

Fine-scale heterogeneity in overstory composition contributes to heterogeneity of wildfire severity in southern boreal forest

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Abstract Wildfire can create a mosaic of impacts of varying severity across the landscape. Although widely recognized, this feature and its causes are little understood or studied in ecology. We studied a 1,200-ha wildfire in the southern boreal forest of the Boundary Waters Canoe Area Wilderness (BWCAW) in northeastern Minnesota, USA, using 275 ground plots (stand-scale) and 1:7,000 scale aerial photographs for the entire burned area (landscape-scale). Fire severity was markedly heterogeneous. Overall, 50% of the burn extent was classified as high burn severity, but patches burned this severely were on average less than 70 m from patches of low severity. As expected, lowlands had lower average fire severity than uplands, but several lowland areas burned, and some upland areas remained unburned. At the landscape scale, pre-fire vegetation type—itsself heterogeneous—and patch size of less flammable cover types influenced fire severity. Crown fire severity in upland areas was lowest in pure aspen–birch and red/white pine stands and highest in jack pine and spruce–fir stands. At the stand-scale, slope position and the density of certain tree species at adjacent plots influenced fire severity. Improved understanding of the severity patterns created by wildfire can help to guide the management of spatial patterns of forested systems. Based on our study, a larger range in disturbance severity at scales of

0.1 to several ha and increasing the average size, and range of sizes, of residual patches would in aggregate better mimic natural disturbance than typical harvests.

Keywords Fire ecology · Fire severity · Forest dynamics · Landscape ecology · Natural disturbance

Introduction

Wildfire is an integral process of many forest systems and historically has shaped plant communities and driven individual species' adaptations. While research on forest fire ecology has a well-established history, variability at a range of scales within a single wildfire event is relatively unexplored (Turner et al. 2003). However, such variability has both ecological and management significance (DeLong and Kessler 2000; Kuuluvainen 2002), and the response may vary depending on the scale at which it is measured (Simard 1991; Johnstone and Kasischke 2005).

Researchers are increasingly finding links between landscape pattern and ecological process (Turner et al. 1997; Peterson 2002). Landscape patterns of disturbance in forests have been documented to influence plant succession (Turner et al. 1999), nutrient and water movement, changes in local climate (Turner et al. 1997), and faunal population, habitat use, and foraging dynamics (Gasaway and Dubois 1985; Merrill et al. 1998; Manolis et al. 2002).

Wildfire itself is influenced by the spatial pattern of the landscape it impacts, including topography, location and arrangement of fire breaks, and distribution and arrangement of fuels (Heinselman 1973; Turner et al. 1997; Peterson 2002). In addition, wildfire behavior, such as rate of spread and crown fire spotting, which are influenced by weather conditions at the time of the fire, also influences

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the spatial pattern of fire severity (Ryan 2002). These factors lead to non-uniform impacts of wildfire, creating a mosaic of fire severity levels across the landscape. In effect, it is possible that the relationship between fire and vegetation results in a feedback system where the type and arrangement of vegetation influence the pattern of fire severity which, in turn, influences post-fire vegetation response (Foster and King 1986; Johnstone and Kasischke 2005).

Understanding wildfire severity patterns on the landscape can also be useful for forest management where the goals include emulating the patterns of natural disturbance (DeLong and Tanner 1996; Bergeron et al. 2002; Hart and Chen 2006). Boreal forest organisms have adapted over millennia to a range of natural disturbance regimes and the resulting landscape patterns (Swanson et al. 1997). Landscape patterns resulting from modern human forest management may deviate from the historical “natural” range of landscape pattern (Harvey et al. 2002). This deviation may negatively impact the persistence and abundance of species sensitive to these landscape patterns (Reich et al. 2001; Friedman and Reich 2005; Hart and Chen 2006). Many current forest management prescriptions have some element of managing for pattern, but these prescriptions are usually for species-specific, esthetic, or economic reasons (Palik et al. 2003). Wildfire patterns may provide an alternative and perhaps better guide for sustainable forest management to help retain integral natural processes and ensure the persistence of species dependent on the spatial pattern of forests.

We studied the fire severity patterns of a 1,200-ha wildfire in the Boundary Waters Canoe Area Wilderness (BWCAW) in northeastern Minnesota, USA (Fig. 1), to address the issue of fire variability in this unique and relatively undisturbed landscape. For this study, we consider two categorical measures of fire severity that are related to

different physical aspects of fire: crown fire severity—the amount of tree crown scorch or canopy foliage consumption; and ground fire severity—known as “depth of burn” or the amount of leaf litter and soil organic matter consumed by fire (Ryan 2002). Although crown fire severity is largely influenced by flaming combustion and is related to the physical measure of fire intensity, ground fire severity is largely influenced by glowing combustion and the duration of heating (Johnson 1992) and may not directly relate to crown fire severity; i.e., high crown severity does not necessarily result in high ground severity, or vice versa (Ryan 2002). Several authors have suggested using ground fire effects as an appropriate method to measure fire severity in boreal or near-boreal systems where there is a relatively thick organic layer (Van Wagner 1983; Schimmel and Granstrom 1996).

For at least 10,000 years, wildfires regularly occurred in the BWCAW as in much of the southern boreal forest in central North America (Heinselman 1973; Ohmann and Grigal 1981). Consequently, the time since last disturbance, disturbance type, and, we hypothesize, disturbance pattern are expected to be critical factors in determining plant community composition in the region (Grigal and Ohmann 1975; Frelich and Reich 1995; Reich et al. 2001; Hart and Chen 2006). The BWCAW presents a unique opportunity to study wildfire processes given its large size (400,000 ha) and minimal human management. Approximately one-half of the area has never been logged and the entire area is now a federal wilderness area where logging is not allowed.

The main objectives of this study were to characterize the patterns and potential causes of fire severity heterogeneity in a single fire event within the BWCAW at the landscape and stand scales. In particular, objectives of this paper were to:

1. Characterize patch composition and structure for different levels of fire severity and pre-fire cover types at landscape and stand scales.
2. Identify the biological and physical parameters influencing levels of fire severity at the landscape and stand scales.
3. Discuss the patterns of this wildfire in comparison to current harvesting recommendations in Minnesota, USA.

Materials and methods

Site description

The study area is within a 1,200-ha wildfire north of Little Gabbro Lake (47°52'N, 91°35'W) that occurred in June

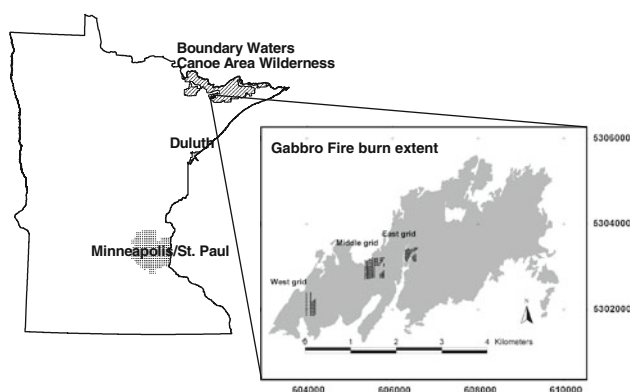


Fig. 1 Gabbro Lake fire of 1995 in the Boundary Waters Canoe Area Wilderness, Minnesota, USA. Burn extent indicates the landscape-scale sample area and the grids indicate the areas sampled at the stand scale

1995 within the BWCAW (Fig. 1). The BWCAW is a low-relief plateau with rocky ridges on the vast Canadian Shield bedrock. The climate is continental with long cold winters, short warm summers, and strong seasonal changes due to the changing intrusions of polar air masses, humid air from the Gulf of Mexico and dry Pacific air masses (Heinselman 1996). The average freeze-free season is around 100 days and the annual precipitation in the central BWCAW is approximately 690 mm (Heinselman 1996).

The upland vegetation type is southern boreal forest composed of a mixture of conifers including white pine (*Pinus strobus* L.), jack pine (*Pinus banksiana* Lamb.), black spruce (*Picea mariana* (Mill) B.S.P.), red pine (*Pinus resinosa* Ait.), and balsam fir (*Abies balsamea* (L.) Mill.), with paper birch (*Betula papyrifera* Marsh.) and trembling aspen (*Populus tremuloides* Michx.). The region historically had a fire-dominated disturbance regime with nearly all tree stands having originated following stand-killing wildfire (Grigal and Ohmann 1975). Overall average fire rotation for stand replacing fire, prior to settlement by Europeans, was 100 years, and this varied from 50 years in jack pine forests to 200–300 years in red and white pine forests (Heinselman 1973). The study area was last burned in 1895 and, while most of the current burn area was never logged, the western one-fifth was logged in 1896 and it is unknown when a burn preceded or followed the logging (Heinselman, unpublished).

Fire description

Lightning started the Gabbro Lake fire on June 6, 1995, following an abnormally dry and hot spring. The fire remained small for 11 days, but then expanded into an intense crown fire for the next 5 days until it was extinguished at the end of day 16 by a cool front with rain showers. The Gabbro fire was the largest wildfire in the BWCAW since a prescribed natural fire policy (now called WFU or Wildland Fire Use) was initiated by the US Forest Service in 1987 (although two larger fires subsequently occurred during 2006 and 2007). The WFU policy allows a wildfire of natural ignition (i.e., lightning) to burn within an area predetermined to not threaten human settlements.

Field methods

Data were collected at the landscape scale and stand scale. The landscape scale encompassed the entire fire extent. Nested within the landscape-scale area were three stand-scale grids of sample points and transects (referred to as east, middle, and west), each covering between 6 and 16 ha and roughly 0.4–0.9 km apart (Fig. 1). The three grids were located in areas with different fire rates of spread that

showed different fire severity and cover type patterns at the landscape scale, and that could be accessed with less than 1 days travel from the wilderness edge. Details of sampling at the two scales follow in the next two sections.

Stand scale

A sample grid was established in each of the three locations with grid points spaced 35 m apart. The 35-m spacing between sample points was a compromise between capturing the heterogeneity of fire severity and covering as large an area as possible within the burn. The number of grid points varied from 57 to 140 among the three locations, with a total of 275 grid points sampled (Fig. 1) and their coordinates recorded using GPS. The middle and east locations had treeless marsh that was not sampled (Fig. 1).

At each grid point, crown and ground fire severity; pre-fire tree abundance, composition and size; slope; aspect; topographic position; and soil depth were measured during August 1995, 2 months following the fire. These variables were assessed on circular plots centered at each grid point, with the radius adjusted for the scale of each variable (e.g., small radii for small trees, larger radii for large trees and topographic position), following standard techniques (e.g., Reich et al. 2001; Battaglia et al. 2009). Ground and crown fire severity were characterized in 2- and 10-m-radius plots, respectively. Ground fire severity (measured at grid points and transects only) included six classes based on an ocular estimate of the amount of surface litter, duff layer, or organic matter consumed (Battaglia et al. 2009; Table 1). Crown fire severity (percent canopy foliage consumed; Sieg et al. 2006) was assigned to one of six classes (Table 1). Topographic variables (slope, aspect, position) were summarized in a 17.5-m-radius plot at each grid point (Table 2). Topographic position described the position of each grid point on the slope using eight categories (Table 2). Soil depth was recorded as the average depth to which a soil probe could be inserted before hitting boulders or bedrock, in five random locations within 2 m of the grid point.

Trees present prior to the fire at each grid point were recorded for species and diameter class. Trees that were alive or scorched could be easily be identified to species, while trees that were severely burned could be identified based on wood and branching characteristics in a large majority of cases. Trees were characterized in a set of nested radius plots depending on tree size as follows: trees of all sizes were recorded in 2-m-radius plots, trees larger than 10 cm dbh were recorded in 2- to 4-m-radius plots (larger radii were used when trees were lower in density to obtain an adequate sample size), and to augment the number of large trees sampled we used a 10 ft²/acre (2.3 m²/ha) prism wedge to record the species and dbh of

Table 1 Descriptions of ground fire severity classes (middle column) based on the amount of surface litter and/or duff consumed and stand-scale and landscape-scale crown fire severity classes (right column) based on the percent canopy foliage consumed

Fire severity class level	Ground fire severity Surface litter and duff layer consumed	Crown fire severity (stand and landscape scale) Canopy foliage consumed (%)
0	Unburned	None
1	Light scorch of surface litter	1–25%
2	1–50% of surface litter	26–50%
3	50–99% of surface litter Some of duff layer	51–75%
4	100% of surface litter Most of duff layer	76–99%
5	Only mineral soil remaining	100%

Table 2 Descriptions of topographic variables at the stand-scale

Topographic position	Aspect class	Slope class
1 = Head	0 = No slope	0 = No slope
2 = Shoulder	1 = 316°–45°	1 = 0–10%
3 = Back	2 = 46°–135°	2 = 11–20%
4 = Foot	3 = 136°–335°	3 = 21+%
5 = Toe	4 = 226°–315°	
6 = Flat, upland		
7 = Valley		
8 = Depression, lowland		

trees, >10 cm dbh, located between 4 and 17.5 m from the sample plot center.

The circular plots centered around each grid point were supplemented with two continuous north–south belt transects between sample points in each location. Along the continuous, north–south belt transects, a 2-m-wide transect was used to record ground fire severity measures and a 20-m-wide transect for crown fire severity measures, using a minimum increment length of 1 m.

Landscape-scale

At the landscape scale, crown fire severity and pre-fire cover types were interpreted from aerial photos and digitized for use in a GIS system (Fig. 2). For crown fire severity, we interpreted 1:7,000 scale aerial-infrared photos, taken 2 months post-fire in August 1995. For pre-fire cover types, we used the National Aerial Photography Program's (NAPP) 1:10,000 scale photos, taken in May 1992 and May 1993. For both coverages, the minimum mapping unit (mmu) was 1 ha, but where clear contrasts existed, exceptions were made to as low as 0.25 ha mmu. Data from the field surveys were used to ground-truth classifications made by aerial photo interpretation.

Crown fire severity was measured as an ocular estimate of the amount of canopy foliage consumed by the fire and assigned to one of 8 classes. These were later combined

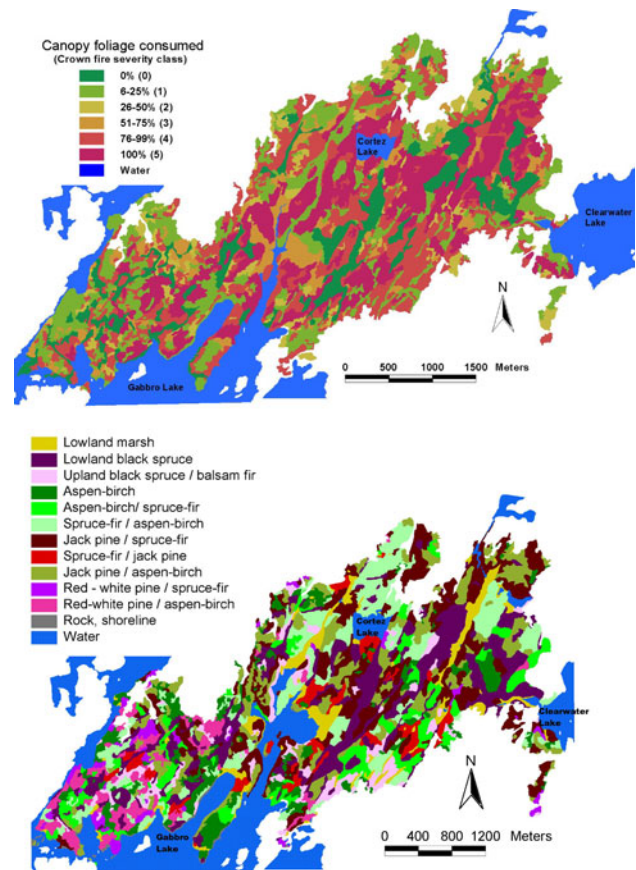


Fig. 2 Burn extent of the Gabbro Lake fire indicating landscape-scale crown fire severity (*top*) and pre-fire cover types (*bottom*)

into 6 classes to match the stand-scale categories (Tables 1 and 3). Pre-fire cover type classes were defined using the relative percentage of the dominant canopy tree species, or other vegetation/non-biotic cover (e.g., rock outcrops) when trees were absent. Since most stands in this region contain a mix of tree species, a cover type designation (i.e., spruce-fir, aspen-birch, etc.) was assigned if the particular species represented >25% of a patch. This resulted in different cover classes with the same tree species, differentiated by their relative composition. For example, if one patch had 50–75% aspen and 25–50% spruce-fir while

Table 3 Landscape-scale summary of severity classes in the entire fire area (upland and lowland) and upland only, and stand-scale patch composition in crown and ground fire severity classes

Crown fire severity class	Landscape-scale								Stand-scale	
	Upland and lowland					Upland only		Stand area	Crown fire severity Stand-scale area (%)	Ground fire severity Stand-scale area (%)
	No. of patches	Total area (ha)	Mean area (ha)	Mean shape index	Fire area (%)	Fire area (%)	Mean area (ha)	Stand-scale area (%)		
0	48	145	3.0	2.1	12.8	4.1	0.1	24.7	44.9	29.0
1	203	263	1.3	1.8	23.1	23.0	0.6	19.0	18.3	5.6
2	83	75	0.9	1.7	6.6	7.6	0.6	2.4	16.3	14.7
3	76	86	1.1	1.7	7.6	7.0	0.5	3.4	10.9	24.1
4	274	295	1.1	1.8	25.9	29.5	0.7	23.5	9.0	22.9
5	66	275	4.2	1.9	24.2	28.8	2.7	26.9	0.7	3.6
Total	750	1,137	1.5	1.8	100	100	1.3	100	100.1	99.9

The column *Stand area* contains the GIS-derived landscape-scale data for the area covered by the three stand-scale sample grids

Mean shape index: $MSI = (P/2)/(\sqrt{(\pi \times A)})$, where P perimeter (m) and A area (m^2) (McGarigal 1995). MSI is 1 if a patch is a perfect circle, and increases as the patch becomes more complicated in shape

Table 4 Landscape-scale, pre-fire cover-type categories, their relative proportions in the total burn area and mean patch area

Cover type	Relative proportions of tree species	No. of patches	Total burn area (ha)	Mean patch area (ha)	Total burn area (%)
Lowland marsh	Not applicable	59	59	1.00	5.39
Lowland black spruce	>50%	97	187	1.93	17.18
Upland black spruce/balsam fir	>50%	50	59	1.17	5.38
Aspen–birch	75–100%	94	93	0.99	8.51
Aspen–birch/spruce–fir	50–75%/25–50%	46	82	1.78	7.49
Spruce–fir/aspens–birch	50–75%/25–50%	77	153	1.99	14.06
Jack pine/spruce–fir	>50%/0–50%	92	193	2.10	17.71
Spruce–fir/jack pine	50–75%/25–50%	25	44	1.75	4.02
Jack pine/aspens–birch	25–75%/25–75%	94	141	1.50	12.93
Red and white pine/spruce–fir	50–100%/0–50%	36	26	0.73	2.40
Red and white pine/aspens–birch	50–100%/0–50%	34	54	1.57	4.91
Total		704	1089	1.55	100

Relative dominance of mixed species stands is indicated by the order of the species names in the cover type name

another patch had 50–75% spruce–fir and 25–50% aspen, then these two patches were classified into different cover types. A total of 49 pre-fire cover classes were identified. For analysis, they were combined into 13 ecologically relevant categories (Table 4).

Analyses

Patch composition and structure of pre-fire cover type and fire severity

Patch characteristics (e.g., size and shape), distribution of fire severity classes, and isolation distances (defined below) were calculated at stand and landscape scales. Summary metrics of fire severity patches were also calculated for a

map comprising a subset of the landscape clipped to the extent covered by three stand-scale grids (called “stand-area”), a map covering upland areas only, and a map with fire severity classes dissolved to 3 categories (mentioned in “Discussion” only).

At the landscape scale, standard landscape metrics, such as fractal dimension and contagion, were calculated using Patch Analyst 3.0 (McGarigal 1995; Riitters et al. 1995) to allow comparison of the size, shape, and arrangement of patches between the overall landscapes of pre-fire cover type and fire severity. For this analysis alone, the cover type classes were collapsed into eight classes to match the full number of fire severity classes in order to make a meaningful comparison. Patch metric values that differed by 10% between the land-cover and fire severity maps were

considered substantive and highlighted for discussion. At the stand scale, average patch radius for ground and crown fire severity classes was calculated as the average transect segment length along the north–south belt transects. Patch radius values at the stand and landscape scales were natural log-transformed and used as response variables with crown and ground fire severity class as ordinal predictors in ANOVA analyses, and followed up by a Tukey test ($\alpha = 0.05$) to determine significant differences in patch radius between the fire severity classes.

Patch isolation describes the minimum distance between high fire severity areas (class 4 or 5) and low fire severity areas (class 0, 1, or 2). For comparison, three distances were calculated that were based on three alternative definitions of low fire severity: unburned (class 0), low (class 0 or 1); moderate (class 0, 1 or 2). At the stand scale, minimum distances of high ground severity plots to areas of low fire severity along the north–south oriented transects were calculated. At the landscape scale, the crown fire severity map (Fig. 2) was converted into 10-m grids and minimum distances of high to low crown fire severity grid cells were calculated.

To allow us to examine the differences in heterogeneity between the two levels of scale, we intersected digitized stand-scale transect data with landscape-scale patches to calculate the percentage of each ground fire severity class within each landscape crown fire severity patch. These values were then averaged for each landscape-scale fire severity class. In order to reduce geo-referencing errors between the landscape-scale GIS data and the stand-scale transect layers, 10-m buffers (the average overlap error calculated from a subset) were clipped from the end of all transect segments that intersected a GIS polygon boundary.

Parameters influencing fire severity level at the landscape and stand scales

We analyzed the influence of pre-fire vegetation and physical variables on levels of fire severity at both the stand and landscape scales. For the landscape scale, we used cover type and patch size of pre-fire cover types as the predictors. Cover type also indicated upland or lowland areas. For the stand scale, pre-fire tree species and density, topographical variables, and soil depth were the predictors.

Influence of pre-fire cover type and patch size on landscape-scale fire severity To determine the average fire severity for each cover-type polygon, we intersected the GIS fire severity layer with the pre-fire cover type layer. Average fire severity for each pre-fire cover type polygon was the response variable in a multiple-least squares regression with cover type and patch area (m^2) as the

predictors. In addition, we calculated the percent area of each cover type that fell in each crown fire severity class.

Influence of pre-fire tree density, topography, and soil depth on stand-scale fire severity We analyzed the influence of pre-fire tree densities and 4 topographical variables (slope, aspect, slope position, and soil depth) on ground and crown fire severity class using multi-category logistic regression (SAS 2001). This was performed on upland plot data only ($n = 157$) because lowland plots had no slope, and hence no aspect or slope position. We used logistic regression because fire severity class is an ordinal variable with non-normal distribution. However, treating fire severity class as a continuous variable using least squares regression yielded similar results.

Pre-fire density (no. of trees/plot) of the six most common tree species, balsam fir, paper birch, jack pine, black spruce, white pine and aspen, was calculated at each plot and as an averaged value for all eight adjacent plots. The initial model was run with 12 density values (plot density and adjacent plot density for six species) and 4 environmental variables. Backwards elimination (Agresti 1996) was used to reduce the model.

Results

Patch composition and structure of fire severity and pre-fire cover type

Visual examination of the map shows that fire severity was considerably heterogeneous (Fig. 2). The pattern of severity appears more complex in the western portion of the burn area while the eastern portion has larger linear blocks of both low and high fire severity. These patterns are also evident in the pre-fire vegetation (Fig. 2).

Patch composition

For the entire landscape within the fire perimeter, roughly half the burn extent was in the highest crown fire severity classes ($>75\%$ canopy foliage consumed), with ca 13% unburned and 37% in the mid-severity classes (Table 3). Patches of the highest and lowest crown fire severity (classes 0 and 5), accounted for only 15% of the 750 fire severity patches, but covered 37% of the fire extent, and were larger in size and more complex in shape than the other classes (Table 3). Upland areas were similar to the overall landscape except that they had a lower percentage of unburned areas (4 vs. 13% for both upland and lowland combined). These patterns were similar for the section of the landscape that covered only the sample grids, although

it had a higher percentage of unburned areas (Table 3, Stand area).

The pre-fire cover type patch composition of the burn area at the landscape scale was almost one-quarter lowland, mainly lowland black spruce (Table 4). Spruce–fir-dominated patches were prevalent in upland areas, covering nearly 25% of the entire landscape (Table 4). Most of the other 50% of the landscape was dominated by jack pine or aspen–birch, while red and white pine patches comprised roughly 7% of the landscape.

Patch size and shape

Mean patch radii were significantly different among fire severity classes for both crown and ground fire severity types at both the landscape and stand scales ($P < 0.0001$, Fig. 3). At the stand scale, crown and ground fire severity patch radius (1/2 transect length at the stand scale) was highest in fire severity class 0, lowest in fire severity class 1, and then gradually increased with fire severity (Fig. 3). Also, mean patch radii were higher for crown fire severity classes than ground fire severity classes in all except fire severity classes 0 and 5 where they were about equal ($P < 0.01$). At the landscape scale, mean patch radii were about one order of magnitude larger than those calculated at the stand scale (Fig. 3), although landscape level patches were still larger than the potential minimum mapping units of 0.25 ha.

Although the mean patch areas of intermediate crown fire severity patches (classes 1–4) were similar to the pre-fire cover type patches, the lowest and highest fire severity patches (0 and 5) were larger on average than the dominant vegetation patches (Tables 3 and 4).

When patch metrics for landscape-scale pre-fire cover type and crown fire severity layers are compared, the two coverages show striking similarity (data not shown). The only difference greater than 10% was the area-weighted mean shape index, indicating that larger fire severity class patches were slightly more complex in shape than larger pre-fire cover type patches.

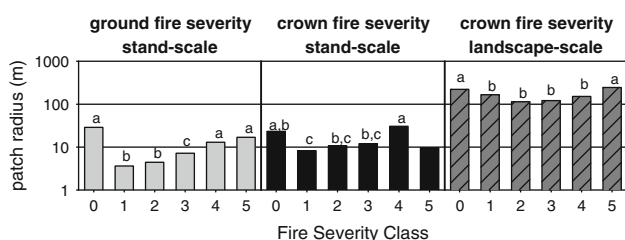


Fig. 3 Mean patch radius for fire severity classes at the stand and landscape scales. Among fire severity classes within a given scale and fire severity type, different letters indicate significant differences (Tukey test $\alpha = 0.05$)

Patch isolation

Mean patch isolation distances of high burn severity patches from lower severity patches were similar between the stand and landscape scales, ranging from 50 to 70 m at the stand-scale and 40 to 100 m at the landscape scale (Fig. 4). Mean isolation distances were larger as the fire severity differences were greater.

Stand-scale variability within landscape-scale patches

Comparing the stand-scale variability within landscape-scale crown fire severity patches reveals marked fire severity heterogeneity at the stand scale (Fig. 5). On average, in unburned landscape-scale patches (crown fire severity class = 0), 60% of the stand-scale transect segments were also unburned (ground fire severity class = 0), but 20% of the transect segment lengths had most or all of the surface litter and duff consumed (ground fire severity classes = 4 or 5). This pattern is similar, but in reverse, for landscape-scale patches with 100% of the crown foliage consumed (crown fire severity class = 5) (Fig. 5). These patterns are similar, but less extreme, for stand-scale crown fire severity (data not shown). These results show that derivation of landscape-scale metrics of low resolution will result in patches that include considerable finer-scale heterogeneity either undetected at or unimportant at the landscape scale, or both.

Parameters influencing fire severity level at the stand and landscape-scales

The degree of landscape-scale fire severity was highly dependent on the pre-fire cover-type ($P < 0.0001$). Across

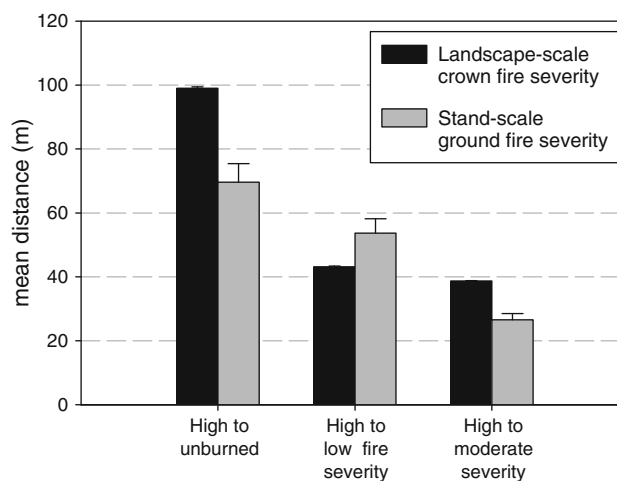


Fig. 4 Distances of high fire severity areas to unburned, low and moderate fire severity areas at both the stand scale and the landscape scale. High, classes 4 and 5; unburned, class 0; low, classes 0 and 1; and moderate, classes 0, 1 and 2 (Table 1). Error bars represent one standard error

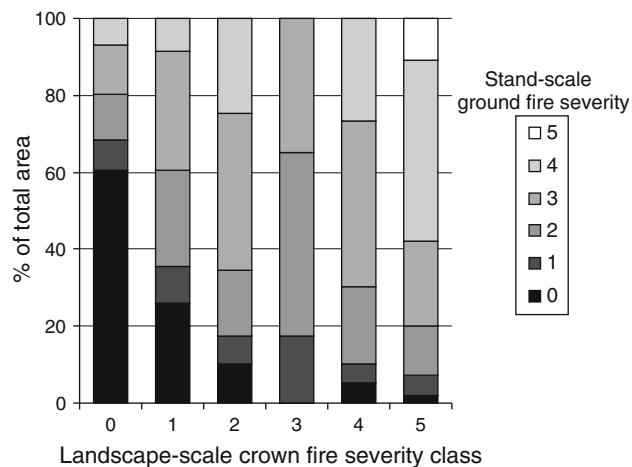


Fig. 5 The proportion of different levels of stand-scale ground fire severity (0, unburned; 5, forest floor completely consumed) within landscape-scale patches of different crown fire severity. Each bar represents the total area of all landscape-scale patches at a given crown fire severity class level (0, unburned; 5, crown completely consumed) (Table 1)

all cover types, landscape-scale median canopy foliage consumed was 78%, equal to about a crown fire severity class of 4. Landscape-scale median canopy foliage consumed was lowest in lowland black spruce at 2%, and was also low in aspen–birch stands (20%) (Fig. 6). In contrast, canopy consumed averaged >90% in forest types dominated by either jack pine or spruce–fir. Aspen–birch stands with a substantial, but minority, component of spruce–fir had canopy consumption rates more similar to spruce–fir than aspen–birch. Red and/or white pine-dominated stands had intermediate levels of median canopy foliage consumed (Fig. 6).

The proportion of the total area occupied by different crown fire severity classes varied among pre-fire cover types (Fig. 7). All cover types experienced 100% canopy foliage consumption (landscape-scale crown fire severity class 5) in some proportion of their area. Roughly half of the 11 pre-fire cover types had the majority of their total area in the highest crown fire severity classes (canopy foliage consumption >75%; Fig. 7), with the strongest trends in spruce–fir- or jack pine-dominated stands. The cover types with a larger proportion of their total area in unburned or low crown fire severity classes included lowland areas and red and white pine stands (Fig. 7).

Regressions predicting landscape-scale crown fire severity from pre-fire cover type and patch area showed that lowland areas, aspen–birch and red and white pine cover types were associated with decreasing landscape-scale crown fire severity while spruce–fir and jack pine cover types were strongly associated with increasing fire severity (Table 5). Further, increasing patch area was

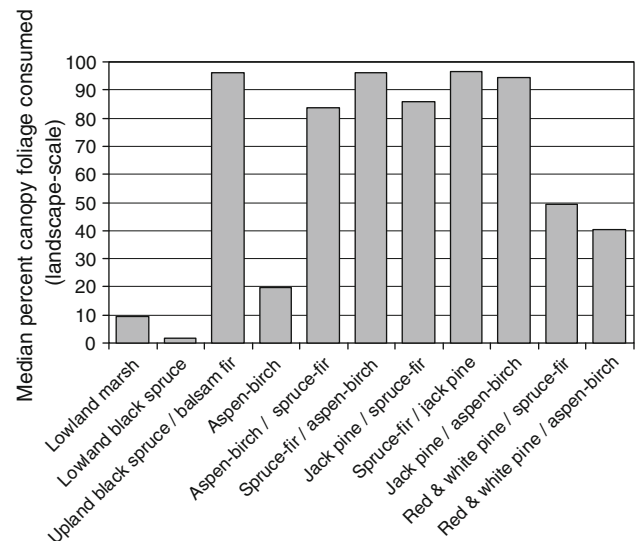


Fig. 6 Median landscape-scale canopy foliage consumed for pre-fire cover-types. Relative dominance of mixed species stands is indicated by the order of the species names in the cover type name (Table 2)

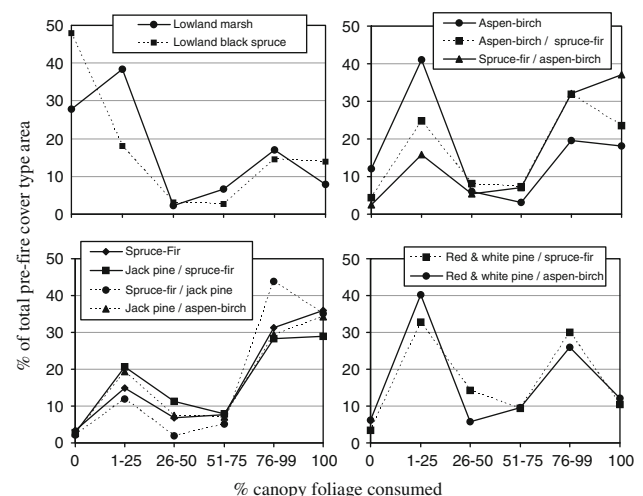


Fig. 7 The proportion of the total area for a given pre-fire cover type in each landscape-scale crown fire severity class (classes 0 through 5 represented on the x-axis as the percent canopy foliage consumed in each; Table 1). Relative dominance of mixed species stands is indicated by the order of the species names in the cover type name (Table 2)

associated with decreasing fire severity in the lowland black spruce and aspen–birch cover types (Table 5).

For upland plots at the stand scale, four variables significantly influenced the multiple logistic regression predicting ground fire severity (Table 6). These were topographic position and the adjacent densities of balsam fir, black spruce, and aspen. For topographic position, ground fire severity increased moving upslope towards the head of the slope. Ground fire severity had a negative

Table 5 Results of regression predicting landscape-scale crown fire severity class by pre-fire cover-type and patch area

Label	Estimate	Std. error	<i>t</i> value	<i>P</i> value ^a
Constant	3.8899313	0.09942	39.13	<0.0001
Lowland marsh	−1.252547	0.218693	−5.73	<0.0001
Lowland black spruce	−0.799951	0.17288	−4.63	<0.0001
Spruce–fir	0.7685443	0.236875	3.24	0.0012
Aspen–birch	−0.540326	0.182707	−2.96	0.0032
Aspen–birch/spruce–fir	0.0033157	0.239104	0.01	0.9889
Spruce–fir/Aspen–birch	0.4664891	0.190371	2.45	0.0145
Jack pine/spruce–fir	0.6687057	0.179247	3.73	0.0002
Spruce–fir/jack pine	1.1696798	0.317371	3.69	0.0002
Jack pine/aspen–birch	0.4129965	0.174714	2.36	0.0184
Red and white pine/spruce–fir	−0.302778	0.399276	−0.76	0.4485
Red and white pine/aspen–birch	−0.594129	0.271679	−2.19	0.0291
Patch area	0.0000085	0.000006	1.54	0.1248
Patch area: lowland black spruce	−0.000016	0.000006	−2.58	0.0101
Patch area: aspen–birch	−0.00002	0.000011	−1.83	0.0678

All results for cover-type are included, but only the significant ($P < 0.10$) patch area \times cover-type interaction terms are shown. Overall model $R^2 = 0.139$, $P = <0.0000$, $n = 990$. See Table 2 for more detailed pre-fire cover type descriptions

^a Bold indicates a significant P value ($\alpha < 0.10$)

Table 6 Stand-scale ground and crown fire severity class in relation to topographic position (Table 3) and the adjacent density of selected tree species

Variable	<i>P</i> value	<i>b</i> ^a
Stand-scale: ground fire severity		
Topographic position	0.003	0.641
<i>Abies balsamea</i> density	0.003	0.000266
<i>Picea mariana</i> density	0.0002	−0.00266
<i>Populus tremuloides</i> density	<0.00001	0.00699
Stand-scale: crown fire severity		
Topographic position	0.0079	0.389
<i>Abies balsamea</i> density	0.0022	0.000272
<i>Pinus banksiana</i> density	0.0044	0.00715
<i>Pinus strobus</i> density	0.0019	0.014
<i>Populus tremuloides</i> density	<0.00001	0.00947

^a *b* is the estimated slope from the linear regression model

relationship with the density of black spruce at points adjacent to the sample point (adjacent density), but had a positive relationship with adjacent densities of balsam fir and aspen. Adjacent densities ranged up to 9,000 stems/ha for balsam fir, 0–500 stems/ha for aspen and 0–1,100 stems/ha for black spruce, so these density values can have a considerable influence on the estimated model response.

Crown fire severity was also predicted by topographic position and the adjacent densities of balsam fir and aspen, and with the same trends as ground fire severity (Table 6).

Moreover, as the adjacent densities of white pine and jack pine decreased, crown fire severity decreased (Table 6). Adjacent density of white pine ranged up to 200 stems/ha and jack pine ranged up to 250 stems/ha.

Discussion

Landscape fire severity patch composition and structure

At the landscape scale, the proportion of the landscape in the highest (>75% foliage consumed, 50–58%; Table 3) and lowest levels (unburned, 4–13% of landscape) of fire severity in this study, were roughly similar to results of other boreal forest studies. A 1971 spring wildfire less than 100 km from our study site also had about half of the upland areas with 50–100% crown browning and 10% unburned (Nordin and Grigal 1976). The Yellowstone wildfires in a pine-dominated ecosystem in 1988 left 28% of the landscape unburned, 16% lightly burned, 25% moderately burned, and 31% in the highest fire severity (Turner et al. 1994). For 69 fires, ranging from 21 to 17,700 ha in size, in northern Alberta, unburned “islands” increased with the size of the fire and about 5% of the area in largest fires (2,001–20,000 ha) remained unburned (Eberhart and Woodard 1987).

Mean isolation distances for high to low severity areas in this study ranged from 50 to 100 m (Fig. 4). These distances are within, or near, the range of dispersal

distances for most boreal forest trees (Zasada et al. 1992; Weyenberg et al. 2004). Studies of the 1988 Yellowstone fires reported isolation distances of 50–200 m (Turner et al. 1994), and nearly three-quarters of burned areas were within 200 m of unburned areas for fires 400–2,000 ha in size in northern Alberta (Eberhart and Woodard 1987). Collectively, these data suggest that isolation is generally not a major limitation for tree regeneration following wildfire in boreal ecosystems. The most likely reason for limited isolation distance is that larger patches with high fire severity are more complex in shape (Eberhart and Woodard 1987). High complexity decreases the amount of interior space and hence average isolation distance, probably due to an increased likelihood of fire encountering some type of fire break (i.e., vegetation or topography) and/or a higher probability of stochastic effects, such as a shift in wind direction.

Inevitably, any classification at a coarser scale will obscure heterogeneity at a finer scale (Simard 1991). Mean patch size was an order of magnitude smaller at the stand scale, for both crown and ground fire severity, than at the landscape scale (Fig. 3). This was probably due in part to the defined minimum mapping unit (MMU) and to the resolution limits of the aerial photos, although the mean patch size at the landscape scale was larger than the MMU. The variability between scales may also reflect an imperfect overlap between the stand-scale transect data and the landscape-scale GIS data, although we attempted to account for this by using a 10-m buffer at the end of each transect segment per GIS fire severity patch (see “Materials and methods”). It is also possible that different mechanisms influence fire severity at different scales.

Influence of pre-fire cover type and tree density, and topography on fire severity

Previous studies of boreal forest wildfires documenting fire severity heterogeneity or, more commonly, the existence of unburned patches, attributed these observations to slight depressions with higher soil moisture (Quirk and Sykes 1971), topography (Van Wagner 1983), variations in wind and the location of wetlands and water bodies (Rowe and Scotter 1973), or being downwind of fuel breaks (Foster 1983). Although we observed a lowland versus upland effect in this study, lowland cover types also burned to some extent. While lowland black spruce had the lowest median fire severity at the landscape scale (2%), nearly 25% of lowland areas had greater than 75% canopy foliage consumed (Fig. 7). Conversely, while stands dominated by spruce–fir and jack pine had less than 5% of their area unburned, no cover type had zero unburned areas (Fig. 7). In other words, not all lowland areas remained unburned, and not all upland areas burned completely. In addition,

analysis of upland stand-scale data showed landscape position significantly influenced fire severity (Table 6) indicating that the pattern of fire severity is more complicated than just a lowland versus an upland contrast.

Among upland areas, those plots at the top (or shoulder) of a slope tended to have higher fire severity than plots with positions lower on the slope (Table 6). This was also reported by Nordin and Grigal (1976). Slope tops tend to be drier, and fire tends to increase in intensity as it moves up a slope.

Why did some lowland areas burn? Field observations revealed that many of these areas were quite dry in spring and summer 1995. However, other lowland areas clearly wet at the time of field sampling in August, and likely also at the time of the fire, had still burned. One possibility is that lowland areas that burned were adjacent to more flammable cover types, thus increasing their likelihood of burning. Heinselman (1996) observed that patches of aspen within a matrix of more flammable vegetation were more likely to burn. Another explanation may be that lowland black spruce forests have areas of contiguous fuels with high canopy bulk density and relatively few breaks caused by topographical features.

At the stand scale, adjacent density of balsam fir, black spruce and aspen had significant influence on ground fire severity at a given plot (Table 6), but results were mixed compared to expectations. Ground fire severity increased as the adjacent density of both balsam fir and aspen increased. The trend was somewhat expected for balsam fir but not for aspen. Black spruce had an opposite effect, which is reasonable if this reflects lowland trees.

Landscape fire severity and pre-fire cover type patch structure: fire, vegetation legacy

The similarities in landscape patch structure between fire severity and pre-fire vegetation are striking (Fig. 2). The fire-created patches are slightly more interspersed and complex in shape than the pre-fire vegetation, but otherwise the patch structure metrics are alike (data not shown). However, all cover types had a wide range of fire severity levels (Fig. 7), suggesting that the fire had a moderate level of independence from vegetation landscape configuration. Mean patch sizes of the lowest and highest fire severity levels (0 and 5) were between 2–3 times larger than the mean patch size of pre-fire cover types, suggesting that these fire severities crossed different pre-fire cover types (Tables 3, 4). Although the exact relationship between pre-fire vegetation structure and fire severity spatial structure is not clear, it is possible that heterogeneity of reproduction after the fire will lead to smaller vegetation patches than fire severity patches. This has happened in other parts of the BWCAW after recent fires, where patches of aspen and

jack pine developed within severely burned areas (e.g., the Roy Lake Fire of 1976; Heinselman 1996).

Management implications

Incorporating natural variability in forest management is often suggested as a coarse filter approach to managing for biodiversity (Hunter 1990). There is increasing evidence that many plant and vertebrate animal species are dependent on the spatial patterns of forests (Merrill et al. 1998; Sillett et al. 2000; Manolis et al. 2002; Palik et al. 2003), and it is well known that plant and wildlife species composition responds to severity of disturbance (Heinselman 1973; Ohmann and Grigal 1981; Bergeron et al. 2002). However, a species by species approach to management can be problematic as the needs of one species may conflict with the requirements of another species. A more holistic ecosystem management approach incorporating or restoring natural ecosystem properties, such as the natural spatial patterns of wildfire and variability in severity, can be one means of managing for biodiversity. Several studies indicate that the spatial patterns of fire may differ from logged landscapes in boreal systems (DeLong and Tanner 1996, Purdon et al. 2004). The fire in this study created complexly-shaped patches of widely differing severity within each cover type, in both uplands and lowlands, and with varying combinations of canopy and ground fire severity. In contrast, harvesting tends to create two levels of severity (harvested and unharvested) within one cover type, and harvesting is often limited to moderate levels of severity uniformly applied to the entire harvested area. However, many harvests in Minnesota are done during winter when the ground is covered with snow, and therefore create disturbance that is uniformly low in ground severity.

The fire in this study created patterns that may differ from the current practices of stand management in terms of the incidence and distribution of residual patches (or individual surviving trees), the size and shape of clearcuts, and the distribution of clearcuts on the landscape. Residual patches in clearcuts have been documented to benefit birds and tree regeneration (Merrill et al. 1998; Palik et al. 2003). Residual patches may be important for species dependent on mature forest, provide important resource niches, and can act as plant recolonization sources for adjacent disturbed areas. Suggestions regarding the distribution of residual trees in clearcuts are generally 0.1 ha of residual patch for every 2 ha of clearcut or 5% of the landscape (Chadwick et al. 1986; MFRC 2005). This study shows that nearly 13% of the landscape was unburned and another 23% had low burn severities, and the average size of these residual patches was 1–3 ha (Table 3), an order of magnitude larger than current management recommendations. Another potentially useful comparison may be to examine

patterns in upland areas as the majority of harvesting in this region occurs in upland cover types. While the percentage of unburned areas in upland areas is low (~4%), 23% of upland areas were classified as low severity (<25% canopy foliage consumed) (Table 3). Residual patch sizes of upland areas with less than 25% canopy foliage consumed averaged 0.43 ha (but varied from 0.09 to 25.2 ha for all fire severity classes and from 0.15 to 34.7 ha for the most severely burned patches) and comprised 27% of the burned landscape (Table 3).

In addition to residual areas, the patch size of clearcuts may influence forest regeneration and other spatially dependent organisms. The average clearcut size in Minnesota in the 1990s was 10 ha (Puettmann et al. 1998), and the Minnesota DNR recommends that maximum clear cut size should be 20 ha (MN DNR 1985). In our study, the mean patch sizes for the highest fire severity classes were 1–4 ha in both upland and lowland areas, 0.7–2.7 ha in upland areas (Table 3), and 2.2 ha if the highest classes (>75% canopy foliage consumed) are lumped together in upland areas (not shown). Overall, 50% (nearly 60% in upland areas) of the landscape experienced fire severities high enough to kill most of the trees. Therefore, although an individual patch size was smaller than current clearcut logging prescriptions, it is likely the density of these patches were much higher than those found in a logged landscape.

Thus, forest managers could make the most progress towards mimicking natural patterns caused by fire, by creating a larger range in ground and canopy disturbance severity at fine spatial scales on the order of 0.1 to a few ha, and increasing the recommended average size, as well as the range of sizes, of residual patches.

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