

The spatial variation of duff reduction is visible in Figure 4.5. Directly at the stem base, the duff layer is reduced by 13 cm on average with large amounts of heat being released close to the cambium layer. Further away from the stem base, duff depth is less and duff consumption incomplete.

4.4 Discussion

Duff depth was significantly reduced by both prescribed fires, with a higher percentage of mineral soil exposed after the dry burn. High duff consumption in the dry burn was produced because of low fuel moisture contents after the dry Sierra Nevada summer (Fig. 4.2). Duff moisture content was close to the historical annual minima during the dry burn (10-12 %; Ziemer 1968). Duff moisture content was significantly higher (30-60 %) in the moist burn because of a single rainfall event one week before the prescribed fire.

Duff depths in areas with less or no overstory cover were shallower, possibly due to lower litter input rates or extended spring snow coverage which would produce higher moisture levels promoting increased microbial activity (Möttönen *et al.* 1999). The high accumulation of litter and duff around the base of overstory trees has been observed in other studies (Ryan and Frandsen 1991).

Our results indicate there is substantial complexity in duff consumption and this should be taken into account when planning a prescribed fire. At the time the dry burn was conducted, there was low variation in duff moisture because of the proceeding dry summer. No significant trend was found in duff moisture with respect to location of the nearest overstory tree. Slight variations in duff moisture did not affect duff consumption, as all duff had a moisture content below 30 % and would be expected to burn completely (Brown *et al.* 1985).

In the moist burn, spatially differentiated duff data are not available, but based on moisture measurement at the stand level and reports of Ziemer (1968), Chrosciewicz (1989), Bouten *et al.* (1992), Otto (1994) and Möttönen *et al.* (1999), we believe duff moisture was approximately 60 % in areas with no canopy cover or in areas under the crown drip-line, and approximately 30 % beneath canopy cover.

From the observations made in this study, the question arises, under what conditions can spatial variation of duff consumption be expected? Previous research has shown the importance of duff moisture and has identified moisture thresholds with regard to smoldering combustion (Brown *et al.* 1985, Sandberg 1980).

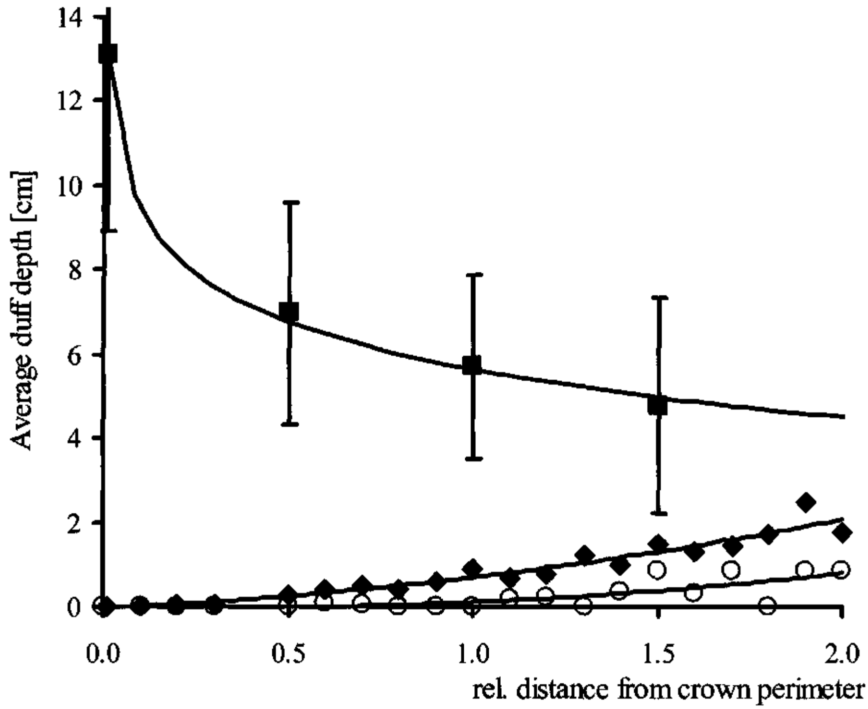


Figure 4.5. *Average duff depth (pre- and post-fire) in relation to relative distance from overstory trees (1.0 representing the edge of the crown). Pre-fire duff depth (filled squares) from the dry burn was modeled with $y = -1.63 \ln(x) + 5.64$ ($R^2 = 0.99$), post-fire duff depth for the moist burn (filled diamonds) with $y = 0.32x^2 + 0.41x - 0.04$ ($R^2 = 0.93$) and for the dry burn (empty circles) with $y = 0.29x^2 - 0.21x + 0.02$ ($R^2 = 0.60$).*

Three general categories of duff moisture are: i) <30 % duff moisture, which results in complete duff consumption ii) 30 to 120 % duff moisture, resulting in incomplete consumption, and iii) >120 % duff moisture, where no duff consumption is possible unless woody fuels are dry enough to sustain combustion on the duff surface (Brown *et al.* 1985, Sandberg 1980). Spatial variation of duff moisture, especially within the moisture range of 30 to 120 %, directly affects smoldering combustion and can create local differences in duff consumption (van Wagner 1972).

In this study, two of these three classes of duff consumption were observed. The dry burn consumed almost all organic material to the mineral soil and duff moisture was so

low at all locations, that smoldering combustion and propagation was not limited (<30 % duff moisture), and no spatial pattern was found. The moist burn most likely encountered spatial variation in duff moisture that stopped smoldering combustion in areas with high moisture content (30 to 120 % duff moisture). As a scenario, one can imagine a situation later in the rainy period, when duff moisture rises above 120 % at all locations in the stand. Under these moist conditions, there would be no duff consumption (only the fast-drying litter fuels might be dry enough to burn).
Although we can present pre-fire spatial duff moisture data only for the dry burn, we assume that duff consumption patterns in this burn (Fig. 4.3) was influenced by spatial differences in duff moisture that were caused by spatial varying throughfall rates. The 25 mm of rain one week before the moist burn was likely intercepted by tree crowns or possibly drained off the crown towards the crown perimeter (Otto 1994, Ziemer 1968). This resulted in low duff moisture in areas close to stem bases of overstory trees, and higher duff moisture in areas below the crown edge or with no crown cover.

Given the slow desorption rate of conifer duff (average timelag of 50 hrs for *P. ponderosa*; Fosberg 1977), this moisture was stored longer in the duff than in the exposed, fast drying litter. In the moist burn, the higher duff moisture content in areas with no canopy cover lead to a high latent heat flux for water evaporation, which finally stopped the smoldering combustion process. This assumption is supported by previous research on throughfall patterns in conifer forests (Ziemer 1968, Chrosciewicz 1989, Bouten *et al.* 1992, Whelan *et al.* 1998), and it would explain the increase in post-fire duff depth in the moist burn at distances of 2 to 4 m from the stem base or at a fraction of 0.75 to 1.25 of the crown perimeter (Fig. 4.4).

The probability of duff remaining after prescribed fire is much lower under crown cover than in areas with no crown coverage. An S-shaped function is suitable to model this relationship (Fig. 4.4). The low probability of duff surviving fire close to trees is due to less moisture and thicker, more continuous duff profiles (Miyanishi and Johnson 2002). The high correlation of our data with the Chapman-Richards equation indicates that this spatial trend in post-fire duff depth can be modeled (Fig. 4.5).
To improve duff consumption prediction on a stand-level we suggest using process-based duff consumption models with spatial differentiation of duff moisture depending on the overstory crown coverage. This would allow more precise predictions of duff moisture, which can have a major influence on fire effects. For example, duff consumption around the stem base is an important variable when predicting post-fire tree mortality (Stephens and Finney 2002).

Spatial variation in duff moisture should be considered when planning prescribed fires. Including this information should improve the accuracy of fire effect predictions. Duff moisture measurements from distinct locations (stem base, beneath the canopy drip-line, in openings) should be used as input for consumption and fire effects models.

Ideally, this approach gives spatial differentiated predictions of duff consumption, and therefore, predictions of direct fire effects, such as tree mortality and mineral soil exposure.

4.5 Conclusions

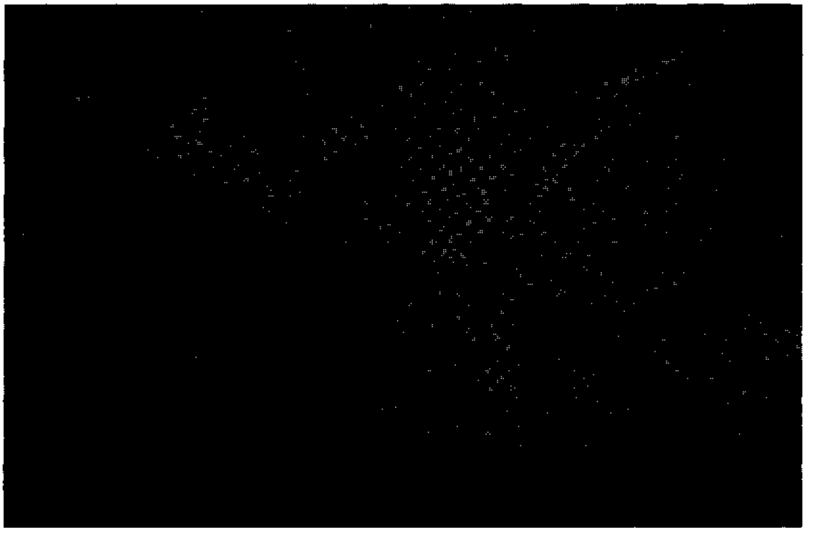
Spatial variation of duff consumption is strongly dependent on duff moisture. Although many of the factors affecting duff moisture and dynamic (such as throughfall, water uptake, weather, etc.), we conclude that duff consumption patterns can be improved by considering crown coverage of the dominant trees. In accordance with the model of decreasing influence of a single tree with increasing distance from it (Zinke 1962, Kuuluvainen *et al.* 1993), we found a strong positive correlation between the distance from the stem base and the probability of duff remaining after prescribed fires. Our aim was not to present a new process-based duff consumption model, but our results show that a spatial component which considers stand structure and spatial duff moisture variation could significantly improve the quality of process-based models in general, and could be used to guide the construction of better models and to test current models for their accuracy.

The S-shaped functions used in this work are suitable to describe the probability of duff remaining after prescribed fire. Spatial interpolation techniques, such as used by Robichaud and Miller (1999) that include overstory stand structure could be linked to process-based duff consumption models to increase the accuracy of duff consumption and consequently, fire effects.

CHAPTER 5

Physiological stress induced by fire

Submitted as: HILLE M, SASS-KLAASSEN U. GROWTH RESPONSE OF SCOTS PINE (*PINUS SYLVESTRIS* L.) TO FOREST FIRE DEPENDS ON FIRE INTENSITY.



Abstract

The effect of surface fires on the radial growth of Scots pine (*Pinus sylvestris* L.) as an indicator for fire-induced physiological stress was studied in four pine stands in Germany. Radial growth was significantly reduced up to seven years after fire in two of the stands, where the fire occurred in the late season and consumed almost the entire humus layer. In the other two stands, which burned in the early season and where the humus layer was charred only superficially, no clear effect on post-fire radial growth was observed.

Regression analysis for all sampled trees on the burned sites show that DBH and the degree of humus consumption have a significant influence on post-fire radial growth of bigger trees were less negatively affected after a fire and extensive humus consumption around the stem bases resulted in reduced growth. Char height, i.e. flame height, does not show any relation to post-fire growth. It is assumed that heat damage to (fine) roots and mycorrhiza as a result of extensive humus consumption caused a reduction in tree vitality reflected in reduced radial growth.

These results can be used to evaluate the effect of surface fires in Scots pine stands by pointing out relevant factors that affect post-fire tree growth and to estimate the physiological stress induced by fire of different intensity.

5 Growth response of Scots pine (*Pinus sylvestris* L.) to forest fire depends on fire intensity

5.1 Introduction

Disturbances in forest ecosystems most often have significant influence on the structure and dynamics of forest ecosystems from the gap- to the landscape-level. Depending on the disturbance intensity and severity, post-disturbance forest dynamics are altered and net primary production (NPP) is often influenced.

NPP of individual trees and stands, reflected in e.g. radial growth, is variable in time because plant growth depends on several internal and external factors, such as site ecology (Schweingruber 1996), the development stage of the forest and the individual tree (Ryan *et al.* 1997) as well as interannual variation caused by climate (Fritts 1976) and different types of disturbances. Disturbance by fire normally affects various parts of the trees, i.e. roots, stem and crown, which leads to injuries and often – through the reduction of the amount of photosynthetic tissue – to growth reduction (Gutsell and Johnson 1995), and tree mortality (Stephens and Finney 2002). In general, the resilience of a tree species and the stand against fire determines the decline in NPP and thus radial growth. Changes in radial growth pattern have been proved to reflect stress-induced changes in tree vitality (Banks 1991, Schweingruber *et al.* 1986, Schweingruber 1996).

Dendrochronology was used in several previous studies to show the impact of different natural and human-induced disturbances on tree growth. After disturbances on a stand- or landscape level, such as air pollution (e.g. Innes 1990; Oleksyn *et al.* 1993; Hornbeck and Smith 1985), insect attack (e.g. Armour *et al.* 2003; Payette *et al.* 2000), mistletoe attacks (Rigling *et al.* 2004) or fire (e.g. Bergeron and Charron 1994; Grissino-Mayer and Swetnam 2000) a reduced growth was observed in most cases.

In contrast to other disturbances, fire can have negative as well as positive effects on the growth of individual trees and stands. Negative effects of fire may be due to damage of living tissue, i.e. the foliage, the cambium and the roots, which affects tree vitality and leads to a reduction in radial growth (Vanninen and Mäkelä 2000). Factors such as scorch height (extent of living foliage killed), char height (indicator of flame height and therefore heat release on the stem; McInnis *et al.* 2004), and humus consumption (as an important indicator of damage on fine roots which are stored in the O_{FH}-layer) indicate the susceptibility of individual trees to fire. Positive effects of low-intensity surface fires are caused by an increase in the amount of plant-available nutrients (Covington and Sackett 1986, Beese and Divisch 1980), the reduction of competition within the tree community (Schmidt *et al.* 2004), and the removal of understory vegetation (Chandler *et al.* 1983). This could significantly increase the resource availability and thus the radial growth of the surviving trees.

For surface fires, both reduced and increased radial growth has been reported. Significant growth depression occurred in *Pinus palustris* Mill. (Boyer 1987), whereas for several fire adapted tree species, such as e.g. *Sequoiadendron giganteum* Lindl. (Mutch and Swetnam 1995) and *Pinus taeda* L. (McInnis *et al.* 2004), a growth release after fire was reported.
For *Pinus sylvestris*, the species under study in this report, a reduction of 17 % in radial growth in the first 10 years after the fire has been reported for Siberian forests (Wirth *et al.* 2002). On the long term no effect of fire on radial growth was recognized. From Scandinavia, a strong correlation between fire intensity and mortality of overstory trees was reported (Linder *et al.* 1998). For Scots pine from Switzerland, an increased radial growth was observed the first two years following the fire (Rigling *et al.* 2004). Besides that, up to now there is not much information available about the resilience of Scots pine to forest fire of different intensity, especially not for Central Europe, where fires occur in pine plantations that have been established on sites naturally occupied by broadleaved tree species (Zerbe and Brande 2003). Given the expected higher fire frequency in the future (Badeck *et al.* 2004), our knowledge about post-fire ecosystem response is rather limited.

The objective of this study is to evaluate the impact of surface fires of low to medium intensity on Scots pine in Central Europe. By analyzing post-fire radial growth patterns both on a single-tree and a stand basis, the aim is to study (1) how fires of different intensity affect stands of Scots pine, and (2) which parameters have the strongest impact on post-fire growth activity of the individual pine trees. It is assumed, that fire intensity reflected by flame height and degree of humus consumption play a major role to explain varying physiological stress levels that are induced to the trees, and therefore differences in post-fire radial growth.

5.2 Material and Methods

5.2.1 Sampling sites

Four previously burned pine stands in Germany were selected for this study. All stands are even-aged, single-species pine plantations (see Tab. 5.1 for inventory data). Parts of the stands were burned during forest fires set by arsonists. The study sites are located in *Brüggen-Bracht* (hereafter referred to as BB; near Mönchengladbach, North-West Germany), *Gandenitz* (GA; near Templin, North-East Germany) and two sites in *Hammer* (HA-old and HA-young; near Königswusterhausen, North-East Germany). On all four sites, pines stock on poor sandy soils, and grow under an average precipitation regime of 690 to 750 mm yr^{-1} and a mean annual temperature of 9.5° C in BB, 540 to 600 mm yr^{-1} and 8.0° C in GA and in HA-old and HA-young, respectively.

In each stand, two adjacent sampling areas were selected, one in the burned part of the stand and the other, as a control, in an unburned part of the same stand. For all four stands, both parts are similar considering stand - and soil characteristics. The presence of absence of fire was the only difference. This approach allows a direct comparison of growth data of the unburned and the burned part of all four stands (Pollanschütz 1966).

5.2.2 Sampling procedure

Sampling was done in late fall 2003. At each subsite, i.e. burned and unburned, of the four sample sites 20 healthy, dominant or co-dominant pines were randomly selected. From each tree two increment cores were taken at 1.3 m above the ground from opposite directions.
For an estimate of fire intensity, humus depth (O_{FH}) was measured with a ruler at four locations 1 m from each sampled tree to calculate humus consumption by fire. Maximum char heights, as the height on the bole up to where the bark was blackened by fire, were also recorded.

5.2.3 Dendrochronological study

Increment cores were mounted, surfaced, measured and cross-dated according to standard dendrochronological techniques (Swetnam *et al.* 1985). Measurement and cross-dating was performed using the programs TSAP (Rinn 1996) and COFECHA (Grissinio-Mayer 2001, Holmes 1983).
From radial growth data of the sampled trees on the control sites, the average radial growth for each year was calculated, and from these averages an overall ring-width value for the years from 1960 to 2003 (RWV_{60-03}) was derived for each of the four control sites (Mutch and Swetnam 1995). In a second step, a ring-width index (RWI) was calculated for each tree and each year following

$$RWI = \frac{RW_x}{RWV_{60-03}} \qquad \text{(Equation 5.1)}$$

with RW_x as the ring-width in year x. The RWI was calculated for both the pines at the control and those at the burned part of the four stands whereby RWV_{60-03} from the trees at the control site were always taken as base for the RWI calculation. This index allows a comparison between the annual radial growth of the trees from the burned and the control sites independently from medium-term growth trends and annual variability of other influencing factors such as climate. In this approach the impact of climate was not included into the data analysis, because this would have caused additional variation between the four sites and the different years of the burns. Instead, a pair-wise

comparison was done between the unburned and the burned parts of the sites, with the factor 'fire' being the only difference in stand history.
To compare post-fire radial growth between the control and the burned area, two-tailed, unpaired t-test were performed with the RWI's of both areas for each year within a period from three years before to seven years after the fire event.

To assess, which fire-related factors mainly affected annual tree-ring width of the pines on the burned area, a regression model was created with the RWIs of the three post-fire years as dependent variable and DBH, char height and humus consumption (as the difference between humus depth around each tree at the burned area and the average humus depth at the reference site) as independent variables.

5.3 Results

Fire intensity, expressed as char height and percentage of humus consumption, varied between the four sites sampled (Tab. 5.1). Highest char heights were found on sites HA-young and BB, with char heights of about 130 cm (Tab. 5.1). Humus consumption varied considerably with values between 10-15 % on sites HA-young and HA-old and 70-72 % on sites BB and GA. Despite the high fire intensity, especially on the sites BB and GA almost all trees survived the fires and the stands are still fully stocked (Tab. 5.1).

Table 5.1. *Stand inventory in fall 2003 and fire-related information for the four burned pine stands (average ± SD).*

Site	Age	Stem density	Basal area	Char height	Humus consumption	Date of fire event
	[yrs]	[N ha^{-1}]	[m^2 ha^{-1}]	[cm]	[%]	
HA-young	45	2050	25	131 (55)	10 (10)	30.04.1999
HA-old	69	950	27	113 (40)	15 (11)	30.04.1999
BB	65	1050	26	133 (58)	70 (14)	10.08.1976
GA	43	1560	23	70 (42)	72 (21)	30.07.1994

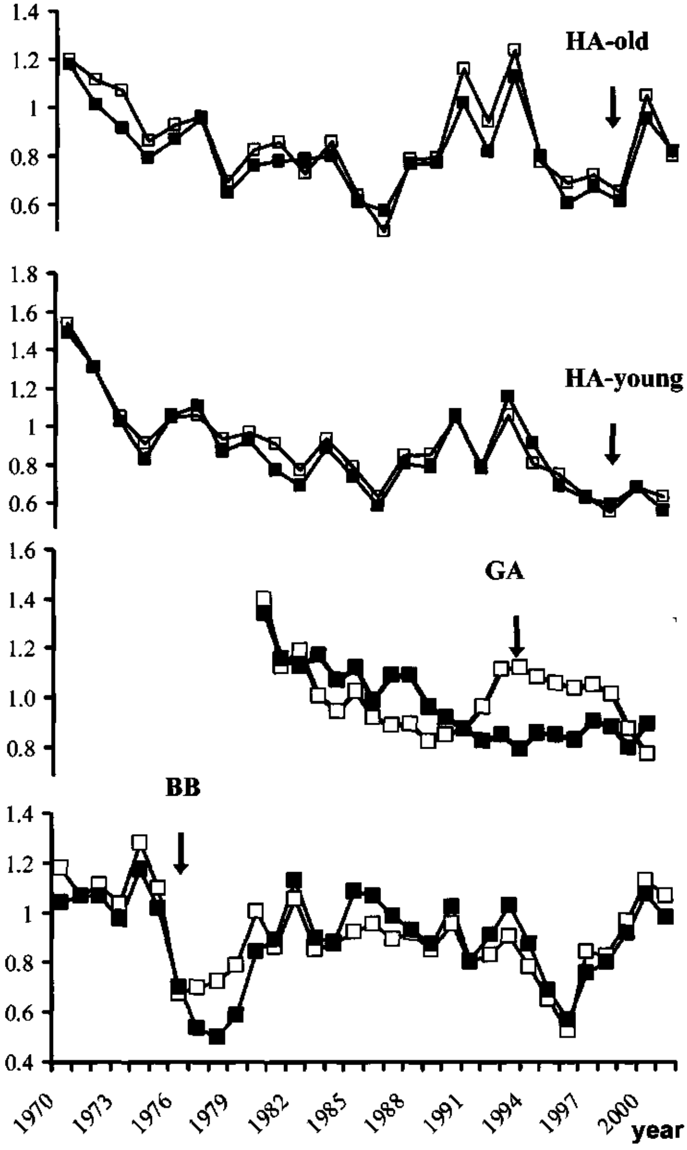


Figure 5.1. *Average radial growth series (indexed) of trees at the control (empty squares) and burned areas (filled squares) of the four sample sites. Fire years are indicated by arrows.*

Mean radial growth of the pines on all four sites varied between 0.5 and 5 mm for the time span analyzed. The ring-width index $RWV_{60\text{-}03}$ was 1.99, 2.64, 1.63, and 2.74 for sites BB, GA, HA-old and HA-young, respectively. The higher RWV-values for GA and HA-young reflect the typical faster juvenile growth of Scots pine compared to older trees. Some significant marker years (drought years such as 1976 or 1999), led to reduced annual growth of pines from all four sites (Fig. 5.1).

For site HA-old, no effect of the fire in 1999 on the average radial growth was observed. The indices calculated for the trees from controlled and burned stands follow the same growth trend for all years following the fire (Fig. 5.1). Comparing the annual RWI for trees from the burned and unburned area at HA-old with a two-tailed, unpaired t-test, revealed higher radial growth for the trees at the control areas for the four years following a fire. However, differences were not significant (Tab. 5.2).

Site HA-young shows a similar impact of the fire on radial growth, with the tendency of higher radial growth of the control trees (except in post-fire year 2), but again, differences were not significant (Fig. 5.1; Tab. 5.2).
For site GA, post-fire radial growth was significantly higher on the control site. Here, radial growth on the control site was at a high level in the five years after the fire event (1994), but not within trees on the burned area (Fig. 5.1, Tab. 5.2).

A reduction in post-fire radial growth was also found in trees stocking on the burned area of site BB (Fig. 5.1). Here, the growth in the first four years following the fire event was significantly reduced (Tab. 5.2). Note that the fire in BB occurred at the end of the growing season, so that growth in the fire year (year '0') was not affected.

Table 5.2. *Capitalized letters C (Control) and B (Burned) indicate where trees grew better in certain years at the four sampling sites. Significance levels, based on paired t-test for each post-fire year, are *= 0.1, **= 0.01 and ***= 0.001; n = 40 for each site.*

Site	Years before and after fire event										
	-3	-2	-1	0	1	2	3	4	5	6	7
HA-young	B	B	B	C	C	B	C	C	-	-	-
HA-old	C	C	B	C	C	C	C	B	-	-	-
GA	B	B	B	C	C^{***}	C^{***}	C^{***}	C^{***}	C^{***}	C^{***}	C^{***}
BB	B	C	C	B	C^{*}	C^{***}	C^{**}	C^{**}	B	B	B

The results of the regression model indicate that tree diameter (DBH) and humus consumption significantly affected post-fire radial growth (Tab. 5.3). Bigger pines within a site were less affected by the fire. Moreover, it became obvious that trees with higher humus consumption around their base show a stronger growth reduction after fire than trees with more humus remaining. Char height, as an indicator for flame height was not a significant factor. Also, no intercept for the regression model was found to be significant. This pattern was consistent for all three years following the fire - the years in which a reduced growth in all four sites was found (Tab. 5.2).

Table 5.3. *Regression model to explain post-fire radial growth for all measured trees (n = 4 sites times 20 trees = 80 trees). The general model consist of growth indices for each of the first three years after fire and the average growth for the first three years (indexed by average radial growth) on the control sites as dependent variables, and DBH, char height and humus consumption around the stem base as independent variables. Intercepts were not significant in any year.*

	F	p	DBH	Char height	Humus consumption	R^2
			[cm]	[cm]	[cm]	
Year 1	2.23	0.1165	0.0177*	n.s.	-0.051*	0.07
Year 2	4.41	0.0166	0.0230**	n.s.	-0.067***	0.14
Year 3	3.94	0.0249	0.0217***	n.s.	-0.039**	0.12
Year 1-3	3.91	0.0256	0.0208**	n.s.	-0.052**	0.12

5.4 Discussion

The results for the four sampling sites show that *Pinus sylvestris* is able to survive surface fires of low- and medium intensity. This holds even true for the trees on sites GA and BB, where the fire burned with flame height of up to 1.3 m on average and removed more than 70 % of the humus layer. This points to the fact that the heat impact on the bark during the fire was not high enough to kill larger parts of the cambium although several fire scars were found at the sites GA and BB. This result is in line with findings of a higher probability for larger pines to survive a surface fire (Rego and Rigolot 1990). Even though no data on pre-fire stand structure is available, it is assumed that fire-induced mortality was very low because all four stands are almost fully stocked and no obvious gaps or openings were found.

However, it is obvious that the fire events had an impact on the radial growth of the pines. The impact was significant at the sites GA and BB where fire intensity was highest, with about 70 % of the humus layer being removed. The pines at these sites show a radial growth reduction during a period of at least four years following the fire. These results differ from observations of an increased growth for Scots pine after fire in Switzerland (Rigling *et al.* 2004) and in sub-fossil Scots pine from England (Lageard *et al.* 2000), but are in line with findings from Siberian Scots pine forests, where a

growth decline was found for the first decade after the fire, followed by a period of increased growth (Wirth *et al.* 2002).
However, at all four sample sites in this study, radial growth rates of the pines were not significantly different between control and the burned areas seven years after the fire. This shows that Scots pine is able to recover soon after fire disturbance.
The observed growth reduction of the pines from the burned areas was linked with two factors, namely DBH and humus consumption. Bigger trees were found to grow relatively better in the first three post-fire years than small trees, which indicates that they are less affected by a fire event. The degree of humus consumption and post-fire radial growth were negatively correlated. Char height, however, had no significant influence on post-fire growth, which shows, that flame heights of 1.3 m probably did not affect the foliage of the pine trees.
For Scots pine, reduced transpiration and thus a generally reduced photosynthetic activity with the consequence of reduced radial growth was observed, when the majority of upper roots was harmed. The symptoms are comparable to the effect of a drought, which lead to reduced radial growth even if substantial water is available for roots in greater depths (Irvine *et al.* 1998). Since the majority of fine roots and mycorrhiza are located in the upper soil horizon (Roberts 1976), they are very susceptible to damage during fire - especially in case of intense humus consumption (Stendell *et al.* 1999, Grogan *et al.* 2000). This explains the relatively strong effect of fire in the sites BB and GA. Root damage and subsequent restoration of the root and mycorrhiza systems together with the consequence that water- and nutrient transport will be limited throughout a certain period after the fire will result in a reduction of NNP and thus radial growth of the pines.
The stronger effect of the fires at the sites GA and BB could also be related to the timing of the fire event. Fires in the early growing season, like at HA-old and HA-young (Tab. 5.1), when the humus layer was still moist from the winter precipitation, where related to less humus consumption (Hille and Den Ouden 2005b). Less humus consumption means less harming of the roots and mycorrhiza, which would explain, that despite high char height (Tab. 5.1), NPP was not affected and growth reduction was not observable.
Similar results were found for Scots pine in the boreal forest, where early season fires did not reduce ring width, but late season fires did (Lehtonen and Kolström 2000).
Another aspect is that extraordinary dry climate conditions, like in 1976, when the fire occurred at site BB, lead to a strong soil-water deficit which results in growth stress for the trees, even on undisturbed sites. This most likely enhanced the negative growth reaction of the pines in the burned area at site BB. Severe ecological impacts like prolonged soil-water deficits in combination with forest fire in a specific year frequently cause physiological responses of the tree that are carried over into subsequent years (Innes 1993). This might be the reason for the reduced radial growth throughout a period of several, up to seven, years after the fire event.

It can be summarized that both, the climate situation in the year of a fire event as well as after the fire event and the season when the fire occurs play an important role the post-fire radial growth of pine.
Taking the results of this study into account different post-fire stress levels to valuate the impact of a fire can be defined:

a) Low-intensity surface fires that occur under moist conditions, i.e. in winter or in the early growing season, when humus moisture is high, lead to restricted humus consumption and thus a limited root and mycorrhiza damage. After such fires, the pines did not show a reduced radial growth (HA-young and HA-old).
b) High-intensity surface fires whereby the majority of the humus layer is consumed cause damage of the root system and induce growth stress to the pines with the consequence of reduced NPP and radial growth (GA and BB).
c) Very high-intensity surface fires whereby the humus layer is completely consumed and large parts of the root system and the cambium are damaged kill the trees or expose it to lethal bark beetle attack (Linder *et al.* 1998).

With this knowledge about fire effects on Scots pine from Central European forests, post-fire stand and single tree reaction can be predicted. This offers the possibility to assess the impact of forest fires on stand dynamics and to predict the fire-induced physiological stress. For prescribed fires, our results can be used to design a prescription to reach a given management goal: for fuel reduction a fire as described under a) should be applied. For stimulation of natural regeneration, where mineral soil should be exposed (Hille and Den Ouden 2004, Mallik 2003), fires under b) would be suitable whereas the c-type reflects stand-replacing fires.

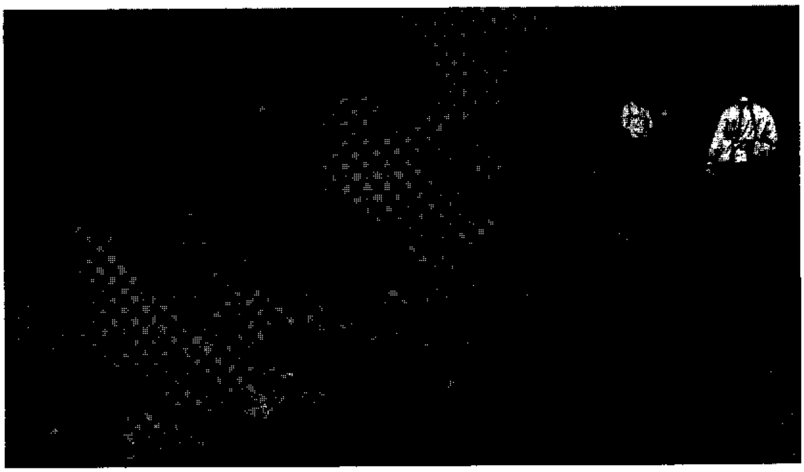
5.5 Conclusions

Surface fires of high intensity cause a radial growth reduction in pine trees, which persists throughout a period of up to seven years following the fire event. Low intensity surface fires, which leave the humus layer and the embedded fine roots untouched, however, have no impact on radial growth. Diameter of the affected trees and especially humus consumption have been found to act as key parameters which determine physiological growth stress and mortality rates of the pines and should be assessed before taking decision about how to deal with a burned stand.
Pinus sylvestris is a species, which faces frequent forest fires in its natural environment (Lehtonen and Kolström 2000, Agee 1998a). This study showed that it has a similar tolerance towards low to medium-intensity surface fires in Central Europe.

CHAPTER 6

Change in stand structure

Published as: SCHMIDT L, HILLE M, VON GADOW K. 2004. ANALYSE EINES BODENFEUERS. VERÄNDERUNGEN DER BESTANDESSTRUKTUR UND DES WACHSTUMS. *ALLG FORST J ZTG* 175: 78-82



Abstract

The emphasis of this paper is to assess the impact of a forest fire on a 43 year old Scots Pine plantation in Eastern Germany. Important results include a partial reduction of the litter and humus layer, a high mortality in the smaller dbh-classes and temporarily lower radial tree growth when compared to an adjacent, unburned stand. The effects of the fire are similar to those observed in a thinning from below.

In this paper, we also try to integrate the observed fire effects into the only existing succession model for pine stands in Germany, known to us. The classification of this fire as a 'non-stand-replacing' disturbance adds a new component to this model and highlights the high variability of possible successional pathways after fire.

6 Analysis of a surface fire – change in stand structure and growth[1]

6.1 Introduction

Modern forest research assesses, describes and analyses forest development with respect to several different disturbance factors. While the consequences of storm disturbances are widely known, the disturbance factor fire is considered relatively seldom. Within the last two decades, the attention for forest fires and their effects has grown worldwide. Especially in boreal, Mediterranean and tropical forests a large body of knowledge on the natural fire regime and ecological effects has accumulated. At the same time, research into fire ecology received only little attention.

The tree species with the highest attention for fire ecology in Central Europe is Scots pine (*Pinus sylvestris* L.). In this species' natural range in continental Eurasia, fire is the main agent determining forest dynamics (Zackrisson 1977, FIRESCAN 1996, Agee 1998a). The only succession model for Central European pine stands that also includes forest fire was built by Otto (1995). His approach is based on observations and results after the large-scale fire in the *Lüneburger Heide* in the 1970's. These fires were all stand-replacing crown fires (Luttermann 1976). In this model, it is assumed, that all stands are completely destroyed and no trees survive the fire. The natural succession starts with a colonization by fungi, mosses, grasses and, at a later stage, pioneer tree species (Butin and Kappich 1980, Jahn 1980). These effects of the fire are classified as catastrophic.

However, other observations show, that forest fire can also be a short-term disturbance which does not lead to the destruction of the burned stand (Goldammer 1979, Mrazek 2002). On some occasions, fire accelerated succession by stimulating natural regeneration in old pine stands (Recke 1928, Chandler *et al.* 1983). The degree of damage in the stand depends mainly on the fire intensity, e.g. expressed as energy release per area. Fire can have an impact on trees by damaging the crown, fine roots or the cambium (Ryan 1982, Stephens and Finney 2002). Each of these factors or a combination of several damage types can induce stress to the trees on the burned area which can lead to reduced growth and vitality and finally to mortality on a stand level. Other parameters, such as humus depth after fire, can be included to predict the consequences of fire (Hille and Stephens 2005).

In this paper, we analyze mortality, change in stand structure and tree growth after fire in a young pine stand. Our goal is to assess the fire-induced stress on a tree- and a stand-level. Additionally, we extend the mentioned post-fire succession model by Otto (1995) with our results and those from other studies.

[1] This chapter is a translation of the original paper.

6.2 Methods

This study is based on a comparison between a burned area and an adjacent unburned area in compartment 105a2 of the communal forest Templin, Brandenburg State, Germany. On the 30th July 1994, a surface fire burned through 1.15 ha of the 4.0 ha stand, a 43-year old even-aged stand. In the following, the burned area is referred to as part A, the unburned part of the stand as part B. This single-species stand with a closed canopy originated from seedlings planted into the berm after plowing. The soil is mesic sandy soil. The forest type is classified as blueberry-Scots pine association (*Myrtillo-Cultopinetum sylvestris*; Hofmann 1997).

In the years before the fire, both parts of the stand were managed identically, so that a direct comparison between part A and B is possible. The fire is the only difference between the two parts (Pollanschütz 1966). According to the forest manager, the fire burned homogenous on part A with flame lengths of approximately one meter.

In circular forest inventory plots of 500 m² the following six parameters were recorded for each tree: x- and y coordinates, dbh, tree height, social tree class and char height on the bole. We located three plots in part A and three in part B on a systematic grid.

To analyze radial tree growth, radial increment cores were taken from dominant trees within each plot at 1.3 m height and analyzed in the laboratory (Athari 1980). To analyze the change in litter and humus depth, we sampled the depth of the O_L and O_{FH} layer on both parts of the stand. Samples were takes in the berm and between berms.

6.3 Results

The fire in the sampled stand was a surface fire of low to medium intensity. The flames did not reach the live foliage, and no scorching was observed. The observed char heights on the boles confirmed the verbal report of flame lengths of approximately 1 m. Average char height was 70 cm, with a range between 10 and 250 cm (Fig. 6.1). There was no significant correlation between dbh and char height.

On almost all trees only the outer bark plates were charred, while the inner bark layers were not blackened. Two pines showed fire scars, indicating that the fire killed the cambium on approximately 25% of the stem circumference. In both cases, the trees had started to overgrow the wound.

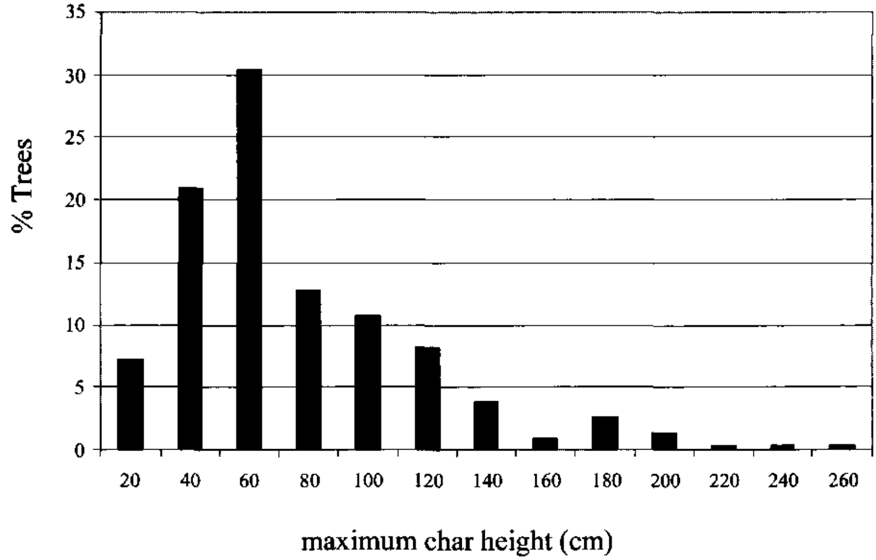


Figure 6.1. *Histogram of maximum char height of all pine on the burned site. On the y-axis the percentage of trees within the char height classes. The x-axis represents upper limits of char height classes.*

The low intensity of the fire became also evident from the only partly consumed humus layer. A thin charcoal layer was found between the humus layer and the litter, which accumulated after the fire. Eight years after the fire, the litter and humus layers were still shallower on the burned area than on the unburned (1,6 cm O_L, 0,9 cm O_{FH} on the burned area; 2,5 cm O_L, 5 cm O_{FH} on the unburned). This difference is significant (unpaired t-test for O_L: $Df = 14$, $t = 4{,}61$ $p < 0{,}001$; O_{FH}: $Df = 14$, $t = 10{,}68$, $p < 0{,}001$). At all locations, some remaining humus was found, the mineral soil was not exposed within the sampled plots.

A further differentiation between sampling locations in the berm and between two berms partly explains the observed humus depth variation. At locations in the berm, a significant lower humus depth was found, on both the burned and the unburned part of the stand. A two-way ANOVA shows the impact of plowing and fire on today's humus depth (Tab. 6.1).

Table 6.1. *Results of the two-way ANOVA, with ‚fire occurrence' (yes/no) and site preparation (berm/exposed soil). Both factors have a significant influence on total humus depth.*

	O_L				$O_{F,H}$			
Factor	Df	SS	F-Value	p	Df	SS	F-Value	p
Plowing	1	0.88	1.49	0.23	1	5.28	4.93	0.034
Fire	1	11.88	20.17	< 0.00	1	157.53	147.07	< 0.000

6.3.1 Stand structure

Stand density on the burned part A (1560 ha^{-1}) is significantly lower than on the unburned part B (1840 ha^{-1}; unpaired t-test with Df = 4, t = 1,51, p = 0,096). Since no harvesting has occurred after the fire and both parts originated from the same planting density, this difference can only be caused by the fire. The dbh-distribution gives an explanation for the lower stand density on the burned area (Fig. 6.2).

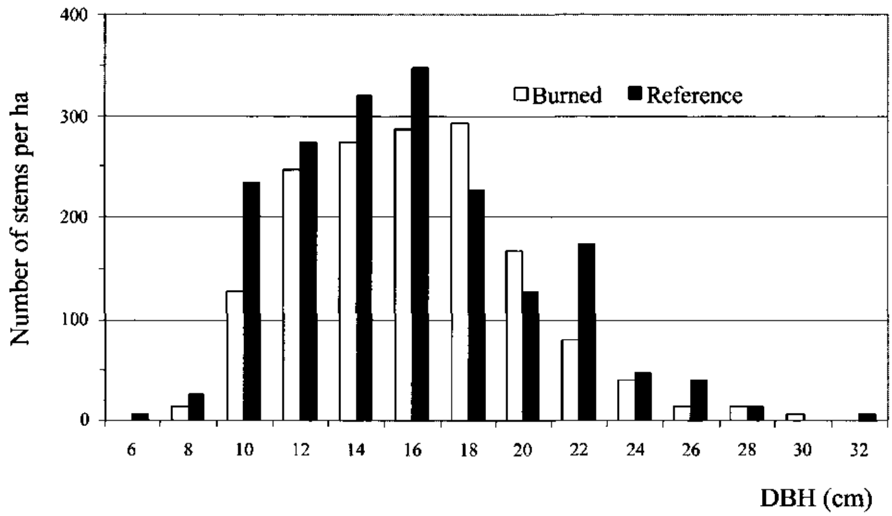


Figure 6.2. *Diameter distribution on the burned (black bars) and unburned area (white bars). More smaller pines are stocking on the unburned part, while the distribution is slightly skewed to higher DBH-classes.*

Compared to the unburned part, less small pines are present in the burned part. In the upper dbh-classes, no difference in stem numbers becomes evident. Consequently, the average dbh on the burned area is higher than on the unburned area, but this difference is not significant. The lower stem number on the burned area also influences basal area. The unburned area has a significantly higher basal area (36 m^2/ha) than the burned area (31 m^2/ha; t-Test, Df = 4, t = 2,04, p = 0,056). However, both parts of the stand are fully stocked and no gaps are evident.
The missing small trees on the burned area also influence the histogram of social tree classes (Fig. 6.3). On the unburned area, a higher portion of trees in Kraft's crown position index (Kraft 1884) classes 4 and 5 (suppressed trees) were found than on the burned area. In contrast, a higher proportion of dominant trees were found on the burned area.

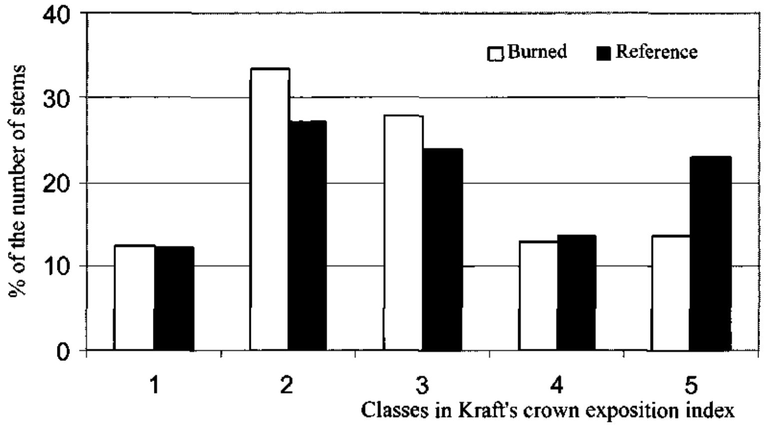


Figure 6.3. Histogram of tree classes on the burned and unburned part of the stand.

6.3.2 Radial growth

Looking at the radial growth of the sampled trees on the burned area, no abrupt change in annual radial growth was observed. From 1981 to 1994 (the time before the fire), annual radial growth declines – after the fire, radial increment is constant around 2.25mm a^{-1} (Fig. 6.4).
However, compared to the unburned area, annual increment on the burned area is lower in the years 1994 to 2001. This difference is largest in 1995, the first year after the fire occurred. In the following years, this differences becomes less. In 2002, no differences were visible.

Before the fire (1981 to 1993), no significant differences in radial increment were found between trees from the two sampled areas. Average year-ring width was 2.65 mm a^{-1} (Tab. 6.2). In the years after the fire (1995-2002), annual growth is significantly lower on the burned (2.25 mm a^{-1}) than on the unburned area (2.55 mm a^{-1}). However, following Figure 6.4, this difference is not caused by a reduced growth on the burned area, but by a temporal increased growth on the unburned area.

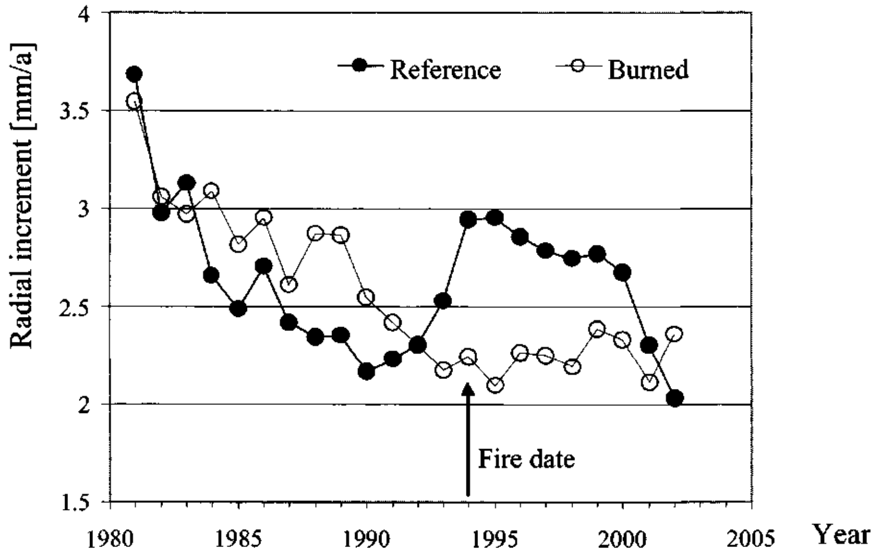


Figure 6.4. *Radial increment for trees on the burned and unburned part ('Reference') of the stand, from 1981 to 2002.*

6.4 Discussion

Forest research assesses, describes and analyses forest dynamics with respect to several different disturbance agents. The impact of storm has been in the focus of research quite extensively. In contrast, forest fire in Central Europe has not received much attention in the past (Gadow 2000). However, the world-wide interest in the interaction between fire and forest dynamics is huge. From our results it becomes clear, that the fire in the sampled pine stand was of low intensity and low severity. This becomes visible e.g. by the remaining humus layer of approximately 1 cm thickness, which reduced heat impact on fine roots. The char heights of 1 m on average proves the relatively short flame lengths.

Table 6.2. *Radial increment on the burned and unburned part of the stand between 1981 and 2002, measured on increment cores of 10 dominant pines.*

Time span	Area	$\bar{x}$ [mm]	Minimum [mm]	Maximum [mm]	t	Df	P
1981-1993	Burn (A)	2,74	1,36	6,10	1,18	212	0,2371
	Reference (B)	2,61	1,53	4,54			
1995-2002	Burn (A)	2,25	0,71	4,68	1,96	142	0,0517
	Reference (B)	2,55	1,13	4,57			

The change in stand structure indicates the impact of the fire on the stand structure. Mortality after fire was selective (Fig. 6.2). Suppressed trees with small dbh were more likely to be killed than bigger trees. These observation confirms previous results (Morris and Mowat 1958, Goldammer 1979). Mortality by fire is correlated to dbh because dbh is positively correlated to bark thickness, which is the most important factor determining cambium mortality (Rego and Rigolot 1990).
The radial growth of surviving trees was only slightly influenced by the fire. The analysis of the sampled increment cores shows that, although there is a significant difference in radial growth, this difference is mainly due to short-term higher increments for trees on the unburned area. There was no growth depression after the fire, which leads to the conclusion, that the fire-induced stress was low. We expected a higher annual growth on the burned area, due to the lower stand density (Johansen 1975, Lageard *et al.* 2000).
Overall, this fire can be compared to a thinning-from-below. The fire killed only suppressed trees. Therefore, our results give an example for a fire, which can not be classified as a ‚disaster' – simply because the impact did not lead to a significant change in stand structure or forest composition. Together with other recent observations after fire, our results can be used to extend the post-fire succession for Central European pine stands (Fig. 6.5).

As shown in Figure 6.5, post-fire stand-development depends mainly on fire severity and the degree of damage in the ecosystem. The fire can be classified as an ecological disaster, as was the case after high severity, stand-replacing fires in the Lüneburger Heide. Succession starts on bare mineral soil, and goes through several stages (fungi, mosses, grasses, herbs, pioneer trees species), as shown on the right side of Figure 6.5 (Butin and Kappich 1980, Jahn 1980).

Several other examples show that a forest fire can also be an event with minor impact on the ecosystem. As shown on the left side of Figure 6.5, forest fire of low severity is similar to a thinning-from-below.

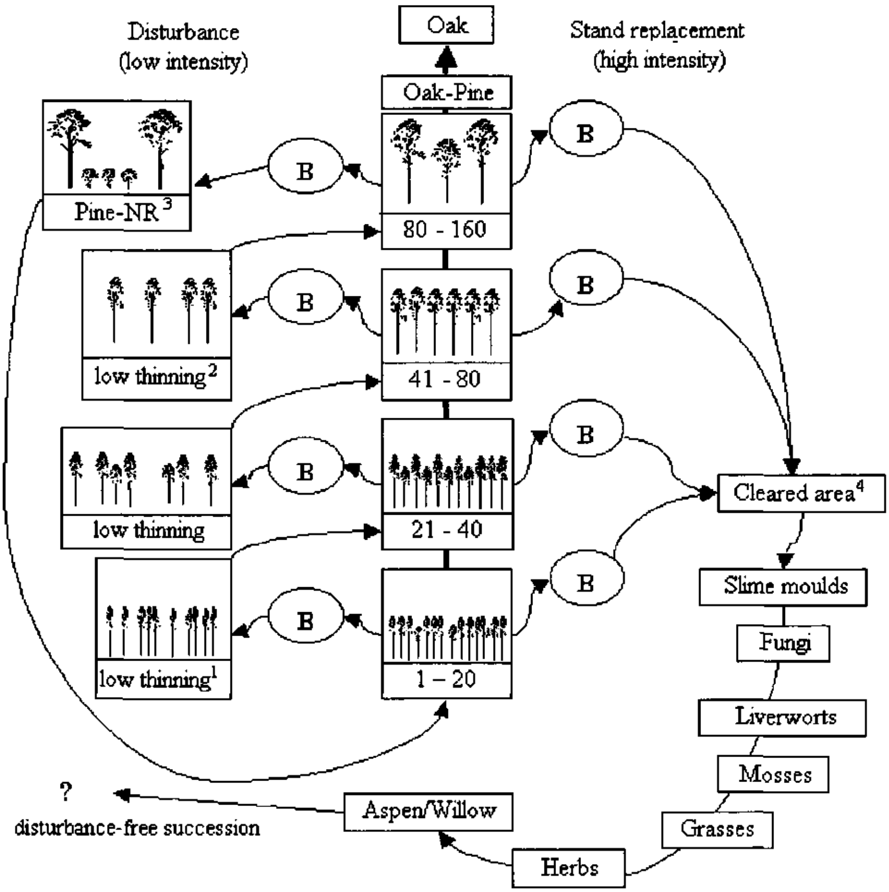


Figure 6.5. *Multi-variable succession in pine stands with regard to fires of different intensity (extended from Otto 1995). Forest fire (B) with stand-replacing intensity destroys the stocking stand in all age-classes and the humus layer, succession will start on a cleared site (right side of the figure). Non stand-replacing fires of lower intensity have effects similar to those of a thinning-from-below in young stands and can stimulate natural regeneration in older stands (left part of the figure). NR = Natural regeneration.*
Authors: 1) Goldammer 1979, 2) this study, 3) Recke 1928, Klein 1964, Hinz 1993, 4) Butin u. Kappich 1980, Jahn 1980

The 'intensity' of the thinning depends mainly on fire intensity – which can also vary within a given stand. Additionally, there are several reports of a spontaneous recruitment of pine following a fire in older pine stands (Recke 1928, Conrad 1925).

Several conditions, which are necessary for a successful establishment of pine seedlings are created by fire, such as the exposure of the mineral soil (Dohrenbusch 1997).

In summary, this studied pine stand was only slightly influenced by the fire. Scots pine has a similar resistance against fire than in this species' natural range in the boreal forest. With respect to future climate change and more frequent forest fires (Gerstengabe *et al.* 1999) there is an urgent need to expand our knowledge on the effects of fire in Central European pine stands.

CHAPTER 7

Germination and fire

Published as: Hille M, Den Ouden J. 2005. Charcoal and Activated Carbon as Adsorbate of Phytotoxic Compounds - a comparative study. *Oikos* 108: 202-207

Abstract

This study compares the potential of natural charcoal from Scots pine (*Pinus sylvestris* L.) and activated carbon to improve germination under the hypothesis that natural charcoal adsorbs phytotoxins produced by dwarf-shrubs, but due to it's chemical properties to a lesser extent than activated carbon. Activated carbon has been used in many bioassays as an adsorbate to clean aqueous solutions.

We used aqueous extracts from young leaves of *Calluna vulgaris* (L.) Hull and *Vaccinium myrtillus* (L.) as phytotoxin sources in two different concentrations (10 and 14 gr. of dried leaves in 100 ml distilled water). Germination of pine seeds was prevented by the higher concentration of both species, while the lower ones did not show significantly reduced germination. Both ericaceous species showed a very similar potential to prevent germination of Scots pine seeds.

Supplemented carbon (activated carbon, powdered or granulated pine charcoal) restored germination in strong extracts. Adding activated carbon resulted in germination of almost 100%. With pine charcoals added, lower germination percentages were observed. The charcoal powder was more effective (60 % for *C. vulgaris*; 28 % for *V. myrtillus*) than the charcoal granulate (30 % and 16 %, respectively) in restoring germination.

Chemical and surface analysis of the three carbon supplements revealed that activated carbon had by far the biggest active surface area (641 $m^2 g^{-1}$), and thus many more cavities to bind phytotoxins than natural charcoal (total surface area of 142 $m^2 g^{-1}$).

We conclude, that charcoal produced by forest fires can have a positive effect on seed germination, but to a much lesser extent than activated carbon. Previous studies, which used activated carbon as an equivalent for charcoal, overestimated the effect of charcoal on germination.

7 Charcoal and activated carbon as adsorbate of phytotoxic compounds - a comparative study

7.1 Introduction

Some forest floor vegetation types, common in Central European pine forests, consist of species which are known for their effectiveness in preventing tree establishment. In particular, *Calluna vulgaris* (L.) Hull, *Vaccinium myrtillus* (L.), *Empetrum hermaphroditum* (Hagerup) and *Pteridium aquilinum* (L.) Kuhn are often successful in maintaining monotonous field layers without trees invading and therefore causing a stagnation in succession (Gimingham 1994, Jäderlund 2001). Mechanisms, which prevent tree establishment include the build-up of organic matter (Ohlberg 1957, Sydes and Grime 1981, Den Ouden and Vogels 1997, Dohrenbusch 1997), strong resource competition (Buchmann *et al.* 1996, Rode 1993, Mallik 1995) and phytotoxic interference (Nilsson 1994). For a review of conifer regeneration problems due to ericaceous understory see Mallik (2003).
Recent research has shown the effects of phytotoxic interference in *V. myrtillus* and *E. hermaphroditum* dominated ecosystems in the boreal zone, especially in terms of reduction of seed germination and seedling growth (Jäderlund *et al.* 1998, Nilsson 1994, Zackrisson *et al.* 1997b). For *C. vulgaris* only the release of potentially toxic polyphenolic compounds and their presence in the humus and the upper mineral soil has been shown (Robinson 1972, Jalal *et al.* 1982, Jalal and Read 1983), but to our knowledge, their effects on germination have not been tested yet. Activated carbon added to water extracts in laboratory experiments and to the soil surface in field experiments reduced the phytotoxic effects by adsorbing phytotoxins (Zackrisson and Nilsson 1992, Nilsson 1994, Jäderlund *et al.* 1996, 1997, 1998; Zackrisson *et al.* 1997b, Nilsson *et al.* 2000). From the results of the studies mentioned a key role of natural charcoal in post-fire succession by phytotoxin removal was postulated. Amounts of charcoal found in burned stands in Sweden ranged from 980 to 2,074 kg/ha, had a homogenous size class distribution on a mass basis and were not correlated to the time since the last fire (Zackrisson *et al.* 1996).
Zackrisson and Nilsson (1992) observed a complete removal of the phytotoxic effect by added activated carbon and concluded, that "fire also creates charcoal, which acts as activated carbon reducing the allelopathic potential of the remaining litter and humus". Zackrisson *et al.* (1996) also used activated carbon in bioassays and concluded that "if we assume, that wildfire charcoal has effects comparable to activated carbon, then it is [...] to cause appreciable ecological effects when added to the surface of humus in forest plots". Charcoal was therefore suggested to be a long-term adsorptive pool with an ecological effect similar to the effect of activated carbon in bioassays. However, the proof that natural charcoal has indeed the same adsorbic potential as activated carbon, is lacking.

We doubt, that results from studies with activated carbon can be used to explain the role of charcoal in the forest floor, especially since the adsorption potential of ordinary charcoal is rather small compared to activated carbon (Cookson 1978, Huang 1978, Gore 1982). The specific surface area is the most important variable for adsorption, while other characteristics (i.e. pore structure, electrophoretic properties and surface acidity) are of minor importance (Huang 1978).

The objective of this study is to compare the magnitude of phytotoxin reduction by natural charcoal and activated carbon. In a comparative study we want to test the hypothesis, that the effectiveness of charcoal as the natural adsorber produced by forest fire to reduce phytotoxic concentration produced by *V. myrtillus* and *C. vulgaris* is much lower.

7.2 Methods and Materials

7.2.1 Toxin adsorption in bioassays

Bioassays with extracts of *V. myrtillus* and *C. vulgaris* were used in this study. Fresh leaves of *V. myrtillus* and *C. vulgaris* were collected in May 2002 at pine-forest sites in eastern Germany (app. 80 km north of Berlin) dominated by one of the two species in the field layer. The plant parts were air-dried at 20° C until no further weight loss was recorded. Extracts were prepared by soaking air-dried leaves in distilled water for 48 hrs at 20° C. Two solutions of different concentration were prepared: 0.1 g leaves per ml distilled water (representing a 10 % concentration) and 0.14 g ml^{-1} (14 %). The solutions were passed through a filter paper (*Schleicher and Schuell Type 595*).
The charcoal was produced from branches and twigs of Scots Pine in an oven with adjustable oxygen inflow. The procedure for charcoal production was to ignite approximately 200 g of air-dried (12 – 15 % moisture content) short branch and twig pieces (5 to 10 cm length) at 450° C and to close the oxygen inflow after 1 minute of combustion. The charred material remained in the oven until it cooled down. The charcoal was powdered using a mortar or granulated to pieces of < 10 mm in diameter (Tab. 7.1). Activated carbon powder (Type 18001, formerly called "*Labasco*") was provided by SIGMA-ALDRICH, Seelze, Germany. This material consists of charcoal from Scots pine wood, which has been cleaned and steam-treated to increase porosity.
Three ml of the extract were put into 70 mm diameter Petri-dishes. Depending on the treatment, 0.5 g of either powdered charcoal, granulated charcoal or activated carbon was added to the dishes, stirred and remained untreated for 24 h. Charcoal (either powdered or granulated), activated carbon or no carbon supplement was added to the solutions.

***Table** 7.1. Size class distribution for granulated and powdered pine charcoal, and activated carbon powder.*

	Fraction (%) of total mass in size class			
	< 1.0 mm	1.0 – 1.7 mm	1.7 - 6 mm	> 6mm
Granulated Pine Charcoal	3	3	26	12
Powdered Pine Charcoal	88	8	8	0
Activated Carbon	100	0	0	0

On top of the carbon, a single sheet of filter paper was added (*Schleicher and Schuell Type 589*), on which 50 pine seeds were located. Scots pine seeds (*Pinus sylvestris* L.) from the federal seed agency of the state of Brandenburg, Germany (Origin: *Waldsieversdorf, Abt. 6154a1*) were used to test germination in the bioassays. The dishes were sealed with parafilm to reduce evaporation. The experiment was set up in a full-factorial design, each treatment was replicated five times. Dishes without toxins nor carbon supplements served as a control.
The bioassays were run parallel in an incubator at 20° C and 70 % RH with a day-night ratio of 17:7. To compensate for evaporation losses, distilled water was added when necessary. The total number of germinated seeds per dish was recorded until no further germination occurred. Seeds were scored as germinated if the radicle exceeded 1 mm in length. The experiment was stopped when no germination occurred for at least three days. Seed germination data were Logit-transformed and analyzed in ANOVA and Tukey's post-hoc test.

7.2.2 Active inner surfaces of different charcoals

C and H – content of the charcoal and the activated carbon was determined at the DSK-laboratory (*Deutsche Steinkohle AG, Ibbenbüren, Germany*). The samples were outgassed at 120° C under vacuum for 12 hrs to clean the surface areas. Inner surface area was determined using multiple-point BET-technology (Brunauer *et al.* 1938). In brief, the active surface of a powder is calculated from the N_2-isotherm, which is observed at the boiling point of liquid nitrogen. The volume is determined from the adsorption curves which corresponds to the quantity of nitrogen that is necessary to form a mono-molecular layer inside the powder. From this value the specific surface of the sample can be determined in $m^2\ g^{-1}$. In a second step, the t-method (Lippens and De Boer 1965) was used to determine pore structure. This method compares adsorption on the porous solid with that of a nonporous reference. It has been widely used to determine the micropore volume of porous solids.

7.3 Results

Seed germination started after three days. As there were no additional seeds germinating after day 21, the experiments were stopped after 25 days (Fig. 7.1). The control bioassay (no toxins and no carbon supplement) had a germination of 80 %. Neither the activated carbon nor the two types of charcoals had any effect on pine seed germination when no toxins were added ($F_{3,16} = 0.35$; $p = 0.79$).

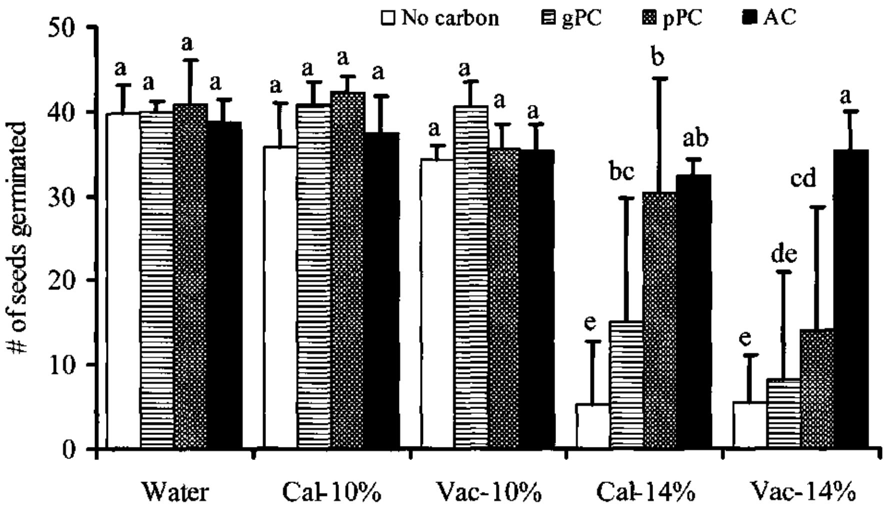


Figure 7.1. *Numbers of germinated pine seeds and significance levels for Logit-transformed seed counts (out of 50 seeds) assuming a binomial distribution. Values with same letter are not significant different in Tukey's post-hoc analysis at the 95 % level. gPC = granulated pine charcoal, pPC = powdered pine charcoal, AC = activated carbon. Analysis based on one-way ANOVA ($F_{19,80} = 12.64$, $p = 0.0001$).*

Low concentrations of aqueous extracts (10 %) from *C. vulgaris* and *V. myrtillus* did not have significant negative effects on germination. Treatments with activated carbon and the two charcoal types showed no significant differences to the control (Fig. 7.1). No significant difference was found between the three types of carbon supplements.

High concentration of the extracts (14 %) indicates the evidence of germination interfering effects of the two shrub species: Germination in aqueous extracts without carbon was reduced to 10 % of the non-toxic control. *C. vulgaris* and *V. myrtillus* had a similar magnitude in preventing germination (Fig. 7.1).

Activated carbon was able to significantly remove the germination inhibition (germination occurred to 90 % of the control in *C. vulgaris* and *V. myrtillus*; Fig. 7.1). Addition of the two pine charcoals increased the germination rate but to a lower extent than activated carbon. Added granulated pine charcoal allowed germination of 30 % of the 50 seeds in *Calluna* and 16 % in *Vaccinium*. Powdered pine carbon reduced toxic concentration so that 61 % for *Calluna* and 28 % for *Vaccinium* of the pine seeds germinated (Fig. 7.1). For *Vaccinium*, germination in granulated pine charcoal was not significant different from non-carbon, but for *Calluna* germination in granules of charcoal was significantly higher than without any carbon supplement.
The results of the post-hoc analysis allow a grouping of combinations of carbon supplements and toxin concentrations in three classes of similar germination rate: i) low concentrations of *C. vulgaris* and *V. myrtillus*, the control and all toxin types with activated carbon added (germination rate of 75 %), ii) high concentrations of *C. vulgaris* and *V. myrtillus* with natural charcoals added (germination rate of 35 %) and iii) high concentrations without carbon or granulated pine charcoal (germination rate of 10 %; Fig. 7.1).

The three carbon types had different levels of pureness of the material. While activated carbon as an industrial product consist mainly of carbon (94 %), the carbon content of the pine charcoals was only 82-84 %, mainly due to the presence of ash in the material.

The nitrogen adsorption / desorption isotherms used for the BET-Method revealed important differences between natural charcoal and activated carbon. For activated carbon, the adsorbed volume increased fast under increasing pressure, while the charcoal showed only a small uptake of nitrogen under higher pressure (Fig. 7.2), indicating that natural charcoal has no mesopores (pores with > 2nm diameter). The two natural charcoals had a very similar nitrogen uptake characteristic.

Calculated surface areas for the three carbon types showed different values: The BET-surface area for activated carbon was around 641 $m^2\ g^{-1}$, with a total pore volume of 0.546 $cm^3\ g^{-1}$. For the powdered and the granulated pine charcoal the values were identical, with 142 $m^2\ g^{-1}$ and 159 $m^2\ g^{-1}$ of surface area, respectively and a similar total pore volume of 0.1 $cm^3\ g^{-1}$ (Tab. 7.2).

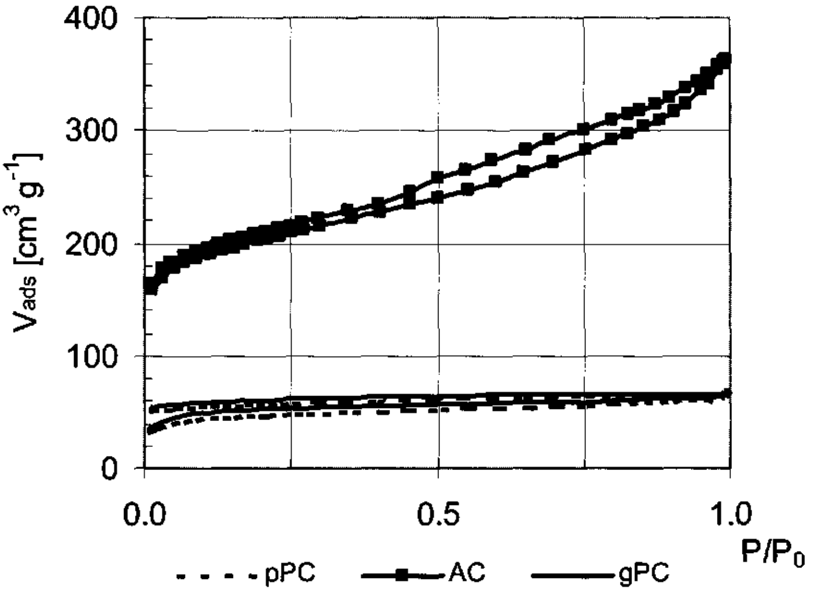


Figure 7.2. *Nitrogen adsorption and desorption isotherms of the powdered pine charcoal (pPC), granulated pine charcoal (gPC) and activated carbon (AC). The graphs show the volume of nitrogen uptake (V_{ads}) by the material under increasing and decreasing pressure (P/P_0).*

The nitrogen isotherms of activated carbon exhibited important adsorption for P/P0 above 0.5 (Fig. 7.2). From this feature, Cao *et al.* (2002) concluded the presence of mesopores. Although the existence of mesopores in activated carbon remains not fully proven by our analysis, the different shape of the adsorption isotherms hints at different pore size distributions for activated carbon and the two charcoals. This presumption is stressed by the results of a pore classification (t-method), where for natural charcoal 75-82 % of the total surface area is provided by micropores, but for activated carbon only 58 % (Tab. 7.2).

Table 7.2. Chemical properties of the powdered, granulated charcoal and the activated carbon. BET-Surface Area^{+} is split up into micropore and mesopore surface area.

	BET-Surface Area^{+}	Micropore Surface Area*		Mesopores Surface Area*		Pore Volume**
	[$m^2\ g^{-1}$]	[$m^2\ g^{-1}$]	% of total surf. area	[$m^2\ g^{-1}$]	% of total surf. area	[$cm^3\ g^{-1}$]
Granulated Pine Charcoal	159	131	82	28	18	0.101
Powdered Pine Charcoal	142	106	75	36	25	0.098
Activated Carbon	641	374	58	267	42	0.546

* determined with the t-method (Lippens and De Boer 1965)
** for pores with radius less than 717 Å at rel. pressure (P/P_o) of 98%.
\+ Brunauer, Emmet and Teller (1938)

7.4 Discussion

Activated carbon used in previous studies reduced phytotoxic impact by adsorbing toxins and from this, conclusions were drawn for the toxin reducing potential of natural charcoal, for example produced by forest fires. Our results also show significantly improved germination of pine seeds with activated carbon added. However, as types of carbon vary, their effectiveness in adsorbing toxins differed in our study. Added activated carbon has a much higher positive effect on seed germination than the natural charcoals. Total germination of pine seeds was significantly increased by natural charcoal when added to 14 % extracts of *C. vulgaris* and *V. myrtillus*, but charcoal could not completely eliminate germination reduction (Fig. 7.1).

Based on our results of the chemical analysis we suggest, that variation in germination rate is mostly due to differences in active surface area of the supplemented charcoal. The large difference in adsorption potential between activated carbon and natural charcoal becomes evident using the BET-method. Activated carbon has a five times higher active surface area and total pore volume and also a different pore structure. While natural charcoal mainly consists of micropores, our results show the existence of mesopores of more than 2 nm diameter within the activated carbon particles (Table 7.2). These results are reasonable, as the pores of the activated carbon used in this study had been widened by steam treatment. These mesopores are important for the adsorption of larger toxin-molecules and for a better accessibility of the inner parts of the particle. Size exclusion limits the adsorption of toxins of a given size and shape if pores are too small. In aqueous systems, size exclusion was observed when the pore

width is smaller than about 1.7 times the second largest dimension of the adsorbate (Kasaoka *et al.* 1989). As the actual sizes of the toxin-molecules produced by *Calluna* and *Vaccinium* are not known, it remains unclear, if in this case pore diameter might play a role in adsorption potential.
In our bioassays, aqueous extracts of *Vaccinium* leaves from Central European origin have negative impact on pine seed germination, but the magnitude of this effect strongly depends on extract concentration. There seems to be a strong line between the 10 % and 14 % concentrations which determines whether seeds germinate or not (Fig. 7.1). These results are in line with previous research, where no germination was observed in concentrations above 10 % (Jäderlund 2001).
The germination inhibiting potential of aqueous extracts from *Calluna* leaves has not been shown before and similar toxic effects as from leachates of *Vaccinium* were found in this study. Not only does *Calluna vulgaris* reduce germination of pine seeds in the same magnitude, it also seems to have a similar threshold of toxicity (between 10 % and 14 %) as *Vaccinium*. The results show, that *C. vulgaris* influences forest floor dynamics not only by strong nutrient competition, N-binding (Buchmann *et al.* 1996, Qualls *et al.* 1996) and reducing decomposition rates (Bending and Read 1996), but also can prevent the germination of pine seeds by toxic leachates in high concentrations.

To estimate the role charcoal plays in the complex interaction of tree regeneration and phytotoxic effects of ericaceous shrubs, it is essential to know more about the size classes and spatial distribution of the charcoal after a fire event. Although the crushing of the charcoal (either into powder or granulates) did not affect pore structure and total surface area (Tab. 7.2), our results show, that finer charcoal powder is more effective in absorbing toxins than bigger pieces (Fig. 7.1). While germination with added granulated pine charcoal was in general not significant different from extracts without carbon supplement, powdered pine charcoal increased germination rate significantly. These differences in toxin uptake might be explained by better accessibility of the finer powder and it's homogenous distribution in the Petri-dish due to smaller particle size (Tab. 7.1). From the significant adsorption potential of even charcoal of different size class, we conclude, that properties of different charges of natural charcoal varies widely and may depend on factors such as wood moisture before the charring process starts, fire intensity and other factors. A general comparison of homogenous activated carbon with natural charcoal is therefore invalid.
In summary, our results show that studies on toxin reduction with activated carbon (e.g. Zackrisson and Nilsson 1992, Zackrisson *et al.* 1996) can not directly be used to estimate adsorption properties of natural charcoal. The potential of charcoal will be overestimated.

CHAPTER 8

Germination and early growth on burned sites

Published as: HILLE M, DEN OUDEN J. 2004. IMPROVED RECRUITMENT AND EARLY GROWTH OF SCOTS PINE (*PINUS SYLVESTRIS* L.) SEEDLINGS AFTER FIRE AND SOIL SCARIFICATION. *EUROPEAN JOURNAL OF FOREST RESEARCH* 123: 213-218

Abstract

The success of seedling recruitment of Scots pine (*Pinus sylvestris* L.) is strongly depending on soil surface properties, such as humus depth and moisture content. In an undisturbed forest floor, seedlings are seldom able to establish due to high incidence of desiccation in the organic soil layer. Means that remove the organic soil layer are often necessary to improve access for radicles to the more stable moisture regime in the mineral soil.

In this study we investigated pine seedling establishment mechanical soil scarification, burning of litter (O_L) and burning of litter and humus ($O_L + O_{FH}$) in two mature pine stands in Germany. The field layer of the first stand was dominated by grasses (*Molinia caerulea* L. and *Deschampsia flexuosa* L.), whereas the field layer of the second stand was dominated by blueberry (*Vaccinium myrtillus* L.).

Pine seeds were placed in experimental plots, and seedling numbers and height were recorded in regular intervals. All treatments that removed organic soil yielded in higher seedling counts than on the undisturbed forest floor. The highest number of seedlings was found on scarified and severely burnt plots, whereas seedling counts were lower on lightly burnt plots. Seedlings were significant tallest on burnt plots.

This study shows that pine regeneration is stimulated by fire not only in boreal forests, but also under Central European conditions. Under the impression of higher fire frequency in the near future due to climatic changes, natural regeneration and succession on burnt sites should get more in the focus of forest management and research.

8 Improved recruitment and early growth of Scots pine (*Pinus sylvestris* L.) seedlings after fire and soil scarification

8.1 Introduction

Natural regeneration of Scots pine (*Pinus sylvestris* L.) was a strongly discussed topic in forest management in Central Europe during the 1930's (Dohrenbusch 1997). Due to its varying success on different sites and the strong dependency on favorable outside conditions (such as sufficient moisture in the upper mineral soil layer by rainfall at regular intervals, a not too dense field layer and high light levels on the forest floor; Mallik 2003, Dohrenbusch 1997, Olberg 1957, Wittich 1955), stand regeneration using natural seeding of Scots pine was not common practice during the 20th century. Instead, a plantation-style *clearcut-replanting* system prevailed in Central Europe until the 1980's. Recently, forest management places more emphasis on a *'close to nature'*-management, and now natural regeneration is the desired way to manage and regenerate pine stands, especially on poor sandy soils (BMF 2001). Silvicultural systems based on heavy shelterwoods and natural regeneration are replacing clearcutting followed by planting.

From an ecological view, the most critical stage in Scots pine recruitment is the germination and establishment of contact to a stable soil moisture regime since the young seedling is highly depending on a sufficient water supply. It is crucial for seedling survival that the radicle is able to penetrate the thick and slowly decomposing ectorganic soil layer (O_{FH}) to reach the mineral soil below (Den Ouden and Vogels 1997, Ahlgren 1974, Wittich 1955). The thicker the ectorganic soil layer, the longer it takes for the roots to reach the mineral soil and thus a more stable soil moisture regime. Once the seedlings have established their roots in the mineral soil, mortality rate drops sharply (Nilsson *et al.* 2002).

The moisture regime in the humus layer is highly variable and dries out quickly, so that the seedling is strongly depending on sufficient and frequent rainfall until it's root reaches the mineral soil (Oleskog and Sahlén 2000). Another factor that is obstructive for the recruitment of pine seedlings is the release of phytotoxins by dwarf-shrubs, which can inhibit germination of Scots pine (Hille and den Ouden 2005a, Zackrisson *et al.* 1997a, Nilsson 1994).

Disturbances, which remove the organic soil layer and field layer vegetation, are required to allow pine seedling establishment. In traditional forest management, mechanical means are used to reduce ectorganic soil layers and forest floor vegetation at least partly. Soil scarification with plows or molding cutters has prevailed as the dominating silvicultural tool to stimulate tree recruitment (Nilsson *et al.* 2002, Dohrenbusch 1997). After removal of the organic soil layer, germinating seeds have direct access to the mineral soil and competition with other vegetation is reduced (Beland *et al.* 2000, Karlsson and Örlander 2000).

The two common natural disturbances, which are able to remove the ectorganic soil layer in central Europe, are windthrow and fire. While windthrow only exposes mineral soil spot-wise (around uprooted stems; Kuuluvainen and Juntunen 1998), fire acts area-wise, but with spatial variation that depends on the local amount of organic material that is consumed (Schmidt *et al.* 2004). The effectiveness of fire to enhance pine regeneration has been observed in many pine-dominated ecosystems in the boreal and Mediterranean zone (Spanos *et al.* 2000, Agee 1998a, Engelmark *et al.* 1998, Kuusela 1990, Sykes and Horrill 1981, Viro 1969, Uggla 1959). Here, fire is an important natural agent to induce forest succession sequences and create floral and faunal biodiversity on a stand and landscape level (Lindbladh *et al.* 2003, Peltzer *et al.* 2000).
Fire plays a significant role in western and central Europe as well, especially in regions where pines grow on poor sandy soils (e.g. *Veluwe*, NL; *Lausitz*, D; *Lüneburger Heide*, D). Current scenarios of climate change predict high summer droughts and thus an even increased fire danger and higher fire frequency can be expected for the near future (Badeck *et al.* 2003). At this moment only anecdotal descriptions of the presence of recruited cohorts a few years after fire in pine stands exist (Klein 1964, Schmidt 1929, Conrad 1925) and fire has never been considered as a management tool to regenerate pine stands.
The objective of this study is to analyze and compare the germination and early growth of Scots pine on scarified, burnt and untreated forest floors. In this paper, seedling establishment and early growth are indicators for the suitability of different soil surface treatments for recruitment and natural regeneration of Scots pine. With this study we want to answer the question, whether a surface fire promotes the natural regeneration of Scots pine in Central Europe, and how the post-burn recruitment and early growth performs compared to scarified and untreated forest floor.

8.2 Methods

We started our experiment in two 75-year old pine stands in April 2002 (Tab. 8.1). Stand A, located in northwest Germany, is classified as "*Rubo-Avenello-Cultopinetum sylvestris*" (Hofmann 1997), dominated by grasses (*Molinia caerulea* L. and *Deschampsia flexuosa* L.) and black berry (*Rubus fruticosus* spc.) in the field layer. This stand is under oceanic influence with average precipitation of 690-750 mm yr^{-1} and a mean annual temperature of 8.5° C. In Stand B, located in northeast Germany, the forest floor was covered mainly by blueberry (*Vaccinium myrtillus* L.), wood sorrel (*Oxalis acetosella* L.) and Schreber's feathermoss (*Pleurozium schreberi* (Brid.) Mitt.) and is classified as "*Oxalio-Myrtillo-Cultopinetum sylvestris*". This stand is under less oceanic influence, expressed by lower average annual precipitation (540-600 mm yr^{-1}) and mean annual temperature of 8.0° C.

Table 8.1. *Stand and forest floor inventory data for the two studied Scots pine stands. The forest floor in stand A is dominated by grasses, and by blueberry in stand B. Both stands stock on sandy soils. Litter (O_L) and humus (O_{FH}) depth are averages of 12 measurements (± SD).*

Stand	Stand Age	Stem density ($N\ ha^{-1}$)	Basal Area ($m^2\ ha^{-1}$)	Height (m)	Yield class*	Litter depth [cm]	Humus depth [cm]
A	75	520	25	20.5	2.0	2.7 ± 0.8	6.4 ± 1.3
B	75	450	28	26	0.6	4.3 ± 1.0	4.7 ± 0.8

* based on the yield tables of Wiedemann (1949)

Both stands are pure pine-stands, even-aged and resulted from the traditional clearcut-replanting system. Stand inventories were conducted on five systematically placed circular plots (500 m² each) within each stand (Tab. 8.1). Litter and humus measurements were taken with a ruler at 12 locations on a 2x2 m grid in each stand.

Three seeding sites were established in each stand, and were located within canopy gaps of at least 7 m in diameter, with 10-20 m distance from each other. Shading understory trees within the gap, such as *Betula sp.* or *Frangula alnus* Mill. were removed. Each site covered app. 16 m² and contained four different forest floor treatments of 1 x 2 m. Treatment areas were adjacent to each other, with a 50 cm buffer between different treatments and had a rectangular shape.

Treatments were a) *no action* (control), b) *removal of the entire organic layer (scarification)*, c) *burning of the litter* (O_L; simulating a low fire severity) and d) *burning of litter and humus* (O_L, O_{F+H}; high fire severity). The two treatments that include burning of the forest floor were performed by burning the litter (c) and litter + humus layer (d) with a gas burner and leaving the ash on site.

Three sampling plots of 25 x 25 cm were established in each treatment with a distance to the border of the treated area of at least 30 cm. A mesh cage (13x13mm mesh) covered the plots to define seed location and to protect seeds and seedlings against predators. Finally, 50 Scots pine seeds (supplied by the federal seed agency of the state of Brandenburg, Germany; Origin: *Waldsieversdorf, Abt. 6154a1*) were sown into the mesh cage. Seed viability in an incubator was 85 %.

The number of seedlings and individual heights were recorded until October 2002 in two-week intervals. To better treat the dichotomous data, seedlings scores were Logit-transformed (Collins *et al.* 1992) before statistical analysis and differences between treatments were analyzed for the two stands, using a nested-ANOVA design with factor '*Treatment*' nested within factor '*Site*'. The nested-ANOVA design was used because we assume that resource availability differs within the three sites (see also Kuuluvainen *et al.* 1993), so that a combination of all levels of the factor '*Treatment*' with all levels of the factor '*Site*' is constrained. The ANOVA was followed by Tukey's post-hoc tests to determine difference between treatments.

8.3 Results

First seedlings were observed two weeks after seeding. Highest seedling counts were found after four weeks in stand A (grass), and after six weeks in stand B (blueberry). All seedling counts afterwards yielded in lower numbers (Fig. 8.1, 8.2), with a linear decrease of seedling numbers over the entire period. At the end of the first vegetation period, seedling counts were on average between four and eight per plot, except on control plots, were no seedlings were found. Early in the following growing season, almost all seedlings on all plots died, probably due to lack of water during the very dry spring of 2003. On average, one seedling was found alive on a scarified or burnt plot.

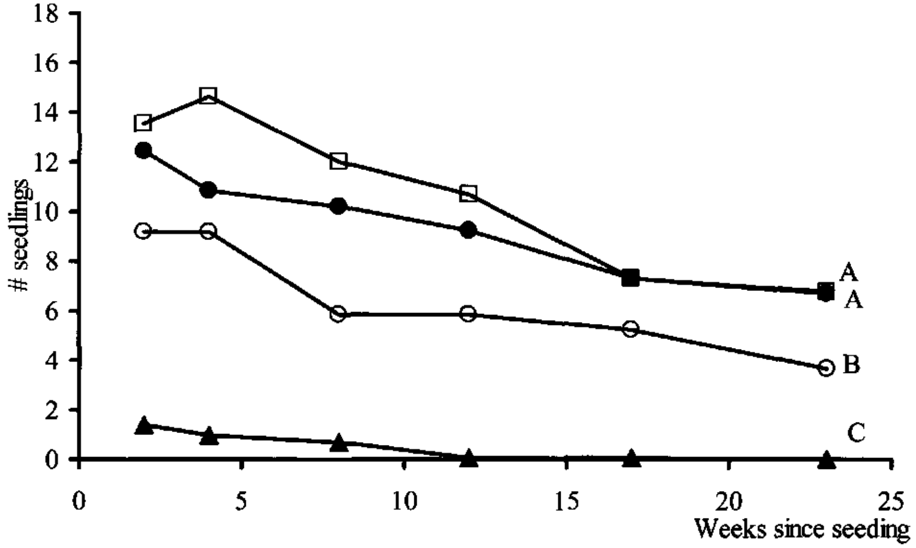


Figure 8.1. *Mean seedling counts for all sites in stand A (grass) after seeding on May 10th. Plot treatments are either scarification (□), burning of the litter (○), burning of litter + humus (●) or no soil disturbance (▲). Final seed scores with the same letter are not significantly different.*

In stand A, scarification (S) and burning of litter + humus (BLH) resulted in almost identical seedling numbers. Plots with burnt litter (BL) had significantly lower seedling numbers. On control plots, no seedlings were found at the end of the first growing season (Fig. 8.1). In stand B, treatment BLH resulted in lower seedling numbers than S, but this difference was not significant. On plots with burnt litter, seedling counts were significant lower than on scarified plots, but not significantly lower than BLH. Like in stand A, no seedlings were found on undisturbed control plots in stand B after 23 weeks (Fig. 8.2).

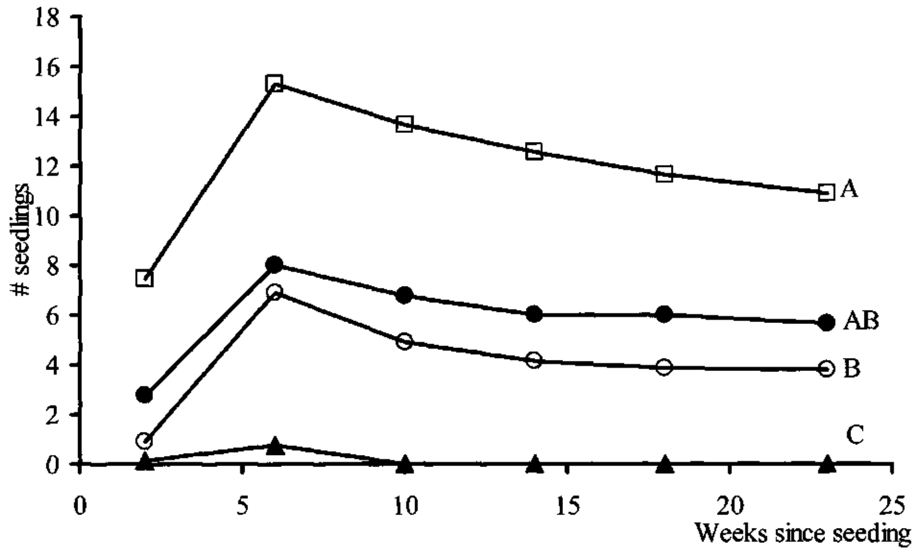


Figure 8.2. *Mean seedling counts for all sites in stand B (blueberry) after seeding on May 11th. Plot treatment are either scarification (□), burning of the litter (○), burning of litter + humus (●) or no soil disturbance (▲). Final seed scores with the same letter are not significant different.*

Weekly mortality rates, calculated by the formula of Hamilton and Edwards (1976), were highest for plots with removal of organic material between week 2 and 4 in stand A and between week 6 and 10 in stand B (from 2 to 5 % week^{-1} and from 0.8 to 3 % week^{-1}, respectively). After that, mortality rates were constant between 1 and 2 % week^{-1} for all treatments and both stands. The highest mortality rate of 20 % week^{-1} were found on control plots in stand A between week 8 and 12, and a 100 % mortality occurred on control plots in stand B between week 10 and 14.

Two-way nested ANOVA with factor *treatment* and *site(treatment)* showed differences in seedlings scores. In stand A, only factor '*treatment*' had a highly significant effect on seedling counts. In stand B, *treatment* and *site* were significant factors for seedling scores 23 weeks after seeding (Tab. 8.2).

Table 8.2. *Nested-ANOVA with factor 'Treatment' and 'Treatment' nested within 'Site' for the Logit-transformed seed scores 23 weeks after seeding.*

Stand	Source	Df	SS	F-Value	P
		Seedling scores			
A (Grass)	Treat	3	2.58	17.16	<0.0001
	Site (Treat)	8	0.08	0.20	0.9883
B (Blueberry)	Treat	3	3.32	40.38	<0.0001
	Site (Treat)	8	0.92	4.22	0.0029
		Height			
A (Grass)	Treat	2	10.56	3.50	0.002
	Site (Treat)	6	39.92	4.42	0.041
B (Blueberry)	Treat	2	10.62	5.38	0.009
	Site (Treat)	6	3.02	0.51	0.796

Average seedlings top height (the average height of the five tallest seedlings per plot) at the end of the growing season was significantly affected by *treatment* and *site* in stand A. In stand B, only *treatment* had a significant influence on height (Tab. 8.2).
Seedlings on plots with burnt litter and humus were taller than on all other treatments but differences were only significant in stand B. Seedlings on plots with burnt litter were also taller than on scarified plots, but the difference was not significant in both stands (Fig. 8.3). Seedling heights were not recorded from control plots, because no live seedlings were found.

8.4 Discussion

This study clearly shows that different soil surface treatments are responsible for variation in germination success and early growth of Scots pine in northern Germany. The need for a substantial disturbance of the forest floor to recruit Scots pine seedlings is shown explicitly in our study. The removal of the ectorganic soil layer and the potentially phytotoxic field layer (Jäderlund *et al.* 1996) by scarification and by fire were found to facilitate seedling establishment in both grass- and dwarf-shrub dominated field layers. The differences in seedling counts were found already two weeks after seeding and the relative differences of seed counts between treatments remained constant for the entire sampling period (Fig. 8.1, 8.2). On control plots, only few seedlings emerged, and after 10-12 weeks none of them were alive. The importance of a suitable germination environment becomes also evident by the fast reduction of seedling numbers from week 6 to 15. Nilsson *et al.* (2002) showed that first stages of seedling establishment (germination, radicle elongation, establishment of roots in the mineral soil) have highest mortality and are therefore crucial for successful pine regeneration. Thereafter, competition with overstory trees for above- and belowground resources could have been another factor that caused high mortality rates (Vickers and Palmer 2000).

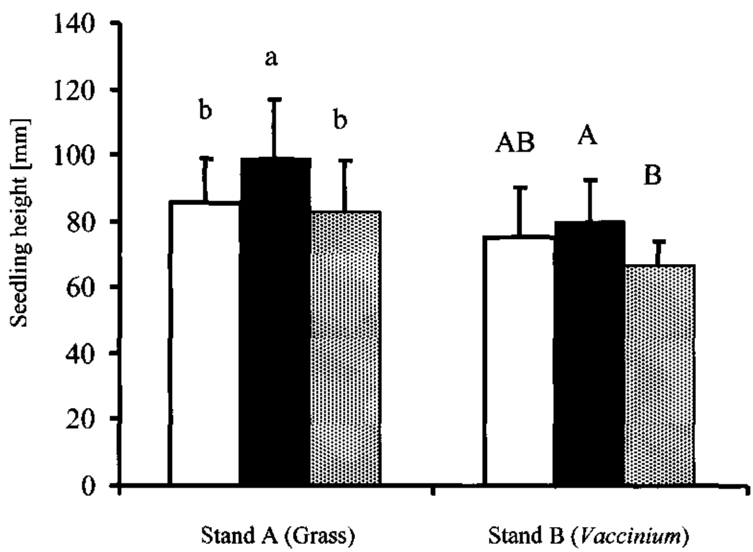


Figure 8.3. *Mean height (± SD) of the five tallest seedlings for all sites in stand A and B, 23 weeks after seeding. Plot treatments are either scarification (░), burning of the litter (□) and burning of litter + humus (■). No living seedlings were found in the controls. Bars with the same letter are not significantly different.*

On plots with a total incineration of the organic material (burnt litter and humus), seedling numbers were higher than on partly burnt plots (burnt litter) in both stands, significant different only in stand A, though. It is likely, that the remaining organic material is still too thick to penetrate for many seedlings and therefore causes higher mortality rates. Recruitment of Scots pine seems to be favored by forest fires of

medium intensity, which leave little organic material behind. Fires of low intensity are not able to remove enough organic material, fire of high intensity and a stand-replacing character would kill seed trees.
Site factors and ground conditions have a strong effect on regeneration success in Scots pine forests (Tengelmark 1998). Within-stand variation of site factors (such as humidity, soil moisture or temperature) can have an impact on recruitment success and seedling growth, too (the main reason why the nested ANOVA design was chosen, Tab. 8.1; Kuuluvainen *et al.* 1993). In our study, the effect of factor *site* within a stand was negligible for stand A, but not for stand B (Tab. 8.1), which might be due to the lower overall moisture level in stand B under a more continental influence than stand A.

Similar to seedling numbers, plot treatment had a significant influence on seedling height in both stands. Data are lacking for plots without disturbed forest floor because no seedlings were alive at the end of the first growing season. However, results from previous studies showed that average heights of Scots pine seedlings were clearly lowest on untreated sites and highest on plowed sites (Sugg 1990).
Mean top heights of seedlings were highest on plots with burnt litter and humus in both stands. One explanation for this can be found in the higher nutrient availability on burnt sites (Viro 1974). Surface fires in old pine stands consume a majority of dead branches and needles, the top litter and, depending on the fire severity, parts of the humus layer. This accelerates the nutrient cycle, as these nutrients which were locked in the humus layer are suddenly released and available for plants (Covington and Sackett 1986, Chandler *et al.* 1983, Beese and Divisch 1980, Braathe 1974). In contrast to mechanical means of site preparation that remove the entire organic material down to the mineral soil, nutrients are not removed from the plots and are made plant-available (Johansson 1994). Especially high amounts of nitrogen, required in initial seedling stages for ectomycorrhizal colonization (Rebane 2001), are made plant-available by fire.
In many pine-dominated forests in other parts of the world, e.g. in the boreal forest, dry mountainous pine forests, the Mediterranean zone or the mixed-conifer zone in the Western US, fire is the main agent which determines regeneration dynamics (Agee 1998a). Key fire effects that promote pine recruitment are the opening of serotinous cones, preparation of a suitable seedbed for pine seeds and inducing mortality in the overstory, thereby increasing resource availability for seedlings. The positive response of Scots pine to burnt forest soils is typical for a species that faces a moderate-severity fire regime in its' natural range (Agee 1998a). Surface fires that are not stand-replacing occur most often in stands of *Pinus sylvestris* and cause overstory mortality only in areas with high fire intensity. In these patches, natural pine regeneration creates the post-disturbance forest (Engelmark *et al.* 1998, Engelmark 1993). Although Central European pine stands, such as the ones used for this study, are not the natural forest vegetation, the stimulating effect of a low and moderate intensity surface fire by preparing good seedbed conditions can be found here.

8.5 Conclusions

Soil disturbance is necessary to stimulate seedling recruitment of Scots pine. As shown in this study, scarification was successful in stimulating natural regeneration of Scots pine, but, as shown in previous studies, also Norway spruce (*Picea abies* L. Karst.), birch (*Betula pubescens* Ehrh.) and beech (*Fagus sylvatica* L.) in southern Sweden (Nilsson *et al.* 2002) and Germany (Dohrenbusch 1997, Huss and Burschel 1972). Fire as the natural disturbing agent, which is able to remove organic soil layers area-wise, has a similar positive effect for seedling establishment of Scots pine in Central Europe, which has not been shown before. In distinction with scarification, fire causes a significantly faster height growth of seedlings the first year after fire, probably due to the higher availability of nutrients in the upper soil layer (Skre *et al.* 1998).

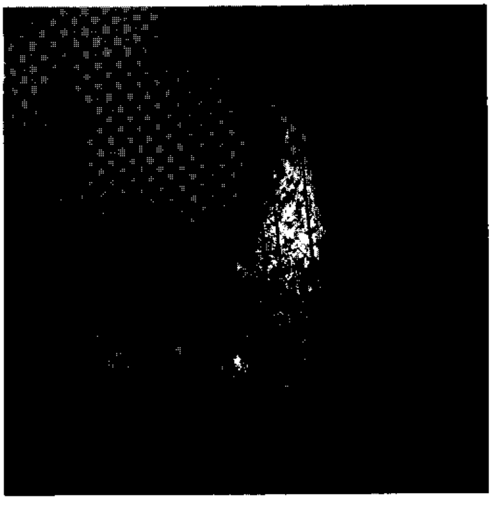
The almost complete mortality of seedlings at the start of the second vegetation period shows that even with intensive soil disturbance recruitment remains dependent on favorable weather conditions.

As shown in this study, fire facilitates seedling recruitment of Scots pine and seedling performance is better than after common forest floor treatments. Forest fire, which is an essential factor for forest dynamics in *Pinus sylvestris*' natural range, has a similar stimulating effect on recruitment in the plantation-style stands in Central Europe. The results of this study strongly encourage the use of fire for natural regeneration of pine stands (given that at least a few seed trees survived the fire), either by leaving burnt areas untouched or even by setting prescribed fires. However, further research is needed to document forest dynamics over longer periods after fire.

CHAPTER 9

Succession after small-scale, high severity fires

Prepared as: HILLE M, DEN OUDEN J, MOHREN F. FOREST SUCCESSION AFTER HIGH SEVERITY, SMALL-SCALE FIRES IN SCOTS PINE (*PINUS SYLVESTRIS* L.) STANDS. TO BE SUMITTED



Abstract

Early forest succession up to 12 yrs after four stand-replacing, small-size fires (< 1ha) within even-aged, single species Scots pine stands in Eastern Germany is described in this study. Forest inventory plots were located within the four burned patches, and seedling species, height and age was sampled.

The post-fire tree cohort after the fire mainly consists of Scots Pine, but also birch and aspen seedlings. Seedling numbers of Scots pine decreased linearly from on average 250,000 ha^{-1} in year 4 to 35,000 ha^{-1} in year 12. For birch and aspen, seedling numbers peaked 6 to 8 yrs after the fires (5,500 ha^{-1} and 3,600 ha^{-1}, respectively), and 12 years after the fires, seedling numbers were 5,100 ha^{-1} for birch and 1,400 ha^{-1} for aspen. Pines recruited only in the first and second post-fire year, aspen and birch established during a period of seven years.

Despite the higher numbers of pine and the slower colonization of the burned sites, the two broadleaved species became pre-dominant in the seedling cohort and overgrew the pine seedlings in height.

Compared to the unburned part of the stand, the organic soil layer was significantly reduced by the fires, and higher moss depths were observed on the burned sites. Compared to succession after other disturbance types in Scots pine stands, such as windthrow or scarification, seedling numbers are higher after small-scale fires by a magnitude of ten.

Controlled forest fires of this type could be used as an additional silvicultural technique to regenerate and transform single-species pine stands into mixed and more natural forests.

9 Forest succession after high-severity, small-scale fires in Scots pine (*Pinus sylvestris* L.) stands

9.1 Introduction

Regions in North-West Europe, where Scots pine (*Pinus sylvestris* L.) grows on poor sandy soils (e.g. the *Veluwe* region, The Netherlands; *Kempen* region, Belgium; *Lausitz* region; *Lüneburger Heide*, Germany) experience forest fires of varying size and intensity each year (Schelhaas *et al.* 2003), and fire is an essential part in natural pine forest dynamics (Niklasson & Drakenberg 2001, Engelmark *et al.* 1998, Engelmark 1993). Current scenarios of climate change predict more summer droughts and thus an even increased fire danger. For the near future, higher fire frequency can be expected and the influence of fire on forest management will rise significantly (Badeck *et al.* 2004). However, at present, not much is known about the ecological consequences of forest fires and how burned sites develop over time under North-Western European conditions. Post-fire succession could almost never be observed, since almost all burned sites were replanted immediately.

The only reports on succession after forest fire in North-Western European pine stands deal with large-scale stand-replacing fire events (> 100 ha burned) with very severe impacts on the ecosystem (Jahn 1980, Meijer zu Schlochtern & Koop 2000). Here, succession of burned sites took long periods and went through many different stages before tree regeneration was observed (fungi, mosses, grasses, pioneer tree species; Jahn 1980). For small burned areas, no data on post-fire development is available, mainly due to the fact that almost no burnt areas were set aside for natural succession, but were replanted immediately after the fire. However, some anecdotal descriptions of ample tree regeneration after small scale surface fires in Central European pine stands can be found (Klein 1964, Recke 1928). More detailed knowledge on post-fire succession dynamics of pine stands is available for the boreal and hemiboreal forest zone, the natural range of *Pinus sylvestris* (Viro 1974, Engelmark 1993, Engelmark *et al.* 1998). Here, germination and early growth of *P. sylvestris* occurs mainly within the first five years after the fire. Afterwards, a steady, but less intense recruitment is observed (Engelmark 1993, Engelmark *et al.* 1998).

A key role for successional pathways is played by the intensity of the fire, which determines the consumption of organic soil organic matter (Mallik 2003, Thomas & Wein 1985), the survival of the seeds bank and propagules (Granström & Schimmel 1993, Flinn & Wein 1977) and mortality in the overstory (Schmidt *et al.* 2004, Stephens & Finney 2002, Linder *et al.* 1998). In contrast to windthrow, fire may, depending on its intensity, not only remove the overstory, but also strongly reduce the organic soil layer and can be one of the most severe disturbances which significantly alters forests and determines succession (Runkle 1985).

Pinus sylvestris is a pioneer species that depends on mineral soil with little or no organic soil cover for successful recruitment (Mallik 2003). It is crucial for seedling survival that the radicle is able to penetrate the thick and slowly decomposing ectorganic soil layer (O_{FH}) to reach the mineral soil below (Hille & Den Ouden 2004, Den Ouden & Vogels 1997, Ahlgren 1974, Wittich 1955). The ectorganic soil layer has a highly variable moisture regime and dries out quickly, so that the seedling is strongly depending on sufficient and frequent rainfall until it's root reaches the mineral soil with a more stable moisture regime (Oleskog & Sahlén 2000). The thicker the ectorganic soil layer, the longer it takes for the roots to reach the mineral soil and thus a more stable soil moisture regime. Once the seedlings have established their roots in the mineral soil, mortality rate drops sharply (Nilsson *et al.* 2002). Therefore, soil scarification to remove the ectorganic soil layer in combination with heavy thinning and a few seed trees on the site has proven to stimulate ample regeneration on poor and mesic sites (Beland *et al.* 2000, Oleskog & Sahlén 2000) and is a common way to regenerate pine forest in Central and Northern Europe (Karlsson & Örlander 2000, Dohrenbusch 1997).
Another factor that is obstructive for the recruitment of pine seedlings is the release of phytotoxins by dwarf-shrubs in the field layer, which can inhibit germination of Scots pine (Hille & Den Ouden 2005a, Zackrisson *et al.* 1997, Nilsson 1994).
Most conifers and broadleaved pioneer species rely on conditions such as exposed mineral soil, high light and moisture levels on the soil surface, for germination, which are created by fire (Beese & Divisch 1980, Viro 1974), and by high soil temperature for biomass allocation (Domisch *et al.* 2001). Many silvicultural regeneration methods create such conditions by a combination of harvesting overstory trees and soil scarification.
Since forest management is adapting new management schemes, which take natural disturbances into account (Wohlgemuth *et al.* 2002, Beck 2000), there is a need to increase our knowledge about the ecological consequences of forest fire, both wildfires and prescribed fires. In modern forest management approaches, natural regeneration is the favored way to regenerate the forest, and is replacing the clear-cut-replacing system. Forest dynamics in the close-to-nature management are aimed to mimic natural forest dynamics, including disturbance regimes and post-disturbance pathways (Bengtsson *et al.* 2000). However, in recently developed successional pathways that depend on site, silvicultural treatment and disturbances (Kint 2003), pathways after fire disturbance are not defined yet, and therefore not applicable in the decision process of forest managers. Due to the frequent occurrence of forest fires in Central European pine stands (Schelhaas *et al.* 2003), there is a need to provide more information about post-fire succession pathways.

The aim of this study is to show the re-colonization after small-scale stand-replacing fire events in pine stands on poor sandy soils in North-West Europe. Based on forest floor measurements and seedling inventories from four set-aside burned areas, we show

how the fires affected the humus layer and post-fire forest floor composition, and show tree re-colonization dynamics up to 12 yrs after the fires.
We hypothesize that after small-scale stand-replacing fires in North-Western European pine plantations on sites with a complete humus consumption, a removed overstory and sufficient seed input from outside the burned area, conditions promote the recruitment of a dense Scots pine seedling cohort. We also expect that the observed recruitment patterns are similar to observations in *Pinus sylvestris*' natural range.

9.2 Methods

9.2.1 Study sites

Seedlings (used as the term for all recruited trees after the fire) on four burned sites in the forest district *Hammer*, app. 80 km south of Berlin, Germany were inventoried (Fig. 9.1). The region is dominated by even-aged Scots pine monocultures stocking on poor sandy soils with continental influence (8.0° C average temperature, 500-600 mm precipitation per yr). The pine stands are classified as *Festuco-Cultopinetum sylvestris* (Hofmann 1997) with few grasses (*Agrostis capillaris* L., *Deschampsia flexuosa* L., *Festuca ovina* L.) and mosses (*Pleurozium schreberi* (Brid.) Mitt. and *Dicranum scoparium* Hedw.) in the field layer. The potential natural vegetation in the region would be a mixed stand of *Quercus petraea* Liebl., *Quercus robur* L., *Betula pubescens* Ehrh. and *Pinus sylvestris*.

The cause of the fires remained unknown, and due to fast response of suppression forces, burned sites were rather small (0.09 to 0.6 ha; Tab. 9.1). The firelines, created during suppression activity, marked the edges of the burns. All burned sites are completely surrounded by mature pine stands, are located within 1,000 m of each other and the fires occurred all within 4 yrs. On all four burned sites, all overstory pines were killed by the fires and merchantable timber was removed the week after the fires. We do not expect that timber removal and skidding had a significant influence on recruitment, since the ectorganic soil was completely combusted. In the same year as the fire, the stands, in which the fires occurred were fenced completely to prevent herbivore influence on succession. Table 9.1 gives an overview of burned sites used in this study.

Figure 9.1. The seedling cohort on burned site 1 in fall 2004, with the burned site surrounded by the even-aged pine forest.

9.2.2 Sampling procedures

Seedling inventories were conducted in 1996, 2000 and 2003 on the burned sites and in 2003 on the reference plots. Maximum length and width of each burned area was measured and sampling points were laid in a systematic 3x3 grid over the site, with a grid cell of 1/4 the length of the burned area. As a reference to the burned sites, we extended the sampling grid into the unburned pine stands and nine additional sampling plots were located around the burned area.

At each sampling point, all recruited seedlings within a circular plot of 2 m radius were recorded. Parameters included species, height and age of the tree. Tree age was estimated by counting annual branch whorls for pine. For aspen and birch, a relationship between height and age was developed in a pre-study (Fig. 9.2).

In that study, 22 birch and aspen were cut on the sites and year rings at the base were counted. With an adj. R^2 of 0.65 we were able to estimate the age of these species with non-destructive sampling methods.

Table 9.1. *Characteristics of the sampled pine stands, which burnt partly in a stand-replacing surface fire.*

Site-Abbrev.	Compartment	Age	Pre-fire basal area	Date of fire	Burn size
		yrs	m^2/ha		ha
1	2219a3	104	26.5	21 Apr 1996	0.6
2	2223a1	72	25	26 Apr 1993	0.09
3	2230a1	99	27	24 May 1992	0.32
4	2230a1	99	27	22 May 1992	0.35

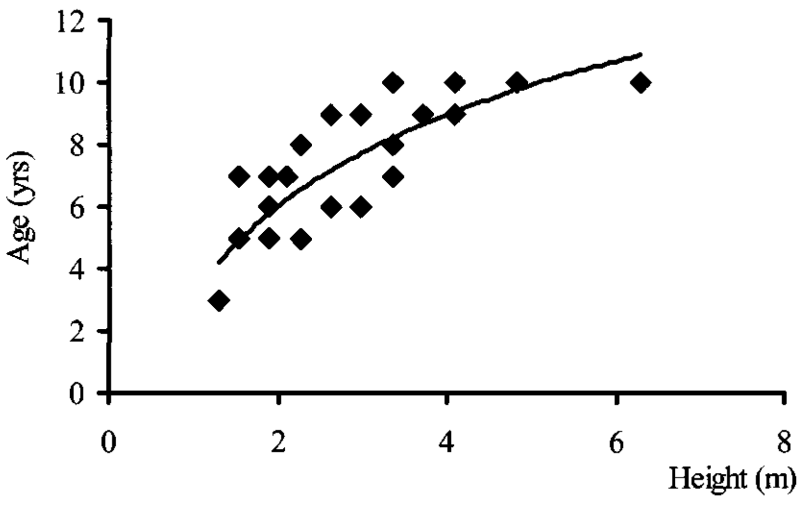


Figure 9.2. *Relationship between height and age of* Betula pubescens *and* Populus tremula, *determined by stem analysis of 22 trees. The regression was used to reconstruct the age for all other saplings of these species on the burned sites. Regression function: Age = 4.25 ln(height) + 3.07 ; R^2=0.65.*

At a distance of 1 m from each sampling point, four measurements of moss (including the fine litter layer, O_L) and humus layer ($O_{F,H}$) depth in North, East, South and West direction were taken.

9.3 Results

On all four sites humus and moss depth was strongly reduced by the fire. Even eight to ten years after the fire event, almost no humus was present and humus depth was significantly different from the surrounding unburned stand, where humus depth was roughly 5 cm (Tab. 9.2). In contrast, a thicker moss layer (mainly *Pleurozium schreberi*, *Polytrichum juniperinum* Hedw. and *Funaria hygrometrica* Hedw.) was found on the burned sites. This difference was highly significant on three of the four sites (Tab. 9.2).

Table 9.2. *Average (n=28 per site) moss and humus depth on the burned area and the surrounding plots in the unburned stand. Means are compared with two-tailed t-tests, means with different letters are significant different (p<0.01).*

Site	Moss depth [cm]		Humus depth [cm]	
	Burned	Reference	Burned	Reference
1	2.98 [a]	1.78 [b]	0.97 [a]	4.98 [b]
2	2.78 [a]	1.06 [b]	0.24 [a]	4.43 [b]
3	3.63 [a]	3.39 [a]	0.04 [a]	5.10 [b]
4	5.26 [a]	3.28 [b]	0.10 [a]	4.69 [b]

Recruitment of *Pinus sylvestris* was intense on all four sites, with variation between 62,000 ha^{-1} on site 1 and 368,000 ha^{-1} on site 4 in year 4. Since the first seedling inventories were conducted 4 yrs after the fire on site 2, 3 and 4, no data are available for early years. However, between year 4 and 12 after the fire on these sites, mortality reduced sapling numbers to 35,000 ha^{-1} on average with a linear rate. For site 1, seedling numbers were lower during the entire observation period (from 62,000 ha^{-1} in year 4 to 13,400 ha^{-1} in year 7; Fig. 9.3). On the reference plots, which were sampled in 2003, 17,500 pine seedlings were found per hectare.

Birch and aspen seedlings were found on all burned sites, despite the fact, that the nearest seed trees of this species were more than 2 km away. In contrast to *P. sylvsetris*, average seedling numbers for the four sites were low the first 3 years after the fire (2,000 ha^{-1}; 3,100 ha^{-1}), but peaked after 4 to 8 yrs (14,200 ha^{-1}; 7,300 ha^{-1} on average) for birch and aspen, respectively. In the following years, average seedling numbers of birch and aspen on all sites were reduced in year 12 to 5,100 and 1,400 ha^{-1}, respectively (Fig. 9.3).
The share of the three species in the seedling cohort was very similar over all four sites. In year 10 after the fire as a reference, pine, birch and aspen represented 83 %, 11 % and 3 % of the total seedling number, respectively.

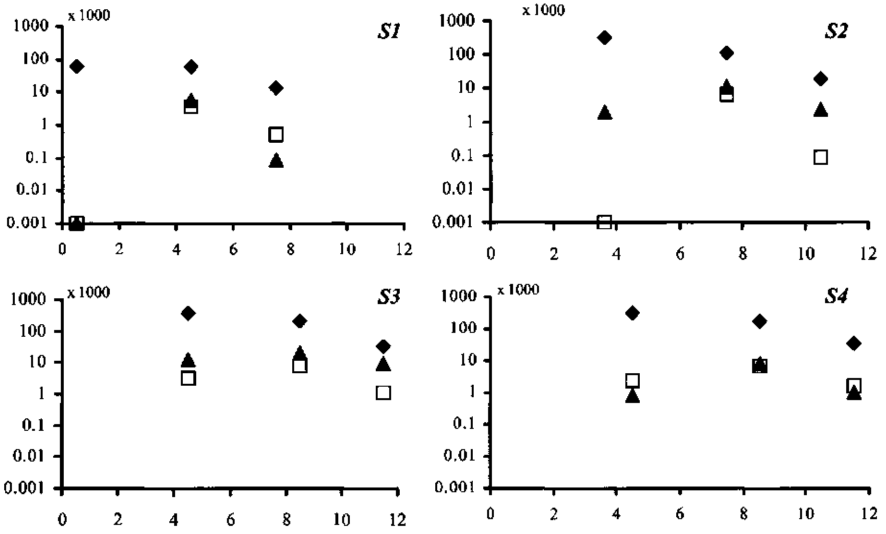


***Figure 9.3.** Mean counts (N ha[-1] on log-transformed y-axis) for seedlings of Scots pine (filled diamonds), birch (filled triangles) and aspen (empty squares) on the burnt areas over time since fire occurrence (years on x-axis) for all four sites (S1 to S4).*

With the estimated age of birch and aspen seedlings (Fig. 9.2) and the counted number of branch whorls for pines, we were able to reconstruct the year of establishment for each seedling. Almost all pine seedlings (97 %) established the first year after the fire on all four sites, only 3 % in the second year. After the second year, no additional pine seedlings established (Fig. 9.4). For aspen and birch, the main recruitment took places with a steady rate within the first 4 yrs after the fire and was less in year 5 to 7. Especially aspen seedlings established mainly in year 3 and 4 after the fire, while birch seedlings were recruited predominantly in the first four years after fire (Fig. 9.4).

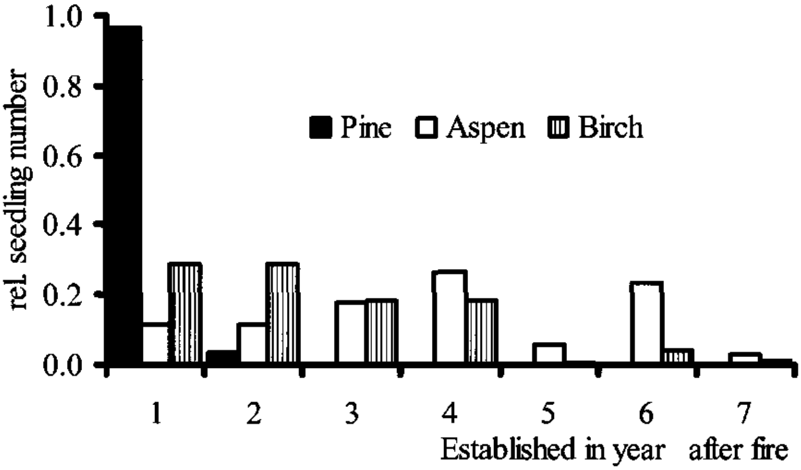


Figure 9.4. *Frequency distribution of age classes of pine, birch and aspen seedlings on the four burned sites.*

Average top height of pine seedlings (calculated as the average height of the five tallest pines per plot) was around 3 m, even for site 4, where the fire burned 3 to 4 yrs later as on the other sites. However, pine seedlings were out-competed in height growth by birch and aspen seedlings, which reached average top heights between 7.6 m on site 1 and 12 m on site 2 (Fig. 9.5).

The height frequency distribution for all seedlings of the three species stresses the height dominance of birch and aspen. Scots pine, which dominates the seedling cohort by number, is mainly present in lower height classes less than 4 m. Both broadleaved species are dominant in the seedling cohort with the majority of seedlings above 4 m.

9.4 Discussion

Fire facilitated light-demanding species and increased alpha-diversity (Wohlgemuth *et al.* 2002, Denslow 1985) on all four burned sites in this study, where fire promoted the recruitment of pine, birch and aspen in small gaps within a single-species dominated forested area. Based on previous results, we assume, that this is attributed to good seedbed conditions after fire for pioneer species by removal of the thick organic soil layer (Tab. 9.2) and phytotoxic compounds (Gallet & Pellissier 1997, Hille & den Ouden 2005a), the high availability of nutrients in the upper mineral soil (Viro 1974), increased soil temperatures (Domisch *et al.* 2001) and moisture conditions (Oleskog & Sahlen 2000) and the removal of the overstory by fire-induced heat damage on the cambium and roots (Rego & Rigolot 1990). Due to the small size of the areas burned, seed dispersal of Scots pine was sufficient to reach every location within the burned site (Dohrenbusch 1997).

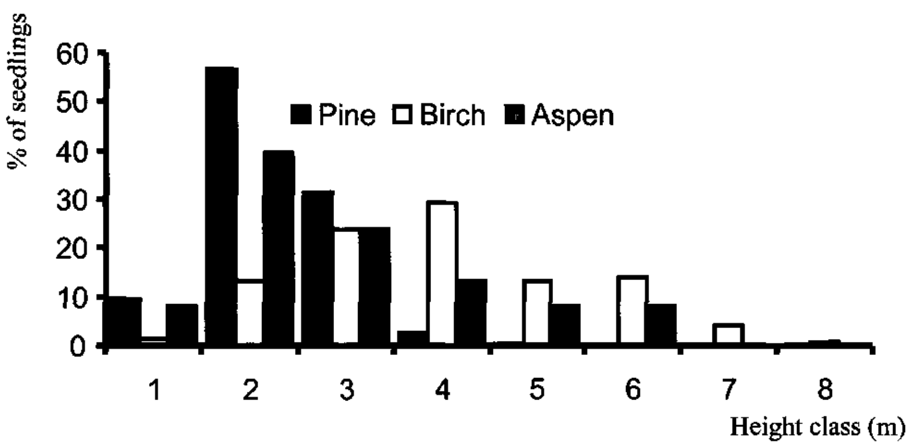


Figure 9.5. *Height frequency distribution of pine (filled bars), birch (empty bars) and aspen (striped bars) seedlings of all seedlings on the four burned sites.*

Establishment of seedlings of many species of woody plants is often limited to a brief period following disturbance (1-5 yrs), particularly when reestablishment of biomass is rapid (Oliver 1981, Canham & Marks 1985). As shown in previous studies (Granström 1987), seeds of *Pinus sylvestris* show no innate dormancy and germinate immediately in the soil. Therefore, this species has no viable seeds stored in the humus layer for more than one year. In our study, the very short recruitment window for pine on the burned sites is striking. Almost all seedlings emerged the first year after the fire. We do not believe that limited seed sources after the second post-fire year caused this pattern. For *Pinus sylvestris*, seed numbers are generally high enough each year for the establishment of a dense seedling cohort. Six-year consecutive seed counts from traps in closed pine forest showed large variation between different stands and years, but at least 22 viable seeds were recorded per m^2 each year, with maxima above 400 seeds per m^2 each year (Dohrenbusch 1997).

Although not shown in this study, we suggest that the thick moss cover plays a major role in determining the length of the regeneration window. The thick layer of

Pleurozium schreberi, *Polytrichum juniperinum* and *Funaria hygrometrica*, which was also observed on large burned areas (Jahn 1980), could prevent germination of pine, but not of birch and aspen after two post-fire growing seasons.

All four study sites were smaller than 0.6 ha and were obviously completely covered with pine seeds from the surrounding seed trees. With increasing distance from the stand edge, seed numbers of *Pinus sylvestris* decrease exponentially, at distances above 120 m only non-viable seeds were found (Dohrenbusch 1997). For birch and aspen, however, seed dispersal is less restricted to the vicinity of seed trees (Leder 1992). Pine seedling numbers were high with up to 368,000 ha^{-1} in year 4 after the fire.
Compared to other disturbance types, the recruitment window after the fires of 1-2 years in this study is very short. For natural regeneration of pine after thinning a 7-yr recruitment window was observed (Dong *et al.* 2003). After windthrow, an even wider age-class range of recruited pine seedlings (2-13 years) was found (Dohrenbusch 1997). Compared to results of studies on succession after fire in boreal pine stands, the recruitment window for *Pinus sylvestris* is slightly longer in Scandinavia (3-5 years; Engelmark *et al.* 1998).

In this study, succession was similar on all four sites, with a linear decrease of numbers of pine seedlings, and a peak of aspen and birch seedlings 4 to 8 yrs after the fire. With 35,000 ha^{-1}, seedling density 12 yrs after the fire is much higher than after windthrow of similar gap size on sites in North-West Germany (15,000 ha^{-1} on average; Dohrenbusch 1997). We believe that the area-wise removal of the litter and humus layer by fire stimulates tree recruitment more than the spot-wise presence of mineral soil after windthrow which leaves most parts of the organic soil layer intact. Despite the higher numbers of pine and the slower colonization of the burned sites, the two broadleaved species overgrew the pine seedlings quickly and became pre-dominant in the seedling cohort. This was also observed after scarification on clear-cut sites (Karlsson *et al.* 2002).

Compared to large-scale fires in North-West Europe, the re-colonization of the burned, small-scale sites in this study was faster and higher seedling densities were found. In a study by Jahn (1980) and Wichmann (1996), which documents succession on a 2.5 ha fenced area within a 8,000 ha burn in the Lüneburger Heide, Germany, is was shown, that post-fire succession goes through several stages (fungi, mosses, grasses) over several years, before any seedlings were found. Two decades after the fire, mainly willow (*Salix* sp.) , aspen and birch were found (49%, 33% and 18% of total seedling number, respectively), and due to the limited range of pine seed dispersal, no pines colonized the site (Tab. 9.3). This comparison shows, that for larger burned areas the spatial distribution of tree species will be determined by their seed dispersal mechanisms and range (Greene & Johnson 1989), and under these conditions, colonization and succession rates are slow, depend on much more in- and external factors and are therefore harder to predict (Turner *et al.* 1998).

After scarification, as a human-induced stimulus for recruitment in Scots pine stands, site re-colonization was fast and ample seedlings were found (7,000 to 80,000 ha^{-1}; Tab. 9.3). *Pinus sylvestris* was dominating the seedling cohort in most cases, except for one study, where scarification took place during a good seed year of birch, resulting in birch dominance (Karlsson *et al.* 2002).

After windthrow, with only spot-wise exposure of the mineral soil, seedling establishment was rather slow, and seedlings established over longer periods up to ten years. Also, spatial seedling distribution was patchy and strongly clustered in disturbed mircohabitats , particularly uprooted pits and mounds (Ulanova 2000, Kuuluvainen & Juntunen 1998). But also areas without any seedling on the wind-thrown site were found. The higher the light levels on the forest floor, the more homogenous were seedling densities and age class distribution (Dohrenbusch 1997). Total seedling numbers of pine were between 2,000 and 20,000 ha^{-1}, with birch and spruce (*Picea abies* L.) present in low numbers in the seedling cohort (Tab. 9.3). Pine seedling numbers were reported to be strongly correlated to site factors, forest floor vegetation type and humus depth (Dohrenbusch 1997).

Without any disturbance (therefore no humus removal, mineral soil exposure and the overstory left impact), only few recruited pine seedling were found. Again, total seedling numbers depended on site factors, but were less than 1,000 ha^{-1} in all other studies summarized in Table 9.3. Only in the reference plots in this study, higher numbers (17 000 ha^{-1}) were found.

Recently, forest management places more emphasis on the usage of successional pathways, including disturbances, as models for silvicultural systems (Wohlgemuth *et al.* 2001, Bengtsson *et al.* 2000). The results of this study show a succession pathway in pine stands after small-scale fire events. The main factors of this pathway are the stand-replacing fire intensity, which removes overstory coverage, the removal of the humus layer, and seed availability on the entire disturbed site. The outcome of this pathway is in conjunction with desired stand structures of a multi-species, multi-storey forest.

Table 9.3. *Succession after different stand-replacing disturbance types in even-aged Scots pine monocultures in Central Europe. Seedling data was collected between four and ten years after a disturbance event, as indicated in column 'Years since disturbance'. In case of 'No disturbance' (overstory not affected, no mineral soil exposure), no specific starting point of recruitment was defined. Dominating species in bold.*

Succession after	Yrs since disturbance	Species in regenerated cohort	Pine seedlings N ha^{-1} (x 1 000)	Reference	Successional rate
Small-scale fire	4	***Pinus sylvestris*** *Betula pubescens* *Populus tremulus*	62-368 14 7.3	this study	Fast, all pines recruited within two yrs after disturbance
Large-scale fire	4	*Salix caprea* *Populus tremula* *Betula pendula*	1.5 1.2 0.7	Jahn 1980	Slow, limited by seed availability and extreme micro-climate
Windthrow	10	***Pinus sylvestris*** *Betula pubescens*	8.5-20 0.7-27	Dohrenbusch 1997	Slow, due to humus layer and dense vegetation, high age range of more than 10 yrs for pine
	4	***Pinus sylvestris*** *Betula pubescens*	2-4	Elgersma 1980 Fanta 1982	
	4	***Pinus sylvestris*** *Betula pendula*	2.7 1.2	Kuuluvainen and Juntunen 1998	
Scarification	4	***Pinus sylvestris*** *Betula pubescens* *Picea abies*	12-80 0.7 0.6	Dohrenbusch 1997	Fast, all pines recruited within 2 yrs
		Betula pubescens *Pinus sylvestris*	19 7.6	Karlsson *et al.* 2002	
		Pinus sylvestris	14	Dong *et al.* 2003	
No disturbance		***Pinus sylvestris*** *Betula pubescens*	0-2.6 0.6	Dohrenbusch 1997	Slow, depending on humus depth, forest floor vegetation and soil type
		Pinus sylvestris *Betula pubescens* *Quercus petraea* *Fagus sylvatica*	<1	Al and Montizaan 1987; Fanta 1982	
		Pinus sylvestris	17	this study	

9.5 Conclusions

The small-scale, high severity fires described in this study induced regeneration of Scots pine, birch and aspen, and increased species diversity within a single-species, even-aged pine plantation. The fires stimulated the dense regeneration of pine, birch and aspen and post-fire succession had a similar rate as in the natural range of *Pinus sylvestris* in the boreal forest (Agee 1998a, Engelmark 1993). The different species show variation in their ability to re-colonize the burned sites in this study, only within the first two years pine was able to establish. In contrast, high severity fires on large scale resulted in a dominance of birch and aspen on the burned site.

From our results and findings of others, we conclude that the combination of small-scale overstory and humus removal, and seed availability for a quick re-colonization favor the recruitment on Scots pine. The small-scale fires, described in this study, prepared seedbed conditions for the recruitment of Scots pine, indicated by seedling numbers which were much higher than after scarification, windthrow or large scale fires.

The succession pathway after fire on sites less than one hectare results in forest structures, which are desired in a modern forest management, and natural succession on burned sites or even the use of prescribed fires for natural regeneration in pine stands should be considered more thoroughly.

CHAPTER 10

Synthesis

10 Synthesis

Despite the high importance of Scots pine (*Pinus sylvestris* L.) in North-West European forestry, almost nothing is known about the ecological impact of fire in these forest types. Fire is the main disturbance agent, which determines forest dynamics in the natural range of *P. sylvestris* (Spies and Franklin 1989, Agee 1998a, b), but has been neglected as a factor in the management of this species in North-West Europe. For natural *P. sylvestris* forests, a fire regime of moderate severity was observed, with a spatial mixture of low intensity surface fires and patches with stand-replacing flare-ups. Fire severity is influenced by slope, climate, nutrient and moisture level of the site and forest floor vegetation, and fire therefore greatly affects landscape diversity in the boreal forests of Europe and Asia (Agee 1998b).

With respect to fire occurrence, *P. sylvestris* forests in North-West Europe are both similar to many other pine forest ecosystem (e.g. in the boreal zone or in the Western US) and at the same time considerably different from these systems. For example, more homogenous stand structures in even-aged single-species pine plantations in North-West Europe are found compared to natural forests in the boreal forests.

The main differences to natural pine forests are found in a strongly human-influenced fire regime. The fire regime (as the combination of the four fire characteristics *area*, *season*, *magnitude* and *frequency*; section 1.2.3) describes the occurrence of fire in a certain region. The fire regime in North-West European pine stands is different from natural pine forests, because almost all fires are ignited by humans and not by lightning. Fire season and frequency is therefore directly affected by human-induced ignitions, fire size depends on the reaction time of fire suppression forces, and less on forest structure. Ecological fire effects, however, are determined by several ecological variables and are similar to that observed in natural pine stands.

Similarities can be found in fuel loading, ecological fire effects and forest dynamics as determined by fire occurrence. **Humus consumption**, determined by **fuel load**, **fire intensity** and humus moisture is the key factor, which determines the magnitude of first- and second-order fire effects. Additionally, several ecological factors, such as the production of **charcoal**, fire-induced **physiological stress** and **mortality** play a role in determining post-fire **succession**.

10.1 Fuel load and humus consumption

The **fuel load** (Chapter 2) in managed stands of *P. sylvestris* is only slightly influenced by the age of the stand. Similar values for fine surface fuels and litter were found in stands from 20 to 120 yrs. This indicates that the production of fine fuels and needles in the crowns is balanced with their decomposition rate. However, the load of larger diameter fuels is highest when stem exclusion affects large trees between 35 and 80 years. Additionally, the presence of large diameter dead wood is directly influenced by the harvest of thinned trees. Dwarf-shrubs, mosses and grasses represent additional fuels. These plants can contribute significantly to the total fuel load (Hille and Goldammer 2002), but are strongly site dependent and have high variation in moisture content, so that they can serve as a heat sink or source during a fire.

For the **humus load**, a logarithmic accumulation over time for a stand age up to 120 years was observed. The O_H and O_F layers added up to 44 t ha^{-1} in 120 yrs old stands – a value which is in the range observed in other pine-dominated forests (Van Wagtendonk *et al.* 1998b).

Fire modeling is an important approach to predict fire behavior, both for suppression activities and to predict the ecological impact of the fire (Chapter 3). There is no process-based fire behavior model available for *P. sylvestris* stands, and given the high variability of fuel and wind, the fire behavior is well met with the custom-made BEHAVE model. Especially for the pine model #23, the calculated spread rates and flame lengths are in range of the observed values. Using the results and the gained experience of the BEHAVE modeling, a FARSITE simulation was created. The FARSITE model is very useful in extreme fire weather situations, where several fire suppression resources have to be positioned at places where they can reach high effectiveness. Additionally, predictions on humus consumption within the burned area are the first step in post-fire succession assessment.

Humus consumption is mainly determined by humus moisture and fuel load (Chapter 2). Humus consumption is based on a two-step process of smoldering, with an endothermic process of char forming (*pyrolysis*), and following, an exothermic process of oxidation (Miyanishi and Johnson 2002). Therefore, propagation of smoldering combustion is dependent on sufficient heat being transferred from the exothermic oxidation zone to the adjacent humus to cause pyrolysis. Factors that affect this heat transfer therefore strongly influence humus combustion.
Humus moisture was found to have the strongest impact on the smoldering combustion process (Frandsen 1987), because stored moisture results in a latent heat flux for water evaporation, which provides an effective heat sink; thus it can slow down or extinguish smoldering combustion. The regression of humus consumption (Chapter 2) is separated into two distinct zones with strong dependence on humus moisture. Due to the concave shape of the regression curve below 120 % humus moisture, small changes in humus

moisture can lead to large changes in the humus consumed, particularly between 50 and 120 % moisture content, which can lead to differences in overstory mortality and impact on seed survival. For humus moisture above 120 % moisture content, almost no smoldering combustion was observed. Spatial and temporal variation in humus moisture can therefore cause strong variation of humus consumed and consequently, with-in site variation in post-fire succession. Humus moisture is mainly determined by rainfall events (Llorens *et al.* 1997), crown coverage (Chapter 4) and type of field layer vegetation (Chapter 2). Clerkx and van Hees (1993) showed the influence of different field layer vegetation types on humus moisture in *P. sylvestris* stands. In their study, humus moisture was lowest at plots without vegetation. For plots with plant cover, their samples beneath grass (*Deschampsia flexuosa* L.) were significantly drier than beneath heather (*Erica tetralix* L.), which was explained by reduced evaporation.
Climate change (e.g. drier summers in North-West Europe; IPCC 2001) and nitrogen inputs, which cause changes in field layer vegetation (e.g. invasion of grasses in many open forest types) are expected and will have a significant effect on humus moisture.

Although several process-based humus consumption models exist, none of them considers the **spatial variation** of humus consumption (Chapter 4). Spatial variation of humus consumption is strongly dependent on humus moisture. Although many of the factors affecting humus moisture are dynamic (such as throughfall, water uptake, weather, etc.; Chapter 2), the prediction of humus consumption patterns can be improved by considering crown coverage of the dominant trees. There is a strong positive correlation between the distance from the stem base and the probability of humus remaining after prescribed fires. Spatial variation of humus moisture, especially within the moisture range of 30 to 120 %, directly affects smoldering combustion and can create local differences in humus consumption (Miyanishi and Johnson 2002, van Wagner 1972; Chapter 2). The spatial humus consumption patterns can be influenced by spatial differences in humus moisture that were caused by spatially varying throughfall rates. Precipitation before the burn can be intercepted by tree crowns and drained off the crown towards the crown perimeter (Otto 1994, Ziemer 1968). This results in low humus moisture in areas close to stem bases of overstory trees, and higher humus moisture in areas below the crown edge or with no crown cover. Therefore, a more intense humus consumption is often observed beneath the crown canopy (Miyanishi and Johnson 2002, Chapter 2).

Spatial interpolation techniques, such as used by Robichaud and Miller (1999) that include overstory stand structure could be linked to process-based humus consumption models to increase the accuracy of humus consumption models and consequently, fire effects. Spatial variation of humus moisture is higher in open conifer forests than in densely stocked plantations (such as in North-West Europe), because precipitation in open stands has a higher variation between locations with and without crown coverage. However, the strong spatial variation of humus moisture has also been observed in old *P. sylvestris* stands (Möttonen *et al.* 1999).

10.2 Fire effects

The degree of humus consumption was found to have a strong influence on post-fire radial growth and mortality. **Physiological stress** induced by fire (Chapter 5) can cause a reduction in post-fire radial tree growth. The degree of humus consumption and post-fire radial growth were negatively correlated. After fires with severe humus consumption, radial growth of *P. sylvestris* declined compared to trees stocking on the unburned part of the stand. The physiological stress is probably caused by damage to fine roots, which are located in the humus and upper mineral soil layer. As a consequence, water uptake is limited.

For *P. sylvestris*, reduced transpiration and thus a generally reduced photosynthetic activity with the consequence of reduced radial growth was observed, when the majority of upper roots was harmed. The symptoms are comparable to the effect of a drought, which leads to reduced radial growth even if substantial water is available for roots in greater depths (Irvine *et al.* 1998). Since the majority of fine roots and mycorrhiza are located in the upper soil horizon (Roberts 1976), they are very susceptible to damage by intense humus consumption (Stendell *et al.* 1999, Grogan *et al.* 2000). Root damage and subsequent restoration of the root and mycorrhiza systems together with the consequence that water- and nutrient transport will be limited throughout a certain period after the fire will result in a reduction of radial growth of the pines.

Fires in the early growing season, when the humus layer is still moist from the winter precipitation, were related to less humus consumption (Chapter 2). Less humus consumption means less harming of the roots and mycorrhiza. Later in the season, when humus moisture reach their annual minimum, humus consumption will be more complete and therefore the fire impact will be more severe. Similar results were found for *P. sylvestris* in the boreal forest, where early season fires did not reduce ring width, but late season fires did (Lehtonen and Kolström 2000).

However, the reduced radial growth was only observed for 3-4 years after the fire. After that, the sampled pines had recovered and showed normal growth.

Three different post-fire stress levels corresponding to a certain degree of humus consumption can be defined:

a) Low-intensity surface fires that occur under moist conditions, i.e. in winter or in the early growing season, when humus moisture is high, lead to restricted humus consumption and thus a limited root and mycorrhiza damage. After such fires, the pines did not show a reduced radial growth.

b) High-intensity surface fires whereby the majority of the humus layer is consumed cause damage of the root system and induce growth stress to the pines with the consequence of reduced radial growth.

c) Very high-intensity surface fires whereby the humus layer is completely consumed and large parts of the root system and the cambium are damaged kill the trees or expose it to lethal bark beetle attack (Stephens and Finney 2002, Linder *et al.* 1998).

Post-fire mortality of certain individuals of the tree population within a burned area leads to a change in stand structure (Chapter 6). After a low-intensity fire (as described in a) mortality after fire is selective - small and suppressed trees have the highest probability of being killed (Linder *et al.* 1998), and post-fire stand structures are similar to that after a thinning from below. Mortality by fire is correlated to dbh because dbh is positively correlated to bark thickness, which is the most important factor determining cambium mortality (Rego and Rigolot 1990).

10.3 Recruitment and succession

Without any disturbance (therefore no humus removal, mineral soil exposure and the overstory left intact), only few recruited pine seedlings are found in *P. sylvestris* stands. With the removal of the hindering humus layer and the production of charcoal, which absorbs phytotoxins, germination and growth conditions are largely improved (Mallik 2003). The fire event is followed by a period of abundant regeneration, but as the ground vegetation recovers, regeneration becomes more difficult and restricted to disturbed microsites created by uprooted spots and decayed wood.
On mesic sites, the limited time-span for regeneration is related to the status of the ericaceous understory. Fire removes the field layer vegetation and the humus layer with all its mechanical and phytotoxic barriers that prevent germination and establishment in the undisturbed forest floor (Mallik 2003, Chapters. 11, 12). Phytotoxins were found in plants, litter and decomposed organic layers and in smaller amounts in the upper mineral soil (Jäderlund 2001, Wallstedt *et al.* 2000). Typical fires in pine stands remove the field layer, the litter and, dependent on fire intensity, parts of the soil O horizon (Agee 1998a). With the combustion of the organic material, toxins are also oxidized and destroyed. So, the toxin-load of the site is strongly reduced for several years, until new ericaceous plants recover from seeds or rhizomes (Granström and Schimmel 1993, Schimmel and Granström 1996). This gives tree seeds the opportunity to germinate and grow without toxic interference. The temporal removal of the toxin source and stored toxin pools (in the litter and decomposed organic layers) can be seen as a first-order fire-effect.

Charcoal has the potential to reduce phytotoxic levels in the forest floor. The charcoal remaining after the fire effects plant establishment in the following growing seasons. It stimulates active soil microbial biomass (Wardle *et al.* 1998) and, as shown in Chapter 7, adsorbs toxins produced by early emerging shrubs. This temporal scale of toxin concentration is important, because the adsorptive potential of charcoal diminishes over time (Zackrisson *et al.* 1996). So, the perfect time for seed germination is directly after the fire event, partly due to the reduced toxin load and the maximum adsorption potential of the charcoal. The adsorption of toxins by natural charcoal can therefore be regarded as a long-term second-order fire-effects.

However, the ecological significance of this effect is overrated (Chapter 7). Previous studies (Zackrisson *et al.* 1996) suggested that charcoal has a significant ecological impact on succession dynamics - however, it should be considered that these results were obtained in lab experiments and that the positive effects on germination do most likely not lead to an additional improvement of germination conditions when the main hindering factor, the humus, is removed. Additionally, Zackrisson *et al.* (1996) assumed, that pine charcoal has the same absorptive capacity as Activated Carbon, but the results in Chapter 7 show, that the inner surface area of charcoal is five times less compared to Activated Carbon.

As shown in Chapter 8, conditions for seedlings improve with higher humus consumption. Number of seedlings emerged and growth rates were higher on severely burned sites than on slightly burned areas. On plots with a total incineration of the organic material (burnt litter and humus), seedling numbers and height growth rates were higher than on partly burnt plots (burnt litter) and scarified plots. It is likely that the remaining humus layer after low-intensity fires is still too thick to penetrate for many seedlings and therefore causes higher mortality rates.
The fire accelerates the nutrient cycle, as these nutrients which were locked in the humus layer are suddenly released and available for plants (Covington and Sackett 1986, Chandler *et al.* 1983, Beese and Divisch 1980, Braathe 1974). In contrast to mechanical means of site preparation that remove the entire organic material down to the mineral soil, nutrients are not removed from the plots and are made plant-available (Johansson 1994). Especially high amounts of nitrogen, required in initial seedling stages for ectomycorrhizal colonization (Rebane 2001), are made plant-available by fire.
Recruitment of *P. sylvestris* seems to be favored by forest fires of medium intensity, which leave little organic material behind. Fires of low intensity are not able to remove enough organic material, fire of high intensity and a stand-replacing character would kill seed trees.

The usual **successional pathway** of pine stands in North-West Europe is based on even-aged monocultures of *P. sylvestris*. These stands face self-thinning or stem-reduction by harvesting, a thick humus layer with increasing stand age (Chapter 2), high light levels on the forest floor in higher stand-ages (81-160 years), which leads to a dense forest floor vegetation on most sites. Eventually, broadleaved species (which are the naturally occurring species in North-West Europe), invade the site (Fanta 1982). Depending on site quality, maple, ash, beech or oak are recruited. On poor sites, birch and aspen are found with *P. sylvestris* playing a major role in the succession.
Without disturbance (therefore no humus removal, mineral soil exposure and the overstory left impact), *P. sylvestris* is not able to recruit new cohorts, and only few recruited pine seedling are found (in most studies less than 1.000 ha^{-1}; Dohrenbusch 1997).

After *scarification*, as a human-induced stimulus for recruitment in *P. sylvestris* stands, site re-colonization is fast and ample seedlings are found (7,000 to 80,000 ha^{-1}). *P. sylvestris* is dominating the seedling cohort in most cases (Karlsson *et al.* 2002).
After *windthrow*, with only spot-wise exposure of the mineral soil, seedling establishment is rather slow, and seedlings establish over longer periods up to ten years. Also, spatial seedling distribution is patchy and strongly clustered in disturbed microhabitats, particularly uprooted pits and mounds (Ulanova 2000, Kuuluvainen & Juntunen 1998). Pine seedling numbers are reported to be strongly correlated to site factors, forest floor vegetation type, and humus depth (Wiedemann 1948, Dohrenbusch 1997).

Two main approaches to describe the structural development of *P. sylvestris* stands have been derived over the last decades. A climax model was developed with a site-specific climax (Fanta 1995, Van Dobben 1994), where disturbances lead to a retrogressive stand dynamic. More recently, a pathway model was created with pathways of structural development depending on site, silvicultural treatment and disturbances (Kint 2003). However, in this model, pathways after fire disturbance are mentioned but not defined.
In general, early post-fire succession depends on life history traits of the available species and variation in fire severity. Moreover, the effect of these two components varies according to environmental conditions, such as site, aspect or soil fertility (Gracia *et al.* 2002, Halpern 1989).
Fire severity strongly affects post-fire succession by determining the magnitude of several first- and second-order fire effects. Three severity classes (low severity, high severity on small or large scale) with specific successional pathways are differentiated for *P. sylvestris* stands:

- Low severity fires. Overstory trees remain alive, fuel and humus load is reduced and occasionally, natural regeneration is induced on the entire burned area.
- Small-scale fires of high severity (< 1ha). Gap regeneration takes place and seeds from the surrounding stands quickly re-colonize the disturbed area. Also other pioneer species can be found (birch, aspen; Chapter 9).
- High severity fires. Overstory trees are killed and the entire humus is removed. Seed availability is limited and depends on seed trees outside the burned area. Species composition on the burned site depends on the size of the burned area, since seed dispersal range of *P. sylvestris* is limited to approximately 100m.

Low severity fires reduce stem numbers by inducing mortality to trees in lower DBH-classes (Goldammer 1979, Chapter 6). These fires also reduce humus depths and eventually stimulate natural regeneration in older pine stands (Recke 1928), where light levels are high enough for seedling growth. The magnitude of fire impact depends on fire intensity, but fire size does not play a significant role on post-fire species composition, since enough seed-carrying overstory trees survive.

After **high severity, small-scale** burns less than 1 ha, fire promoted the recruitment of pine, birch and aspen in small gaps within a single-species dominated forested area (Chapter 9). This is attributed to good seedbed conditions after fire for pioneer species by removal of the thick organic soil layer (Chapter 8) and phytotoxic compounds (Gallet and Pellissier 1997, Chapter 7), the high availability of nutrients in the upper mineral soil (Viro 1974), increased soil temperatures (Domisch *et al.* 2001) and moisture conditions (Oleskog and Sahlen 2000) and the removal of the overstory by fire-induced heat damage on the cambium and roots (Rego and Rigolot 1990). Due to the small size of the areas burned, seed dispersal of *P. sylvestris* was sufficient to reach every location within the burned site (Dohrenbusch 1997). Pine seedling numbers were high with up to 368 000 ha^{-1} in year 4 after the fire and almost all seedlings emerged the first year after the fire. Despite the higher numbers of pine and the slower colonization of the burned sites, the two broadleaved species (aspen and birch) overgrew the pine seedlings quickly and became pre-dominant in the seedling cohort.
Compared to other disturbance types, the recruitment window after the fires of 1-2 years was very short. For natural regeneration of pine after thinning a 7-yr recruitment window was observed (Dong *et al.* 2003). After windthrow, an even wider age-class range of recruited pine seedlings (2-13 yrs) was found (Dohrenbusch 1997). Compared to results of studies on succession after fire in boreal pine stands, the recruitment window for *P. sylvestris* is slightly longer in Scandinavia (3-5 yrs; Engelmark *et al.* 1998).

In studies by Jahn (1980) and Wichmann (1996), which document succession on a 2.5 ha fenced area within a 8.000 ha **high severity, large scale** fire in the Lüneburger Heide (Germany), post-fire succession went through several stages (fungi, mosses, grasses) over several years, before any tree seedlings were found. Large-scale fires with high severity remove seed trees over large areas, and therefore limit seed availability at certain locations within the burned area due to the limited seed dispersal distances of *P. sylvestris*. Pioneer species with wind-dispersed seeds are enabled to colonize the burned site and will dominate the new stand (Wichmann 1996). Two decades after the fire, mainly willow (*Salix* spc.), aspen and birch were found (49 %, 33 % and 18 % of total seedling number, respectively), and due to the limited range of pine seed dispersal, no pines colonized the site. This shows, that for larger burned areas the spatial distribution of tree species is determined by their seed dispersal mechanisms and range (Greene and Johnson 1989), and under these conditions, colonization and succession are slow, depend on much more in- and external factors and are therefore harder to predict (Turner *et al.* 1998).
It is also possible, that a combination of a low severity fire with local flare-ups with stand-replacing severity leads to gaps where the overstory cohort is killed, leading to ample regeneration on the entire burned site. After the initial site re-colonization, the subsequent deterioration of possibilities for regeneration and seedling survival depends on suitable microsites created by more recent small-scale disturbances, such as the fall of trees killed by the fire (Kuuluvainen and Rouvinen 2000).

These three different pathways after forest fire can be compared to fires occurring under natural fire regimes in different forest types. The low severity fires are similar to fires, which frequently occurred in ponderosa pine or mixed-conifer forests in the Western US (Agee 1998b, Tappeiner and McDonald 1996; Stephens and Collins 2004). High severity fires are more often found in boreal pine stands with slow decomposition rates and large fuel build-up, which lead to high fire intensities. Depending on the size of the burn, these sites have a much slower recovery in general, mainly due to the large size of the burned area and the lack of seed sources (Turner *et al.* 1997, Turner *et al.* 1998, Castro *et al.* 1989).
As mentioned earlier, the fire regime plays the key role in determining, which pathway succession will take. Since the regime is strongly human-influenced in North-West Europe, all three pathways are observable. Under dry conditions, high severity fires will occur – the size of the burn will largely depend on fire suppression effectiveness. Under moist conditions, fires will have a lower severity.

10.4 Fire and forest management

Fire is, next to windthrow and insect attack, the most common natural disturbance agent which occurs in Central Europe (Schelhaas *et al.* 2001). Especially in *P. sylvestris* stands, which are replacing the broadleaved forests which would occur on most sites naturally, the fire hazard is high during spring and summer (Lex and Goldammer 2001). In the natural range of *P. sylvestris* in the boreal zone of Europe, fire is the key agent that determines forest dynamics (Agee 1998b). Fire causes mortality in the overstory, removes the ectorganic soil layer (Chapter 2) and therefore prepares suitable seedbed conditions for pine seeds (Chapter 7, 8, 9). However, no approach to include fire in forest management under North-West European conditions has been developed.

With a recently developed stronger emphasis on biodiversity in forest management, the use of **fire** in the **management** of *P. sylvestris* forests should be considered more thoroughly. Especially under the impression of global change a flexible, innovative management approach which considers fire is needed (Lindner 2000). Climate change likely causes a higher fire frequency in North-West Europe in the near future (Badeck *et al.* 2004), and nitrogen inputs through the air will cause higher amounts of grasses in open pine stands, which create an easily ignitable fuel load. Additionally, recent social changes which cause migration from rural areas will lead to a lack of wildfire suppression forces in areas, where they are needed most.
As shown above, post-fire forest dynamics depend on fire severity, but can be predicted accurately. At a given burned site within a *P. sylvestris* stand, forest managers should consider to take no action and leave the burned site to natural succession. Additionally, the use of prescribed fires with a designed severity can be applied to reach certain silvicultural goals, such as the stimulation of natural regeneration.

Fields for the **potential use of fire** in North-West European forestry could be the reduction of fuel hazards and pre-commercial thinnings, the restoration of ecosystems and the preparation of seedbeds and sites for regeneration.

10.4.1 Fuel load and fire hazard reduction

Fuel loads in Central European pine stands are as high as in other conifer-dominated forests in the world. Litter and humus loads increase with stand age up to 23 t ha^{-1} and 44 t ha^{-1}, respectively. Fuel reduction, especially if areas would be burned that could serve as shaded fuel breaks afterwards, would reduce the hazard of crowning fires in case of a wildfire.

High stand-densities, resulting from the traditional clearcut-replanting system with very high densities of trees planted (up to 15.000 ha^{-1}) are an even higher threat than fuels on the ground. There is a high change for surface fires to develop into crown fires in these stands, so that a combination of fuel reduction and thinning would be an appropriate measure to reduce fire hazard in young pine stands.
It has been shown in the past, that both unplanned arsonist-set fires and prescribed burns reduced fuel load and stand density in dense *P. sylvestris* plantations without damaging overstory trees (Goldammer 1979, Chapter 6).

10.4.2 Restoring ecosystems

Several ecosystem types faced frequently fires in the past. Heathlands, for example, gained their uniqueness by frequent use for grazing and the periodical removal of the biomass. Shepherds used fires to generate new *Calluna*-shoots, which are preferred by their sheep (Pyne 1997). The fire stopped the succession of these areas into forest by scorching tree seedlings, maintaining the unique character and ecological value of these areas (Muhle and Röhrig 1979, Klaus 1993). Without periodic disturbances, these ecosystems will success into forests and will therefore loose their unique flora and fauna.

Traditional forest uses included growing cereals within burned patches inside the closed forest. These techniques lead to diverse forest structures and habitats for fauna and flora (Pyne and Goldammer 1997).

10.4.3 Preparing seedbeds and sites for regeneration

Anecdotal reports by Recke (1928) and Conrad (1925) observed the beneficial use of surface fire in pine forests in Europe. The stimulating effect of fire on pine recruitment

bases mainly on the removal of the humus layer, which hinders seedling establishment mechanically (Den Ouden and Vogels 1997) and chemically by stored phytotoxins (Jäderlund 2001, Chapter 7). Surface fire can expose the mineral soil area-wise and stimulates germination and seedling establishment (Chapter 8). Current forest regeneration techniques, such as mechanical scarification, mimic these effects.
However, the large variation in fire intensity and severity makes it essential to differentiate between different fire types. There is no doubt that uncontrolled fires under extreme conditions yield in undesired conditions and are an ecological catastrophe (e.g. the large scale fires in 1976). Small fires of high severity (Chapter 9) and low intensity surface fire yield in desired post-fire pathways.

It is often argued, that the application of fire is not controllable and too dangerous within densely populated Central Europe. However, examples from other regions, but also from Europe, where prescribed fires are already used on small-scale fires to restore and maintain heathlands, show that fire is controllable. The examples of increased biodiversity and desired natural regeneration of pine stands show, that fire under certain conditions create conditions, which are in line with current forest management goals. However, prescribed burning in Central Europe is still in the 'research stage', but results so far are promising that a broader application will follow soon.

CHAPTER 11

Summary

11 Summary

This thesis studies the ecological consequences of forest fire in North-West European Scots pine (*Pinus sylvestris*) forests. The focus is on post-fire succession, and the factors and mechanisms that influence the successional pathways after fire.
Fuel load, humus consumption and size of the burn are the most important factors that play a role in the impact of forest fires in North-West European pine stands. While the size of the burn is mainly determined by the reaction time of suppression forces and the weather situation, the occurrence of fire has a strong impact on several key ecological processes that play a role in forest ecosystem development.

Fuel load and fuel moisture

The fuel load and the fuel moisture - in combination with external weather factors - determine the intensity of the fire. In combination with humus moisture, this affects the degree of humus consumption. High fuel loads and low humus moistures are positively correlated with humus consumption.

Field data show a steady accumulation of humus in Central European Scots pine stands (up to 45 t ha^{-1} in 120 yr old stands). Amounts of litter remain constant over the different stand ages (~15 t ha^{-1}). The consumption of this fuel load by fire is strongly dependent on moisture. The relationship between humus moisture and humus consumption was tested in a controlled laboratory experiment. Above moisture levels of 120%, humus was reduced by a constant 10-15% on a dry mass basis. When the moisture level was below 120% there was a parabolic increase in humus consumption with decreasing moisture levels. Increasing the amount of fuel load led to an increase in humus consumption.
Because of the strong relationship between humus moisture and humus consumption, spatial heterogeneity in humus moisture may cause distinct spatial patterns in humus consumption under field conditions. This was studied in a mixed-conifer forest in the north-central Sierra Nevada, California. One site was burned under dry conditions, and here almost all duff (the O_f and O_h layers of the ectorganic soil profile) was consumed, with only small quantities remaining in canopy gaps. Another site was burned under moist conditions a few days after the first annual rains. This 'wet' burn produced strong spatial patterns of duff consumption; with increasing distance from the base of the nearest canopy tree, the probability of duff remaining significantly increased. Canopy crown coverage was clearly correlated to spatial patterns in duff consumption. There is strong evidence that variation in precipitation throughfall resulted in a higher duff moisture content in gaps, and lower duff moisture beneath tree crowns, causing spatial variation in duff consumption in the stand.

These studies show that including a spatial component in a process-based duff consumption model would improve the accuracy of fire-effect predictions. Furthermore, these results can be used to predict humus consumption in Scots pine stands, and to build a fire severity and post-fire succession model for Scots pine stands in Central Europe.
In this thesis, a contribution is made to the development of such models in cooperation with the Global Fire Monitoring Center in Freiburg, Germany. By adapting established fire behavior simulation models (BEHAVE, FARSITE), for the first time a fire behavior model has been applied for the specific conditions of pine forests in the eastern, continental part of Germany. In this model the interspersed heathlands were included that constitute an important carrier of a wildfire at landscape level. The predicted fire behavior was in good agreement with results from an experimental fire. Besides the model, contributions were made to an automated fire detection system that was able to detect all fires arising in an area of approximately 1000 km^2. These are elements in the development of a fire information – decision support system that may use remote sensing data to forecast fire danger.

Direct fire effects

Humus consumption influences tree mortality, post-fire overstory tree growth and site re-colonization. The higher the degree of humus consumption, the higher the induced physiological stress on trees inside the burned area. Under severe conditions, this may even kill the trees.

A surface fire in a 43 year old Scots pine plantation in Eastern Germany caused a partial reduction of the litter and humus layers, a high mortality in the smaller dbh-classes and temporarily lower radial tree growth when compared to an adjacent, unburned stand. The effects of this fire were similar to those observed in a thinning from below. Since commonly post fire succession models only include stand replacing fires, this 'non-stand-replacing' disturbance adds a new component to this model and highlights the high variability of possible successional pathways after fire.
In four pine stands in Germany that had surface fires up to 30 years ago, radial growth patterns of the surviving trees was studied. In two of the stands, where the fire occurred in the late season and had consumed almost the entire humus layer, the radial growth was significantly reduced up to seven years after the fire. In the other two stands, which burned in the early season and where the humus layer was charred only superficially, no clear effect on post-fire radial growth was observed. Overall, radial growth reduction was less in large diameter trees and in trees that experienced less humus consumption around their stem bases. It is assumed that heat damage to (fine) roots and mycorrhizas as a result of extensive humus consumption caused a reduction in tree vitality reflected in reduced radial growth.

Germination and establishment after fire

The consumption of the ectorganic soil layers causes the exposure of mineral soil. This favors the establishment and early growth of seedlings of pine and other small-seeded pioneer tree species. Furthermore, the charcoal produced by the fire improves germination conditions by reducing the levels of phytotoxins in the forest floor produced by ericaceous species.

The establishment of pine seedlings in relation to mechanical soil scarification, burning of litter (OL) and burning of litter and humus (OL + OFH) was studied in two mature pine stands in Germany. All treatments that removed organic soil yielded in higher seedling counts than on the undisturbed forest floor. The highest number of seedlings was found on scarified and severely burnt plots, whereas seedling counts were lower on lightly burnt plots. Seedlings were significantly taller on burnt plots. On the undisturbed forest floor, pine seedlings are seldom able to establish due to high incidence of desiccation in the ectorganic soil layer.

The potential of charcoal to detoxify phytotoxic compounds produced by *Vaccinium myrtillus* and *Calluna vulgaris* were studied in a bioassay. Aqueous extracts were made of young leaves of the two species in two concentrations (10 g and 14 g of dried leaves in 100 ml distilled water respectively). The germination of pine seeds was prevented by the high concentration extract, while the lower concentration extract did not significantly reduce germination. Supplemented carbon (activated carbon, powdered or granulated pine charcoal) restored germination in strong extracts. Adding activated carbon resulted in germination of almost 100%. With pine charcoals added, lower germination percentages were observed. The charcoal powder was more effective (60 % for *C. vulgaris*; 28 % for *V. myrtillus*) than the charcoal granulate (30 % and 16 %, respectively) in restoring germination.

Chemical and surface analysis of the three carbon supplements revealed that activated carbon had by far the largest active surface area (641 $m^2 g^{-1}$), and thus many more cavities to bind phytotoxic compounds than natural charcoal (total surface area of 142 $m^2 g^{-1}$). Charcoal, produced by forest fires, can thus have a positive effect on seed germination, but to a much lesser extent than activated carbon. Previous studies, which used activated carbon as an equivalent for charcoal, therefore overestimated the effect of charcoal on germination.

Post-fire succession

Post-fire succession in Scots pine stands depends on overstory mortality and seedbed conditions (both determined by humus consumption), but also the size of the burned area.

Three fire types with different ecological post-fire pathways were distinguished:

- High severity fires on a small-scale with ample recruitment of Scots pine, but also birch and aspen.
- High severity fires on a large-scale where re-colonization is mainly limited by seed availability, therefore mainly the wind-dispersed birch and aspen are found in the post-fire cohort.
- Low severity fires in old pine stands with high light levels on the forest floor lead to ample regeneration on the entire burned area (shelterwood regeneration).

In the last part of this thesis, special attention was given to high severity, small-scaled (<1 ha) fires. Early forest succession up to 12 y after fire occurrence was studied in four Scots pine stands in Eastern Germany. The post-fire tree cohort mainly consisted of Scots pine, but also birch and aspen seedlings. Seedling numbers of Scots pine decreased linearly from on average 250,000 ha^{-1} in year 4 to 35,000 ha^{-1} in year 12. For birch and aspen, seedling numbers peaked 6 to 8 yrs after the fires (5,500 ha^{-1} and 3,600 ha^{-1}, respectively), and 12 years after the fires, seedling numbers were 5,100 ha^{-1} for birch and 1,400 ha^{-1} for aspen. Pines recruited only in the first and second post-fire year, aspen and birch established during a period of seven years. Compared to the unburned part of the stand, the organic soil layer was significantly reduced by the fires, and higher moss depths were observed on the burned sites. Compared to succession after other disturbance types in Scots pine stands, such as windthrow or scarification, seedling numbers are higher after small-scale fires by a magnitude of ten.

The use of fire in forest management

Post-fire succession is diverse in species composition and includes several different pioneer species. The stand structures following low severity and high severity fires of small extent are in line with current forest management objectives of multi-species stand composition and horizontal and vertical stand structures. Burned sites not necessarily have to be re-planted immediately; often it would be better to leave the site to natural succession.

The controlled use of fire in the management of Scots pine stands should be reconsidered, especially for regeneration purposes and fuel load reduction in areas with increased fire hazard. Controlled forest fires could be used as an additional silvicultural technique to regenerate and transform single-species pine stands into mixed and more natural forests. However, further research is necessary to find out more about the conditions, under which prescribed fires can be applied and which ecological consequences can be expected.

REFERENCES

References

Agee JK. 1993. Fire ecology of Pacific Northwest forests. Island Press, Washington, D.C.

Agee JK. 1998a. Fire and pine ecosystems. In: Richardson DM (ed) Ecology and Biogeography of Pinus, Cambridge University Press, pp 193-218

Agee JK. 1998b. The landscape ecology of Western forest fire regimes. *Northwest Science* 73: 24-33

Agren J, Isaksson L, Zackrisson O. 1983. Natural age and size of *Pinus sylvestris* and *Picea abies* on a mire in the inland part of Northern Sweden. *Holartic Ecology* 5: 228-237

Ahern F, Goldammer JG, Justice C (eds). 2001. Global and regional vegetation fire monitoring from space: Planning a coordinated international effort. SPB Academic Publishing bv, The Hague, The Netherlands

Ahlgren CE. 1974. Effects of Fires on Temperate Forests. North Central United States. In: Kozlowski TT, Ahlgren CE (eds): Fire and Ecosystems. Physiological Ecology. Academic Press, pp 195-223

Al E, Montizaan M. 1987. Spontane bosontwikkeling onder scherm van grove den op droge zandgrond. Thesis, Landbouw Universiteit Wageningen, The Netherlands. pp. 77

Anderson HE. 1990. Moisture diffusivity and response time in fine forest fuels. *Canadian Journal of Forest Research* 20: 315-325

Angelstam P. 1998. Maintaining and Restoring Biodiversity in European Boreal Forests by Developing Natural Disturbance Regimes. *J of Vegetation Science*, no. 9, pp. 593-602

Armour H, Straw N, Day K. 2003. Interactions between growth, herbivory and long-term foliar dynamics of Scots pine. *Trees* 17: 70-80

Athari S. 1980. Untersuchung über die Zuwachsermittlung rauchgeschädigter Fichtenbestände. Dissertation Fakultät für Forstwissenschaften und Waldökologie, Georg-August-Universität Göttingen

Badeck F-W, Lasch P, Hauf Y, Rock J, Suckow F, Thonicke K. 2004. Steigendes klimatisches Waldbrandrisiko. *AFZ/Der Wald* 59: 90-93

Banks JCG. 1991. A review of the use of tree rings for the quantification of forest disturbances. *Dendrochronologia* 9: 51-70

Barret SW, Arno SF, Key CH. 1991. Fire regimes of western larch-lodgepole pine forest in Glacier National Park, Montana. *Can J For Res* 21:1711-1720

Beck W. 2000. Silviculture and stand dynamics of Scots pine in Germany. Invest. Agr.: Sist. Recur. For.: Fuera de Serie n° 1-2000, pp 200-212

Beese F, Divisch RF. 1980. Zum Stoffaustrag eines durch Waldbrand beeinflußten Braunerde-Podsols unter Kiefer. *Forstwissenschaftliches Centralblatt* 99: 257-273

Beier C, Hansen K, Gundersen P. 1992. Spatial variability of throughfall fluxes in a spruce forest. *Environmental Pollution* 81: 257-267

Beland M, Agestam E, Ekö M, Gemmel P, Nilsson U. 2000. Scarification and seedfall affects natural regeneration of Scots pine under two shelterwood densities and a clear-cut in southern Sweden. *Scand J For Res* 15: 247-255

Bending GD, Read DJ. 1996. Effects of the soluble polyphenol tannic acid on the activities of ericoid and ectomycorrhizal fungi. - *Soil Biol Biochem* 28: 1595-1602

Bengtsson J, Nilsson SG, Franc A, Menozzi P. 2000. Biodiversity, disturbances, ecosystem function and management of European forests. *For Ecol Manage* 132: 39-50

Bergeron Y, Charron D. 1994. Postfire stand dynamics in a southern boreal forest (Quebec): A dendroecological approach. *Ecoscience* 1: 173-184

Bergeron Y, Harvey B. 1997. Basing silviculture on natural ecosystem dynamics: an approach applied to the southern boreal mixedwood forest of Quebec: *For Ecol Manage* 92: 235–242

Biswell HH. 1989. Prescribed burning in California wildlands vegetation management. University of California Press

BMF (Bundesministerium der Finanzen). 2001. Naturnaher Waldbau - Die Waldentwicklungstypen der Bundesforstverwaltung. BMF, Berlin

Bourgeau-Chavez LL, Harrell PA, Kasischke ES, French NHF. 1997. The detection and mapping of Alaskan wildfires using a spaceborne imaging radar system, *Int. J. Remote Sensing* 18 (2): 355-373

Bouten W, Heimovaara TJ, Tiktak A. 1992. Spatial patterns of throughfall and soil water dynamics in a douglas fir stand. *Water Resources Research* 28: 3227-3233

Boyer WD. 1987. Volume growth loss: a hidden cost of periodic prescribed burning in longleaf pine. *South J Appl For* 11: 154-157

Braathe P. 1974. Prescribed Burning in Norway - Effects on Soil and Regeneration. Proc. Annual Tall Timber Fire Ecology Conference, 13: 211-222

Brose P, Schuler T, Van Lear D, Berst J. 2001. Bringing fire back. The changing regimes of the Appalachian mixed-oak forests. *J For* 99: 31-35

Brown JK. 1974. Handbook for inventorying downed woody material. USDA Forest Service Intermountain Forest and Range Experiment Station General Technical Report INT-16. Ogden, Utah

Brown JK, Marsden MA, Ryan KC, Reinhardt ED. 1985. Predicting duff and woody fuel consumed by prescribed fire in the Northern Rocky Mountains. USDA Forest Service Intermountain Forest and Range Experiment Station General Research Paper INT-337. Ogden, Utah

Brown JK, Reinhardt ED, Fischer WC. 1991. Predicting humus and woody fuel consumption in northern Idaho prescribed fires. *Forest Science* 37: 1550-1566

Bruckner A, Kandeler E, Kampichler C. 1999. Plot-scale spatial patterns of soil water content, pH, substrate-induced respiration and N mineralization in a temperate coniferous forest. *Geoderma* 93: 207-223

Brunauer S, Emmett PH, Teller E. 1938. Adsorption of gases in multimolecular layers. - *J Am Chem Soc* 60: 309-319

Buchmann N, Gebauer G, Schulze ED. 1996. Partitioning of N-15-labeled ammonium and nitrate among soil, litter, below- and above-ground biomass of trees and understorey in a 15-year-old Picea abies plantation. *Biogeochemistry* 33: 1-23

Burgan RE, Rothermel RC. 1984. 'BEHAVE: fire behavior prediction and fuel modeling system – fuel subsystem.' USDA Forest Service Intermountain Forest and Range Experiment Station General Technical Report INT-167. Ogden, Utah

Burger LW, Hardy C, Bein J. 1998. Effects of prescribed fire and midstory removal on breeding bird communities in mixed pine-hardwood ecosystems of southern Mississippi. Tall Timbers Fire Ecology Conference Proceedings, 20: 107-114

Burrough PA. 1986. Principles of Geographical Information Systems for Land Resources Assessment, Oxford University Press, Oxford

Burrough PA, McDonnel RA. 1998. Principles of Geographical Information Systems, Oxford University Press, Oxford

Busing RT and White PS. 1997. Species diversity and small-scale disturbance in an old-growth temperate forest: a consideration of gap partitioning concepts. *Oikos* 78: 562-568

Butin H, Kappich I. 1980. Untersuchungen zur Neubesiedlung von verbrannten Waldböden duch Pilze und Moose. - *Forstw. Cbl.* 99: 283-296

Byram GM. 1963. Combustion of forest fuels. In: Forest fire. Control and use (K.P. Davis, ed.), 61-89. McGraw Hill, New York

Canham CD, Marks PL. 1985. The response of woody plants to disturbance: patterns of establishment and growth. In: Picket, S.T.A. & White, P.S. (eds.) The ecology of natural disturbance and patch dynamics, pp. 197-216. Academic Press, London

Cao C, Lam N. 1997. Understanding the scale and resolution effects in remote sensing and GIS, In: D. A. Quattrochi and M. F. Goodchild (eds), Scale in Remote Sensing and GIS, CRC Lewis Publishers, Boca Raton, Florida, pp. 57-72

Cao NZ, Darmstadt H, Soutric F, Roy C. 2002. Thermogravimetric study on the steam activation of charcoals obtained by vacuum and atmospheric pyrolysis of softwood bark residues. - *Carbon* 40: 471-479

Castro J, Bento J, Rego F. 1989. Regeneration of Pinus pinaster forests after wildfire. In: Fire in ecosystem dynamics, edited by Goldammer and Jenkins, SPB Academic Publishing

Chandler C, Cheney P, Thomas P, Trabaud L, Williams D. 1983. Fire in forestry, Volume I: Forest fire behavior and effects. John Wiley & Sons, New York

Chrosciewicz Z. 1974. Evaluation of fire-produced seedbeds for jack pine regeneration in central Ontario. *Can J For Res* 4: 455-457

Chrosciewicz Z. 1989. Prediction of forest-floor moisture content under diverse jack pine canopy conditions. *Can J For Res* 19: 1483-1487

Chuvieco E, Congalton RG. 1989. Application of remote sensing and geographic information systems to forest fire hazard mapping, *Remote Sens Environ* 29: 147-159

Clark B. 1994. Soils, water and watersheds. In: Fire effects guide. National Wildfire Coordinating Group, US Department of the Interior, NFES #2394

Clerkx APPM, Van Hees AFM. 1993. 'Het vochtgehalte in de strooisellaag oder verschillende vegetaties in twee grove-dennenopstanden.' IBN-rapport 040, Instituut voor Bos- en Natuuronderzoek. Wageningen, The Netherlands

Collins J, Mancilulli M, Hohlfeld R, Finch D, Sandri G, Shtatland E. 1992. A random number generator based on the Logit transform of the logistic variable. *Computers in Physics* 6: 630-632

Conrad A. 1925. Das Bodenfeuer als Freund des Forstmanns. *Forstl. Wochenzeitschrift Silva* 13: 139-141

Cookson JT. 1978. Adsorption mechanisms: The chemistry of organic adsorption on activated carbon. - In: Cheremisinoff PN, Ellerbusch F (eds.), Carbon adsorption handbook. Ann Arbor Science

Covington WW, Sackett SS. 1986. Effect of periodic burning on soil nitrogen concentrations in ponderosa pine. Soil Science Society of America Proceedings 50:452-457

Covington WW, Fule PZ, Moore MM, Hart SC, Kolb TE, Mast JN, Sackett SS, Wagner MR. 1997. Restoring ecosystem health in ponderosa pine forests of the southwest. *J. For.* 95: 23-29

Chrosciewicz Z. 1974. Evaluation of fire-produced seedbeds for jack pine regeneration in central Ontario. *Canadian Journal of Forest Research* 4: 455-457

Chrosciewicz Z. 1989. Prediction of forest-floor moisture content under diverse jack pine canopy conditions. *Canadian Journal of Forest Research* 19: 1483-1487

Crutzen P, Goldammer JG (eds.) 1992. Fire in the environment. Dahlem Workshop Reports, Wiley Publishers

DeBano LF. 1981. Water repellent soils: A state-of-the art. USDA, For. Serv. Gen. Tech. Rep. PSW-46. Pacif. Southw. For. and Range Exp. Sta., Berkeley, CA

Deeming JE. 1990. Effects of Prescribed Fire on Wildfire Occurrence and Severity. – In: Walstad JD *et.al.* (eds.) Natural and Prescribed Fire in Pacific Northwest Forests, Corvallis OR DeMers MN. 1997. Fundamentals of Geographic Information Systems, John Wiley & Sons, New York

Den Ouden J, Vogels D. 1997. Mechanical resistance by an ectorganic soil layer on root development of seedling *Pinus sylvestris*. *Plant and Soil* 197: 209-217

Denslow JS. 1985. Disturbance-mediated coexistence of species. In: Picket, S.T.A. & White, P.S. (eds.) The ecology of natural disturbance and patch dynamics, pp. 307-327. Academic Press, London

Dohrenbusch A. 1997. Die natürliche Verjüngung der Kiefer (*Pinus sylvestris* L.) im nordwestdeutschen Pleistozän. Schriften aus der Forstlichen Fakultät der Universität Göttingen und der Niedersächsichen Forstlichen Versuchsanstalt, Band 123. Sauerländer´s Verlag Frankfurt am Main

Domisch T, Finér L, Lehto T. 2001. Effects of soil temperature on biomass and carbohydrate allocation in Scots pine (*Pinus sylvestris*) seedlings at the beginning of the growing season. *Tree physiology* 21: 465-472

Dong PH, Diep DQ, Schüler G. 2003. Kiefern-Naturverjüngung im Pfälzerwald - Natural regeneration of Scots Pine in the Pfälzerwald. *Forst und Holz* 58, 83-86

Dyrness CT, Norum RA. 1983. The effects of experimental fires on black spruce forest floors in interior Alaska. *Can J For Res* 13: 879-893

Elgersma AM. 1980. De natuurlijke verjonging von grove den op stormvlaktes op de Veluwe. Thesis, Landbouw Universiteit Wageningen, The Netherlands. pp 33

Elliott KJ, Hendrick RL, Major AE, Vose JM, Swank WT. 1999. Vegetation dynamics after a prescribed fire in the southern Appalachians. *For Ecol Manage* 114: 119-213

Endtmann KJ, Mai DH, Lange E. 1991. Die Kiefer (*Pinus sylvestris*). Berichte aus Forschung und Entwicklung 24. Forschungsanstalt fuer Forst- und Holzwirtschaft Eberswalde

Engelmark O. 1993. Early post-fire tree regeneration in a Picea-Vaccinium forest in northern Sweden. *J Veg Sci* 4: 791-794

Engelmark O, Hofgaard A, Arnborg T. 1998. Successional trends 219 years after fire in an old Pinus sylvestris stand in northern Sweden. *J Veg Sci* 9: 583-592

Esseen PA, Ehnström B, Ericson L, Sjöberg K. 1997. Boreal Forests. *Ecological Bulletin* 46: 16-47

Fanta J. 1982. Natuurlijke verjonging van het bos op droge zandgronden. Rijksinstituut voor onderzoek in de bos- en landschapsbouw "De Dorschkamp". Rapport nr. 301

Fanta J. 1995. Walddynamik in Flugsandgebieten des niederländischen Teiles des nordwesteuropäischen Diluviums. Forstarchiv 66: 128-132

Finney MA. 1998. FARSITE: Fire Area Simulator- model development and evaluation. Res.Pap.RMRS-RP-4, Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station

FIRESCAN Science Team. 1996. Fire in ecosystems of boreal Eurasia: The Bor Forest Island Fire Experiment, Fire Research Campaign Asia-North (FIRESCAN). In: Biomass burning and global change. Vol.II (J.S.Levine, ed.), 848-873. The MIT Press, Cambridge, MA

Flinn MA, Wein RW. 1977. Depth of underground plant organs and theoretical survival during fire. *Can J Bot* 55: 2550-2554

Fosberg MA. 1977. Heat and water transport properties in conifer humus and humus. USDA Forest Service Rocky Mountain Forest and Range Experimental Station Research Paper RM-195. Ft. Collins, CO

Frandsen WH. 1987. The influence of moisture and mineral soil on the combustion limits of smouldering forest humus. *Can J For Res* 17: 1540-1544

Franklin JE, Shugart HH, Harmon ME. 1987. Tree death as an ecological process. *Bioscience* 37: 550-556

Fritts HC. 1976. Tree rings and climate. Academic Press, London

Fritze H, Pennanen T, Pietikaeinen J. 1993. Recovery of soil microbial biomass and activity from prescribed burning. *Can J For Res* 23: 1286-1290

Gadow KV. 2000. Evaluating risk in forest planning models. *Silva Fennica* 34 (2): 181-191

Gallet C, Pellissier F. 1997. Phenolic compounds in natural solutions of conifer forests. *J Chem Ecol* 22: 2401-2412

Gerstengabe F-W, Werner PC, Lindner M, Bruschek G. 1999. Estimation of Future Forest Fire Development in the State of Brandenburg. Int. *Forest Fire News* 21: 91-93

Gimingham CH. 1994 Lowland heaths of West Europe: Management for conservation. *Phytocoenologia* 24: 615-626

Gliessman SR. 1978. The establishment of bracken following fire in tropical habitats. *Am Fern J* 68: 41-44

Goldammer JG. 1979. Der Einsatz von kontrolliertem Feuer im Forstschutz. *Allg Forst- u J Ztg* 150: 41-44

Goldammer JG. 1982. Die Anwendung von Schaumlöschmitteln in der Waldbrandbekämpfung und die Versuchsergebnisse mit EXPYROL W 1. *Allg Forst Z* 37: 1094-1097

Goldammer JG, Di X. 1990. The role of fire in the montane-boreal coniferous forest of Daxinganling, Northeast China: A preliminary model. In: Fire in ecosystem dynamics. Mediterranean and northern perspectives (J.G. Goldammer and M.J.Jenkins, eds.), 175-184. SPB Academic Publishing, The Hague

Goldammer JG, Hille M, Held A. 2001. Feuerexperiment Brandenburg (2001). Forschungsvorhaben A2, Deutsches Forschungsnetz Naturkatastrophen (DFNK). http://www.fire.uni-freiburg.de/dfnk/zwischen bericht.htm.

Gore WT. 1982. Charcoal production and properties. - CSIR special report Hout / National Timber Research Institute. Pretoria. No. 262

Gracia M, Retana J, Roig P. 2002. Mid-term successional patterns after fire of mixed pine-oak forests in NE Spain. *Acta Oecologica* 23: 405-411

Graham RT, Harvey AE, Jain TB, Tonn JR. 1999. The Effects of Thinning and Similar Stand Treatments on Fire Behavior in Western Forests. General Technical Report PNW-GTR-463, USDA Forest Service, Pacific Northwest Research Station, Portland, OR

Granström A. 1987. Seed viability of fourteen species during five years of storage in a forest soil. *Journal of Ecology* 75: 321–331

Granström A. 1996. Fire ecology in Sweden and future use of fire for maintaining biodiversity. In: Goldammer JG, Furyaev VV (eds). Fire in ecosystems of boreal Eurosia. Kluwer Academic Publishers. pp 445-452

Granström A, Schimmel J. 1993. Heat effects of seeds and rhizomes of a selection of a selection of boreal forest plants and potential reaction to fire. *Oecologia* 94: 285-293

Granström A. 2001. Fire Management for Biodiversity in the European Boreal Forest. *Scandinavian J Forest Res.* Suppl. 3: 62-69

Green RN, Trowbridge RL, and Klinka K. 1993. Towards a taxonomic classification of humus forms. *For. Sci. Monogr.* 29: 1–49

Greene DF, Johnson EA. 1989. A model of wind dispersal of winged and plumed seeds. *Ecology* 70: 339-347

Grissino-Mayer HD, Swetnam TW. 2000. Century-scale climate forcing of fire regimes in the American Southwest. *The Holocene* 10: 213-220

Grissino-Mayer HD. 2001. Evaluating cross-dating accuracy: A manual and tutorial for the computer program COFECHA. *Tree-Ring Res* 57: 205-221

Grogan P, Baar J, Bruns TD. 2000. Below-ground ectomycorrhizal community structure in a recently burned bishop pine forest. *J Ecology* 88: 1051-1062

Gutsell SL, Johnson EA. 1995. How fire scars are formed: coupling a disturbance process to its ecological consequences. *Can J For Res* 26: 166-174

Halpern CB. 1989. Early successional patterns of forest species: interactions of life history traits and disturbance. *Ecology* 70: 704-720

Hamilton DA, Edwards BM. 1976. Modeling the probability of individual tree mortality. USDA Forest Service Res Pap, INT-185. Intermountain Forest and Range Experimental Station, Ogden, Utah, USA

Hetsch W. 1980. Bodenphysikalische und bodenchemische Auswirkungen eines Waldbrands auf einen Braunerde-Podsol unter Kiefer. *Forstwissenschaftliches Centralblatt* 99: 257-273

Hille M, Goldammer JG. 2002. Dispatching and modelling of fires in Central European pine stands: New research and development approaches in Germany. In: Improving dispatching for forest fire control. Proceedings of a workshop held in the Mediterranean Agronomic Institute of Chania (MAICh), Crete, Greece, 6-8 December 2001 (General Secretariat for Civil Protection, Greece, and MAICh, Greece, ed.), 59-74. MAICh, Chania

Hille M, Den Ouden J. 2004. Improved recruitment and early growth of Scots pine (*Pinus sylvestris* L.) seedlings after fire and soil scarification. *European Journal of Forest Research*, 123: 213-218

Hille M, Den Ouden J. 2005a. Charcoal and activated carbon as adsorbate of phytotoxic compounds - a comparative study. *Oikos*, 108: 202-207

Hille M, Den Ouden J. 2005b. Fuel load, humus consumption and humus moisture dynamics in Central European Scots pine stands. *Int J Wildland Fire*, 14: 1-7

Hille MG, Stephens SL. 2005. Mixed conifer forest duff consumption during prescribed fires: Tree crown impacts. *Forest Science*, 51 in press.

Hinz R. 1993. 600 Jahre Domstiftsforst Seelensdorf. BFL 1993. S. 101-102

Hofmann G. 1997. Mitteleuropäische Wald- und Forst-Ökosystemtypen in Wort und Bild. AFZ/Der Wald Sonderheft, 91p

Hokkanen TJ, Järvinen E, Kuuluvainen T. 1995. Properties of top soil and the relationship between soil and trees in a boreal Scots Pine stand. *Silva Fennica* 29: 189-203

Holmes RL. 1983. Computer-assisted quality control in tree-ring dating and measurement. *Tree-Ring Bull* 43: 69-78

Hornbeck JW, Smith RB. 1985. Documentation of red spruce growth decline. *Can J For Res* 15: 1199-1201

Huang CP. 1978. Chemical interactions between inorganics and activated carbon. - In: Cheremisinoff PN, Ellerbusch F (eds), Carbon adsorption handbook. Ann Arbor Science

Husari SJ, McKelvey KS. 1996. Fire management policies and programs. pp. 1101-1118 in Sierra Nevada Ecosystem Project: Final report to Congress. Vol. II Assessments and Scientific Basis for Management Options. Wildland Resources Center Report No. 37, Centers for Water and Wildland Resources, University of California, Davis

Huss J, Burschel HP. 1972. Förderung der Buchennaturverjüngung mit verschiedenartigen Bodenbearbeitungsverfahren. Ergebnisse langfristiger Beobachtungen. Forstarchiv 43: 233-239

Innes JL. 1990. General aspects in the use of tree rings for environmental studies. In: E.R. Cook and L.A. Kairiukstis (eds) Methods of Dendrochronology: Applications in the Environmental Sciences. International Institute for Applied Systems Analysis, Kluwer Academic Publishers, Boston, pp 224-229

Innes JL. 1993. Forest health: its assessment and status. CAB, Wallingford, pp 677

IPCC (Intergovernmental Panel on Climate Change) 2001. Climate Change 2001: Working Group II: Impacts, Adaptation and Vulnerability. Geneva, Switzerland

Irvine J, Perks MP, Magnani F, Grace J. 1998. The response of *Pinus sylvestris* to drought: stomatal control of transpiration and hydraulic conductance. *Tree Physiology* 18: 393-402

Jäderlund A, Zackrisson O, Nilsson MC. 1996. Effects of bilberry (*Vaccinium myrtillus* L.) litter on seed germination and early seedling growth of four boreal tree species. - *J Chem Ecol* 22: 973-986

Jäderlund A, Zackrisson O, Dahlberg A, Nilsson MC. 1997. Interference of *Vaccinium myrtillus* on establishment, growth, and nutrition of Picea abies seedlings in a northern boreal site. - *Can J For Res* 27: 2017-2025

Jäderlund A, Norberg G, Zackrisson O, Dahlberg A, Teketay D, Dolling A, Nilsson MC. 1998. Control of bilberry vegetation by steam treatment - effects on seeded Scots pine and associated funghi. *For Ecol Manage* 108: 275-285

Jäderlund A. 2001. Bilberry (*Vaccinium myrtillus* L.) in a boreal forest ecosystem - effects on tree seedling emergence and growth. Thesis, Silvestria 188. Swedish University of Agricultural Sciences

Jahn G. 1980. Die natürliche Wiederbesiedlung von Waldbrandflächen in der Lüneburger Heide mit Moosen und Gefäßpflanzen. *Forstwissenschaftliches Centralblatt* 99: 297-324

Jakubuskas ME, Lulla KP, Mausel PW. 1990. Assessment of vegetation change in a fire-altered forest landscape, *Photogramm Eng Remote Sens* 56(3): 371-377

Jalal MAF, Read DJ, Haslam E. 1982. Phenolic composition and it's seasonal variation in *Calluna vulgaris*. - *Phytochemistry* 21: 1397-1401

Jalal MAF, Read DJ. 1983. The organic acid composition of *Calluna* heathland soil with special reference to phyto- and fungitoxicity. I. Isolation and identification of organic acids. - *Plant Soil* 70: 257-272

Johansen RW. 1975. Prescribed burning may enhance growth of young slash pine. *Journal of Forestry* 73 (3): 148 - 149

Johansson MB. 1994. The influence of soil scarification on turn-over rate of slash needles and nutrient release. *Scand J For Res* 9: 170-179

Johnson EA. 1992. Fire and vegetation dynamics: studies from the North American boreal forest. Cambridge University Press

Johnson DW, Susfalk RB, Dahlgren RA, Caldwell TG, Miller WW. 2001. Nutrient fluxes in a snow-dominated, semi-arid forest: Spatial and temporal patterns. *Biogeochemistry* 55: 219-245

Kaila L, Martikainen P, Punttila P. 1996: Dead trees left in clearcuts benefit saproxylic Coleoptera adapted to natural disturbance in boreal forests. *Biodiversity and Conservation* 6: 1-18

Karlsson C, Örlander G. 2000. Soil scarification shortly before a rich seed fall improves seedling establishment in seed tree stands of *Pinus sylvestris*. *Scand J For Res* 15: 256-266

Karlsson C, Nilsson U, Örlander G. 2002. Natural regeneration in clear-cuts: effects of scarification, slash removal and clear-cut age. *Scand. J. For. Res*. 17: 131-138

Kasaoka S, Sakata Y, Tanaka E, Naitoh R. 1989. Design of molecular-sieve carbon. Studies on the adsorption of various dyes in the liquid phase. - *Int Chem Eng* 29: 734-742

Kasischke ES, French NHF, Harrell P, Christensen NL Jr., Ustin SL. 1993. Monitoring wildfires in boreal forests using large area AVHRR NDVI composite image data, *Remote Sens Environ* 45: 61-70

Kasischke E, Stocks B (eds.) 2000. Fire, Climate Change and Carbon Cycling in the Boreal Forest. Springer Verlag

Kint V. 2003. Structural development in ageing Scots pine (*Pinus sylvestris* L.) stands in Western Europe. PhD Thesis, Ghent University, Belgium. 194p

Klaus S. 1993. Das Birkhuhn (*Tetrao tetrix*) als Nutznießer von Bränden und anderen Katastrophen im Wald – Materialien zu Naturschutz u. *Landschaftspflege* 1: 19-25

Klein E. 1964. Bemerkungen zur Kiefernnaturverjüngung. Die Sozialistische Forstwirtschaft. Januar 1964, 14. Jahrgang

Kolström T, Kellomäki S. 1993. Tree survival in wildfires. *Silva Fenn* 27: 277-281

Koutsias N, Karteris M. 1998. Logistic regression modelling of multitemporal Thematic Mapper data for burned area mapping, *Int J Remote Sens* 19 (18): 3499-3514

Kraft, G. 1884. Beiträge zu Lehre von den Durchforstungen, Slagstellungen und Lichtungshieben. Hannover.

Kührt E, Behnke T, Jahn H, Hetzheim H, Knollenberg J, Mertens V, Schlotzhauer G, Gotze B. 2000. Autonomous early warning system for forest fires tested in Brandenburg (Germany), *Int Forest Fire News* 22: 84-90

Kullman L. 1986. Late Holocene reproductional patterns of *Pinus sylvestris* and *Picea abies* at the forest limit in central Sweden. *Canadian Journal of Botany*, 64: 1682-90

Kuuluvainen T, Hokkanen TJ, Järvinen E, Pukkala T. 1993. Factors related to seedling growth in a boreal Scots pine stand: a spatial analysis of a vegetation-soil system. *Can J For Res* 23: 2101-2109

Kuuluvainen T, Juntunen P. 1998. Seedling establishment in relation to microhabitat variation in a windthrow gap in a boreal *Pinus sylvestris* forest. *J Veg Sci* 9: 551-562

Kuuluvainen T, Rouvinen S. 2000. Post-fire understory regeneration in boreal Pinus sylvestris forest sites with different fire histories. *J Veg Sci* 11: 801-812

Kuusela K. 1990. The dynamics of boreal coniferous forests. SITRA 112. Helsinki

Lageard JGA, Thomas PA, Chambers FM. 2000. Using fire scars and growth release in subfossil Scots pine to reconstruct prehistoric fires. *Palaeogeography, Palaeoclimatology, Palaeocology* 164: 87-99

Lam N, Quattrochi DA. 1992. On the issues of scale, resolution and fractal analysis in the mapping sciences, *Prof. Geogr.* 44 (1): 88-98

Lawson BD, Dalrymple GN, Hawkes BC. 1997. Predicting forest floor moisture contents from duff moisture code values. Canadian Forest Service, Technology Transfer Notes 6

Leblon B, Kasischke E, Alexander M, Doyle M, Abbott M. 2002. Fire danger monitoring using ERS-1 SAR images in the case of Northern Boreal forests, *Nat Hazards* 27: 231-255

Leder B. 1992. Weichlaubhölzer - Verjüngungsökologie, Jugendwachstum und Bedeutung in Jungeständen der Hauptbaumarten Buche und Eiche. Schriftenreihe der Landesanstalt für Forstwirtschaft Nordrhein-Westfalen

Lehtonen H, Kolström T. 2000. Forest fire history in Viena Karelia, Russia. Scandinavian Journal of Forest Research 15: 585-590

Lembcke G, Knapp E, Dittmar O. 1975. 'DDR-Kiefern-Ertragstafel 1975.' IFE-Berichte aus Forschung und Entwicklung (Eberswalde, Germany)

Lex P, Goldammer JG. 2001. Fire situation in Germany. International forest fire news 24: 22-30

Lindbladh M, Niklasson M, Nilsson SG. 2003. Long-time record of fire and open canopy in a high biodiversity forest in southeast Sweden. *Biological Conservation* 114: 231-243

Linder P, Jonsson P, Niklasson M. 1998 Tree mortality after prescribed burning in an old-growth Scots pine forest in northern Sweden. *Silva Fennica* 32: 339-349

Lindholm T, Vasander H. 1987. Vegetation and stand development of mesic forest after prescribed burning. *Silva Fennica* 21(3): 259-278

Lindner M. 2000. Developing adaptive forest management strategies to cope with climatic change. *Tree Physiology* 20: 299–307

Lippens BC, De Boer JH. 1965. Pore systems in catalysts. V. The t-method. - *J Catalysis* 4: 319-323

Llorens P, Poch R, Latron J, Gallart F. 1997. Rainfall interception by a *Pinus sylvestris* forest patch overgrown in a Mediterranean mountainous abandoned area I. Monitoring design and results down to the event scale. *Journal of Hydrology* 199: 331-345

Lust N, Geusdens G, Olsthoorn AFM. 2000. Scots pine in Belgium and The Netherland. Invest. Agr.: Sist. Recur. For.: Fuera de Serie n° 1-2000: 213-230

Luttermann K. 1976. Die große Waldbrandkatastrophe. Schaper Verlag, Hannover

Mallik AU. 1995. Conversion of temperate forests into heaths: Role of ecosystem disturbance and ericaceous plants. *Environmental Management* 19: 675-684.

Mallik AU. 2003. Conifer regeneration in boreal and temperate forests with ericaceous understory: role of disturbance, seedbed limitation, and keystone species change. *Critical Reviews Plant Science* 22: 341-366

Martin RE, Dell JD. 1978. Planning for prescribed burning in the inland northwest. USDA For. Serv. Gen. Tech. Rep. PNW-GTR-66

Mason WL, Alia R. 2000. Current and future status of Scots pine (*Pinus sylvestris* L.) forests in Europe. Invest. Agr.: Sist. Recur. For.: Fuera de Serie n° 1-2000: 317-333

McArthur AG, Cheney NP. 1966. The characterization of fires in relation to ecological studies. *Australian Forest Research* 2: 36-45

McInnis LM, Oswald BP, William HM, Farrish KW, Unger DR. 2004. Growth response of Pinus taeda L. to herbicide, prescribed fire, and fertilizer. *For Ecol Manage* 199: 231-242

Meijer zu Schlochtern M, Koop HG. 2000. Effecten van brand in bos op arme zandgronden. Alterra-rapport 160. Alterra, The Netherlands

Metz LJ. 1958. Moisture held in pine litter. *Journal of Forestry* 56: 148

Michaelis H, Behnke T, Tschentscher M, Mottola S, Neukum G. MOSES - A modular sensor electronics system for space science and commercial applications. SPIE's

International Symposium on Optical Science, Engineering and Instrumentation, Denver, USA, July 1999, SPIE The International Society for Optical Engineering, Materials and Electronics for High-Speed and Infrared Detectors, 115-121

Miller M, Findley J. 1994. Plants. In: Fire effects guide. National Wildfire Coordinating Group, US Department of the Interior, NFES #2394

Mißbach K. 1973. 'Waldbrand- Verhütung und Bekämpfung.' pp. 126. (VEB Deutscher Landwirtschaftsverlag Berlin)

Miyanishi K, Bajtala MJ, Johnson EA. 1999. Patterns of humus consumption in *Pinus banksiana* and *Picea mariana* stands. In 'Proceedings of Sustainable Forest Management Network Conference, Science and Practice: Sustaining the Boreal Forest'. pp. 112-115 (Sustainable Forest Management Network, Edmonton, Canada)

Miyanishi K. 2001. Humus consumption. In 'Forest Fires - Behavior and Ecological Effects'. (Eds Johnson EA, K Miyanishi) pp. 437-475 (Academic Press)

Miyanishi K, Johnson EA. 2002. Process and patterns of duff consumption in the mixedwood boreal forest. *Can J For Res* 32: 1285-1295

Morris WG, Mowat EL. 1958. Some effects of thinning a ponderosa pine thicket with a prescribed fire. *Journal of Forestry* 56: 203-209

Möttönen M, Järvinen E, Hokkanen TJ, Kuuluvainen T, Ohtonen R. 1999. Spatial distribution of soil ergosterol in the organic layer of a mature Scots pine (*Pinus sylvestris* L.) forest. *Soil Biology and Chemistry* 31: 503-516

Mrazek F. 2002. Alemanns Kiefern - Nachkommen. AFZ 4/2002, S. 174ff

Muhle O, Röhrig E. 1979. Untersuchungen über die Wirkungen von Brand, Mahd und Beweidung auf die Entwicklung von Heide-Gesellschaften. Schriften aus der Forstlichen Fakultät der Universität Göttingen und der Niedersächsischen Forstlichen Versuchsanstalt. Band 61

Mutch RW, Gastineau OW. 1970. 'Timelag and equilibrium moisture content in reindeer lichen.' USDA Forest Service Intermountain Forest and Range Experiment Station General Technical Report INT-76. Ogden, Utah

Mutch RW. 1994. Fighting fire with prescribed fire – a return to ecosystem health. *Journal of Forestry* 92 (11): 31-33

Mutch LS, Swetnam TW. 1995. Effects of fire severity and climate on ring-width growth of giant sequoia after fire. In: Brown JK, Mutch RW, Spoon CW, Wakimoto RH (tech coord) Symposium on fire in wilderness and park management: past lessons and future opportunities. March 30- April 1, 1993. USDA Forest Service Gen Tech Rep, INT-320, Ogden, Utah, USA

Neary DG, Klopatek CC, DeBano LF, Ffollitt PF. 1999. Fire effects on belowground sustainability: a review and synthesis. *For Ecol Manage* 122: 51-71

Nelson RM. 1969. Some factors affecting the moisture timelag of woody material. USDA Forest Service South East Research Paper SE-44

Nieuwstadt MGL, Sheil D, Kartawinata K. 2001. The ecological consequences of logging in the burned forests of East Kalimantan, Indonesia. *Cons Biol* 15: 1183-1186

Niklasson M, Drakenberg B. 2001. A 600-year tree-ring fire history from Norra Kvills National Park, southern Sweden: implications for conservation strategies in the hemiboreal zone. *Biol Conserv* 101: 63-71

Nilsson MC. 1994. Separation of allelopathy and resource competition by the boreal dwarf-shrub *Empetrum hermaphroditum* Hagerup. *Oecologia* 98: 1-7

Nilsson MC, Zackrisson O, Sterner O, Wallstedt A. 2000. Characterisation of the differential interference effects of two boreal dwarf shrub species. *Oecologia* 123: 122-128

Nilsson U, Gemmel P, Johansson U, Karlsson M, Welander T. 2002. Natural regeneration of Norway spruce, Scots pine and birch under Norway spruce shelterwoods of varying densities on a mesic-dry site in southern Sweden. *For Ecol Manage* 161: 133-145

NWCG (National Wildfire Coordinating Group). 1989. Fireline Handbook. PMS 410-1, NFES 0065, Boise ID

NWCG (National Wildfire Coordinating Group). 1996. 'Glossary of Wildland Fire Terminology.' PMS 205 (NFES 1832)

Oertel D, Briess K, Halle W, Neidhardt M, Lorenz E, Sandau R, Schrandt F, Skrbek W, Venus H, Walter I, Zender B, Zhukow B, Goldammer JG, Held AC, Hille M, Brueggemann H. 2003. Airborne forest fire mapping with an adaptive infrared sensor. *Int J Remote Sensing* 24: 3663-3682

Ogee J, Brunet Y. 2002. A forest floor model for heat and moisture including a litter layer. *Journal of Hydrology* 255: 212-233

Oinonen E. 1967. Sporal regeneration of bracken (*Pteridium aquilinum* (L.) Kuhn) in Finland in the light of the dimensions and the age of it's clones. *Acta For. Fenn.* 83(1): 1-96

Olberg A. 1955. Die standörtlichen Voraussetzungen und die waldbauliche Ausnutzung der Kiefernnaturverjüngung. Schriftenr. Forstl. Fak. Univ. Göttingen. Bd. 18

Olberg A. 1957. Beiträge zum Problem der Kiefernnaturverjüngung. Schriften aus der Forstlichen Fakultät der Universität Göttingen und der Nds. Forstlichen Versuchsanstalt, Band 18

Oleksyn J, Fritts HC, Mughes MK. 1993. Tree-ring analysis of different *Pinus sylvestris* provenances, *Quercus robur*, *Larix decidua* and *L. decidua* x *L. kaempferi* affected by air pollution. *Arboretum Kornickie* 38: 87-111

Oleskog G, Sahlén K. 2000. Effects of seedbed substrate on moisture conditions and germination of *Pinus sylvestris* seeds in a clearcut. *Scand J For Res* 15: 225 - 236

Oliver CD. 1981. Forest development in North America following major disturbances. *Forest Ecology and Management* 3: 352-354

Oliver CD, Larson BC. 1990. Forest stand dynamics. McGraw-Hill, New York

Oliver CD, Larson BC. 1996. Forest stand dynamics. Update Edition. John Wiley & Sons, Inc., New York

Otto HJ. 1994. 'Waldökologie.' Ulmer, Stuttgart, Germany

Payette S, Bhiry N, Delwaide A, Simard M. 2000. Origin of the lichen woodland at its southern range limit in eastern Canada: The catastrophic impact of insect defoliators and fire on the spruce-moss forest. *Can J For Res* 30: 288-305

Peltzer DA, Bast ML, Wilson SD, Gerry AK. 2000. Plant diversity and tree responses following contrasting disturbances in boreal forest. *For Ecol Mngmt* 127: 191-203

Pollanschütz J. 1966. Verfahren zur objektiven Abschätzung (Messung) verminderter Zuwachsleistung von Einzelbäumen und Beständen. *Mitt Forstl Bundes-Versuchsanstalt* 73: 129-191

Punttila PH, Haila Y. 1996. Colonisation of a burned forest by ants in the southern Finnish Boreal Forest. *Silva Fennica* 30: 421-435

Pyne S. 1997. Vestal Fire. An Environmental History, Told through Fire, of Europe and Europe's Encounter with the World. University of Washington Press, Seattle and London

Pyne SJ, Goldammer JG. 1997. The culture of fire. An introduction to anthropogenic fire history. - In: Sediment records of biomass burning and global change (J.S.Clark, H.Cachier, J.G.Goldammer and B.J.Stocks, eds.). Springer-Verlag, Berlin-Heidelberg-New York pp. 71-114

Pyne S. 1998. Strange fire: The European encounter with fire. Tall Timbers Fire Ecology Conference Proceedings 20: 1-12

Qualls RG, Haines BL, Swank WT. 1991. Fluxes of dissolved organic nutrients and humic substances in a deciduous forest. - *J Ecol* 72: 254-266

Rebane A. 2001. Root function and morphology of *Pinus sylvestris* L. seedlings and its application in forest renewal. Licentiate thesis, Swedish University of Agricultural Sciences, Report No 17, Uppsala

Recke Graf vd. 1928. Naturverjüngung der Kiefer durch Brandkultur. *Der Deutsche Forstwirt* 10, 103: 652-653

Rego F, Rigolot E. 1990. Heat transfer through bark - a simple predictive model. In: Fire in ecosystem dynamics. Mediterranean and northern perspectives (J.G.

Goldammer and M.J. Jenkins, eds.), 157-162. SPB Academic Publishing, The Hague

Reinhardt ED, Brown JK, Fischer WC, Graham RT. 1991. Woody fuel and humus consumption by prescribed fire in northern Idaho mixed conifer logging slash. USDA Forest Service Intermountain Forest and Range Experiment Station Research Paper INT-443. Ogden, Utah

Reinhardt ED, Keane RE, Brown JK. 2001. Modeling fire effects. *International Journal of Wildland Fire* 10: 373-380

Richards FJ. 1959. A flexible growth function for empirical use. *J Expt Botany* 10: 290-300

Richards GD. 1990. An elliptical growth model of forest fire fronts and it's numerical solution. *Int J Numer Methods Eng* 30: 1163-1179

Richards GD. 1995. A general mathematical framework for modeling two-dimensional wildland fire spread. *Int J Wildland Fire* 5(2): 63-72

Riek W, Strohbach B, Siewert C. 2002. Veränderung chemischer Eigenschaften von Waldböden durch Feuereinwirkung - Ergebnisse eines Waldbrandversuches in der Lausitz. Unpublished data, LFE Eberswalde, Germany

Rigling A, Weber P, Cherubini P, Dobbertin M. 2004. Bestandesdynamik zentralalpiner Waldföhrenwälder aufgezeigt anhand dendroökologischer Fallstudien aus dem Wallis, Schweiz. *Schweiz Z Forstwes* 155: 178-190

Rind D, Goldberg R, Ruedy R. 1989. Change in climate variability in the 21st century. Climatic Change 14, 5-37

Rinn F. 1996. TSAP -Time Series Analysis Program. For applications in forestry, tree-ring laboratories and dendrochronology. Rinntech (Heidelberg)

Roberts J. 1976. A study of root distribution and growth in a Pinus sylvestris L. (Scots pine) plantation in East Anglia. *Plant Soil* 44: 607-621

Robichaud PR, Miller SM. 1999. Spatial interpolation and simulation of post-burn humus thickness after prescribed fire. *International Journal of Wildland Fire* 9: 137-143

Robinson RK. 1972. The production by roots of *Calluna vulgaris* of a factor inhibitory to growth of some mycorrhizal fungi. *J Ecol* 60: 219-224

Rode MW. 1993. Leaf-nutrient accumulation and turnover at 3 stages of succession from heathlands to forest. *J Veg Sci* 4: 263-268

Rothermel RC. 1972. A mathematical model for predicting fire spread in wildland fuels. Res. Pap. INT-115. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station

Runkle JR. 1985. Disturbance regimes in temperate forests. In: Picket, S.T.A. & White, P.S. (eds.) The ecology of natural disturbance and patch dynamics, pp. 17-33. Academic Press, London

Rutter AJ. 1963. Studies in the water relations of *Pinus sylvestris* in plantation conditions. *Journal of Ecology* 51: 191-203

Ryan KC. 1982. Evaluating potential tree mortality from prescribed burning, pp. 167-178. In: David M. Baumgartner (comp.). Symp. - Site preparation and fuels management in steep terrain. Wash. State Univ. Coop. Extension, Pullman

Ryan KC, Frandsen WH. 1991. Basal injury from smoldering fires in mature *Pinus ponderosa* Laws. *International Journal of Wildland Fire* 1: 107-118

Ryan MG, Binkley D, Fownes J. 1997. Age-related decline in forest productivity: pattern and process. *Adv Ecol Res* 27: 213-262

Sandberg DV. 1980. Duff reduction by prescribed underburning in Douglas-fir. United States Department of Agriculture, Forest Service Research Paper PNW-272. 18pp

Schaap MG, Bouten W, Verstraaten JM. 1997. Forest floor water content dynamics in a Douglas fir stand. *Journal of Hydrology* 201: 367-383

Scheffer M, Carpenter S, Foley JA, Folke C, Walker B. 2001. Catastrophic shifts in ecosystems. *Nature* 413: 591-596

Schelhaas MJ, Varis S, Schuck A. 2001. Database on Forest Disturbances in Europe (DFDE), European Forest Institute, Joensuu, Finland,

http://www.efi.fi/projects/dfde/

Schelhaas M-J, Nabuurs G-J, Schuck A. 2003. Natural disturbances in the European forests in the 19th and 20th centuries. *Glob Change Biol* 9: 1620-1633

Schimmel J. 1993. On fire. Fire behavior, fuel succession and vegetation response to fire in the Swedish Boreal Forest. Dissertation in Forest Vegetation Ecology 5. Swedish University of Agricultural Sciences, Dep of Forest Veg Ecology

Schimmel J, Granström A. 1996. Fire severity and vegetation response in the Boreal Swedish forest. *Ecology* 77: 1436-1450

Schmidt W. 1929. Waldbrandboden und Keimling! *Der Deutsche Forstwirt* 11, Nr. 37: 193-194

Schmidt L, Hille M, von Gadow K. 2004. Analyse eines Bodenfeuers. Veränderungen der Bestandesstruktur und des Wachstums. *Allg Forst J Ztg* 175: 78-82

Schweingruber FH, Albrecht H, Beck M, Hessel J, Joos K, Keller D, Kontic R, Lange K, Niederer M, Nippel C, Spang S, Spinnler A, Steiner B, Winkel-Seifert A. 1986. Abrupte Zuwachsschwankungen in Jahrringfolgen als ökologische Indikatoren. *Dendrochronologia* 4: 125-183

Schweingruber FH. 1996. Tree Rings and Environment. Dendroecology. Swiss Federal Institute for Forest, Snow and Landscape Research, Birmensdorf and Berne

Skre O, Wielgolaski FE, Moe B. 1998. Biomass and chemical composition of common forest plants in response to fire in western Norway. *J Veg Sci* 9: 501-510

Smit A. 1999. The impact of grazing on spatial variabitity of humus profile properties in a grass-encroached Scots pine ecosystem. *Catena* 36: 85-98

Spanos IA, Daskalakou EN, Thanos CA. 2000. Postfire, natural regeneration of Pinus brutia forests in Thasos island, Greece. *Acta Oecologica* 21: 13-20

Spies TA, Franklin JF. 1989. Gap characteristics and vegetation response in coniferous forests of the Pacific Northwest. *Ecology* 70:543-545

Spurr DH, Barnes BV. 1980. Forest Ecology. 3. Auflage New York

Stendell ER, Horton TR, Bruns TD. 1999. Early effects of prescribed fire on the structure of the ectomycorrhizal fungus community in a Sierra Nevada ponderosa pine forest. *Mycol Res* 103: 1353-1359

Stephens SL. 2001. Fire history of adjacent Jeffrey pine and upper montane forests in the Eastern Sierra Nevada. *Int J Wildland Fire* 10: 161-176

Stephens SL, Finney MA. 2002. Prescribed fire mortality of Sierra Nevada mixed conifer tree species: effects of crown damage and forest floor combustion. *Forest Ecology and Management* 162: 261-271

Stephens SL, Collins BM. 2004. Fire regimes of mixed conifer forests in the north-central Sierra Nevada at multiple spatial scales. *Northwest Sci* 78: 12-23

Stephens SL, Finney MA, Schantz H. 2004. Bulk density and fuel loads of ponderosa pine and white fir forest floors: impacts of leaf morphology. *Northwest Sci* 78: 93-100

Sugg A. 1990. Seedling establishment results from a site preparation study in southern Sweden: The first four years survival and growth of Scots pine (*Pinus sylvestris* L.) and Norway spruce (*Picea abies* L. Karst). Swedish University of Agricultural Sciences, Department of Silviculture, Umea, Sweden

Sweeney JR, Biswell HH. 1961. Quantitative studies of the removal of litter and duff by fire under controlled conditions. *Ecology* 42: 572-575

Swetnam TW, Thompson MA, Sutherland EK. 1985. Using dendrochronology to measure radial growth of defoliated trees. Spruce Budworms Handbook. USDA Forest Service, Agriculture Handbook No. 639, pp 271

Sydes C, Grime JP. 1981. Effects of tree leaf litter on herbaceous vegetation in deciduous woodland. I. Field investigations. *J Ecol* 69: 237-248

Sykes JM, Horrill AD. 1981. Recovery of vegetation in a Caledonian pinewood after fire. *Transactions of the Botanical Society of Edinburg*. 43: 317-325

Tappeiner JC, McDonald PM. 1996. Regeneration of Sierra Nevada forests. In: Sierra Nevada Ecosystem Project: Final report to Congress, vol. III, chap. 12. Davis: Univ. Calif., Centers for Water and Wildland Resources

Tengelmark DO. 1998. Site factors as multivariate predictors of the success of natural regeneration in Scots pine forests. *For Ecol Manage* 109: 231-239

Thomas PA, Wein RW. 1985. The influence of shelter and hypothetical effect of fire severity on the postfire establishment of conifers from seed. *Canadian Journal of Forest Research* 15: 148-155

Thonicke K. 2003. Fire disturbance and vegetation dynamics - analysis and models. PhD-Diss, University of Potsdam, Germany, 96

Thonicke K, Cramer W. 2004. Long-term trends in vegetation dynamics and forest fires in Brandenburg (Germany) under a changing climate. *Natural Hazards* (in press)

Tolonen K. 1983. The post-glacial fire record. In: Wein, Ross W.; MacLean, David A., eds. The role of fire in northern circumpolar ecosystems. Scope 18. New York: John Wiley & Sons: 21-44

Tonn JR, Jurgensen MF, Mroz GD, Page-Dumroese DS. 2000. Miller Creek: ecosystem recovery in a western Montana forest 30 years after prescribed burning and wildfire. Pages 67-73 in W. K. Moser and C.F. Moser (eds.). Fire and forest ecology: innovative silviculture and vegetation management. Tall Timbers Fire Ecology Conference Proceedings, No. 21. Tall Timbers Research Station, Tallahassee, FL

Trabaud L. 1983. Prescribed fires and their application to a Quercus coccifera L. garrigue. In: DFG-Symposium "Feuerökologie", Freiburger Waldschutz-Abhandlungen, Band 4, 301

Turner MG, Romme RH, Gardner RH, Hargrove WW. 1997. Effects of patch size and fire pattern on early post-fire succession on the Yellowstone Plateau. *Ecol Monogr* 67: 411-433

Turner MG, Baker WL, Peterson CJ, Peet RK. 1998. Factors influencing succession: Lessons from large, infrequent natural disturbances. *Ecosystems* 1: 511-523

Ubysz B, Szczygiel R. 2002. Fire situation in Poland. *International forest fire news* 27: 38-64

Uggla E. 1959. Ecological effects of fire on north Swedish forests. Almqvist and Wiksells, Sweden

Ulanova NG. 2000. The effects of windthrow on forests at different spatial scale: a review. *For Ecol Manage* 135: 155-167

Van Dobben HF, Vocks MJ, Jansen E, Dirkse GM. 1994. Veranderingen in de ondergroei van het Nederlandse dennenbos over de periode 1985-1993. IBN-rapport 085, Institut voor Bos- en Natuuronderzoek, Wageningen, Netherlands

Van Lear DH. 2000. Recent advances in the silvicultural use of prescribed fire. Proceedings of the Tall Timbers Fire Ecology Conference 21: 183-189

Van Wagner CE. 1972. Duff consumption by fire in Eastern Pine Stands. *Can J For Res* 2: 34-39

Van Wagner CE. 1987. Development and structure of the Canadian forest fire weather index system. Canadian Forestry Service, Forestry Technical Report 35 (Petawawa National Forestry Institute, Chalk River, Ontario)

Van Wagtendonk JW, Benedict JM, Sydoriak W. 1996. Physical properties of woody fuel particles of Sierra Nevada conifers. *Inter. J. Wildland Fire* 6: 117-123

Van Wagtendonk JW, Sydoriak WM, Benedict JM. 1998a. Heat content variation of Sierra Nevada conifers. *International Journal of Wildland Fire* 8: 147-158

Van Wagtendonk, JW, Benedict JM, Sydoriak WS. 1998b. Fuel bed characteristics of Sierra Nevada conifers. *West J Appl For* 13: 73-84

Vanha-Majamaa L, Lahde E. 1991. Vegetation changes in a burned area planted by Pinus sylvestris in northern Finland. *Annales Botanici Fennici*, 28: 161-70

Vanninen P, Mäkelä A. 2000. Needle and stem wood production in Scots pine (*Pinus sylvestris*) trees of different age, size and competitive status. *Tree Physiology* 20: 527-533

Vega JA, Bara S, Mal del Carmen Gil. 1983. Prescribed Burning in pine stands for fire prevention in the N.W. of Spain: Some Results and Effects. In: DFG-Symposium "Feuerökologie", Freiburger Waldschutz-Abhandlungen, Band 4, 301

Vickers AD, Palmer SCF. 2000. The influence of canopy cover and other factors upon the regeneration of Scots pine and its associated ground flora within Glen Tanar National Nature Reserve. *Forestry* 73: 37-49

Viro PJ. 1969. Prescribed burning in forestry. Metsan tutkimuslaitoksen Julkaisuja. 67: 1-49

Viro PJ. 1974. Effects of forest fire on soil. In: Kozlowski, T. T.; Ahlgren, C. E., eds. Fire and ecosystems. New York: Academic Press: 7-45

Wade DD. 1993. Societal influences on prescribed burning. In: Hermann, S. (ed.), Proc. 18th Tall Timbers Fire Ecology Conf., Tall Timbers Res. Stat. pp. 351-355

Wagle RF, Eakle TW. 1979. A controlled burn reduces the impact of a subsequent wildfire in a ponderosa pine vegetation type. *Forest Science* 25:123-129

Wallstedt A, Nilsson M-C, Zackrisson O, Odham G. 2000. A link in the study of chemical interference exerted by Empetrum hermaphroditum: Quantification of Batatasin III in soil solution. *Journal of Chemical Ecology* 26: 1311-1323

Wardle DA, Zackrisson O, Nilsson M-C. 1998. The charcoal effect on boreal forests: mechanisms and ecological consequences. *Oecologia* 115: 419-426

Weber MG, Stocks BJ. 1998. Forest fires and sustainability in the Boreal Forests of Canada. *Ambio Vol* 27 No. 7: 545-550

Whelan MJ, Sanger LJ, Baker M, Anderson JM. 1998. Spatial patterns of throughfall and mineral ion deposition in a lowland Norway Spruce (*Picea abies*) plantation at the plot scale. *Atmospheric Environment* 32: 3493-3501

Whittle CA, Duchesne LC, Needham T. 1997. The impact of broadcast burning and fire severity on species composition and abundance of surface vegetation in a jack pine (*Pinus banksiana*) clear-cut. *Forest Ecology and Management* 94: 141-148

Wichmann I. 1996. Sukzession auf alten Waldbrandflächen im Forstamt Lüß. Diplomthesis University Göttingen, Germany. 92p

Wiedemann E. 1948. Die Kiefer - waldbauliche und ertragskundliche Untersuchungen. Verlag M&H Schaper, Hannover, Germany

Wiedemann E. 1949. Ertragstafeln der wichtigen Holzarten. Verlag M&H Schaper, Hannover, Germany

Wirth C, Schulze ED, Schulze W, Stünzer-Karbe D, Ziegler W, Jiljukova IM, Sogatchev A, Varlagin AB, Panvyorov M, Grigoriev S, Kusnetzova W, Siry M, Hardes G, Zimmermann R, Vygodskaya NN. 1999. Above-ground biomass and structure of pristine Siberian Scots pine forest as controlled by competition and fire. *Oecologia* 121: 66-80

Wirth C, Schulze ED, Kusznetova V, Milyukova I, Hardes G, Siry M, Schulze B, Vygodskaya NN. 2002. Comparing the influence of site quality, stand age, fire and climate on aboveground tree production in Siberian Scots pine forests. *Tree Physiology* 22: 537-552

Wittich W. 1955. Die standörtlichen Bedingungen für die natürliche Verjüngung der Kiefer und für ihre Erziehung unter Schirm. *Allg. Forst- u. Jagdztg.* 126: 109-117

Wittich KP. 1998a. 'Waldbrandgefahrenvorhersage, Teil 1: Streufeuchtemodell.' Beiträge zur Agrarmeteorologie (DWD intern, Offenbach am Main)

Wittich K-P. 1998b. Waldbrandgefahrenvorhersage des Deutschen Wetterdienstes. Allg. Forst Z. - *Der Wald* 53: 321-324

Wohlgemuth T, Bürgi M, Scheidegger C, Schütz M. 2002. Dominance reduction of species through disturbance - a proposed management principle for central European forests. *For Ecol Manage* 166: 1-15

Wright HA, Bailey AW. 1982. Chapter 5, Grasslands, pp. 80-137 In: Fire Ecology - United States and Canada. John Wiley, New York

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

Zackrisson O. 1980. Forest fire history: ecological significance and dating problems in the north Swedish boreal forest. In: Stokes, Marvin A.; Dieterich, John H., technical coordinators. Proceedings of the fire history workshop; 1980 October

20-24; Tucson, AZ. Gen. Tech. Rep. RM-81. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 120-125

Zackrisson O, Nilsson MC. 1992. Allelopathic effects by *Empetrum hermaphroditum* on seed germination of two boreal tree species. - *Can J For Res* 22: 1210-1319

Zackrisson O, Nilsson MC, Wardle DA. 1996. Key ecological function of charcoal from wildfire in the Boreal forest. *Oikos* 77: 10-19

Zackrisson O, Nilsson MC, Dahlberg A, Jaederlund A. 1997a. Interference mechanisms in Conifer-Ericaceae-feathermoss communities. *Oikos* 78: 209-220

Zackrisson O, Norberg G, Dolling A, Nilsson MC, Jäderlund A. 1997b. Site preparation by steam treatment: effects on forest vegetation control and establishment, nutrition, and growth of seeded Scots pine. - *Can J For Res* 27: 315-322

Zasada JC, Norum RA, Van Velhuizen RM, Teutsch CE. 1983. Artificial regeneration of trees and tall shrubs in experimentally burned upland black spruce/feather moss stands in Alaska. *Canadian Journal of Forest Research* 13: 903-913

Zerbe S, Brande A. 2003. Woodland degradation and regeneration in Central Europe during the last 1000 years - a case study in NE Germany. *Phytocoenologica* 33: 683-700

Ziemer RR. 1968. Soil moisture depletion patterns around scattered trees. USDA Forest Service RN-PSW-166

Zinke PJ. 1962. The pattern of influence of individual forest trees on soil properties. *Ecology* 43: 130-133

Acknowledgements

This dissertation was completed after Marco Hille passed away in the aftermath of an accident on December 4, 2004. His dissertation shows his scientific commitment and his professional knowledge and skills in the implementation of his talents.

All who knew Marco appreciated his frankness and warmth, as well as his view of important things in life. He motivated many people with whom he worked, and fought untiringly for his point of view when he was convinced of it. His family often feared for their son, brother, grandson, nephew and cousin during his numerous travels and adventures around the world and yet they lost him only a few hundred meters away from his parents' house. Therefore our first thanks, as best as possible in Marco's name, go to his family who gave him incredible support and trust, which allowed him to study and work as he chose. A very special tribute to his parents, Herbert and Monika Hille.

I also want to include Kathy Doyle in connection to his family. Even though she is not a blood relative, Marco regarded her as his 'American mum'.

Saana Järvenpää Deichsel gave Marco a lot of zest for life. She motivated him even more in his work and in their time together. On behalf of Marco we thank Saana for her love and support. Thanks also to Saana for editing several parts of text in this thesis.

"Burn to live, live to burn" describes Marco's motto towards living life to the fullest. In Germany, Marco was one of the few advocating the use of controlled fires to protect and preserve important natural resources in our present time. The responsibility and energy with which Marco took on this quest demonstrates Marco's compassion for the environment and for forestry. For the encouragement and support of this work in the field we thank the colleagues from 'Bundesforstamt Lausitz" in Brandenburg, especially Mr. Noack and Mr. Brunn who, in Marco's words "started a new movement for the use of prescribed fires in Central Europe."

Marco's work would not have been possible without his experience in the USA, through his educational year and subsequent stays at the University of California, Berkeley, USA. Marco himself said that "Professor Scott Stephens from the University of California awoke the fire passion in me." Through the successful cooperation with the Stephen's lab in Berkley, a close friendship developed. In the field, Jason Moghaddas, Domenico, and Matt were Marco's compadres in the Sierra Nevada, California, and in all the campfire talks in Mexico and Bridgeport.

The Global Fire Monitoring Centre of the Max Planck Institute for Chemistry in Freiburg, under the guidance of Professor Goldammer, gave invaluable support to

Marco's introduction to forest fire research and monitoring in Europe. Without this connection, it would not have been possible for Marco to establish himself in this field of research. Many thanks to everyone at the Max Planck Institute, and at the University of Freiburg who worked together with Marco and helped him to succeed.
Marco found his scientific home at Wageningen University, in the Forest Ecology and Forest Management Group under the leadership of Professor Mohren. Here he was offered infrastructure and a working environment with colleagues who were always willing to listen to him and discuss his scientific questions. Despite his numerous stays in foreign countries he was always fully integrated with the team, so that he was able to use the group's contacts and reputation. A special mention goes to Frits Mohren, Jan den Ouden and Joke Jansen-Klijn.

Finally, we wish to thank the Robert Bosch Stiftung for funding this research project.

Thank you very much for everyone who helped and worked together with Marco on his research. Keep the fire burning!!! And, as Marco ended his first draft of these acknowledgements: "It's been a good time."

Lars Schmidt MSc
Institute of Forest Management (IWW)
Faculty of Forest Sciences and Forest Ecology
Georg-August-University Göttingen, Germany

Curriculum vitae

Marco Hille was born on the 29th of March 1976 in Rheine (Westfalen), Germany. His love of nature was inspired by the family move to the countryside and his father's passion for hunting. After graduating from high school at the Episcopal Fürstenberg Gymnasium in Recke in 1995, he was drafted into the military service for one year. In October 1996 he started to study forestry at the Georg-August University Göttingen. After internships and a research project in South Africa, from July to November 1998, he wrote his Master thesis on 'Endsplitting in *Eucalyptus grandis* sawlogs', supervised by professor von Gadow, at the Institute of Forest Management. Afterwards he was employed there as a research assistant. From August 1999 to May 2000 he received a post-master scholarship from Göttingen University and the University of California, in Berkeley. There, he took courses in forest operation management, forestry computing and leadership studies. At the University of California he also was employed as an assistant at the College of Natural Resources and worked on forestry biometrics and remote sensing. During this time, he was introduced to controlled burning of forests, which increasingly became the focus of his studies. Marco became so excited about the subject that he wrote his own research proposal about controlled burning in Europe in order to apply for a PhD position by the Robert Bosch foundation at the University of Wageningen, the Netherlands. He was very happy when he was invited to start this project in the summer of 2001. His knowledge of fire ecology was then deepened when he worked with professor Goldammer in Freiburg at the Global Fire Monitoring Centre and by conducting his own experiments on the burning of heathland. He became a passionate scientist, always on the road between Wageningen, Berkeley, Freiburg, Schale, and his beloved field-experiments in the forests of East Germany, in Gandenitz. He did all of this while also presenting at conferences and symposiums in Greece, Italy, Spain and Germany and field trips to the Sierras in California and to Mexico.
This promising, hopeful career as a scientist ended on December 4, 2004 in an incomprehensible way. While cutting down trees, he was so severely injured by a falling tree he had cut himself, that attempts to save his life proved in vain the same day.

His parents, Monika and Herbert Hille
and his sister Katrin
Schale, Germany