Factors Influencing Occurrence and Impacts of Fires in Northern European Forests

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Päätalo, M-L. 1998. Factors influencing occurrence and impacts of fires in northern European forests. Silva Fennica 32(2): 185–202.

The return interval and number of fires vary, depending on the geographical location in interaction with climate, topography and amount of fuel. During recent decades, in northern Europe the number and severity of fires have been insignificant compared with Mediterranean region, in which fire return intervals may be 15–35 years, compared to the average of 60–120 years for boreal forests. This is partly due to the efficient system of fire protection in northern Europe, but is mainly due to the less favourable climate for fire and the smaller human impact on ignition of forest fires.

The consequences of fire are related to both site and stand characteristics, site being the most important factor controlling the stability of stands. Dry sites being more flammable and likely to ignite are associated with high risk of fire. In northern Europe, due to the interaction between species and site, the role of species difference in risk of fire damage is not clear. In southern Europe, fire risk cannot be explained by differences between tree species. There, other vegetation (shrubs, etc.) is of major importance for the risk of fire.

Management of forests can, to some degree, alter the risk and the occurrence of fire. In northern Europe, logging may have compensated for fire occurrence by decreasing the amount of fuel. In addition, forest roads act as fire-breaks and facilitate fire-fighting. On the contrary, in southern Europe the risk of fire has been found to increase because the traditional forest uses and management have decreased, which increases the accumulation of fuel. However, it is not yet possible to quantify and compare the effect of management in absolute terms.

Currently, some tools, such as fire-risk indices, remote sensing and GIS-based techniques, are available for prediction of fire risk in some areas. For example, fire-risk indices are most suitable for areas, like northern Europe, which have a low fire risk. In high-risk areas, such as southern Europe, more sophisticated techniques are needed for assessment of the risk. In the future, assuming global warming at northern latitudes ($2 \times CO_2$ climate), the risk of fire damage could also increase in northern Europe. Therefore, to allow the various locational and silvicultural factors to be assessed on the European level, an integrated risk model is needed.

Keywords fire, climate, site, tree species, risk assessment
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Received 13 June 1997 **Accepted** 11 May 1998

1 Introduction

When a forest fire occurs, it changes the physical, chemical and biological conditions of forests, especially in southern Europe (Goldammer and Jenkins 1990, Trabaud et al. 1993, Wein 1993). In boreal forests, in particular, forest fires also have an important effect on forest succession by regularly controlling forest dynamics (Heinselman 1981, Barney and Stocks 1983, Chandler et al. 1983, Wein and MacLean 1983, Engelmark 1984, Antonovski et al. 1992, Goldammer and Furyaev 1996). Fires affect the species composition and age-class distribution of stands (Zackrisson 1977, Barney and Stocks 1983, Zackrisson and Östlund 1991). Often recurring fires favour shade-intolerant Scots pines (*Pinus sylvestris* L.) and birches (*Betula* spp.), while reduction of fire occurrence favours development of stands dominated by Norway spruce (*Picea abies* (L.) Karst.) (Viro 1974, Bradshaw 1993, Sepponen 1989, Oliver and Larson 1990, Granström 1991a).

In northern Europe, especially after small fires, the mosaic of ecosystems is developed and biological diversity increases (large fires, on the other hand, may unify landscapes) (Zackrisson 1977, Zackrisson and Östlund 1991, Binkley et al. 1993). In addition, forest fire may occur at an inconvenient time in the rotation, compensating for logging; thus an increase in the occurrence of forest fires may result in an unfavourable age-class distribution and lead to lack of a desirable timber assortment (Suffling 1992). On the other hand, fire (if it recurs often enough) may interrupt the post-fire successions, which may lead to a degenerative phase (Sirén 1955, Zackrisson 1977).

Forest fires affect soil conditions (temperature, dryness, ash toxicity, humus layer and frosts) and may cause a lack of parent trees and low seed production after year of the fire, which may also endanger regeneration (Kohh 1975, Granström 1991a, Sirois 1993). On the other hand, intensive fire increases the near-ground air temperature, which may in turn lengthen the growing season, releases nutrients, making them available for uptake by trees and other vegetation, and decreases acidity and the organic layer (Viro 1974, Kohh 1975, Oliver and Larson 1990). In southern Europe, typical ground or surface fires also affect soil erosion and nutrient losses, which may in turn endanger growth and recovery of trees and other vegetation (Goldammer and Jenkins 1990, Vasconcelos 1995). In these conditions, large fires, which are typical for southern Europe, may unify landscapes and decrease biological diversity (Binkley et al. 1993).

In Europe, fire damage has affected nearly 8 million hectares of forests in an estimated 660 000 forest fires between 1980 and 1992 (Table 1), the average size of the burned area being 12 hectares. Depending on geographical location, there are significant differences in the scale and frequency of fire damage (Fig. 1, 2, Table 1). In Spain in a single year (1990) wildland fires (i.e. not only forest fires) damaged over 2 000 000 hectares of land and cost almost 680 million US dollars. In

Table 1. Forest fires in some European countries and in Europe during 1980–1992 with respect to wooded areas (Forest fire statistics... 1993, Europe's environment. Statistical... 1995).

Region	Number of fires			Burned area (km^2)		
	Total period	Yearly max	Yearly min	Total period	Yearly max	Yearly min
Finland	6556	852	171	51.9	10.8	1.0
Norway	7069	975	286	110.6	28.5	0.9
Germany	19639	2111	1071	133.6	21.6	7.1
UK	7212	1288	98	63.1	17.3	0.6
Portugal	131578	23251	2349	10905.6	1618.5	224.4
Spain	138883	20384	4880	29948.2	4863.3	970.7
Europe	664446	81907	34790	77459.6	10679.7	3970.7

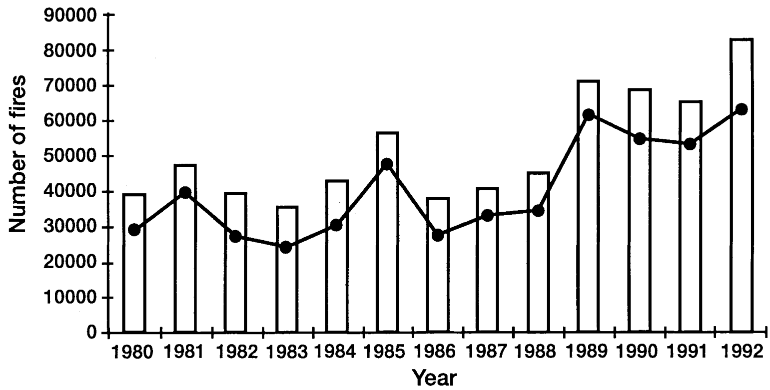

Fig. 1. Number of fires in Europe between 1980 and 1992. Columns show the number of fires in the whole of Europe and the line indicates the number of fires in the Mediterranean region (i.e. in Albania, Bulgaria, Cyprus, France, Greece, Italy, Portugal, Spain and former Yugoslavia).

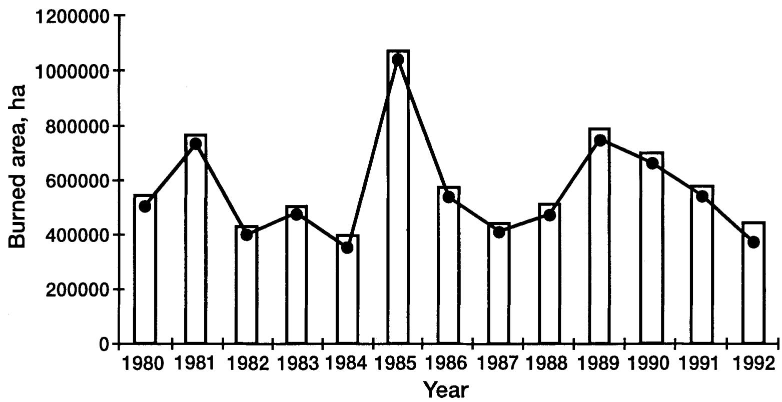

Fig. 2. Area burned in fires between 1980 and 1992 in Europe (hectares). Columns represent the area burned in the whole of Europe and the line represents the area burned in the Mediterranean region.

Italy, Portugal, Greece and the Mediterranean parts of France, forest fires are also a serious problem. However, it is difficult to find any clear trend from year to year (Forest fire statistics... 1993). It has been suggested that, due mainly to human activities, in the future, problems linked with wildfires will increase in the Mediterranean region (Goldammer and Jenkins 1990, Stocks and Trollope 1993). There erosion is a serious consequence of fires, increasing both ecological and economic losses. In such areas, forest management for soil protection should be one of the highest priorities (Stanners and Bourdeau 1995).

In northern Europe, forest fires are now rare (Chandler et al. 1983, Engelmark 1984). In Finland, due to efficient forest management, the annual number of fires is about 400–500; and only 0.002–0.004 % of the forest burns each year (Suffling 1992, Aarne 1993, Forest fire statistics... 1993, Parviainen 1996). In Norway and Sweden the average areas of national forest land burned each year are 0.05 and 0.02 %, respectively; and burned areas have increased as a result of human impacts (Suffling 1992). On the whole, in northern European countries the mean size of fires is nowadays less than one hectare (Zackrisson and Östlund 1991, Forest fire statistics... 1993, Granström 1993). Even so, fires are not as frequent in central European temperate forests as in northern Europe (Chandler et al. 1983, Forest fire statistics... 1993). In the future, however, with elevated temperature the risk of fire may increase in northern Europe even in boreal forests, because summers may become longer and drier (Flannigan and Van Wagner 1991, Woodward and Diament 1991, Hogenbirk and Sarrazin-Delay 1995). This review summarises the literature available on factors affecting fire occurrence and ecological impact, with special attention to northern Europe and comparison with southern Europe. In addition, the tools available for reducing the risk of fire damage are discussed and the need for further work is identified.

2 Effects of Climatological Factors on Forest Fires

2.1 Effects of Weather and Climatological Factors

In the long run, *climate* determines the growth of forests and thus the amount of fuel they contain. In warmer regions, such as around the Mediterranean, growth is rapid, which leads to more fuel production than in colder regions, although drought may restrict growth (Chandler et al. 1983). At higher latitudes, litter decomposes more slowly, and thus large amounts of fuel accumulate on the ground (Oliver and Larson 1990, Wein 1993). Chandler et al. (1983), however, have claimed that decay may also be extremely slow in Mediterranean climates where rainfall is largely confined to the winter months. On the other hand, in both boreal and Mediterranean region conditions, climate determines the length and severity of the fire season (Chandler et al. 1983) as well as fire return interval in interaction with fuel accumulation and soil-site conditions.

Weather over a short time-scale, especially temperature, precipitation and wind speed, control fire behaviour and intensity due to its effects on fuel flammability in various soil-site conditions. Weather determines such fuel properties as moisture and the ability to ignite and burn (Fig. 3) (Chandler et al. 1983, Schimmel and Granström 1991). For example, in some Canadian boreal forests, critical fire weather is associated with a characteristic persistent 50 kPa longwave ridge and its breakdown. This seems to cause a sequence of fuel drying and ignition plus wind to produce favourable conditions for the occurrence of fire (Johnson 1992).

Temperature controls fuel moisture and thus fuel ignition as well as the rate at which fuel burns (Chandler et al. 1983, Johnson 1992). In summer, heating is usually greatest at midday. In northern Europe, the long summer days may add to the fire risk by increasing the temperature (Wein and MacLean 1983). Consequently, the levels of biomass moisture drop, allowing fire to spread rapidly (Wein 1993). Fires are most evident when the summer air temperature is above the mean summer temperature (Zackrisson 1977, Chandler et al. 1983). Therefore, climate change is a risk. A temperature increase of 3–5 °C may increase the fire area by 15 to 50 times (Suffling 1992). In addition, above-normal July temperatures seem to have a significant effect on occurrence of forest fires (Franssila 1959, Engelmark 1984). In northern Europe, on average, weather conditions are less favourable for fire to ignite and spread than in the Mediterranean region where the summers are long, hot (30 °C on average) and dry (Naveh 1975, Tárrega and Luis-Calabuig 1990, Stanners and Bourdeau 1995).

In northern European boreal forests, which represent low average annual *precipitation* (250–600 mm), fire will probably ignite only if there is enough dry fuel (Chandler et al. 1983, Granström 1991b). Thus, low *air humidity*, which is affected by both precipitation and temperature, increases the risk of fire (Saari 1923, Chandler et al. 1983). According to Van Wagner (1983), to increase the probability of fire in northern forests, rainless periods of at least one or two weeks are needed.

In the Mediterranean area the total annual precipitation varies considerably from year to year, ranging from 500 to 1500 mm. Furthermore, rainfall is common in winter, whereas summers

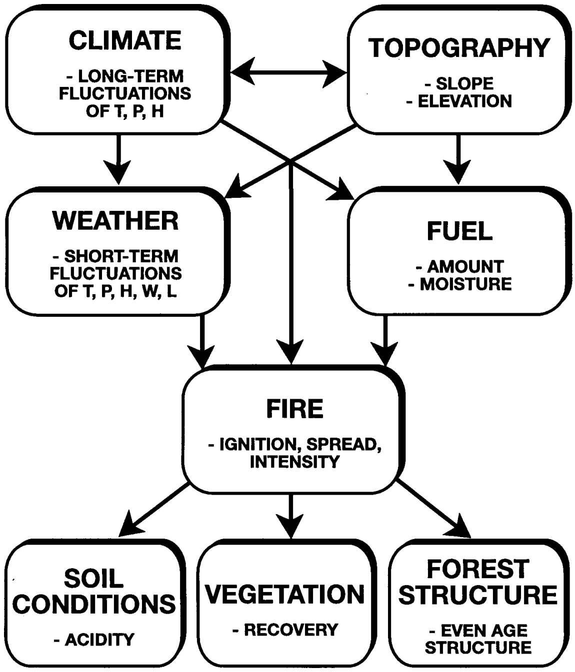

Fig. 3. Factors affecting fire risk and the consequences of fire occurrence (as examples). Key to the letters used in the diagram: T = temperature, P = precipitation, H = air humidity, W = wind, L = lightning.

are dry (average air humidity 50–60 %) with high fire risk from May to October (Naveh 1975, Chandler et al. 1983). On the other hand, even a few millimetres of light rainfall is enough to saturate litter and temporarily lower the fire risk (Chandler et al. 1983).

Drying high-speed *winds* (e.g. the foen winds in the Mediterranean area) make the vegetation more flammable. In addition, wind maintains and increases combustion by constantly transporting new air to the fire (Chandler et al. 1983). Furthermore, it very strongly influences the spread of fire once a fire has been ignited (Franssila 1959, Chandler et al. 1983, Oliver and Larson 1990, Johnson 1992). Wind speed determines how rapidly fire is spread (Franssila 1959), i.e. the rate of spread will double for each 4 m/s increase in wind speed (Chandler et al. 1983).

According to Granström (1991a), wind speed also determines how far from ground level flames and smoke will rise. However, the stronger the wind speed and the lower the temperature (at that height), the longer it takes for fire and smoke to reach a certain height. In intensive fires, damage can occur higher up (Granström 1991a). Furthermore, wind can carry burning embers to ignite spot fires ahead of the main fire. The direction of fire spread is determined by wind direction (Chandler et al. 1983).

Weather generates the principle source of ignition for natural fires, i.e. *lightning* (Johnson 1992). Before human activities, forest fires were caused by lightning (Zackrisson 1977, Haapanen and Siitonen 1978, Heinselman 1981, Barney and Stocks 1983, Engelmark 1984, Zackrisson and Östlund 1991). Nowadays, in northern Europe 5–20 % of all forest fires are ignited by lightning (Suffling 1992). In Finland, 20 % of all fires are ignited by lightning (Aarne 1993); in Sweden, the corresponding percentage is lower

(Barney and Stocks 1983, Granström 1993). In the Mediterranean region, depending on the country, lightning is estimated to account for only 0–6 % (ca. 2 % on average) of the area burned annually (Naveh 1975, Garty 1990, Trabaud et al. 1993).

Occurrence of lightning is essentially dependent on weather (Franssila 1959). In northern Europe, dry high-pressure thunderstorms are important for the occurrence of forest fires (Zackrisson 1977, Engelmark 1984, Granström 1991b, Granström 1993). Lightning ignitions peak in early July (Granström 1993). However, lightning ignition cannot be translated into fire frequency (Granström 1993), because the rain that accompanies thunderstorms may extinguish fires naturally (Wein and Moore 1977, Van Wagner 1983). The occurrence of lightning depends on the region, as well as on the specific location within a single country (Zackrisson 1977, Engelmark 1984, Granström 1991b, Johnson 1992, Granström 1993).

In addition to temperature, the occurrence of lightning and thunderstorms is related to differences in landforms and vegetation (Franssila 1959, Heinselman 1981). Dead standing trees are the most frequent objects of lightning strikes (Kourtz 1967). Fires ignited by lightning cause the most problems in remote areas, where fire-fighting is difficult. In many cases, fires caused by lightning are larger than those caused by humans (Van Wagner 1983, Suffling 1992, Fosberg et al. 1993).

2.2 Topographical Factors

The *general topography* of the landscape and vegetation act as fire paths or, alternatively, as fire-breaks (e.g. Engelmark 1987, Karjalainen 1994). In northern latitudes, convex landscape features burn more frequently than those with concave features (Zackrisson 1977). However, mosaics of forests, peatlands and lakes act as natural fire-breaks and are typical for northern European boreal forests, differing from those in Canada and Siberia (Granström 1993). Artificial fire-breaks, like forest roads, also prevent the spread of fire (e.g. Zackrisson and Östlund 1991). Unlike northern Europe, the uneven topography of the Mediterranean region causes fires to spread rapidly, also making fire-fighting difficult (Chandler et al. 1983).

In boreal forests near the arctic hills of northern Europe, fire has been of only minor importance (Zackrisson and Östlund 1991). On the other hand, outside of the arctic-hill area, hills are isolated and there the fire incidence changes with the *height above sea level*. About 100–200 meters below the forest limit is the turning point for stands that have never burned, i.e. below that point are stands that are strongly affected by fire (Zackrisson and Östlund 1991). At higher elevations, Norway spruce is more abundant; and in lower stands, Scots pine dominates (Engelmark 1987).

Furthermore, due to differences in *latitude*, topography and proximity to large lakes, etc., the fire climate differs in severity and occurrence throughout the year. In continental areas, the fire season is usually quite severe, relatively short and peaks during the summer (Chandler et al. 1983). This is partly due to the long summer days at high latitudes (Wein and MacLean 1983). Closer to the equator, the fire season tends to be longer (Chandler et al. 1983).

Two extremes, *north-facing and south-facing slopes*, have been assumed to play an important role in the frequency of forest fires in the boreal forest zone (Zackrisson 1977, Goldammer and Di 1990). Warmer, south-facing slopes are, in general, more frequently affected by fire than north-facing slopes (as much as three times more often), because on southern slopes the conditions are more susceptible to ignition and spread of fire (Zackrisson 1977). On the other hand, more frequent agriculture may also increase the fire risk on south-facing slopes (Zackrisson 1977). Engelmark (1987), for example, found no difference between northern and southern slopes. However, due to slower decomposition of litter on northern slopes, accumulation of fuel is also greater; and in suitable conditions, fire can be severe (Artsybashev 1985).

On the whole, the likelihood of lightning strikes is greatest on exposed ridges, summits, slopes and other convex surfaces (Engelmark 1987). The steeper the slope, the faster the fire spreads uphill. Of course, the spread on a given slope is also dependent on the type of fuel available there

(Chandler et al. 1983, Oliver and Larson 1990). For example, in northern Sweden during the last 600 years most fires have occurred on steep slopes, on continuous uplands and at low altitudes (Engelmark 1987).

3 Site and Tree Effects on Forest Fires

3.1 Site

In certain geographical locations, the severity of fire damage is related to the site, fire being more frequent on drier sites than on moister ones (Keeley 1981, Angelstam and Rosenberg 1993). Depending on soil and climate conditions, the return intervals of fires occurring in northern Europe may vary from one fire each 40 years to one in 450 years, compared to Mediterranean ecosystems, where fire cycles may be as short as 15–35 years (Zackrisson 1977, Chandler et al. 1983, Engelmark 1984, Hyvärinen and Sepponen 1988) (Table 2). Where fire cycles are shorter, fires are not usually serious (Chandler et al. 1983). Many species of the Mediterranean vegetation types are adapted to repeated fires, and there recovery occurs quickly, due mainly to vegetative survival organs (Trabaud and Chanterac 1985, Trabaud 1987, Trabaud 1990). However, ecosystems in the Mediterranean region are very complicated and different from those in northern Europe; and fire occurrence and impacts vary considerably. Forests are scarce and are often situated in mountains, because many forests were long ago destroyed by humans and replaced by dense shrublands with a few scattered evergreen trees (Chandler et al. 1983, Trabaud et al. 1993).

Table 2. Examples of return interval of fires in Europe.

Region	Vegetation type	Return interval of fires, years
Northern Europe	All	40–160 [10]
	Lichen-*Calluna* type	52 ± 26 [1]
	Vaccinium vitis-idaea type	58 ± 40 [1]
	Vaccinium myrtillus type	91 ± 47 [1]
	Vaccinium uligonosum–Aconitum septentronale type	160 ± 84 [1]
	Lichen-dwarf shrubs and *Vaccinium vitis-idaea* types	110 [5]
	Vaccinium vitis-idaea–Vaccinium myrtillus type	60–100 [7]
	Moist herb-rich site	> 100 [7]
	Herb-rich swamps	Evidently do not burn [7]
	Sub-alpine birch stands	Do not burn [7]
	Hylocomium–Vaccinium myrtillus type	450 [6, 8]
	Empetrum–Vaccinium myrtillus type	111 (40–235) [2]
	Empetrum–Calluna type	112 (44–183) [2]
	Vaccinium vitis-idaea–Vaccinium myrtillus type	128 (18–372) [2]
	Hylocomium–Vaccinium myrtillus	119 (25–260) [2]
	Spruce-dominated types	238 ± 48 [3, 8]
Central Europe	Mountain forests	6000 [4]
	Heaths	5–15 [4, 11, 12]
Southern Europe	All	15–35 [4]
	Garrigue	5–7 [9]
	Maquis	10–15 [9]
	Forest	30–50 [9]

References used in table: 1) Zackrisson 1977, 2) Haapanen and Siitonen 1978, 3) Tolonen 1978, 4) Chandler et al. 1983, 5) Engelmark 1984, 6) Hyvärinen and Sepponen 1988, 7) Zackrisson and Östlund 1991, 8) Kolström and Kellomäki 1993, 9) Trabaud et al. 1993, 10) Hörnsten et al. 1995, 11) Aldhous and Scott, 12) Hobbs and Gimingham 1984.

Fuel type, continuity, amount and quality as well as moisture and other factors constitute important local and regional controls for fire regimes on various sites (Holling 1981, Van Wagner 1983, Schimmel and Granström 1991, Campbell and Flannigan 1996). Fuels consist of living and dead plant materials above the mineral soil (Chandler et al. 1983, Van Wagner 1983). Even with the same type of fuel, the rate of spread will double if the fuel loading doubles, although the rate of spread also depends on fuel size (Chandler et al. 1983, Van Wagner 1983, Schimmel and Granström 1991). In fine-textured fuels, doubling of the fuel loading triples the rate of spread; and in large-textured fuels, spread is affected only slightly by loading (Chandler et al. 1983). Duff with moistures below 30 % dry weight burns unassisted after ignition, while duff with moisture contents above 140 % generally do not burn at all (Johnson 1992). In general, fine-textured fuels do not burn if the moisture content exceeds 15 % (Chandler et al. 1983), and litter burns if the moisture content is below 25–30 % (Van Wagner 1983). Thus, the intensity of the fire *regime* can differ, not only between sites but also on different parts of the same site (Schimmel and Granström 1991).

In northern Europe, dry forest sites, such as *Cladina* and *Calluna* sites, where Scots pine is a dominant tree species, burn easily (Zackrisson 1977, Zackrisson and Östlund 1991) and even seem to need repeated fires for maintenance (Engelmark 1987). Fire damage may result in uneven-aged forest structure (Chandler et al. 1983, Engelmark 1987). On average, on the driest sites, fires occur in 40–65 year cycles because complete fuel accumulation and lichen recovery is needed to support the next fire (Kohh 1975, Engelmark 1987). On these sites, *Cladonia* spp. lichens and *Pleurozium* and *Hylocomium* mosses are typical flammable fuels. When they cover 30–40 % of the site, a fire is possible. *Cladonia* lichens may burn well with about 40 % moisture and *Pleurozium* and *Hylocomium* mosses with 20 % moisture, whereas other mosses do not burn in even dryer conditions (Schimmel and Granström 1991). The consistency of moss and lichen species, together with litter properties, also determines fire intensity (Schimmel and Granström 1991). However, there are also arctic hill *Cladina* sites where there has never been a fire (Zackrisson 1977, Zackrisson and Östlund 1991).

On *Vaccinium*-sites (”dryish” forest sites) dominated by Scots pine, fires have occurred at intervals of 80–120 years (Kohh 1975, Engelmark 1984, Schimmel and Granström 1991) because the amount of well-burning mosses is at its maximum about 100 years after a fire (Table 3). Typical surface fuels on these sites are dwarf shrubs (e.g. *Empetrum* and *Vaccinium* species and *Calluna vulgaris*), which burn well (Engelmark 1987, Schimmel and Granström 1991); but these species differ from each other in their susceptibility to fire and their recovery afterwards (Table 3). Species like *Calluna vulgaris* are very flammable, whereas *Vaccinium* species may survive (Kohh 1975, Schimmel and Granström 1991). Thus, forest fire clearly represents one of the selective factors that influences development of the vegetation within different forest habitats (Zackrisson 1977).

On moist sites, i.e. *Vaccinium myrtillus* sites (often dominated by Norway spruce), the fire rotation is assumed to be about 100 years (Zackrisson 1977). Even longer rotations have been suggested; for example, in northern Finland the intervals between fires have been as long as 450–600 years (Siren 1955, Hyvärinen and Sepponen 1988). On these sites, herbs and grasses dominate the surface layer and may, due to their high moisture content, even prevent the spread of fire (Schimmel and Granström 1991). Where fire has not occurred for a long time, an accumulation of inflammable plant debris and fuels is built up, which increases the risk of forest fire (Engelmark 1984). Because of shallow roots, species like *Deschampsia flexuosa* suffer much of intensive, deeply burning fires in the organic layer. On the other hand, species with coppicing ability can survive fires (Kohh 1975). After fires, however, many pioneer species on the burned site, such as *Rubus idaeaus, Epilobium angustifolium* and the so-called fire mosses, recover rapidly (Yli-Vakkuri 1961, Granström 1991b). This may be due to advantageous competition and an increase in pH, which favour plants with the ability to fix nitrogen.

Bog forests are those forests that are least likely to burn (Schimmel and Granström 1991, Zackrisson and Östlund 1991), but alpine birch for-

Table 3. Response of vegetation to fire in northern Europe.

Layer	Species or genus	Recovery time	References
Ground layer	*Cladonia* spp.	> 20–40 years	Kohh 1975, Schimmel and Granström 1991
	Hylocomium splendens	> 10–20 years (100 years)	Kohh 1975, Schimmel and Granström 1991
	Pleurozium schreberi	> 10–20 years (100 years)	Kohh 1975, Schimmel and Granström 1991, Engelmark 1987
Field layer	*Senacio vulgaris*	Rapid, 1–3 years	Granström 1991b
	Carex pilulifera	Rapid, 1–3 years	ibid.
	Moehringia trinervia	Rapid, 1–3 years	ibid.
	Galeopsis tetrahit	Rapid, 1–3 years	ibid.
	Vicia spp.	Rapid, 1–3 years	ibid.
	Epilobium angustifolia	Rapid, 1–3 years	ibid.
	Geranium spp.	Rapid, 1–3 years	ibid.
	Calluna vulgaris	Slow, 10–60 years	Schimmel and Granström 1991
	Vaccinium myrtillus	Rapid, 2–50 years	Schimmel and Granström 1991, Sepponen 1989
	Vaccinium vitis-idaea	Rapid 2–50 years	Schimmel and Granström 1991, Sepponen 1989
Shrub layer	*Rubus idaeaus*	Rapid, 2–3 years	Kohh 1975, Schimmel and Granström 1991

ests are also seldom affected by fires (Zackrisson 1977, Zackrisson and Östlund 1991). According to Engelmark (1987), rather than being victims of fire, peatlands function as fire-breaks. Above the timber line, in the tundra, fires are rare but can be devastating because it may take more than a century for a burned area to recover (Chandler et al. 1983).

3.2 Stand Characteristics

Northern European forests consist mainly of Scots pine, Norway spruce and birch. The number of tree species in the Mediterranean region is much greater; and many of these are evergreen, bearing small, hard, thick and tough leaves (i.e. sclerophyllous). Because of their adaptation to dry climates, the moisture content of these trees is low and thus they are highly flammable (e.g. *Pinus* spp. and *Eucalyptus* spp.) (Chandler et al. 1983, Trabaud et al. 1985, Castro et al. 1990). Mediterranean tree species have several adaptations that increase their resistance, e.g. thick bark and strong root systems (Chandler et al. 1983). Some species are able to recover due to underground organs (Naveh 1975, Chandler et al. 1983, Trabaud 1990), while others are able to sprout or seed after a fire (Trabaud 1987, Casal et al. 1990). In addition, *Pinus halepensis* and *P. brutia* have serotinous cones, i.e. the cones remain closed unless the heat produced by fire opens them (Trabaud and Chanterac 1985).

However, northern tree species (Scots pine, Norway spruce and birch species) also differ in their susceptibility to fire damage (e.g. Haapanen and Siitonen 1978, Kolström and Kellomäki 1993). Scots pine is considered to be the most fire-resistant tree species in European boreal forests (see Table 4). In northern Sweden 500- to 600-year-old Scots pines have been found that have survived several fires due to their thick and heat-insulating bark (Zackrisson 1977). Scots pine with its higher living crown, survives better than Norway spruce, because it is not as susceptible to surface fires (Zackrisson 1977, Granström 1991a, Kolström and Kellomäki 1993). Furthermore, during a fire the deep root system

Table 4. Effects of the characteristics of some tree species on risk of fire damage.

Characteristic	Risk	Species example
Bark		
Thin	High	Norway spruce (Birch spp.)
Thick	Low	Scots pine
Heat insulating	Low	Scots pine
Crown depth		
Long	High	Norway spruce
Short	Low	Scots pine (Birch spp.)
Root system		
Shallow	High	Norway spruce (Birch spp.)
Deep	Low	Scots pine
Age		
Young	High	All species
Mature	-	-
Overmature	-	-

of Scots pine is often damaged only slightly (Zackrisson 1977). In addition, Scots pine has some ability to repair fire damage with resin and by lateral growth of new wood, which also increase resistance to pathogens and insects (Zackrisson 1977, Engelmark 1984, Gref and Ericsson 1985, Engelmark 1987, Granström 1991a). On dry sites, however, Scots pine is susceptible to burning due to site conditions (Saari 1923, Zackrisson 1977). Furthermore, young Scots pines have been found to be liable to fire damage because of their thin bark (Kolström and Kellomäki 1993). Due to its thin bark, deep crown and shallow root system, Norway spruce is less resistant to fire damage (Ryan and Reinhardt 1988, Oliver and Larson 1990, Granström 1991a). In general, deciduous tree species, like birch spp., are not as flammable as conifers, because their foliage is not capable of supporting a crown fire (Saari 1923, Heinselman 1981, Van Wagner 1983, Parviainen 1996). In addition, the moisture of deciduous leaves is usually more than 150 % of the dry weight, whereas the moisture of coniferous needles is less than 100 % of the dry weight (Johnson 1992).

According to Van Wagner (1983), conifer stands are more prone to crown fires at a young age, the most intense crown fires occurring when trees are less than 10 years old. Intensive damaging fires can occur in young dense stands of Scots pine in which the trees have not yet self-pruned (Granström 1991a). In Norway spruce stands, fire is more easily spread from the ground to the crown (Granström 1991a); but in northern Europe, crown fires are not common. In Norway spruce stands, fire spread is often restricted because of topographical features (Zackrisson 1977, Zackrisson and Östlund 1991). Fully mature stands with a thin short canopy are at the least flammable stage. On the other hand, over-mature stands with dying trees may be in an ideal state for burning (Van Wagner 1977, Van Wagner 1978, Van Wagner 1983).

Mixed stands and pure deciduous stands are not as susceptible to fire as coniferous stands are (Saari 1923, Heinselman 1981). The more deciduous trees there are in a stand, the fewer fires occur and the lower is the rate of spread (Johnson 1992, Karjalainen 1994). The rate of spread is greatest in mixed Scots pine–Norway spruce stands (Karjalainen 1994). The understory in coniferous stands is also more prone to fire because lichens and mosses are more abundant and flammable. Moreover, decomposition is slower there, which increases the amount of fuel (Johnson 1992). In addition, stands with shrubs, high dwarf shrubs and trees (especially undergrowth of Norway spruce) less than 2 meters in total height or branches on the lower trunk are important for transmitting fire to the crowns of the larger trees. Twigs and smaller branches are moist and will not burn unless they are heated for a longer time or directly bathed in flames (Chandler et al. 1983, Karjalainen 1994). Furthermore, fire cannot propagate from crown to crown unless there is sufficient wind or the ground is sufficiently steep to tilt the flames from one burning tree into the foliage of the next (Chandler et al. 1983).

3.3 Management of Forests

Management of forests can to some degree alter the risk of fire. In northern Europe, forest management seems to help prevent fires. On the contrary, in the Mediterranean region, the risk of fire is increasing because the traditional uses and the management of local forests has decreased, which has in turn increased accumulation of fuel (Goldammer and Jenkins 1990, Naveh 1990, Stocks and Trollope 1993, Trabaud et al. 1993, Stanners and Bourdeau 1995). Plantations (*Eucalyptus* spp. and *Pinus* spp.) established in Mediterranean countries have turned out to be more flammable than the natural forests and have thus in recent years been able to sustain large wildfires (Castro et al. 1990, Naveh 1990).

In northern Europe, natural forest fires are no longer evident, due to wider spacing in plantations than in naturally regenerated areas. In addition, loggings compensate for the effects of fire; in harvested forests there is usually not enough fuel to burn (Suffling 1992). On the other hand, on peat bogs drained for forestry, ground fires tend to be more severe and frequent than in natural peat bogs (Suffling 1992). When slash, logging residue and snags are removed, less fuel is accumulated in forests (Zackrisson and Östlund, Granström 1993). On the other hand, because natural fires are lacking, more fuel is accumulated, which puts forests at risk for more severe fires (Wein and Moore 1977, Heinselman 1981, Holling 1981, Binkley et al. 1993). Failing to remove wind falls and logging residue may, however, lead to more fuel accumulation and risk of fire (Flannigan et al. 1989). Forest roads, on the other hand, act as fire-breaks and make fire-fighting easier (Heinselman 1981, Zackrisson and Östlund 1991).

4 Prediction of Fire Risk

In recent decades, various types of fire-control systems have been developed which reduce the numbers of burned areas and probably lengthen the fire cycles, especially in northern Europe (Zackrisson 1977, Heinselman 1981, Kilgore 1981, Aarne 1993, Parviainen 1996). The systems available are various kinds of warning systems or fire-risk indices, remote sensing or GIS-based techniques. Estimation of fire risk involves estimation of the probability of fire ignition and of probable fire behaviour. In practice, fire managers and researchers use fire-rating systems (Van Wagner 1987, Vasconcelos 1995). In any case, risk assessment and danger rating require the same basic information on weather, vegetation, terrain, fire occurrence and human activity, all of which vary in space and time (Vasconcelos 1995). Management of fire risk includes knowledge of fire risk (distribution in time and space), ability to anticipate changes in risk, and decision-making in order to minimise risk. Fire-risk maps provide basic information and are used in forest management to help decide on and schedule stand treatments and placement of fire-breaks (Vasconcelos and Pereira 1990, Vasconcelos 1995).

In Finland, fire-risk prediction is based on observations available for daily values of precipitation, air temperature, relative air humidity and cloudiness. Observations from 50 weather stations are compared with statistics on forest fires and weather observations of previous years. Fire-risk warning is defined for each province using fire-risk indices. The value of the index is calculated based on the percentage moisture in the surface layer of the ground and according to the terrain most susceptible to ignition. Because wind speed does not affect ignition, it is not taken into account. In the United Kingdom, the assessment of fire danger is similar. However, there it also takes into account the composition and condition of the ground vegetation and wind speed (and in addition, in more detail, fire risk by hour and season and short-term weather forecasts) rating the fire danger into five classes. Fire risk is assessed daily for each forest district and can be calibrated for national or regional circumstances. In addition, social factors can be included (Aldhous and Scott 1993). The system of fire-risk index seems to be adequate for areas with low fire risk, as is the case in northern and central Europe.

In areas with a high risk of fire, such as southern Europe, more sophisticated systems are needed. Applications of satellite remote sensing are aimed at obtaining spatially distributed information on fire risk for situations where there is a lack of other relevant spatially distributed information such as fire-occurrence statistics and weather data (Vasconcelos 1995). Remote sensing can be used to estimate reliable temporal and spatial dimensions of fire risk in terms of fuel maps for various types and states of vegetation (López et al. 1991, Justice et al. 1993, Vidal and Devaux-Ros 1995, Vasconcelos 1995). Satellite images offer a convenient way to determine and map the physiological conditions of vegetation cover, greenness and moisture (fuel characteristics) throughout the year in order to ascertain the conditions most likely to lead to fire risk (López et al. 1991, Vasconcelos 1995). The normalised difference vegetation index (NDVI) can be used to monitor evolution of vegetation and the biomass produced. In areas with high temperature and low precipitation, the images show strong decrements in NVDI, while irrigated land areas show minimum decrements or positive increments (López et al. 1991).

From the standpoint of operational applications, satellite remote-sensed data should be obtained at least during spring and summer. Weekly information on the condition of the vegetation would clearly play an important role in modelling susceptibility to fire. One advance in NOAA/AVHRR is the availability of daily global data at reasonable cost. Large areas can also be covered by a single image. The spatial resolution of the AVHRR data integrates local variations and gives an average response that can be of great interest for large-scale projects. A disadvantage seems to be several possible errors, for example, due to clouds (López et al. 1991, Justice et al. 1993, Caetano et al. 1995).

Another example is the Water Deficit Index (WDI), which is based on the thermal infrared (IR) data for estimating the daily evapotranspiration of forested areas. WDI is a combination of vegetation index and the difference between surface and air temperature obtained from Landsat TM images and meteorological data. Methods like WDI can be used for locating areas with high fire risk and also as a tool for reducing survey operations dedicated to fire prevention. WDI can be used to improve forecasting of fire-start locations (Vidal and Devaux-Ros 1995). The limits for Landsat TM-data are the high cost of the data and the repeat cycle of 16 days. Thus, these data can be used for local sampling rather than for regional monitoring (Justice et al. 1993). Landsat TM images have usually been used for larger scale mapping of fuel and vegetation and for assessment of water stress rather than for NOAA/AVHRR data (Burgan and Shasby 1984, Vasconcelos 1995).

When remote-sensing techniques are used for management of fire risk; certain problems need to be solved (Vasconcelos 1995, Caetano and Pereira 1996). Understory vegetation, which often carries and spreads fires in southern Europe, cannot be detected by traditional remote-sensing techniques as an alternative to assessing canopy conditions. The impossibility of detecting the understory might lead to obvious errors in assessment of fire risk and modelling of fire spread; e.g. two pine stands may have different understories and thus different fire risk, although they are mapped as containing the same fuel types. In contrast, pine and eucalyptus stands may have similar understories but be mapped as different types of fuel. On the other hand, remote sensing is crucial for the development of a truly operational information system for fire-risk management (Vasconcelos 1995).

GIS-based techniques can be used to integrate spatial fuels and topographic data with temporal information on weather and wind and initial fuel moisture. For prediction of fire risk, the fuel classification of the area according to vegetation cover can be based on satellite images and field data (Green et al. 1995). The outputs of the models are perimeters with spotting potential, time of arrival, heat, fireline intensity, rate of spread and flame length (Green et al. 1995). The GIS linked with the nonspatial fire-growth model can also be used to integrate ignition risks, land-use values and suppression costs to improve decisions concerning natural fire, fire prevention and suppression responses. The advantage of GIS is that it can handle large, multilayered, heterogeneous databases and queries about the existence, location and properties of a wide range of spatial objects in an interactive way. Thus, it can be an efficient system for managing and displaying the information required for fire-risk management. It can facilitate and improve integration, interpolation and visualisation of different types of available data and support rapid generation of new relevant information (Vasconcelos 1995). The usefulness of remote sensing and GIS for regional and national level assessment of fire risk has been demonstrated (Burgan and Shasby 1984, Burgan and Hartford 1988, Chuvieco and Congalton 1989).

Integration of remote sensing and GIS provides tools for supporting fire-management activities such as risk management and monitoring (Vasconcelos 1995). In an operational system, NVDI data should be included in a geographical information system with digital Terrain Model for slopes and exposure models, and with climatic data updated by automatic stations in forest zones (López et al. 1991). Standard fire-risk models may also be integrated. In GIS, multivariate regression can be used to generate probability distributions of wild fires and ignition-probability maps based on thematic layers of factors that affect fire (Chou 1992, Vasconcelos et al. 1994). However, GIS lacks many relevant spatial-analysis capabilities and does not include procedures for handling time but factors affecting fire risk vary in time and need to be monitored and updated continuously (Vasconcelos 1995). However, fire (growth) simulation requires extensive data collection, as well as import and export of data to and from GIS. Even with these limitations, simulation of fire spread is becoming an important application of GIS. The improvements and modification of GIS-based models developed elsewhere are needed for practical applications in southern Europe. During recent decades, knowledge about forest fires has increased in southern Europe; but in terms of protection, the results have not been favourable.

5 Discussion and Conclusions

Within the European Community, fire damage affects 500 000–550 000 hectares of forested land every year, causing significant economic losses for forest owners. Regionally, the scale of fire damage can vary from less than one hectare of burned area to forest damage over tens or even hundreds of hectares (Forest fire statistics... 1993). The problems caused by fires are most serious in southern Europe, where an important consequence of fires is erosion.

Fire damage in forests depends on the interaction between meteorological conditions, topography, and site and stand characteristics; the latter are controlled by management regimes and forest operations. It is, however, very difficult, to weigh the relative importance of the different factors. Cross-comparisons between regions can also be difficult due to the different climatologies, fuels and practices – hence the differences in risk between southern and northern Europe.

Management of fire risk includes knowledge of this risk in terms of the probability of fire ignition and probable fire behaviour in time and space, ability to anticipate changes in risk and decision making to minimise risk. Information on weather, vegetation, terrain, fire occurrence and human activity, all of which vary in space and time, are needed for prediction of fire risk. This information can facilitate decisions about forest management, scheduling stand treatments and placement of fire breaks (Vasconcelos and Pereira 1990, Vasconcelos 1995). Currently, some tools, such as fire-risk indices, remote sensing and GIS-based techniques, have been developed for prediction of fire risk on certain locations. Fire-risk indices seem to be suitable for areas with low risk of fire, like northern Europe. In high-risk areas, such as southern Europe, however, more sophisticated techniques are needed for assessment of fire risk.

In the future, assuming the global change in climate, in northern latitudes the mean temperature may increase (Kettunen et al. 1987, Carter et al. 1995). Thus, even in northern Europe, the risk of fire damage could increase compared to the present risk, especially if air humidity decreases in summer. The predictions of increased occurrence of droughts in a $2 \times CO_2$ climate indicate an increase in the length of the fire season (Flannigan and Van Wagner 1991, Fosberg et al. 1996, Goldammer and Furyaev 1996). Furthermore, the fire-risk season in northern Europe could lengthen, because elevated temperature may decrease the duration of soil frost (Flannigan and Van Wagner 1991, Peltola et al. 1997).

Knowledge of disturbance ecology and the role of disturbance ecosystem dynamics is important in understanding the consequences of management options and is also a basis for conserving biodiversity in forests. Currently, final quantification and comparison of the absolute effect of various factors affecting fire risk within Europe are not yet possible, although some efforts have recently been made to model fire risk. In the future an understanding of the relative importance of factors affecting fire damage and the link between fire-risk assessment and forest management may be more important than it is now. Therefore, an integrated risk model is needed which will allow the various locational and stand-management factors to be assessed with respect to the probability of fire damage, especially in southern Europe, but also in northern Europe.

Acknowledgements

This study was funded by the European Community (AIR programme) within the scope of the research project “Silvicultural strategies for predicting damage to forests from wind, fire and snow: integrating tree, site and stand properties with geographical information systems and regional environmental models to evaluate options for forest management” (AIR3-CT94-2392). The work has been carried out at the Faculty of Forestry, University of Joensuu, under the leadership of Prof. Seppo Kellomäki. The support of the Faculty of Forestry, University of Joensuu, and the Department of Ecological and Environmental Sciences in Lahti, University of Helsinki, are gratefully acknowledged. The author would especially like to thank Mr. Mario Caetano, Prof. Seppo Kellomäki and Dr. Heli Peltola for valuable comments and constructive criticism.

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