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Can landscape-level ecological restoration influence fire risk? A spatially-explicit assessment of a northern temperate-southern boreal forest landscape

Douglas J. Shinneman [a,b,*], Brian J. Palik [b], Meredith W. Cornett [a]

[a] The Nature Conservancy, 394 Lake Ave. South, Duluth, MN 55802, United States
[b] USDA Forest Service, Northern Research Station, 1831 Hwy. 169 East, Grand Rapids, MN 55744, United States

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ABSTRACT

Management strategies to restore forest landscapes are often designed to concurrently reduce fire risk. However, the compatibility of these two objectives is not always clear, and uncoordinated management among landowners may have unintended consequences. We used a forest landscape simulation model to compare the effects of contemporary management and hypothetical restoration alternatives on fire risk in northern temperate and southern boreal forests of the Border Lakes Region in Minnesota, USA, and Ontario, Canada. Six main model scenarios simulated different combinations of timber harvest, fire exclusion, wildland fire use, and prescribed fire. Mean fire risk values were calculated as a function of high risk fuel type occurrence, fire events, and windthrow events over model time, and were compared among scenarios and among major management areas. Our model results indicate that a continuation of contemporary management, with limited wildland fire use, would increase fire risk over time and lead to greater continuity of high-risk fuel types in parks and wilderness areas. Compared to the contemporary management scenario, greater use of wildland fire in a historical natural disturbance scenario and three alternative restoration scenarios resulted in less spatially aggregated high-risk fuels over time and lower long-term fire risk in parks and wilderness. Outside of parks and wilderness, prescribed fire with logging was effective at reducing fire risk on portions of the landscape in two restoration scenarios, largely by maintaining deciduous tree dominance and fire-tolerant red and white pine stands, and timber harvest alone maintained patches of less fire-prone deciduous forests in some scenarios. However, forest restoration and fire risk objectives were not always compatible, especially when restoration of fire-prone forest conflicted with the goal of reducing risk of large, severe fires. Both fire risk reduction and forest restoration objectives will benefit from spatially coordinated, landscape-level planning among landowners.

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1. Introduction

Reintroduction of fire is central to many forest restoration efforts, both as a tool to achieve desired objectives and ostensibly to minimize fire risk through reductions in fuel loads and fire-prone fuel types (Allen et al., 2002). Large conservation reserves containing fire-dependent ecosystems may provide practical opportunities for the use of wildland fire to meet restoration objectives (Baker, 1994; Kneeshaw and Gauthier, 2003), while adjacent, intensively-managed or human-dominated landscapes may require silvicultural or prescribed fire strategies (Lindenmayer et al., 2006). However, disparate forest management activities among landowners or management areas can create sharply contrasting landscape patterns of forest composition (Tinker et al., 2003) and fuel types (Drobyshev et al., 2008). Unintended consequences of spatially uncoordinated activities can detract from meeting forest restoration and fire management objectives at landscape scales and may limit restoration options (Lytle et al., 2006).

It may be particularly important to consider landscape-scale interactions between management activities and spatial arrangement of fire-dependent forest types (Sturtevant et al., 2009a). For instance, restoration of fire-prone ecosystems in parks and wilderness may conflict with objectives to reduce risk of wildfire on adjacent developed areas or commercial timberlands (Radeloff et al., 2005; Suffling et al., 2008). These conflicting objectives may be especially prone in landscapes historically shaped by high-severity, stand-replacing fire regimes, such as boreal forests. Although fire behavior models applied at landscape scales have indicated that strategic modification of fire-prone forest structures through

* Corresponding author. Current address: U.S. Geological Survey, Forest and Rangeland Ecosystem Science Center, Snake River Field Station, 970 Lusk St., Boise, ID 83706, United States. Tel.: +1 208 426 5206; fax: +1 208 426 5210.
E-mail address: dshinneman@usgs.gov (D.J. Shinneman).

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timber harvest or prescribed fire may reduce susceptibility to severe fire (e.g., Suffling et al., 2008; Beverly et al., 2009), these studies assume single ownership objectives for the landscape in question. Few studies (e.g., Sturtevant et al., 2009a) have specifically assessed fire-risk across multi-landowner landscapes as a response to disparate ecological restoration strategies.

The goal of this research was to assess the potential effects of regional-scale restoration strategies on fire risk in the heavily forested Border Lakes Region (BLR) of northern Minnesota and northwestern Ontario. We used a forest landscape simulation model to assess the degree to which contemporary management and forest restoration alternatives, as modeled and presented previously for the BLR by Shinneman et al. (2010), might differ over time in their potential influence on three fire variables: fire occurrence, fuel type distributions, and mean fire risk (the latter defined by potential interactions between fire occurrence and fuel types). To investigate the response of these three variables, we compared six alternative management scenarios that included various combinations of contemporary forest harvest, restoration activities, and wildfire use. For two restoration scenarios, we also tested the potential to reduce fire risk in portions of the landscape, where stand-replacing fire is less desirable, by using prescribed fire in hypothetical management zones that straddle boundaries of park and wilderness areas adjacent to developed areas and more intensively harvested timberlands. Comparing the potential effects of alternative management and restoration scenarios may be particularly useful in the BLR, where several major landowners are seeking to move fire-prone forest ecosystems toward their ranges of natural variability via different strategies (Ontario Ministry of Natural Resources, 2001; Minnesota Forest Resources Council, 2003), while reducing fire risk to timberlands and developed areas (USDA Forest Service, 2004).

2. Modeling approach and methods

The ~2.1 million ha Border Lakes Region (BLR) in northern Minnesota and northwestern Ontario (Fig. 1) occupies a transition zone between northern temperate and southern boreal forests, with warm, short summers and long, cold winters (Heinselman, 1996). An area of modest topographic relief, the shallow soils of the BLR are underlain by glacially-scoured Precambrian bedrock of the Canadian Shield. Freshwater lakes are a prominent feature of the landscape. Common conifer tree species include jack pine (*Pinus banksiana*), black spruce (*Picea mariana*), white spruce (*Picea glauca*), balsam fir (*Abies balsamea*), red pine (*Pinus resinosa*), white pine (*Pinus strobus*), white cedar (*Thuja occidentalis*), and tamarack (*Larix laricina*). Deciduous trees mainly include paper birch (*Betula papyrifera*), aspen (*Populus tremuloides*, *P. grandidentata*), balsam poplar (*P. balsamifera*), red maple (*Acer rubrum*), black ash (*Fraxinus nigra*) and, in the southwestern portion, northern pin oak (*Quercus ellipsoidalis*).

Prior to EuroAmerican settlement, a stand-replacing fire regime supported fire-adapted, early-successional species, including jack pine, aspen, and paper birch. Stand replacing fire sizes were generally 400–4000 ha, but some likely exceeded 100,000 ha (Heinselman, 1973, 1996). Mean fire rotation was ~50–75 years for jack pine-black spruce forests and ~75–150 years for wetland (e.g., spruce bogs) and mixed-wood (aspen-birch-spruce-fir) forest types (Heinselman, 1973; Beverly and Martell, 2003). Some areas experienced longer fire-free intervals that supported late-successional forests of spruce, fir, and cedar (Heinselman, 1973; Frelich and Reich, 1995). Old white pine and red pine stands were likely maintained by smaller (40–400 ha), low- to moderate-severity fires that occurred every 5–100 years on average, but also experienced severe crown fires every 150–350 years (Heinselman,

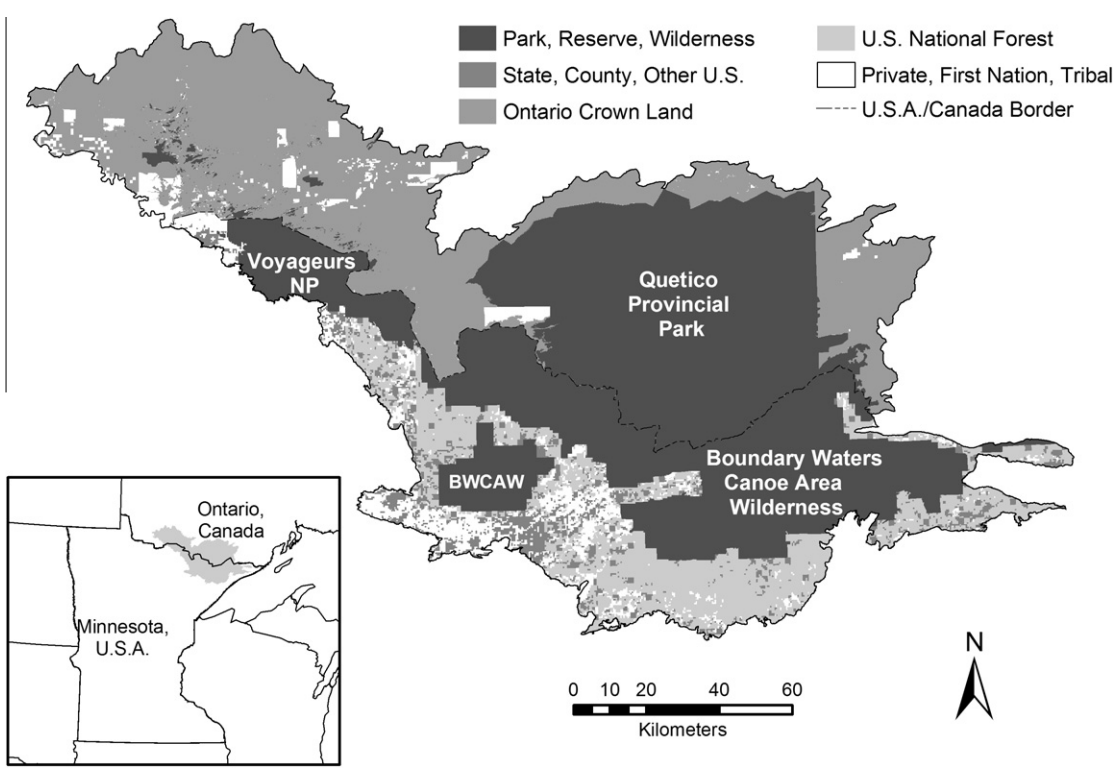

Fig. 1. The Border Lakes Region and major land ownership. NP = National Park; BWCAW = Boundary Waters Canoe Area Wilderness. Parks and wilderness areas are largely undeveloped, private lands within the BLR have generally scattered or lakeshore development, including a few small towns, and most of the remainder of the region is primarily managed for timber harvest.

1973, 1996). Since the early 20th century, fire exclusion nearly eliminated large crown-fires (Heinselman, 1996; Beverly and Martell, 2003) until several, recent, large (>10,000 ha) fires in the Quetico Provincial Park-Boundary Waters Canoe Area Wilderness (QPP-BWCAW) region. Windthrow events were generally small and infrequent, but a rare "blowdown" event in 1999 disturbed >200,000 ha (Frelich, 2002).

Roughly 93% of the BLR is publicly-owned, with 43% within the QPP-BWCAW and Voyageurs National Park conservation area complex (Fig. 1). Although historical logging occurred in portions of the QPP-BWCAW, fire exclusion shifted forest composition from an extensive mix of early- to mid-successional jack pine and aspen to extensive mixed-age, multi-species stands transitioning to late-successional spruce and fir (Frelich and Reich, 1995). Outside of protected areas, extensive timber harvest has generally increased shade-intolerant, aspen-dominated forests (Heinselman, 1996; Friedman and Reich, 2005). Throughout the BLR low- to moderate-severity fires historically maintained old red and white pine stands, but overstory dominance by these species has been substantially reduced by early logging and slash fires, followed by recent large fires and the 1999 blowdown event. Although fire exclusion is a primary management objective across most of the BLR, limited wildland fire use (i.e., allowing some naturally ignited wildfires for management objectives) is permitted in remote portions of parks and wilderness. Timber harvest activities range from no logging in parks and wilderness to short-rotation, even-age harvest using large clearcuts outside of protected areas (Shinneman et al., 2010).

2.1. The forest landscape simulation model

We simulated changes in forest composition, landscape pattern, and disturbance regimes resulting from forest management and disturbance scenarios using LANDIS-II (version 5.1), a spatially-explicit forest landscape simulation model that simulates seed dispersal, species establishment, succession, and natural and anthropogenic disturbance events (Scheller et al., 2007). Similar to other LANDIS models (Mladenoff et al., 1996), LANDIS-II is a raster-based model that simulates interactions among processes and tracks species age cohorts over broad temporal and spatial scales. Each cell in the model represents uniform solar radiation conditions, and cells are aggregated into ecoregions representing consistent climate and soil conditions. Successional pathways are nondeterministic, based on species cohort interactions, response to disturbance events, and growing conditions. LANDIS-II simulates disturbance and successional processes via various model extensions that require user-specified input parameters for each ecoregion (Scheller et al., 2007). We used the age-only succession, base fire, base wind, and base timber harvest extensions. Timber harvest parameters include frequency, species age-cohort removal targets, patch size targets, and post-harvest planting (Gustafson et al., 2000). Base wind requires inputs for windthrow frequency, severity, and size (Scheller and Mladenoff, 2004).

The base fire extension uses several input parameters, including fire spread age, fire size (min, mean, max), and ignition probabilities for each user-defined fire region to calibrate fire frequencies, size-class distributions, and fire rotations (He and Mladenoff, 1999). Fire ignition events are stochastically generated at each time step, by comparing ignition probabilities to a randomly selected number. Successful ignition and subsequent fire spread are determined by a probabilistic relationship between time since last fire for a cell and mean fire-spread age (fire return interval) for the fire region. Thus, successful fire ignition and spread are more likely within regions parameterized for higher ignition probabilities and shorter fire return intervals, and will become more likely as time since fire approaches or surpasses the mean fire return interval. Random wind speed and direction influence spread to adjacent cells, and a fire event will either attain a randomly selected size or extinguish due to insufficient suitable cells to burn.

Fire and wind curve tables are parameterized to approximate the effects of fuel accumulation and decay rates on fire severity since last disturbance, such that greater accumulation (or decay) within a cell leads to more (or less) severe fire. Species cohort mortality from fire depends on the fire severity class (1–5, least to most severe), cohort age, and species fire tolerance (He and Mladenoff, 1999). All cohorts are killed in a class 5 fire, otherwise progressively younger cohorts are most vulnerable as fire severity decreases, with precise mortality distributions based on user-defined relationships between fire severity, cohort age, and species' fire tolerance.

2.2. Model inputs and scenarios

Detailed descriptions of non-fire model inputs are provided in Shinneman et al. (2010) and are only described generally here. An initial forest input map was created at a 100 m × 100 m (1 ha) resolution to represent extant BLR forest communities (Bauer et al., 2009) that were further delineated by growth stages derived from Frelich (2002). Each community type-growth stage was then assigned a list of species-age cohorts (in 10-year age classes). The resulting initial forest map reflected coarse-scale patterns of common forest community type-growth stages. Tree species successional and reproductive traits (longevity, seed dispersal, shade tolerance, fire tolerance, and ability to sprout vegetatively) were delineated based on previous LANDIS models and other relevant sources (Table 1). LANDIS-II requires tree species establishment probabilities (SEPs) that range from 0.0 to 1.0 and that can vary for each species among user-defined ecoregions (He et al., 1999). We assigned SEPs to land type associations using a soils-based ecosystem model (Pastor and Post, 1986) that required input parameters for species attributes, monthly climate data, soil conditions, and geographic location.

Six model scenarios were developed to investigate the potential to restore or move forest conditions closer to the range of natural variability (Shinneman et al., 2010), while also exploring the influence of management activities on fire risk under varying degrees of cross-boundary coordination. A 10-year time step was used to simulate all processes, and each scenario was simulated for 200 years to observe potential, long-term, cumulative effects of management on fuel types and fire risk across a large landscape. Validation for each scenario followed a calibration-based approach, as described in Shinneman et al. (2010). We modeled five replicates per scenario to allow variability in the patterns created by disturbance differences among runs but, because disturbance regimes were tightly parameterized to achieve desired rotations and size class distributions, the variance in disturbance and forest composition response variables was not substantial (and not reported) among replicates. Thus, even though disturbance spatial patterns varied among replicates, there was only minor differences in total area affected over time (e.g., the coefficient of variation for total area burned among replicates ranged from 2.2% to 11.2% among scenarios). For all scenarios, windthrow parameters were set at 1 ha, 93 ha, and 3600 ha for minimum, mean, and maximum sizes respectively, with a rotation of 1000 years (based on Frelich, 2002).

Disturbance regimes for the six management scenarios are described here and in Table 2, with detailed harvest descriptions and parameters provided in Shinneman et al. (2010). The *contemporary management scenario* simulated timber harvest for private, tribal, county, state, provincial, and federal forests, derived from management plans and harvest reports. Fire size and rotation parameters among six distinct fire regions (Table 3, Fig. 2) reflected limited wildland fire use in parks and wilderness and fire exclusion

Table 1
Species life history attributes and range of species establishment probabilities (SEPs) among the 30 upland and four wet forest ecoregions.

Tree species	Longevity (years)[b]	Sexual maturity (years)	Shade tolerance[c]	Fire tolerance[c]	Seed dispersal distance (m)[d]		Vegetative reproductive probability	Sprouting age		Post-fire regeneration[e]	SEP range	
					Effect-ive	Maxi-mum		Min	Max		Upland (range)	Wet forest
Abies balsamea	180	25	5	1	30	160	0.00	0	0	None	0.35–0.62	
Acer rubrum	150	10	4	1	100	200	0.50	0	100	None	0.01–0.19	
Betula papyrifera	120	30	2	2	200	5000	0.50	0	70	None	0.50–0.62	
Fraxinus nigra	250	30	4	1	70	140	0.10	0	70	None	0.00–0.58	0.0–0.9
Larix laricina	180	40	1	1	40	60	0.00	0	0	None	–	0.0–0.9
Picea glauca	200	25	4	2	30	200	0.00	0	0	None	0.55–0.62	
Picea mariana[a]	200	25	4	1	80	200	0.00	0	0	None	0.65–0.99	0.0–0.9
Pinus banksiana	120	15	1	3	20	100	0.00	0	0	Serotiny	0.66–0.98	
Pinus resinosa	270	35	2	4	20	275	0.00	0	0	None	0.14–0.91	
Pinus strobus	270	40	3	3	60	210	0.00	0	0	None	0.03–0.62	
Populus tremuloides	120	20	1	1	200	5000	0.90	0	100	Resprout	0.59–0.62	
Quercus ellipsoidalis	250	35	2	2	30	3000	0.75	30	250	None	0.00–0.20	
Thuja occidentalis[a]	300	30	5	1	45	60	0.50	50	300	None	0.25–0.62	0.0–0.9

[a] Modeled as two separate species, both as a wetland and upland type.
[b] Typical life expectancy without disturbance.
[c] Rank-ordered (1 = highly intolerant, 5 = highly tolerant).
[d] Effective = 95% of dispersal.
[e] Serotiny refers to the opening of cones and release of seeds due to heat from fire.

Table 2
Overview of disturbance and timber harvest dynamics simulated in each of the six scenarios (greater number of X-symbols indicate greater area affected by each process among scenarios).

Process	ND	CND	CM	RM1	RM2a	RM2b
Post-harvest planting			X	X	X	XX
Timber harvest			X	X	X[b]	X[b]
Low/mid-severity fire	XX			X	X	X
High-severity fire	XXX	X	X	XX	XX	XX
Prescribed fire[a]					X	X
Windthrow	X	X	X	X	X	X

Note: See text for windthrow parameters; see Table 3 for fire parameters; see Shinneman et al. (2010) for timber harvest and planting parameters. *Abbreviations for scenarios:* CM, current management; RM, restoration management (refer to text for differences between 1, 2a, and 2b); CND, contemporary natural disturbance; ND, historical natural disturbance.
[a] Scenarios RM2a and RM2b were modeled both with and without prescribed fire.
[b] Timber harvest occurred across ownership boundaries.

policies elsewhere. *Restoration management scenario 1* used the same timber harvest parameters as the contemporary management scenario, but differed by emulating pre-EuroAmerican fire regimes in parks and wilderness, using relatively short rotation, stand-replacing, crown fire regimes in most forest types, and low-severity fires (Table 3) in large patches of red and white pine (Fig. 2). Low severity fire regimes were parameterized to simulate class 1 fires (least severe) on short rotation, and low-severity fire locations were static in the model, representing areas deliberately managed for red and white pine restoration. *Restoration management scenario 2a* simulated pre-EuroAmerican fire regimes as in restoration scenario 1, but also simulated timber harvest that ignored ownership boundaries and emulated natural disturbance patterns and severities via partial harvest in red and white pine stands and large clearcuts in boreal forest types. *Restoration management scenario 2b* was identical to restoration management 2a, with the exception of increased rates of post-harvest planting of conifers in clearcuts. The restoration management scenarios 2a and 2b were also simulated with and without the use of prescribed fire, as described below. The total rate of harvest via each cutting technique in restoration scenarios was comparable to rates in the contemporary management scenario. To gauge the potential influence of fire alone (without the effects of timber harvest) on landscape patterns and fuel types, the *contemporary natural disturbance scenario* simulated fire exclusion and limited wildland fire use via the six distinct fire regions used in the contemporary management scenario. The *historical natural disturbance scenario* also simulated fire without logging, using fire parameters for the entire BLR similar to those used for park and wilderness areas in the restoration scenarios, but maximum fire size was increased to 30,000 ha and low-severity fire regions were extended 100 m beyond extant red pine and white pine patches. Fire in the historical natural disturbance scenario was calibrated for the entire 200 year period to approximate historical fire regimes, while fire in other scenarios was calibrated for the first 100 years only, to allow fire behavior to respond to management.

Restoration management scenarios 2a and 2b were both modeled with and without a 10-km wide prescribed-fire zone that straddled boundaries of parks and wilderness occurring within 5 km of paved or primary gravel roads or developed areas (Fig. 2). This resulted in an "inner" and "outer" moderate-severity prescribed-fire zone that existed inside and outside of large parks and wilderness and constituted 16.6% and 17.8 % of the BLR, respectively. To simulate a realistic burning schedule and to avoid burning the prescribed fire zones all at the same time, the zones were further divided by their spatial overlap with the primary fire regions (Fig. 2). Only one of the overlapping areas was prescribe-burned at each 10-year time step, with each overlapping area "activated" for prescribed fire four times (fifty-year intervals) during the 200-year model period. Prescribed fire was parameterized to burn most of the area activated, resulting in a 75-year prescribed-fire rotation over the duration of the model (Table 3). The fire curve table was adjusted in these zones (fire severities 2, 3 and 4 were allowed to occur 10, 60, 120 years after last fire, respectively) to achieve desired prescribed fire mortality effects, including: $\leqslant$50% mortality in lower severity fires for mid- to older-aged red and white pine stands; generally 50% to 85% in moderate to moderately-high severity fires for older red and white pine stands; and 85–100% for fire-intolerant species (e.g., aspen). When not activated for prescribe-fire, the primary fire region parameters were reactivated, and fuel curves reflected a stand-replacing fire regime again (with fire severity classes 4 and 5 able to occur 10 and 20 years after fire, respectively).

Table 3
Target fire parameters for each fire region, with the percent land area of each fire region by scenario.

Fire region[a]	Target mean rotation	Target max fire size	% land area by Scenario[c]				
			ND	CND	CM	RM1	RM2 (a&b)
BLR-wide high severity[b]	130	20,000–30,000	93.2			41.7	41.7
Pine regions low severity	50	2000	6.8			3.0	3.0
NE	750	5000		8.4	8.4	8.4	8.4
NW	2000	6200		21.6	21.6	21.6	21.6
QPP/BWCAW	630	22,000		44.4	44.4		
VNP	1050	200		2.8	2.8	2.5	2.5
SE	15,000	700		8.4	8.4	8.4	8.4
SW	2000	1500		14.4	14.4	14.4	14.4
Inner Rx zone/moderate severity	75	2000					16.6
Outer Rx zone/moderate severity	75	2000					17.8

Note: Abbreviations for scenarios: CM, current management; RM, restoration management; CND, contemporary natural disturbance; ND, historical natural disturbance.
[a] See Fig. 2 for fire region locations and acronyms.
[b] Maximum fire sizes were 20,000 and 30,000 for restoration (RM1, RM2) and Natural Disturbance scenarios, respectively.
[c] All columns total to 100%; comprising all land area (1,526,560 ha) in fire regions, except for RM2a & b, in which the prescribed fire zones were superimposed onto the other fire regions. See text for details and full names for scenario abbreviations.

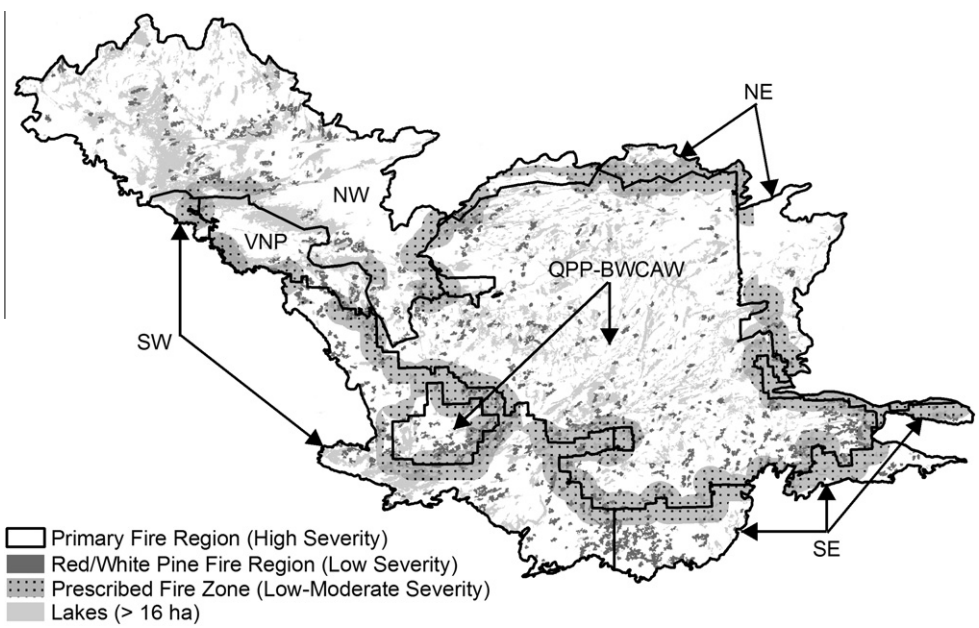

Fig. 2. Fire regions (non-forest cover types are not shown). High-severity fire regions are delineated by solid lines and labeled by their position (e.g., NE) or management name acronym. For the historical natural disturbance scenario, all of the low-severity fire regions were used (i.e., fire was simulated). For restoration scenarios, only the low-severity fire regions within the Quetico Provincial Park/Boundary Waters Canoe Area Wilderness (QPP/BWCAW) and Voyageurs National Park (VNP) fire regions were used. For the current management and contemporary natural disturbance scenarios, no low-severity fire regions were used. For the historical natural disturbance scenario, the entire BLR forest area was treated as a single high-severity fire region. Prescribed fire zones were only used in the alternative versions of restoration scenarios 2a and 2b. Table 3 provides corresponding fire region parameters respective to each scenario.

2.3. Analysis methods

Fire occurrence, fuel types, and mean fire risk were compared spatially and temporally among scenarios, between parks/wilderness and the remainder of the study area (primarily managed for timber), and for the prescribed fire zones in the restoration management scenarios 2a and 2b. Similar to Gustafson et al. (2004), mean fire risk was calculated for each cell as the sum of 10-year periods that experienced high severity fire (severity classes 4–5), an occurrence of a high risk fuel type, or a windthrow event within the last 20 years, divided by the total number of model periods. Thus, 10-year mean fire risk values reflect additive or potential interactive contributions of fire, fuel type, and windthrow to long-term fire risk. Potential mean values range from 0 (no high risk conditions over 200 years) to 3 (all three high risk conditions every ten years over 200 years). For instance, a cell with a fire-prone forest type that experienced windthrow during a given model period obtained a higher fire risk value (=2) than a cell with just a fire-prone forest type (=1). To represent inherent fire risk in all forested cells, and to avoid computational difficulties for statistical analysis, we reassigned cells with a calculated mean fire risk value of zero to a value of 0.01. High risk fuel types were based on species composition and age structures that would most likely support a crown fire, and included stands dominated by boreal conifers (jack pine, boreal spruce, balsam fir), young (<81 years old) red and white pine stands, and older pine stands with a presence of balsam fir (>40 years old) that could serve as a ladder fuel. To assess spatial patterns of high risk fuels among scenarios, an aggregation index (He et al., 2000) was calculated using APACK (Mladenoff and Dezonia, 2004) for all cells with persistent (occurring >75% of model time) high-risk fuel types.

To compare fire risk among different management alternatives, mean fire risk values were derived from a random sample representing 0.5% of the entire forested landscape cells ($n = 7631$).

Sample cells were selected from a grid of cells systematically spaced 1 km apart, in order to minimize potential influence of spatial autocorrelation. These same random cell locations were used for each averaged fire risk map generated by each scenario. Because mean fire risk values could not be transformed to normal distributions using standard procedures (Sokal and Rolf, 1995), we performed non-parametric comparisons and accounted for unequal variances (Ruxton 2006). We used ANOVA on rank-ordered mean fire risk values for each of the six scenarios, coupled with pairwise *t*-tests for unequal variances. Welch's *t*-tests were used on rank-ordered values for the random cells in the inner ($n = 642$) and outer ($n = 1144$) prescribed fire zones to test for differences between use of prescribed fire and no use (for restoration scenarios 2a and 2b). All test statistics were performed with the R statistical package (version 2.14.1; R Development Core Team, 2011) and assessed for significance at $\alpha = 0.01$.

3. Results

3.1. Temporal and spatial patterns of fire occurrence and high-risk fuel types

Regularly occurring large fires in the historical natural disturbance and restoration management (1, 2a, 2b) scenarios resulted in a relatively large total area burned by severe fires (severity classes 4–5) during the first 20–30 years, as extant older forests were particularly vulnerable to fire, followed by a decreased but variable rate of burning (Fig. 3) as forest age class distributions approached quasi-steady states on the landscape. In contrast, fire exclusion in the current natural disturbance and contemporary management scenarios substantially increased the rate of burning after year 60 (Fig. 3; a threefold increase in area burned from the first to second 100-year period), caused by an increase in older, unburned, fire-prone forest.

As expected due to parameterization, total area burned by severe fires (classes 4, 5) over the 200-year model period varied greatly among scenarios. Total area burned in the restoration scenarios was nearly double that of the contemporary management and current natural disturbance scenarios; however, the actual proportion of the landscape affected by fire was only 10–14% greater because the total area includes cells that burned more than once (Table 4). The historical natural disturbance scenario burned a total of ~2.1 million ha, nearly twice that of the restoration scenarios, and more than three times that of the contemporary management and current natural disturbance scenarios (Table 4). The historical natural disturbance scenario also had the largest portion (82.6%) of the landscape affected by fire, with approximately 46% burning more than once, compared to ~19–26% of the landscape area burning more than once in the restoration and <1% in the contemporary management and current natural disturbance scenarios (Table 4). The total area burned was evenly distributed among parks/wilderness and timber-managed areas in the historical natural disturbance scenario, while ~60% of the total area burned was in parks and wilderness in the contemporary management scenarios, and ~80% in the restoration scenarios (Table 4).

The total area with high risk fuel types (Table 5) varied considerably among scenarios over the 200-year model period. High-risk fuel types that persisted in a cell for more than 75% of the total model period occupied 59% of the forested area in the current natural disturbance scenario, ~43% in the historical natural disturbance and contemporary management scenarios, and from 33.5% to 40.6% in the restoration scenarios. The restoration scenarios had the greatest proportion of cells with least persistent (<25% of model period) high risk fuel types. The proportion of the entire BLR within high risk fuel types in parks and wilderness for more than 75% of model time was highest (~30%) for the contemporary management and current natural disturbance scenarios, where limited wildland fire use occurred. Cells with persistent high risk fuel types were least spatially aggregated across major management areas in the restoration scenarios, most spatially aggregated across the entire BLR in the current natural disturbance scenario, and most variably aggregated among major management areas in the contemporary management scenario (Table 5).

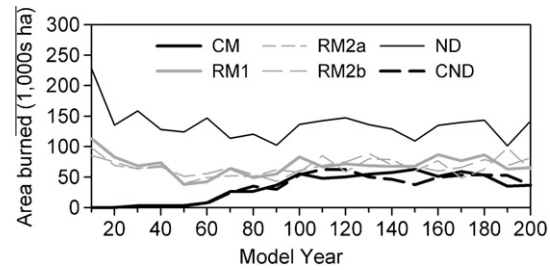

Fig. 3. Total area burned over time in severe fires (severity classes 4–5) by scenario. Abbreviations for scenarios: CM, current management; RM, restoration management; CND, contemporary natural disturbance; ND, historical natural disturbance.

3.2. Mean fire risk

Overall mean fire risk among scenarios for the entire BLR (Fig. 4) was driven largely by variability in fire risk for timber-managed areas (Fig. 5a). For the entire BLR, the current natural disturbance scenario (no timber harvest) and restoration management 2a scenario (harvest that produced large aspen patches) had the highest and lowest mean fire risk, respectively (Fig. 5a). Mean fire risk in wilderness was lower in the restoration and historical natural disturbance scenarios compared to contemporary management and current natural disturbance (i.e., fire exclusion) scenarios. Areas managed for timber had lower mean fire risk than parks/wilderness among all scenarios (Fig. 5a). Similarly, regardless of whether prescribed fire use was simulated in restoration scenarios 2a and 2b, mean fire risk in the outer prescribed fire zone where timber harvest also occurred was substantially lower than the inner prescribed fire zone in parks and wilderness (Fig. 5b). However, fire risk did decrease slightly in both zones in both restoration scenarios when prescribed fire was simulated, significantly so ($t = -4.03$, df = 2286, $p < 0.001$) for restoration scenario 2b in the outer zone and approaching significance ($t = -1.88$, df = 2276, $p = 0.06$) in the outer zone for restoration scenario 2a.

4. Discussion

4.1. Potential influence of forest restoration on spatial patterns of fire risk

A key consideration for restoration management and fire suppression activities is the spatial arrangement of fire prone or fire-dependent forest types (Sturtevant et al., 2009a). Forest landscape simulation models can be readily used to project long term effects of planned management activities on broad-scale fire risk patterns (Gustafson et al., 2004; Sturtevant et al., 2004), whereas field-based research may be impractical for such objectives. In our model, scenarios that simulated extensive fire exclusion or limited wildland fire use, including the current natural disturbance and contemporary management scenarios, produced increasing area burned over time (Fig. 3), more area within high risk fuel types that were more aggregated on the landscape (Table 5), and the highest mean fire risk where fires were suppressed (Figs. 4 and 5). Without other major disturbance events, continued fire exclusion will likely

Table 4
Percent area and area burned for each scenario over the 200-year model period (% in parks and wilderness demonstrates relative distribution of fires between more developed and less developed portions of the study area).

Number of times burned	Percent of entire BLR by scenario					
	ND	CND	CM	RM1	RM2a	RM2b
1	37.4	41.9	41.4	33.3	35.6	34.7
2	37.3	0.6	0.4	18.6	16.7	17.6
3	7.5	0.0	0.0	2.7	2.1	2.2
4	0.8	0.0	0.0	0.2	0.2	0.2
Total	83.0	42.5	41.8	54.8	54.5	54.7
	Total area burned in BLR by scenario					
Area (ha)	2104,102	657,029	645,299	1212,982	1159,331	1178,101
% in parks/wilderness	50.5	57.4	61.5	81.0	79.9	80.4

Note: The total percent of the entire study area represents all cells that burned at least once (i.e., irrespective of how many times); the total area (ha) burned in all fires is the sum of all areas of all fires; and the percent of area burned within parks and wilderness is derived from the total area burned in all fires. Data represent means of all model replicates for each scenario. *Abbreviations for scenarios:* CM, current management; RM, restoration management; CND, contemporary natural disturbance; ND, historical natural disturbance.

Table 5
Percentage of the study area within high-risk fuel-types by categories of persistence (i.e., number of model periods), and aggregation index (AI) values for highly-persistent (>75% of model period) high-risk fuel types, delineated by major management areas for each scenario over the 200-year model period.

No. model periods[a] in high-risk fuel	Percent of entire BLR (% in wilderness areas) by scenario[b]					
	ND	CND	CM	RM1	RM2a	RM2b
<6	21.1 (9.4)	9.6 (4.1)	19.1 (4.0)	23.1 (8.1)	27.5 (8.1)	21.3 (8.7)
6–10	15.4 (6.9)	9.6 (5.0)	17.9 (5.4)	19.6 (7.3)	21.9 (8.3)	16.9 (7.3)
11–15	20.1 (8.4)	21.9 (9.5)	19.8 (8.5)	19.3 (8.0)	17.2 (7.4)	21.2 (8.5)
16–20	43.5 (22.5)	59.0 (28.7)	43.2 (29.3)	38.0 (23.8)	33.5 (23.4)	40.6 (22.8)
Major mgt. area	Aggregation index (AI)[c] for persistent, high-risk fuel types					
Entire BLR	0.66	0.82	0.78	0.66	0.66	0.68
Parks/wilderness	0.67	0.82	0.81	0.66	0.66	0.66
Timber-managed	0.65	0.81	0.66	0.67	0.62	0.72

Note: Abbreviations for scenarios: CM, current management; RM, restoration management; CND, contemporary natural disturbance; ND, historical natural disturbance.
[a] $n = 20$ (10-year periods).
[b] Numbers not in parentheses total to ~100% (the total BLR study area), and number in parentheses total to ~47.2% (the portion of the BLR in parks and wilderness), not adjusted for rounding error.
[c] AI values range from 1 (highest level of aggregation, pixels in a class share the most possible edges) to 0 (complete disaggregation, pixels in a class share no edges). Data represent means of all model replicates for each scenario.

lead to greater dominance by late-successional conifers at landscape scales in the BLR (Scheller et al., 2005; Shinneman et al., 2010), and greater aggregation of high risk fuel types (Table 5) could lead to larger and more severe fires. Similar changes have already occurred in the BLR due to historical fire exclusion and represent a successional shift away from pre-EuroAmerican forest conditions at landscape scales (Frelich and Reich, 1995). In contrast, shorter fire rotations reflective of the pre-EuroAmerican period, as simulated in the historical natural disturbance scenario and for parks and wilderness in the restoration scenarios (1, 2a, 2b), resulted in restoration of forest composition and structure, including greater interspersion of fire-prone jack pine patches with less fire-prone aspen patches (Shinneman et al., 2010). Thus, compared to fire-suppressed areas, there is less continuity of high risk fuels (Table 5), which may result in lower long-term fire risk (Fig. 5a). However, although short-rotation, stand-replacing fire regimes may be acceptable in remote wilderness, large and severe fires are typically suppressed on developed and timber-managed landscapes, as well as within park and wilderness border zones to prevent fire escape (Suffling et al., 2008).

Timber harvest outside of parks and wilderness has increased aspen dominance beyond historical levels in the BLR (Friedman and Reich, 2005), and aspen stands are generally less fire prone than conifer (Li, 2000). Restoration management scenario 2a, which simulated large clearcuts with modest post-harvest conifer planting rates, created and maintained large patches of aspen dominated forests (Shinneman et al., 2010) and produced the lowest mean fire-risk (Fig. 5a). In contrast, restoration scenario 2b, with greater rates of post-harvest conifer planting, produced large patches of fire-prone jack pine and spruce forests and the highest mean fire risk in timber-managed areas (Fig. 5a). Thus, although restoration scenario 2b better reflects ecological restoration objectives (Shinneman et al., 2010), the resulting fire-prone forests on large portions of the landscape may conflict with objectives to reduce fire risk. Sturtevant et al. (2009a) used LANDIS to arrive at similar conclusions in a northern Wisconsin forest landscape, determining that conflicts between preventing fire in an expanding wildland urban interface and using fire to restore fire-dependent pine and oak forests could limit restoration options without careful spatial management considerations, including intentional redistribution of fire-prone forests.

4.2. Potential influence of prescribed fire on fire risk

In our model, severe fire risk was reduced slightly further when prescribed fire was used, though only significantly so outside of parks and wilderness where timber harvest also occurred (Fig. 5b), mainly by maintaining or increasing area within fire-tolerant red and white pine stands and younger, post-harvest, deciduous stands (Shinneman et al., 2010). Prescribed fire or timber harvest could be used to maintain early, deciduous-dominated successional stages, and to restore older red and white pine forests historically shaped by low- to moderate-severity fires, making these stands less vulnerable to crown fire (Beverly and Martell,

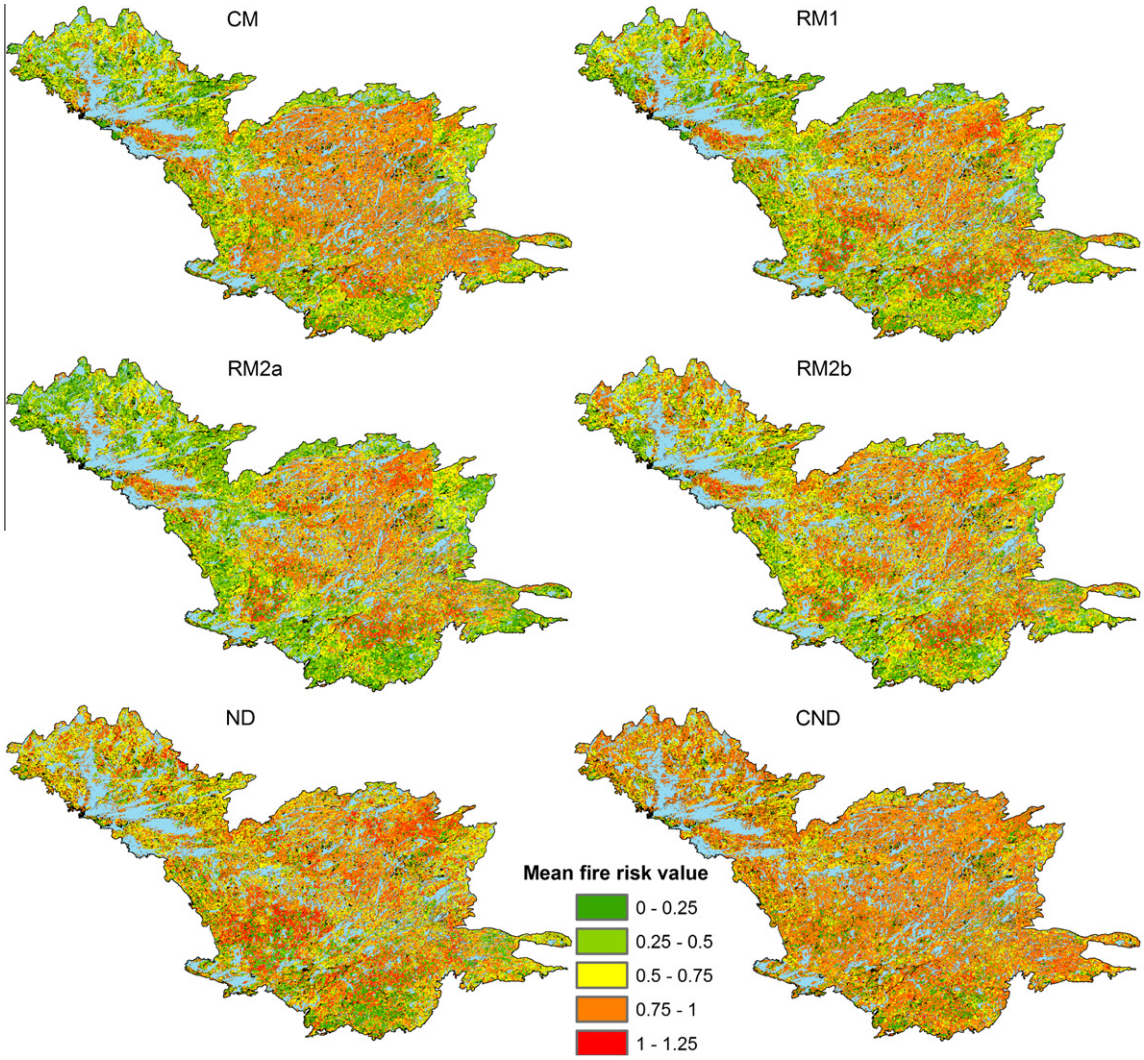

Fig. 4. Mean fire risk by scenario over 200 year model period (blue color represents lakes). Abbreviations for scenarios: CM, current management; RM, restoration management; CND, contemporary natural disturbance; ND, historical natural disturbance.

2003). Beverly et al. (2009) used a fire-landscape simulation model to predict that prescribed fire could effectively reduce wildfire susceptibility on portions a western boreal forest landscape, depending on topography, conifer composition, and spatial arrangement of fuels. Prescribed fire might also help control fire spread. Suffling et al. (2008) used a spatially-explicit fire behavior model to demonstrate that various configurations of prescribed fire management zones within Quetico Provincial Park could theoretically create fire breaks that reduce the probability of wildfire escaping onto surrounding commercial timberlands.

4.3. Model limitations

While our modeling approach provides a first ever examination of restoration and fire risk interactions across multiple ownerships in the BLR, there are a few caveats worth highlighting. First, the model does not represent realistic individual fire behavior. The base fire extension for LANDIS-II used in our model does not directly track fine- or coarse-fuel loads (Scheller et al., 2007) and lacks dynamic fuel and fire-weather interactions that affect fire behavior. A dynamic fire extension is available for LANDIS-II that simulates fire-fuel-weather interactions (Sturtevant et al., 2009b), but the primary goal of this study was to focus on broad-scale effects of land use on coarse-scale patterns of fire risk, and to avoid introducing unnecessary model complexity via realistic simulations of individual fire behavior.

Second, the model does not represent a comprehensive assessment of practical strategies to reduce fire risk on developed lands, as analyzed elsewhere (e.g., Suffling et al., 2008). Careful spatial considerations and cost-benefit analyses will likely be required, especially for extensive prescribed fire use or for major shifts in timber-harvest objectives; thus, potentially restricting the most ambitious strategies to landscapes where severe fire is least acceptable or ecological restoration is most desirable. Moreover, as a strategic consideration, extreme fire weather conditions in boreal forests may override the influence of fuel loads on crown fire spread (Podur and Martell, 2009) despite fuels reduction efforts. For instance, burning used to reduce fuels after the 1999 blowdown had mixed success in preventing two large wildfires in 2006 and 2007 from spreading beyond the BWCAW and into developed areas, likely due to differences in forest composition, fuel moisture, seasonality, and weather events, rather than differences in fuel loads for each fire (Fites et al., 2007).

Third, key potential disturbance interactions and effects were not simulated in our model, including cyclical spruce budworm outbreaks that may temporarily increase fire risk in the short term,

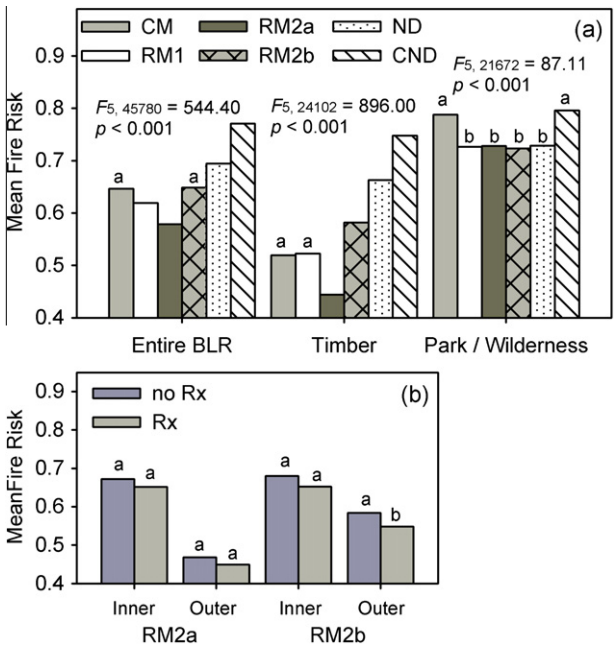

Fig. 5. Comparison of mean fire risk values for major management areas for all scenarios (a), and for prescribed fire zones with and without prescribed fire ("Rx") for the restoration 2a and 2b scenarios (b). Rank-order ANOVA results and statistical comparisons are among scenarios within each management area (pairwise *t*-tests using un-pooled variances and Holm *p*-value adjustment) and between use and no-use of prescribed fire within each prescribed fire zone (Welch's *t*-tests). Mean values that do not differ significantly ($\alpha = 0.01$) share the same letter. Abbreviations for scenarios: CM, current management; RM, restoration management; CND, contemporary natural disturbance; ND, historical natural disturbance.

but may also decrease balsam fir dominance and high-risk fuel types over the long term (Bergeron and Leduc, 1998; Sturtevant et al., in press). Potentially negative effects of timber harvest and prescribed fire were not assessed; neither completely mimics natural disturbance effects, such as post-disturbance patterns of biomass and succession (McRae et al., 2001), and may result in excessive area in young forest (Bergeron et al., 2002; Kneeshaw and Gauthier, 2003), fragmented wildlife habitat (Vors et al., 2007), or deviation from pre-EuroAmerican landscape structure (Baker, 1994). Finally, our model may not be generalizable to northern boreal forests, where fire-weather conditions more than fuel loads are often key determinants of fire behavior and area burned (Podur and Martell, 2009), or to northern temperate forests, where clear-cutting can favor fire-prone species and lack of disturbance favors fire-resistant hardwoods (Gustafson et al., 2004).

Finally, climate change will also likely influence forest communities and long term fire risk in the region (Flannigan et al., 2005) and was not simulated here. For instance, Frelich and Reich (2010) speculated that shifts in moisture availability, increasingly frequent and large disturbance events (e.g., fire and windthrow), and warming temperatures could transform BLR forests to savanna-like woodlands. In contrast, in adjacent portions of northeastern Minnesota, Ravenscroft et al. (2010) used LANDIS-II to project climate-induced shifts away from boreal forest species toward less fire-prone temperate forests dominated by maple. We recognize that further modeling and empirical studies are needed to better predict future, climate-driven forest-fire dynamics, and we are currently engaged in developing new models to investigate these interactions. However, as long as northern forest types dominate the BLR landscape, and given uncertainty regarding the direction and magnitude of future forest dynamics under climate change, the results presented here should prove instructive to land managers engaged in spatially-explicit forest planning meant to restore forests and reduce fire risk.

5. Conclusions

Forest restoration strategies often focus on restoring fire regimes and reducing fuel loads, and the compatibility between these two objectives is often promoted through government policies aimed at reducing fire risk (e.g., the U.S. Healthy Forests Initiative). Yet, forest restoration and fire management objectives are not always compatible, especially in landscapes historically shaped by large, stand-replacing fires. In a previous application of these modeled scenarios, Shinneman et al. (2010) suggested that wildland fire use and timber harvest that emulated natural disturbance patterns and severities across ownership boundaries may produce forest landscape structures and age distributions that reflect the region's range of natural variability. Stand-replacing wildfire use achieved restoration objectives in parks and wilderness, while timber harvest and prescribed fire achieved some restoration goals in more intensively managed landscapes. Yet, results presented here also underscore potential incompatibility between reducing fire risk and achieving forest restoration objectives. Careful consideration must be given to which portions of the landscape should be restored to fire-prone community types and which areas should be managed primarily to reduce fire risk. Cross-boundary coordination among landowners will likely increase opportunities for both successful restoration and meeting fire risk objectives.

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