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Research article

Influence of forest management alternatives and land type on susceptibility to fire in northern Wisconsin, USA

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Abstract

We used the LANDIS disturbance and succession model to study the effects of six alternative vegetation management scenarios on forest succession and the subsequent risk of canopy fire on a 2791 km^2 landscape in northern Wisconsin, USA. The study area is a mix of fire-prone and fire-resistant land types. The alternatives vary the spatial distribution of vegetation management activities to meet objectives primarily related to forest composition and recreation. The model simulates the spatial dynamics of differential reproduction, dispersal, and succession patterns using the vital attributes of species as they are influenced by the abiotic environment and disturbance. We simulated 50 replicates of each management alternative and recorded the presence of species age cohorts capable of sustaining canopy fire and the occurrence of fire over 250 years. We combined these maps of fuel and fire to map the probability of canopy fires across replicates for each alternative. Canopy fire probability varied considerably by land type. There was also a subtle, but significant effect of management alternative, and there was a significant interaction between land type and management alternative. The species associated with high-risk fuels (conifers) tend to be favored by management alternatives with more disturbances, whereas low disturbance levels favor low-risk northern hardwood systems dominated by sugar maple. The effect of management alternative on fire risk to individual human communities was not consistent across the landscape. Our results highlight the value of the LANDIS model for identifying specific locations where interacting factors of land type and management strategy increase fire risk.

Introduction

Managers of public and industrial forestlands are keenly interested in reducing the susceptibility of their forested landscapes to unintended wildfire. Attention has recently focused on the risk that fire poses to people living in the so-called wildland-urban interface, where human communities are located in proximity to large blocks of undeveloped land (Cardille et al. 2001a). The primary tools used to mitigate fire susceptibility are the reduction of fuel loads by thinning, manual fuels removal, and prescribed burning (Mutch 1994). There is also considerable interest in modified forest management practices to produce forest landscapes that are less susceptible to fire ignition and spread. In addition to timber production, managers seek to provide for multiple uses of the forest to varying degrees. Industrial forests are managed to also provide hunting, fishing, and other recreational opportunities. Public lands typically have management goals that include conservation of biodiversity and a wide range of recreational opportunities. In some cases these multiple use goals may add constraints to the feasible management options to reduce fire risk. Because vegetation treatments and natural disturbance interact to determine how fires spread across landscapes, it is critical to understand these interactions in a spatial context when managing fire risk (Cumming 2001).

The risk of ignition and spread of unintended fire in a landscape is determined by complex interactions among vegetation treatments, the legacy of natural disturbance, and the effects of the abiotic environment (e.g., land type) (Heinselman 1981; Frelich and Lorimer 1991; Zhang et al. 1999; Cardille et al. 2001a). A fire may ignite in a forest stand that is susceptible to fire, but it may subsequently spread to less susceptible stands. Applying fuel reduction treatments in individual stands without considering spatial context may produce little change in the spread and severity of fires at landscape scales.

The National Forest Management Act of 1976 mandates that US National Forests develop strategic management plans. The planning process requires that each National Forest develop a number of strategic alternatives and evaluate their projected impacts on a suite of forest values, including economic, ecological and recreational values (Morrison 1994). One impact that is difficult to assess is the effect of the alternatives on the risk of unintended fire. Part of the alternative development process involves the delineation of spatially explicit management units (Management Areas) with specific objectives for each unit. For example, the Chequamegon-Nicolet National Forest in northern Wisconsin has formulated alternative management strategies as part of the Forest Plan revision process. Specific forest management activities are prescribed for each Management Area, and fire mitigation may not be the primary objective. However, because portions of this and other National Forests in the Great Lakes states contain historically fire-driven ecosystems, the effect of management alternatives on the susceptibility of these landscapes to fire is a critical question, both here and throughout the region.

The complex interactions among factors that determine fire susceptibility at landscape scales make simulation models useful tools to investigate the impacts of alternative management scenarios on fire risk. Stochastic process models simulate the mechanisms that drive ecological and physical processes. When various process components are developed independently, the behavior of the resulting simulated system is an emergent property of the simulation. We have developed such a stochastic forest disturbance and succession model (LANDIS) to allow us to study the effects of vegetation management and natural disturbance on the susceptibility of landscapes to fire (Mladenoff et al. 1996; Mladenoff and He 1999; Gustafson et al. 2000). Because LANDIS is a spatially explicit model operating at landscape scales, it is ideally suited to study the interaction of vegetation management alternatives and fire.

The objectives of our study are to 1) simulate the management alternatives proposed for the Chequamegon National Forest, 2) produce spatially explicit estimates of fire susceptibility across the landscape, 3) determine the relative effect of land type and vegetation treatment alternatives on fire susceptibility, and 4) assess the vulnerability of human communities near the National Forest to wildfire under each alternative.

Methods

Study area

We conducted simulations for the Washburn and Great Divide Ranger Districts (RD) of the Chequamegon-Nicolet National Forest (CNNF), located in northern Wisconsin, USA (Figure 1). Quaternary geology and mesoclimatic gradients are the primary determinants of environmental variation in the region. Land types are spatial zones that are relatively homogeneous with respect to environmental factors such as climate, soils, and natural disturbance. Soils information and monthly average temperature and precipitation data were used to quantitatively classify the study area into 8 land types (Host et al. 1996), as shown in Figure 2 and described in Table 1. The northern portion of the Washburn RD is within the Bayfield Sand Plains Subsection (Keys et al. 1995), and is characterized by well-drained outwash sand deposits and jack pine and red pine forests. Several natural barrens are found here, and fire has historically been a dominant driver of ecosystem processes. The southern portion of the Washburn RD, and the Great Divide RD are located mostly within the Winegar Moraine and Central Wisconsin Loess Plain Subsections, char-

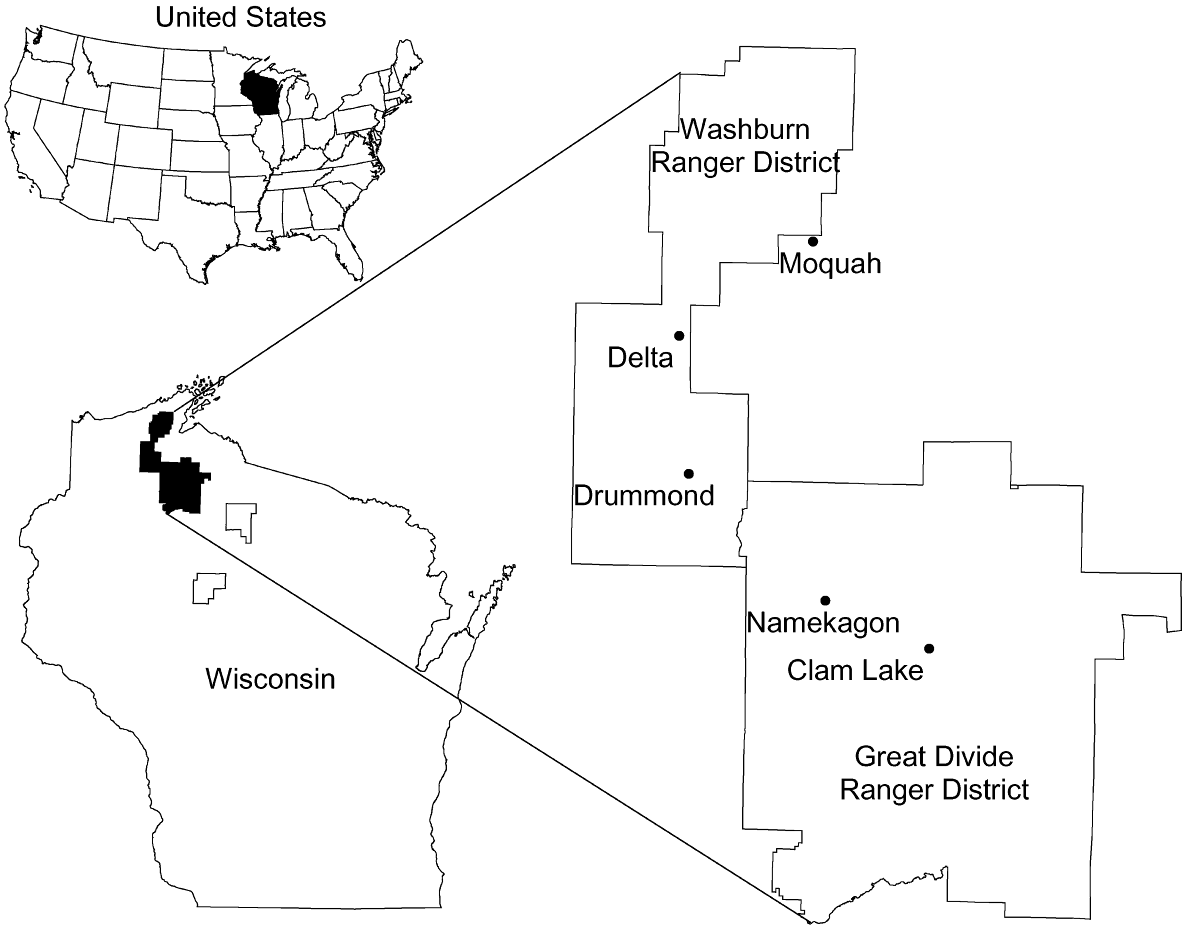

Figure 1. Map of the study area on the Chequamegon National Forest. The large polygons indicate the two Ranger Districts, and the closed circles show the locations of the towns within the wildland-urban interface that were evaluated for fire risk.

acterized by glacial till and mixed deciduous and hemlock forests. Fire was historically less common in these subsections. Fires are routinely suppressed in the region, but disturbance by high winds is a regular occurrence (Canham and Loucks 1984).

Study overview

We used LANDIS v3.6 (described below) to generate 50 replicate simulations of 250 years of vegetation management, forest succession, fire and wind disturbance under six alternative management scenarios for the study area. We chose 250 years to allow the effects of forest management on forest succession and fire to fully manifest themselves. We specifically studied the risk of canopy fires, which were defined as fires occurring in stands containing high-risk fuels (defined below). Each replicate produced maps showing where fires occurred and the presence or absence of high-risk fuels for each cell. For each cell we counted the number of times fire occurred on that cell when conditions (e.g., the tree species age composition of that cell) were appropriate to result in a canopy fire during all 250 years of each of the 50 replicates, to provide a spatially explicit representation of the probability of canopy fire under each alternative. We used these cumulative measures to evaluate the relative effect of land type and management alternative on fire response, as described below.

LANDIS model

The LANDIS model simulates spatial forest dynamics including forest succession, seed dispersal, species establishment, various disturbances, and their interactions (Mladenoff and He 1999; Gustafson et al. 2000).

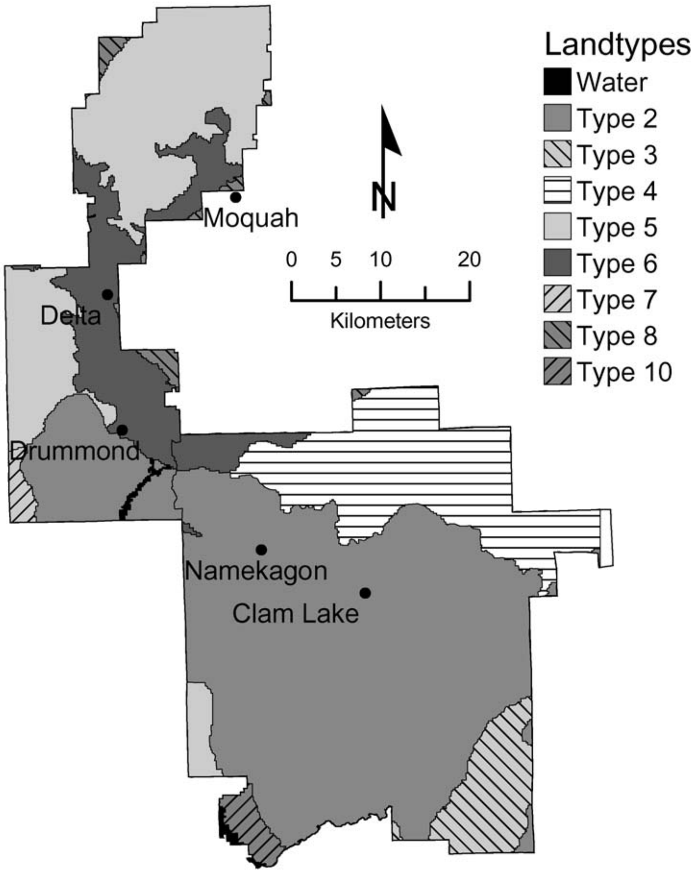

Figure 2. Map showing land type boundaries within the study area. See Table 1 for descriptions.

Table 1. Characteristics of the land types found within the study area.

Land Type	Climate Zone[1]	Soil type	Fire return interval (yrs)[2]	Fire ignition coefficient[2]	Fire probability coefficient[2]	Time since last fire (yrs)[3]
2	I	Mid-well drained silt	200	0.92	128	94
3	I	Somewhat poorly drained silty clay	220	0.88	141	112
4	I	Mid-well drained fine sandy loam	660	0.5	292	130
5	II	Mid-well drained sand	100	1.0	250	47
6	II	Mid-well drained silt	200	0.92	133	94
7	I	Mid-well drained slty loam	320	0.8	136	130
8	III	Mid-well drained silty clay	140	0.95	160	74
10	IV	Mid-well drained silty clay	180	0.93	126	89

[1]Climate zones were delineated using multiple regression techniques (Host et al. 1996). From zone I to IV, both January mean temperature and March mean precipitation (30 year average) tend to increase; [2]Refer to text, and to He and Mladenoff (1999a) for definitions; [3]Value assigned to each cell within the land type, as an initial condition.

The purpose of LANDIS is to simulate the reciprocal effects of disturbance processes (fire, wind, vegetation management) and patterns of forest vegetation on each other across large (10^4 − 10^7 ha) landscapes and long time scales (50 − 1000 years). The model operates on a raster (grid) map, where each cell contains information on the presence (and absence) of tree species and their 10-year age-cohorts (species – age list), but not information about the number or size of individual stems. The model requires mapped land types, and parameters for species establishment, fire characteristics, and fuel accumulation regimes for each land type.

The model simulates differential reproduction, dispersal, and succession patterns using the vital attributes of species, and incorporates effects of disturbance and environmental heterogeneity interacting spatially across the landscape (Mladenoff and He 1999). There is feedback between disturbance and species response. For example, windthrow events may contribute to fuel accumulation on a site, increasing the severity of subsequent fire events and altering the species composition relative to sites without windthrow. Forest succession is simulated based on seed dispersal, seedling establishment (and sprouting), competition, growth, and mortality characteristics of species as described in the literature. The design and behavior of the succession components of the model, and model test results, are described in detail elsewhere (He et al. 1999a; He et al. 1999b; He and Mladenoff 1999; Mladenoff and He 1999).

The forest harvest module of LANDIS allows simulation of disturbance by vegetation management activity. Harvest activity is specified independently for each Management Area. The LANDIS data structure is rich in site information, allowing the heterogeneity of stands to be expressed as heterogeneity both within cells and among the cells that comprise a stand. This structure allows flexible simulation of a wide range of management activities. The user specifies the details about how timber management activities selectively remove age-cohorts of each species on harvested cells. The order in which stands are selected for harvest is based on ranking algorithms that can be related to specific management goals. These features provide the ability to simulate an almost unlimited variety of vegetation management activities to achieve various management goals. Succession on harvested cells is simulated based on the residual species and age classes both on the cell and on dispersal from other cells. Prescribed fire may also be simulated using the harvest module. The timber harvest module of LANDIS is described in detail by Gustafson et al. (2000).

The LANDIS model simulates wind and fire disturbance regimes based on user-specified parameters for wind and fire events on each land type. These parameters are spatially implemented on the landscape using a stochastic algorithm to approximate a desired return interval across the land type over a long-temporal scale (e.g., ≥ 100 years) (He and Mladenoff 1999). LANDIS sequentially simulates windthrow, fire, harvesting, and forest succession at each 10-yr time step.

The fire simulation algorithm in LANDIS is based on the observation that fire appears to be stochastic for a single site, but has repeated patterns in terms of ignition rates, location, size, and shape at landscape scales. Simulation of fire ignition in LANDIS involves selection of random locations and stochastic ignition attempts at those sites. The number of cells selected for ignition attempts (*IgN*) is determined by the ignition coefficient (*IgN* = *ignition coefficient* × *total cell number*). The ignition coefficient sets the proportion of cells in which ignition is attempted, and it can be adjusted to reflect the ignition frequency characteristics of the study area. A successful ignition may or may not occur, depending on the probability of fire (*P*) computed using the mean fire return interval (*MI*) for the land type, and the time since last fire (*lf*) on the cell:

$$P = B \cdot lf \cdot MI^{-(e+2)}$$

B is the *fire probability coefficient*, and it is used for model calibration to ensure that stochastically simulated fire events follow a known historical or empirical distribution (He and Mladenoff 1999). Ignitions are more likely to occur on cells with shorter *MI* and as *lf* increases. Once an ignition is successful, the probability of having subsequent ignitions decreases exponentially (He and Mladenoff 1999).

Fire spread is a process that integrates 1) fire probability (*P*), 2) fire susceptibility (forest age in the cell, is a surrogate for fuel accumulation on a given land type), 3) the ability of a species to survive a fire (fire tolerance class), and 4) the spatial configuration of forest conditions. If a species is sufficiently old or fire tolerant, it will not die if fire severity is low. If no trees on a cell are susceptible to fire, the cell will not burn. Once a fire ignition has occurred, the fire spreads randomly to adjacent sites based on their susceptibility to fire, with a bias toward the wind direction randomly chosen at the time of ignition. Once a fire spreads to a given cell, the cell may or may not be ignited depending on whether a randomly generated fraction (*Pr*) is larger than the fire probability (*P*) of the cell. Fire spreads until either the randomly generated fire size is reached, or the surrounding cells cannot burn ($P<Pr$). Fires are more likely to spread to cells with high *P*, and can spread across land type

Table 2. Description of the management alternatives simulated using LANDIS. The alternatives were developed for the Chequamegon-Nicolet National Forest Plan revision process, except the 'no-harvest' baseline alternative (G), which was developed for comparative purposes for this study.

Alternative	Management objective
A	Emphasize early-successional habitat (aspen)
B	Decrease aspen and increase hardwoods
C	Emphasize ecosystem restoration
D	Emphasize saw timber (pine and hardwoods)
E	Increase pine and decrease aspen
F	Increase hardwoods and restore ecosystems
G	No harvest (baseline alternative)

boundaries where *P* changes. As a result, fire shape is not deterministic, but is the result of interactions among species, fuel accumulation, fire size, and fire probability. The amount of fuel accumulation is assumed to increase with age (time since last fire (*lf*)), with the rate of increase varying according to land type characteristics, and is used to determine the fire severity class on the cell. The species- and age-cohorts killed by fire on a cell are determined by interactions among the species-specific fire tolerance classes of the species found on the cell, the fire susceptibility (determined by age classes), and fire severity (He and Mladenoff 1999).

Simulation Inputs

The six management alternatives we simulated were a subset of the nine draft alternatives developed by the Plan Revision Team of the CNNF as of early 2001 (Table 2). The alternatives consist of spatially delineated Management Areas (MAs) with specific management objectives for each unit (Table 3). The objectives are to be achieved through generic management prescriptions, which include specific combinations of vegetation treatments (Table 3). The alternatives differ in the amount and spatial arrangement of the various MAs (Table 4), but the prescriptions are the same for a particular MA designation across alternatives (i.e., only the MA maps differ among alternatives; see Figure 3). We also simulated a "no-harvest" alternative that was not proposed by the CNNF, for comparative purposes.

Input maps for LANDIS were derived from existing spatial databases, and were gridded to a 60 m cell size. We used GIS coverages provided by the CNNF to generate input maps of stand and MA boundaries. Initial forest composition maps (spatially explicit species and age-cohort data) and land type maps were based upon those used by He et al. (1999a). The spatial location of dominant species was derived from a classified TM image (Wolter et al. 1995), and then randomly assigned age classes and associated species (by land type) to match the statistical distributions found in Forest Service Forest Inventory and Analysis (FIA) data (Hansen 1992) as described by He et al. (1999a). However, rather than assigning initial conditions on a cell by cell basis, we initialized all cells in a stand with the same initial condition, with the condition of each stand randomly assigned from the FIA statistical distributions. The land type characteristics (Table 1) were partly based on values developed by He et al. (1999a). For example, the probabilities of species establishment on cells within a land type were derived by He et al. (1999a) using the LINKAGES model (Pastor and Post 1986). The demographic parameters (e.g., longevity, dispersal range, age of first reproduction) of the 23 tree species found on the study area were also developed for that previous study. Windfall return intervals were derived from a regional historical and empirical study (Canham and Loucks 1984). Our fire disturbance values were not based on the pre-settlement fire regime (He et al. 1999a), but rather a modern fire regime reflecting current fire suppression practices. We used fire records from 1928-1969 for a portion of our study area, and modern fire records over a larger area from 1985 to 1995 (Cardille et al. 2001b) to estimate the average number and size of fires per decade (e.g., mean fire return interval) in each land type (Table 1). We developed LANDIS parameters from these data to simulate a modern fire regime. The time since last fire (*lf*) in each land type was set equal to half of the mean return interval used (He et al. 2000). However, for land types where *lf* exceeded 130 years (4 and 7) we set $lf = 130$ because virtually the entire study area was logged and burned in the late 19th century (Stearns 1997). Finally, the fire probability coefficient (*B*) values were adjusted iteratively for each land type to produce a simulated mean return interval that matched the modern fire return interval (Cardille et al. 2001a) using the procedures of He and Mladenoff (1999). The result was a set of fire probability coefficients that produced an average pattern of burns most closely matching the historical data.

Table 3. Management Area (MA) objectives and the percentage of each MA that was treated in each decade by various silvicultural treatments. MA boundaries for each alternative are shown in Figure 3. MA designations were developed by the CNNF Planning team.

MA	Management objective	Clear-cut	Shelter wood	Selection-60 m gaps	Selection	Prescribed fire
1A	Early Successional Aspen	9.0	1.0	1.0	6.0	0.0
1B	Early Successional Aspen-Conifer	6.5	1.0	1.0	4.0	0.0
1C	Early Successional Aspen-Hardwood	4.5	2.0	2.0	12.0	0.0
2A	Uneven-aged Northern Hardwoods	0.5	0.5	4.0	23.0	0.0
2B	Uneven-aged Northern Hardwoods Interior	0.1	0.0	0.0	32.0	0.0
2C	Uneven-aged Northern Hardwoods Early Successional	2.0	1.0	4.0	16.0	0.0
3B	Even Aged Hardwoods: Oak-Pine	1.0	2.0	4.0	10.0	1.5
3C	Even Aged Hardwoods: Oak-Aspen	3.0	3.0	3.0	7.0	0.75
4A	Conifer: Red-White-Jack Pine	4.0	2.0	2.0	5.0	0.5
4B	Conifer: Natural Pine-Oak	1.0	2.0	0.5	2.0	1.0
4C	Surrogate Barrens Jack Pine-Aspen	13.0	2.0	0.5	1.0	0.8
5	Wilderness	0.0	0.0	0.0	0.0	0.0
6A	Semi-Primitive Non-Motorized	0.05	0.0	0.0	0.0	0.0
8C	Moquah Barrens and Riley Lake	0.5	0.0	0.0	0.0	10.0
8D	Wild Scenic and Recreational Rivers	0.0	1.0	0.5	5.0	0.0
8E	Research Natural Areas	0.0	0.0	0.0	0.0	1.25
8F	Small Natural Areas	0.0	0.0	0.0	0.0	1.0
8G	Old Growth Areas	0.0	0.0	0.0	2.0	2.0

Table 4. Percent of the study area designated to each Management Area (with associated prescriptions) under each alternative.

MA	Alternative					
	A	B	C	D	E	F
1A	11.65	10.60	10.39	10.21	9.94	10.37
1B	7.26	3.40	3.24	4.75	3.78	3.24
1C	8.24	5.35	5.17	6.30	6.13	5.17
2A	2.90	8.79	6.87	11.07	12.36	7.29
2B	1.59	10.54	12.70	7.95	7.64	14.12
2C	28.20	14.69	13.57	13.48	9.91	13.61
3B	0.93	2.51	1.91	0.93	2.48	2.47
3C	7.95	4.75	4.75	8.15	8.42	4.75
4A	14.76	14.88	13.40	15.93	14.45	13.40
4B	0.00	1.15	2.17	0.00	1.13	2.17
4C	1.67	1.67	1.67	1.67	1.67	1.67
5	1.66	1.65	1.65	1.68	1.66	1.66
6A	1.54	4.48	5.10	2.44	3.03	2.67
8C	1.92	1.84	1.84	1.84	1.84	1.84
8D	1.02	1.02	1.00	1.02	1.00	1.00
8E	3.15	3.03	3.03	3.03	3.03	3.03
8F	4.30	4.31	4.33	4.30	4.33	4.33
8G	1.26	5.37	7.21	5.26	7.21	7.21

Simulation approach

Our objective was to examine the likelihood that fires occurred on cells where conditions were appropriate for ground fires to convert into canopy fires (probability of canopy fire). However, LANDIS v3.6 is designed to simulate fire severity solely as a function of time since the last fire. In our study area, fires typically are either a ground fire or a canopy fire, dependent more on the presence or absence of flammable conifers and blowdowns than on the time since the last fire. To achieve our objectives, we designed an unconventional application of LANDIS and relied on extensive post processing of LANDIS output to calculate the risk of canopy fire. We first simulated surface fires to generate a pattern of fire occurrence, and then linked the location of fires to a map delineating cells containing "high-risk fuel" as described below. This approach evaluated the risk of surface fires entering cells containing fuels expected to result in a canopy fire. However, LANDIS is not typically used to simulate the spread of fires sustained by understory vegetation or tree litter. We added a "grass" species to every cell so that surface fires would propagate even when no trees susceptible to fires were present. The "grass" species was always susceptible to burning and it did not limit the establishment or competitive ability of any tree species. We assigned a common, flat fuel accumulation curve to all land types so that fuel loads were independent of time since last fire, and allowed all tree species (excluding grass) to survive any fire. This insured that all of the simulated fires spread to their maximum size solely as a probabilistic function of the fire characteristics defining the land type (Table 1).

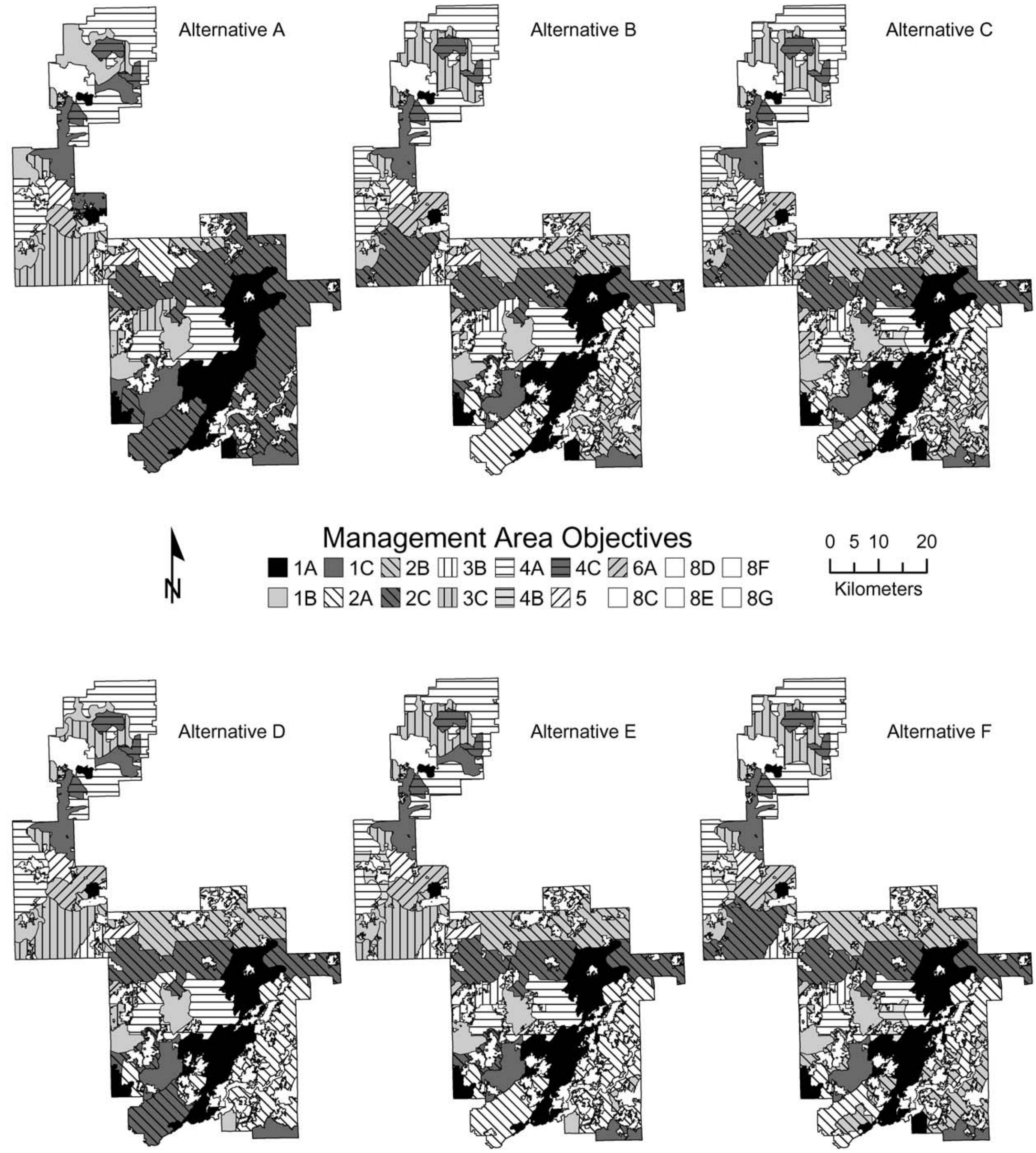

Figure 3. Map of the Management Area boundaries for each alternative.

To calculate the probability of crown fire we used the pattern of the simulated fires to sample a map of sites containing "high-risk fuels" where conditions were appropriate to support a canopy fire. This high-risk fuels map was generated by post-processing LANDIS output of species composition, age structure, and wind damage. High-risk fuels sites included all sites affected by recent (within the last 30 years) wind disturbance, and/or the presence of any conifer that might transfer a low intensity surface fire into the canopy. However, the susceptibility of conifer species to crown fire varied by their characteristic branching

Table 5. Non-parametric analysis of variance of the response of the probability (per decade) of canopy fire (ranks) to the main effects. F-tests were computed for the imbalanced random-effects model according to the Satterthwaite approximation (Steel and Torie 1980). $R^2=0.22$

Source of variation	df	Type III SS	F	Prob $>$F	Expected mean square
Land type (l)	7	12358989072	203.45	<0.0001	$\sigma_e^2+137.75\ \sigma_{la}^2+964.27\ \sigma_l^2$
Management alternative (a)	6	74903097	2.12	0.049	$\sigma_e^2+24.68\ \sigma_{la}^2+197.45\ \sigma_a^2$
l*a	42	364489712	1.65	0.005	$\sigma_e^2+137.75\ \sigma_{la}^2$
Error (e)	9219	48632313035			σ_e^2
Total	9274	61957720256			

structure. Jack pine, balsam fir, white spruce (*Picea glauca*), and northern white cedar (*Thuja occidentalis*) retain low branches that are always susceptible to crowning during a fire. However, red (*Pinus resinosa*) and white pine (*P. strobus*) are self-pruning and can therefore “escape” fire damage if trees are of sufficient age, which we assumed was 80 years. Eastern hemlock (*Tsuga canadensis*) can also escape fire damage (Burns and Honkala 1990), but only if trees are much older (>200 years). We also assumed that the youngest cohort (<10 years) of any conifer would not create enough heat to transfer a fire into the canopy.

We worked with silviculturists familiar with management practices on the CNNF, and with information contained in the 1986 Land Management Plan (USDA Forest Service 1986) and the 1999 draft Management Area Prescriptions and Standards and Guidelines (USDA Forest Service 1999) to develop LANDIS harvest parameters to simulate the harvest prescriptions for each MA. Details of the development of the parameters to simulate the prescriptions are found elsewhere (Zollner et al. in prep). We simulated harvest on the non-National Forest inholdings within the boundaries of each Ranger District because a lack of harvest activity would produce large blocks of late successional sugar maple that could unrealistically alter burning patterns. In the absence of data on harvest practices on these lands, we assumed that management practices on these non-National Forest lands are similar to those on adjacent National Forest land.

Analysis of model outputs

To predict the spatial variation in susceptibility to canopy fires, we produced maps showing the combined results of all 50 replicates of each alternative as an indicator of fire probability. Fire probability maps recorded how often each cell experienced both fire and the presence of high-risk fuel over the 250 simulated years and across the 50 replicate simulations. Our analysis assumes that these values are proportional to the susceptibility of each cell to canopy fire under the alternative management plans. To generate a crude estimate of the relative risk of fire in the human communities in proximity to the CNNF, we calculated the mean probability of canopy fire on the cells within 5 km of the center of 5 selected towns in the wildland-urban interface within the study area, for each alternative. These communities either appear on the Federal list of communities at high risk from wildfire, or are in close proximity to CNNF lands (Figure 1).

To determine the significance of any differences in the probability of canopy fire among alternatives, we conducted an analysis of variance with land type ($n=8$), and management alternative ($n=7$) as the main random effects. We fitted the imbalanced random-effects model using the Satterthwaite approximation (Steel and Torie 1980). We used the cell as the unit of analysis, and randomly selected 0.2% of the cells ($n=1325$) in the study area to satisfy assumptions of independence of observations for linear regression models. We sampled the same 1325 cells (defined by row and column) for each alternative. We were unable to produce a normal distribution of the probability of canopy fire using common data transformations (Sokal and Rohlf 1969), so we conducted a non-parametric analysis of variance on the ranked values of the probability of canopy fire (SAS Institute 1990, p. 1196).

Results

The probability of canopy fire varied considerably by land type (Table 5). This was expected because fire parameters varied by land type. Tukey’s tests ($\alpha=0.05$) showed that the fire-resistant land type (4) had a significantly lower canopy fire probability than most other land types, that the fire-prone land type (5)

Table 6. Mean probability per decade of a canopy fire by land type and management alternative. Probabilities were derived from a sample of 0.2% of the cells in the study area over 50 replicates. Tukey groupings with the same letter are not significantly different ($\alpha = 0.05$).

	Probability of canopy fires			Tukey
	N	Mean	Std Dev	groupings
Land type				
2	3983	0.00173	0.00162	ab
3	427	0.00157	0.00144	b
4	1533	0.00031	0.00060	d
5	1918	0.00220	0.00204	a
6	1148	0.00061	0.00100	c
7	42	0.00305	0.00046	cd
8	112	0.00144	0.00132	b
10	112	0.00067	0.00101	c
Management alternative				
A	1325	0.00155	0.00163	a
B	1325	0.00144	0.00161	ab
C	1325	0.00142	0.00165	b
D	1325	0.00149	0.00168	ab
E	1325	0.00144	0.00165	ab
F	1325	0.00144	0.00165	ab
G	1325	0.00120	0.00177	c

had a significantly higher canopy fire probability than most other land types, and the intermediate land types clustered into two groups that were significantly different from the others (2,3,8) and (6,7,10) (Table 6). The effect of Management alternative was subtle, but significant (Table 5). Tukey's tests showed that alternative A (aspen emphasis) had a significantly higher risk of canopy fire than the others, and that alternative G ('no harvest') had a significantly lower probability of fire than the CNNF alternatives. The interaction between land type and management alternative was also significant, suggesting that the impact of alternatives varied by the land type on which they were implemented. We provide the mean probability of canopy fire by levels of the main effects to illustrate the relative magnitude of the effects (Table 6). Variation within land types was caused by the composition and age structure of the forest resulting from interactions among initial conditions, management activities (Management Area prescriptions) and succession. The pattern of this variation differed only slightly among CNNF management alternatives (refer to Figure 3 for alternative Management Area patterns).

The spatial pattern of the probability of canopy fire reveals the pattern of the underlying land types (Figure 4, Figure 2). Land types with short fire return intervals and high ignition coefficients tended to have a higher probability of canopy fire (Table 1, Table 6). For example, land type 4 had the longest mean fire return interval (660 yrs, Table 1), and exhibited the lowest probability of canopy fire (Figure 4).

The human communities in the wildland-urban interface face varying fire risks, depending primarily on the land types surrounding the community (Figure 2, Figure 4). The effect of management alternative on the fire risk to human communities was relatively small (Table 7), with Alternative A usually producing a slightly higher risk, and the "no harvest" alternative usually the lowest risk. However, the "no-harvest" alternative has similar risk to the other alternatives for the town of Moquah, and the town of Delta had a relatively higher risk of canopy fires under the "no harvest" alternative. We believe that the Delta exception was related to the harvest prescription. Under the aspen emphasis prescription (1C) around Delta, moderately shade tolerant conifers did not persist because of short rotation clear-cutting. The "no-harvest" alternative interacted with the initial conditions around Delta (young balsam fir present) and blowdowns in the aging, uncut forest that were more severe, allowing the persistence of conifers. In other parts of the land type (Figure 2) under "no harvest," the young balsam fir cohorts were not initially present.

Discussion

Long-term landscape-level fire risk in managed forests is complicated by spatial and temporal interactions among multiple ecological and anthropogenic processes. In this study, we investigated the risk of crown fire under a fire regime that was consistent at the landscape scale. Under this simplifying assumption, the cell-level risk of crown fires became a function of 1) the presence/absence of high-risk fuels on a given cell, and 2) the likelihood of a fire occurring on that cell. Because the fire regime was consistent across all simulated management alternatives, the observed differences in crown fire risk was an emergent property of the various factors affecting successional patterns – specifically, the presence or absence of either windthrow mortality or conifer cohorts capable of transmitting a surface fire into the canopy. These factors included both nonspatial processes (e.g., succession as a function of shade and species establishment probabilities), and spatial processes (e.g., seed dispersal and harvesting decisions). In examining the

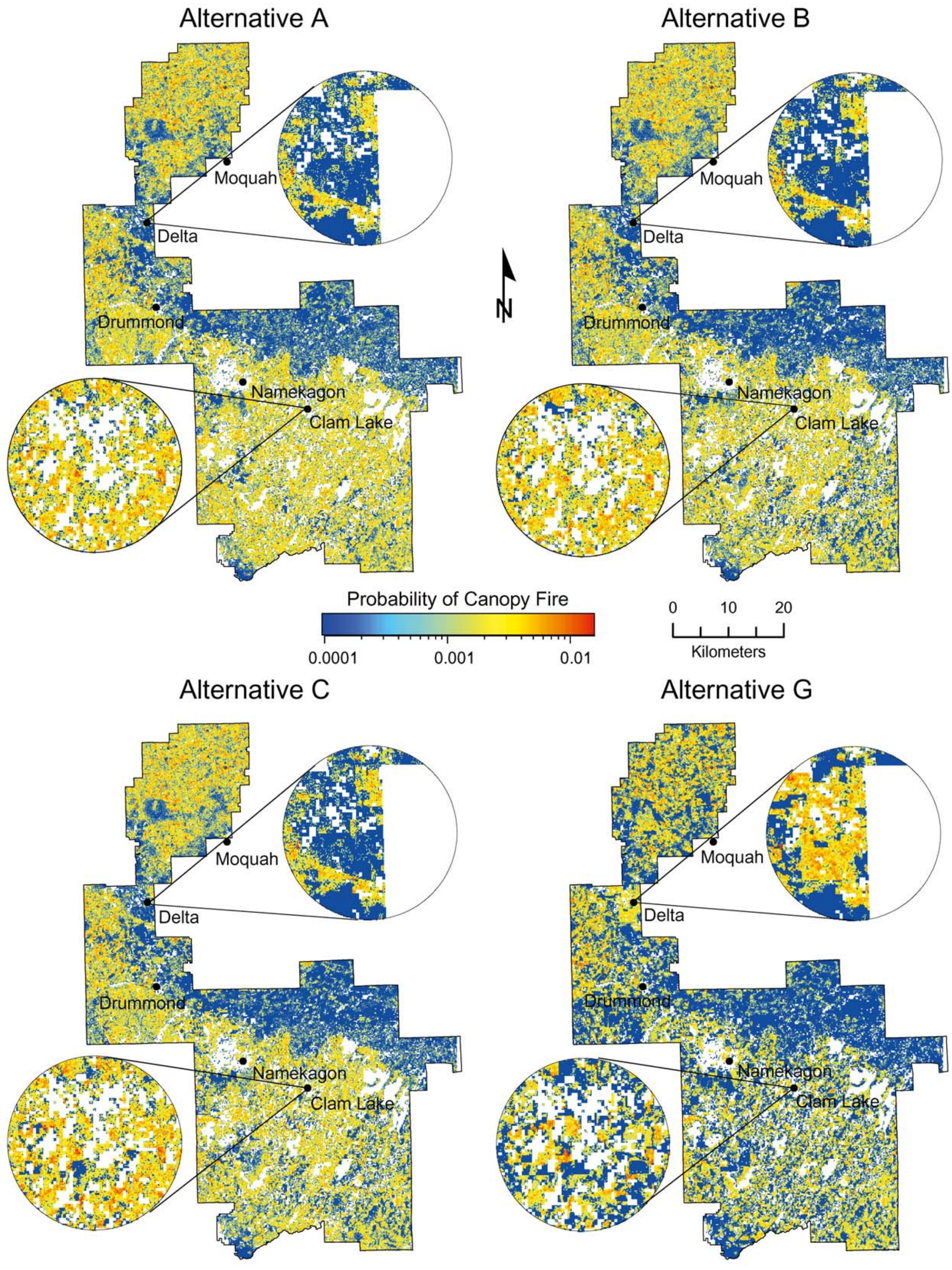

Figure 4. Maps showing the probability (per decade) of canopy fire across the 50 replicate simulations for four alternatives, selected to show the range of canopy fire risk. The alternatives are displayed left to right and top to bottom in descending order of mean fire probability (see Table 6). Insets show the 5 km radius area analyzed around two towns in the urban-wildland interface having markedly different response to the 'no-harvest' alternative.

Table 7. Mean probability (per decade) of canopy fire by management alternative (A-G) within 5 km of communities in the wildland-urban interface in the study area. Probabilities represent the proportion of times (across 50 replicates) that each cell within 5 km of town center burned in the presence of high-risk fuels over the 250-year simulation.

	Alternative													
	a		b		c		D		E		F		G	
Community	Mean	Std. dev.	Mean	Std. dev.	Mean	Std. dev.	Mean	Std. dev.	Mean	Std. dev.	Mean	Std. dev.	Mean	Std. dev.
Moquah	0.0011	0.0014	0.0010	0.0013	0.0010	0.0013	0.0010	0.0013	0.0010	0.0013	0.0010	0.0013	0.0011	0.0016
Delta	0.0007	0.0012	0.0007	0.0011	0.0007	0.0012	0.0007	0.0012	0.0007	0.0012	0.0007	0.0011	0.0016	0.0017
Drummond	0.0013	0.0015	0.0013	0.0015	0.0013	0.0015	0.0013	0.0014	0.0014	0.0015	0.0013	0.0015	0.0009	0.0014
Clam Lake	0.0020	0.0015	0.0019	0.0015	0.0019	0.0015	0.0019	0.0015	0.0019	0.0015	0.0019	0.0015	0.0012	0.0016
Namekagon	0.0017	0.0015	0.0015	0.0014	0.0015	0.0014	0.0015	0.0014	0.0015	0.0014	0.0015	0.0015	0.0012	0.0016

details of our simulation results, we found that 1) management alternatives can modify the abundance of high-risk fuel conditions to significantly change the landscape-level risk of canopy fires, and 2) land type can influence the impact of management strategies as it relates to fire risk. In a previous study using LANDIS, the aging of forests across a landscape resulted in larger and more severe fires that spread from more fire-prone land types into less fire-prone land types (He and Mladenoff 1999). Radeloff et al. (in review) used LANDIS to investigate how different silvicultural practices can be altered to best mimic patterns of natural fire disturbance. Taken together, these results suggest that management activities that control age and composition patterns can change the risk of fire.

Forest managers seek strategies to mitigate fire risk without compromising other benefits and services expected by society from public and industrial forests. In our simulations we found that land type exerted the dominant effect on fire risk, but the management alternatives also had a significant effect on risk of canopy fires. Alternative A had the highest probability of canopy fire (Table 6) because its emphasis on even-age cutting prescriptions reduced sugar maple and other species that do not readily support canopy fires. The mean probability of canopy fire appears to rank the alternatives in order of intensity of disturbance created by the prescriptions (Table 6). The species associated with high-risk fuels (conifers) tend to be favored by disturbance, and the primary deciduous competitor (i.e., sugar maple, a low-risk species) requires low disturbance levels (See also Sturtevant et al. this volume). On most land types, the "no-harvest" scenario (Alternative G) resulted in stands dominated by sugar maple, which ultimately excluded most conifer species. The CNNF alternatives were not designed specifically to moderate fire risk, so the limited difference among them is not surprising. A positive implication of this result is that none of the alternatives exhibited unexpectedly high risk of canopy fire across the study area. We believe that management strategies specifically designed to mitigate fire risk could be quite effective, and that LANDIS could reliably predict the spatial distribution of risk.

We assessed the risk of crown fire by post-processing LANDIS output of species composition, age structure, and wind damage, and canopy fires were not actually simulated. Therefore, our methodology influenced successional pathways on sites where fires should have been stand replacing. However, because crown fires are rare in this ecosystem relative to harvest activities, we are confident that land type and management effects, rather than model artifacts, dominate our results. Our quantitative results are consistent with our understanding of how forest communities respond to silvicultural treatments, and how resistant those communities are to canopy fires. For example, timber harvest maintained shade intolerant tree species that are resistant to surface fire mortality (e.g., oak, aspen) but promote canopy fires (e.g., pines). The “no-harvest” scenario encouraged community transition to shade tolerant species that are vulnerable to surface fire mortality but do not promote canopy fires (e.g., sugar maple). These interactions between forest management, succession and canopy fires suggest that harvest activity can influence the impact of fires in forests within the region.

Simulations in this study were parameterized to emulate the modern fire regime, where the total area burned has decreased an order of magnitude from presettlement estimates due to an effective fire suppression policy (MacLean and Cleland in press). Recent studies in the Lake States suggest that, unlike in drier western forest ecosystems, fire suppression may actually reduce the risk of fire over successional time by decreasing the flammability of forests as they transition from pine to northern hardwoods (Frelich and Reich 1999). Sturtevant et al. (this volume) found that fire suppression had less influence on the distribution of conifers than timber harvest, and that the response of conifers to either fire suppression or harvesting depended on environmental conditions (i.e., land type). More empirical and modeling studies are needed to fully address the subtle interactions between forest harvest, fire suppression, and fire risk in the northern Lake states.

LANDIS is a stochastic model and replication of simulations is an important means to estimate variability. The model was not designed to predict particular spatial events. However, the spatial patterns derived from long-term simulations are the outcome of the complex interaction of spatial processes (dispersal, fire disturbance, and forest harvesting), landscape configuration, and the initial spatial distribution of each species. At the land type or landscape level, these results reveal the expected outcomes of various management strategies in a spatially explicit manner. The large number of replicates used in our study served to identify areas where risk of canopy fire should be greater as a function of forest successional response to management alternatives. The discovery of the interaction between management prescription and land type for the risk of canopy fire around the town of Delta is a good example of how well replicated simulation model runs can provide insights that would otherwise remain hidden. Furthermore, threats to specific communities in the wildland-urban interface under management alternatives can be identified.

An important focus of our study was to study the interactions between the abiotic environment and management strategies, and how they affect fire risk in specific parts of the landscape (e.g., wildland-urban interface). We found that the interaction is statistically significant, but more importantly, the spatially explicit results produced by LANDIS allowed us to visualize how these interactions were manifest across the study area. For example, Figure 4 shows an increase in the probability of canopy fire near the town of Delta under Alternative G, but not under the other alternatives. This is the result of an interaction among the initial conditions, the management prescription of Alternatives A-F (lots of clear-cutting) and wind disturbance. The harvest prescription reduced the severity of blowdowns, and also discouraged the persistence of the highly flammable balsam fir, which reduced fire risk. With “no harvest” simulated under Alternative G, the existing balsam fir was not eliminated, and blowdowns allowed it to persist. Results such as these highlights the value of LANDIS for identifying specific locations where interacting factors of land type and management strategy increase fire risk, providing important information to guide the development of land management policy.

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