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Simulating restoration strategies for a southern boreal forest landscape with complex land ownership patterns

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ABSTRACT

Restoring altered forest landscapes toward their ranges of natural variability (RNV) may enhance ecosystem sustainability and resiliency, but such efforts can be hampered by complex land ownership and management patterns. We evaluated restoration potential for southern-boreal forests in the ~2.1 million ha Border Lakes Region of northern Minnesota (U.S.A.) and Ontario (Canada), where spatially distinct timber harvest and fire suppression histories have differentially altered forest conditions (composition, age–class distribution, and landscape structure) among major management areas, effectively resulting in forest landscape "bifurcation." We used a forest landscape simulation model to evaluate potential for four hypothetical management and two natural disturbance scenarios to restore forest conditions and reduce bifurcation, including: (1) a current management scenario that simulated timber harvest and fire suppression practices among major landowners; (2) three restoration scenarios that simulated combinations of wildland fire use and cross-boundary timber harvest designed to emulate natural disturbance patterns; (3) a historical natural disturbance scenario that simulated pre-EuroAmerican settlement fire regimes and windthrow; and (4) a contemporary fire regime that simulated fire suppression, but no timber harvest. Forest composition and landscape structure for a 200-year model period were compared among scenarios, among major land management regions within scenarios, and to six RNV benchmarks. The current management scenario met only one RNV benchmark and did not move forest composition, age–class distribution, or landscape structures toward the RNV, and it increased forest landscape bifurcation between primarily timber-managed and wilderness areas. The historical natural disturbance scenario met five RNV benchmarks and the restoration scenarios as many as five, by generally restoring forest composition, age–class distributions, and landscape structures, and reducing bifurcation of forest conditions. The contemporary natural disturbance scenario met only one benchmark and generally created a forest landscape dominated by large patches of late-successional, fire-prone forests. Some forest types (e.g., white and red pine) declined in all scenarios, despite simulated restoration strategies. It may not be possible to achieve all objectives under a single management scenario, and complications, such as fire-risk, may limit strategies. However, our model suggests that timber harvest and fire regimes that emulate natural disturbance patterns can move forest landscapes toward the RNV.

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1. Introduction

As human activities move ecosystems beyond their ranges of natural variability (RNV), natural resources, ecological services (e.g., water supply, pest suppression), and native species diversity may be threatened (Christensen et al., 1996; Poiani et al., 2000), while the frequency of uncharacteristically severe disturbances may increase (Swetnam et al., 1999). Thus, managing ecosystems within their RNV has been proposed as a key strategy for promoting long-term resource use and ecological sustainability (Aplet and Keeton, 1999; Landres et al., 1999). However, defining and achieving RNV benchmarks may be problematic for several reasons, including uncertainty about historical conditions, effects of climate change, and potential conflicts with resource use and suppression of natural disturbance events (Hobbs and Norton, 1996; Landres et al., 1999; Nonaka and Spies, 2005). Moreover, effective restoration of ecosystem components, structures, and processes that define RNV typically requires a multi-scale approach operating within a landscape or regional context (Poiani et al., 2000; Lindenmayer and Franklin, 2002; Lindenmayer et al.,

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Table 1
Target RNV benchmarks; estimated from previous research.

Attribute	Source
Forest cover type composition (based on %cover of three major forest types: aspen–birch, jack pine, and red/white pine)	Estimated from: Heinselman (1973, 1980, 1996), Swain (1980), Friedman and Reich (2005)
Age–class distribution for southern boreal forest (negative exponential distribution for all forest types combined)	Van Wagner (1978), Bergeron et al. (2002)
Proportion of total forest area in largest patch size by forest type (calculated for two major forest types: red/white pine and jack pine)	Estimated from: Heinselman (1973, 1980, 1996)

2006). For example, restoring or emulating the effects of natural disturbance regimes for forest restoration purposes requires meeting both landscape-level (e.g., distribution of disturbance patches) and within-stand (e.g., live-tree retention) objectives (Bergeron et al., 2002; Mitchell et al., 2002).

Restoration at landscape and regional scales is often further hampered by spatially complex land ownership and management patterns that can create unintended forest patch mosaics and disturbance dynamics (Mladenoff et al., 1993; Nonaka and Spies, 2005) and can constrain potentially useful management options (e.g., fire use) across boundaries (Ward et al., 2005). For instance, landscapes composed of both conservation reserves and timber-managed lands can develop spatially bifurcated forest compositional and structural conditions (e.g., Tinker et al., 2003). Sharply contrasting patterns of forest conditions over a given landscape can result in divergent trends in ecological processes over time and space (Turner et al., 2001), including wildlife population dynamics, disturbance regimes (Franklin and Forman, 1987), and biogeochemical cycles (Valett et al., 2002), and may impede cooperative, multi-ownership management (Sample, 1994; Lytle et al., 2006). Thus, a strategic hurdle for restoration of large forested landscapes is to develop approaches that account for patterns of ownership and management (Lindenmayer and Franklin, 2002; Thompson et al., 2006). For instance, wildland fire may achieve restoration objectives in large conservation reserves with fire-dependent ecosystems (Baker, 1989, 1994; Kneeshaw and Gauthier, 2003), but silvicultural or prescribed fire strategies may be required in human-dominated landscapes (Lindenmayer et al., 2006).

Understanding spatial and temporal interactions among disturbances and disparate management activities may be critical to effectively meet ecological restoration and other management objectives, including sustainable forestry, biodiversity conservation, and wildfire control (Gustafson et al., 2004; Thompson et al., 2006; Syphard et al., 2007). Spatially explicit, dynamic forest landscape simulation models (FLSMs) can elucidate potential effects of alternative management strategies and disturbance interactions on forest composition and landscape structure over large landscapes and long time periods (Mladenoff, 2005). Model outcomes can be examined in relation to desired ecological restoration objectives (Scheller et al., 2005; Shifley et al., 2006; Xi et al., 2008). However, due to challenging parameterization and data requirements (Shifley et al., 2006), only a few forest modeling studies have simulated forest management practices stratified across complex ownership and management patterns at regional scales (Mehta et al., 2004; Nonaka and Spies, 2005; Thompson et al., 2006; Gustafson et al., 2007). Moreover, quantitative comparisons between modeled management scenarios and RNV benchmarks for restoration are problematic, as RNV conditions are often only qualitatively defined or even lacking for many ecoregions (but see Wimberly, 2002; Tinker et al., 2003). In the one published modeling study we are aware of that quantitatively assessed multi-owner land management policies in relation to RNV, Nonaka and Spies (2005) determined that current forest policies in the Oregon Coast Range would not restore forest landscape structure over time, due in part to management constraints among owners, and that several centuries would be required to achieve RNV benchmarks using a wildfire-only policy.

In the present study, we used a spatially explicit, dynamic FLSM to explore restoration options in the southern boreal and northern-mixed forests of the Border Lakes Region (BLR), a multi-ownership landscape in northern Minnesota and northwestern Ontario. Despite complex ownership and management patterns, ranging from large, protected wilderness to intensively managed timberlands, there is a common desire among major landowners to move forest ecosystems toward their RNV in order to meet ecological sustainability objectives (Ontario Ministry of Natural Resources, 2001; Minnesota Forest Resources Council, 2003; USDI National Park Service, 2002; USDA Forest Service, 2004). However, meeting these objectives requires developing appropriate, regional-scale targets for restoration and assessing the feasibility of achieving these targets across complex ownership patterns.

To assess the potential to move the BLR forest landscape toward its RNV, we simulated forest dynamics over a 200-year period under six different scenarios that reflected unique fire and timber harvest regimes. We hypothesized that a restoration management scenario simulating wildland fire within large conservation reserves, and two scenarios that simulated cross-boundary timber harvest emulating natural disturbance patterns, would most effectively move the landscape toward the RNV compared to a current management scenario reflecting fire suppression and no cross-boundary harvest. To distinguish the effects of fire versus timber harvest on forest landscapes, we also modeled a pre-EuroAmerican fire regime scenario with short-rotation fire cycles and a contemporary fire regime scenario reflecting fire suppression and no timber harvest. We speculated that the pre-EuroAmerican fire regime and contemporary fire regime scenarios would move the landscape toward and away from the RNV, respectively. RNV measures included six estimated benchmarks for forest composition, age–class distribution, and patch size that potentially capture key characteristics of the pre-EuroAmerican forest landscape (Table 1). We also determined the effect of each scenario on the spatial bifurcation of forest conditions between wilderness versus timber-managed areas, by comparing forest type composition, age–class distribution, and landscape structure among major land management areas.

2. Study area

The Border Lakes Region (BLR) covers ~2.1 million ha in northern Minnesota and northwestern Ontario (Fig. 1) and represents an integration of U.S. and Canadian ecological land classifications (Superior Mixed Forest Ecoregional Planning Team, 2002). The region has a cool-continental climate, with warm, short summers and long, cold winters (Heinselman, 1996). Elevations range from 335 to 701 m above sea level, and landforms are characterized by glacially scoured bedrock uplands and rock outcrops of Precambrian origin. Soils are generally thin loamy sands to sandy loams, with scattered deposits of lacustrine and organic soils (Anderson and Grigal, 1984; Ecological Stratification Working Group, 1995). Freshwater lakes occupy nearly 20% of the region (see Appendix A). Forest communities are transitional

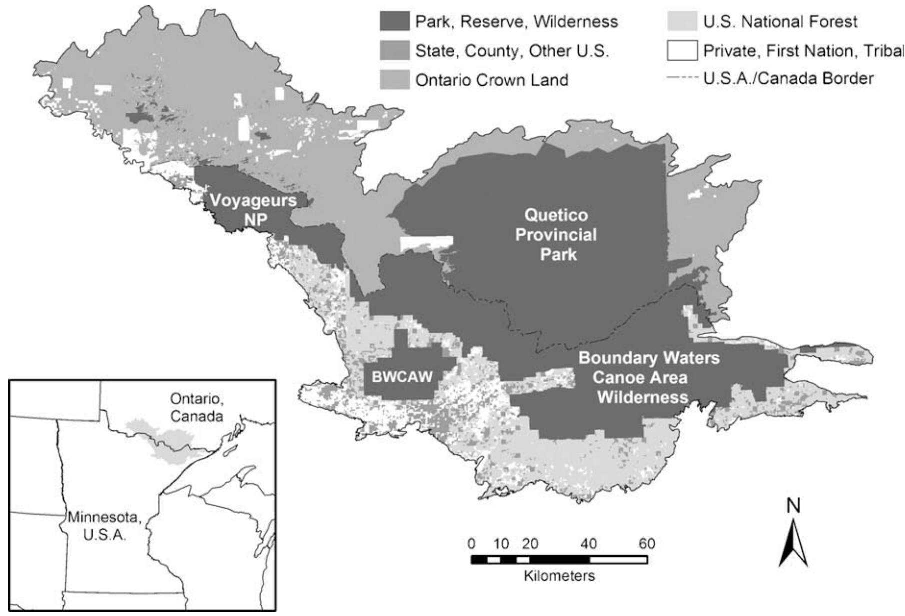

Fig. 1. The Border Lakes Region and major land ownership. NP: National Park; BWCAW: Boundary Waters Canoe Area Wilderness.

between north temperate and boreal forests (Heinselman, 1973). Primary 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*). Primary deciduous species include paper birch (*Betula papyrifera*), aspen (*Populus tremuloides*, *Populus grandidentata*), balsam poplar (*Populus balsamifera*), red maple (*Acer rubrum*), black ash (*Fraxinus nigra*) and, in the southwestern portion, northern pin oak (*Quercus ellipsoidalis*).

Prior to EuroAmerican settlement, stand-replacing fire regimes generated large areas of even-aged, fire-adapted, early-successional species, including jack pine, aspen, and paper birch. Stand replacing fire sizes generally ranged between 400 and 4000 ha, but some may have exceeded 100,000 ha (Heinselman, 1973, 1996). Fire rotation was ~50–75 years for jack pine-black spruce forests and ~75–150 years for wetland and mixed-wood forest types (Heinselman, 1973; Woods and Day, 1977; Beverly and Martell, 2003). Portions of the landscape experienced longer fire-free intervals and supported late-successional forests of spruce, fir, and cedar (Heinselman, 1973; Frelich and Reich, 1995). Smaller (40–400 ha), low- to moderate-severity surface-fires with mean return intervals of 5–100 years maintained older stands of white pine and red pine, with severe crown fires every 150–350 years on average (Heinselman, 1973, 1981). Windthrow events were generally small, with 1000–2000 year return intervals on average, although a rare windthrow event in 1999 disturbed over 500,000 ha (Frelich, 2002).

The unique management histories of protected areas (parks and wilderness) and timber-managed areas has resulted in a corresponding divergence of forest composition across much of the region. Today, roughly 95% of the BLR land area is forested, 93% is publicly owned, and ~43% is protected within the Boundary Waters Canoe Area Wilderness (BWCAW), Voyageurs National Park, and Quetico Provincial Park (Fig. 1). Outside of these conservation areas, early 20th century logging of mature conifers, followed by extensive slash fires and continued logging, have reduced the dominance of conifers in the overstory since EuroAmerican settlement, while shade-intolerant deciduous species, especially aspen, have increased in abundance and are often perpetuated by contemporary logging practices (Friedman and Reich, 2005; Schulte et al., 2007). In contrast, fire suppression in parks and wilderness areas has changed composition from predominantly early- to mid-successional jack pine and aspen forests to mixed-age, multi-species stands trending toward late-successional spruce and fir (Frelich and Reich, 1995; USDI National Park Service, 2002). Across the region, white pine has been substantially reduced as a dominant overstory species due to historical logging (Heinselman, 1996; Friedman and Reich, 2005) and a loss of low- to moderate-severity surface-fire regimes that maintained old stands and provided favorable regeneration conditions for both white and red pine (Heinselman, 1973, 1996).

Contemporary disturbance dynamics and forest landscape structures also reflect the influence of broad-scale patterns of ownership and management. Since the early 20th century, fire exclusion has nearly eliminated large fires (Heinselman, 1996; Beverly and Martell, 2003) until several, recent, large (>10,000 ha) crown-fires in the Quetico-BWCAW region. Wildland fire use is currently allowed in remote portions of parks and wilderness areas, while fire suppression is a primary management objective elsewhere, and prescribed burning is typically used as a short-term response to temporarily increased fire risk (e.g., due to windthrow) or is limited to small-scale restoration projects (USDI National Park Service, 2002; USDA Forest Service, 2001). Modern timber harvest practices are highly variable among landowners, ranging from no logging in parks and wilderness to industrial timberlands with relatively short-rotation, even-age harvest regimes. Moreover, managed forest landscape structure is also generally different on either side of the international border. In northern Minnesota, complex ownership patterns and harvest regulations have generally promoted small-sized cut-blocks (White and Host, 2008), whereas much of northwestern Ontario is dominated by Crown Land ownership and relatively large clearcut patches (Ontario Ministry of Natural Resources, 2001).

3. Model description and methods

3.1. The forest landscape simulation model

We simulated sequential changes in forest composition, landscape pattern, and disturbance regimes resulting from our forest management and disturbance scenarios using LANDIS-II (version 5.1), a spatially explicit, FLSM 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; He and Mladenoff, 1999; Mladenoff and He, 1999), LANDIS-II is a raster (cell) based FLSM that simulates interactions among processes and tracks species age cohorts over broad temporal and spatial scales. Each cell in the model represents uniform light conditions, and cells are aggregated into ecoregions with consistent climate and soil conditions. Successional pathways are nondeterministic, based on tree species cohort interactions, response to disturbance events, and influence of growing conditions. User-defined forest cover types can be assigned to each cell at each time step via a reclassification procedure that utilizes outputs of species age-cohort composition (Mladenoff et al., 1996).

LANDIS-II can be used with various succession and disturbance extensions; we used the age-only succession, base fire, base wind, and base timber harvest extensions, all of which were derived from previous LANDIS models (Scheller et al., 2007). The base fire extension requires input parameters for each user-defined fire region, including fire spread age, fire size distribution, and ignition probability. The probability of fire initiation and spread increase as time since fire exceeds the fire spread age, and species cohort mortality from fire depends on fire severity, cohort age, species fire tolerance, and potential interactions with wind disturbance (He and Mladenoff, 1999). 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).

LANDIS models have been validated for internal logic and tested using sensitivity analysis of key parameters (He and Mladenoff, 1999; He et al., 1999a; Mladenoff and He, 1999; Scheller et al., 2007). Due to input data limitations and stochasticity of the model, strict validation of outcomes derived from simulated disturbance-successional dynamics was not possible. Validation for each scenario followed a calibration-based approach used in other LANDIS models (e.g., He and Mladenoff, 1999; Gustafson et al., 2000), focusing on iterative adjustment of parameters to reflect expected species and community responses, based on previous research.

3.2. Model inputs

LANDIS-II requires an initial forest map, inputs for tree species life-history traits, and disturbance regime parameters. General methods are described below, and detailed descriptions are provided in Appendices A–C. A 100 m × 100 m (1 ha) resolution was used for all input maps in the model, and a 10-year time step was used to simulate all processes.

The initial forest input map must represent each forested cell as a list of tree species age-cohorts, but no such dataset existed for the entire region, and existing stand inventory datasets varied greatly in terms of attributes, resolution, and coverage. Thus, we created a uniform map of extant BLR forest communities by integrating a broadly defined, cover-type (e.g., spruce–fir, aspen–birch) map derived from 2000 Landsat imagery (Bauer et al., in press) with detailed stand inventories from government agencies. We further delineated each forest community into commonly used growth stages (Frelich, 2002) based on stand age, and assigned tree species to each community type-growth stage using stand inventory data. Ten-year age cohorts were then assigned to each species in each community type-growth stage, based on published descriptions of forest community age structures. These methods, explained in more detail in Appendix A, provided an initial forest map that lacked fine-scale accuracy but reflected coarse-scale patterns of common forest community type-growth stages defined by tree species age cohorts.

Tree species successional and reproductive traits (longevity, seed dispersal, shade tolerance, fire tolerance, and ability to sprout vegetatively) were delineated based on previous LANDIS model inputs and other relevant sources (see Appendix A for input parameters and data sources). LANDIS also requires spatially explicit species establishment probabilities (SEPs) that reflect the probability of establishment (ranging from 0.0 to 1.0) for each tree species within user-defined ecoregions (He et al., 1999b). We used land type associations (LTAs) available for Minnesota (Hanson, 2002) as ecoregions, and created new LTAs (or extended Minnesota LTAs) for Ontario, where comparably scaled ecological classification units were not available. Wet forest and non-forest polygons were added to create a final ecoregion input map. SEPs for each ecoregion (Appendix A) were estimated using calculations derived from a soils-based ecosystem model (Pastor and Post, 1986) with input parameters for species attributes, monthly climate data, soil conditions, and geographic location. SEPs for wet forest ecoregions were estimated separately, based on inputs used in previous LANDIS models.

3.3. Model scenarios

One current management, three restoration management, and two natural disturbance scenarios were modeled over a 200-year period. The natural disturbance scenarios were used to gauge the potential effectiveness of fire alone in shaping ecosystems and landscape patterns and to distinguish between the effects of timber harvest and fire. General scenario differences are summarized in Tables 2 and 3, specific fire and harvest parameters are summarized in Appendices B and C, respectively, and each scenario is described in detail below. For all scenarios, windthrow parameters were set at 1, 93, and 3600 ha for minimum, mean, and maximum sizes respectively, with a rotation of 1000 years (based on Frelich, 2002). Potential climate change effects were not simulated in our model, because our focus was on hypothetical restoration strategies for contemporary forest landscape conditions derived from empirical data. Each scenario was replicated 5 times to generate stochastic variability, but due to highly calibrated disturbance rates and size distributions, standard errors were small for replicate outputs (e.g., mean forest area by cover type) and were not reported.

Table 2
Summary of disturbance dynamics simulated in each of the seven scenarios: a check means the disturbance type was simulated; a check-plus for high-severity fire indicates a shorter rotation was simulated (in parks and wilderness for the restoration scenarios, and for the entire landscape in the historical natural disturbance scenario) relative to the current management scenario; and a check-plus for planting indicates all clearcuts in the largest size class were planted with jack pine and black spruce.

Scenario	Wind-throw	Fire		Timber harvest		
		High severity	Low severity	WB	XB	Planting
CM	✓	✓		✓		✓
RM1	✓	✓+	✓	✓		✓
RM2a	✓	✓+	✓		✓	✓
RM2b	✓	✓+	✓		✓	✓+
CND	✓	✓				
ND	✓	✓+	✓			

WB: harvest occurs within ownership boundaries only and XB: harvest occurs across ownership boundaries.
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. Refer to Appendices B and C for specific fire and timber harvest parameters for each scenario.

Table 3
General comparison of timber harvest targets between the current management (CM) and restoration management (RM) scenarios.

Harvest type	Target forest types/species	Target harvest patches		Target %forest area (per 10-years)	
		CM	RM	CM	RM
Clearcut	Jack pine, aspen, mixed-wood, some lowland black spruce	Generally 1–100 s ha in U.S. Generally 10–1000 s ha (max of 10,000 ha) in Canada	Generally 10–1000 s ha (max of 10,000 ha) throughout BLR	8.53	8.50
Shelterwood, Seed tree, and Pine restoration	Red/white pine	Generally 1–100 ha; either removes all but oldest trees or removes all trees (after second entry)	100–200 ha; leave mix of old pines and other trees to emulate low- to moderate-severity fire	0.55	0.58
Commercial thin	Red pine and aspen; some mixed-wood	Generally 1–100 s ha	Generally 1–100 s ha	0.50	0.50
Partial harvest	Aspen–birch	Generally 1–100 s ha	Generally 1–100 s ha	0.31	0.30
Uneven-age	Aspen–birch, white spruce, and white pine	Generally 1–100 s ha	Generally 1–100 s ha	0.02	0.03
			Total %land area	9.91	9.91

Refer to Appendix C for detailed harvest parameters.

3.3.1. Current management (CM) scenario

The CM scenario simulated contemporary disturbance dynamics. Fire was simulated to reflect land ownership and associated fire policies, which are often important determents of modern fire regimes (Cardille and Ventura, 2001). Using agency fire occurrence databases, six contemporary fire regions were delineated, each with relatively distinct fire regimes, based on fire rotation and size class distribution (Appendix B). For the first 100 model years, fire was calibrated to within 10% of actual rotation (Appendix B) or within 100 ha of mean annual area burned, and the distribution of area burned by fire size-class was matched as closely as possible to contemporary trends, using the mean of five model replicates. During the second 100-year period, an increase in forest area that exceeded the user-defined fire spread age parameter required for each fire region by LANDIS (He and Mladenoff, 1999) resulted in roughly a three-fold increase in area burned by severe fires (classes 4 and 5); thus, the second 100-year behavior described here is an emergent property of the modeled fire regime that effectively simulated the potential effects of fire suppression. Prescribed fire was not modeled, given its limited use in management. Contemporary timber harvest practices were simulated to reflect typical harvest techniques by major landowner (Table 3), with specific harvest prescriptions and post-harvest planting simulated for 51.4% of the BLR forest area within six major ownerships (see Appendix C for more details). For U.S. National Forest, Ontario Crown Land, and Minnesota State Forest, harvest parameters represented simplified versions of current or proposed forest management plans. Harvest parameters for private non-industrial, private industrial, First Nation, and county lands were estimated from published summaries. Total area harvested over model time was calibrated to within 1% or 100 ha of the planned harvest area for each management area.

3.3.2. Restoration management (RM) scenarios

The three RM scenarios (RM1, RM2a, RM2b) simulated fire and timber harvest strategies designed to restore key forest conditions. In timber-managed areas, fire was simulated to reflect current fire policies using the parameters of the CM scenario, and in park and wilderness areas the parameters reflected an approximation of pre-EuroAmerican fire regimes (Heinselman, 1973, 1981, 1996; Bergeron et al., 2002) of both stand-replacing and a low-severity fires (Appendix B). The stand-replacing fire regime included a targeted mean rotation of 130 years, a maximum fire size of 20,000 ha, and a negative exponential distribution of area burned by fire size class. The low-severity fire regime simulated a mean rotation of 50 years, a maximum fire size of 2000 ha, and mortality for only the youngest pines and mid-aged to old cohorts of fire-sensitive species (e.g., aspen). Low-severity fire was limited to extant red and white pine patches with interiors >200 m from patch edge (Appendix B).

The RM1 scenario used the same settings as the CM scenario for timber harvest. The RM2a and RM2b simulated forest harvest that more closely emulated pre-EuroAmerican fire size distributions, patterns, and rotations across management boundaries, including: (1) randomly placed clearcuts up to 10,000 ha in size that targeted jack pine, aspen, and mixedwood (e.g., spruce–fir–aspen–birch) stands; (2) low- and moderate-severity harvest techniques in stands that contained red or white pine; and (3) six other harvest types (two types of commercial thinning, tamarack seed tree harvest, partial harvest, uneven age harvest, and lowland forest clear cutting) that collectively targeted only 1% of the total forest area per 10 years and merged similar harvest techniques used among different landowners in the CM scenario (Table 3; see Appendix C for harvest prescription details). For each harvest type (e.g., clear-cutting), similar targets for species, stand age, and harvest area per 10-year period were set to match the CM scenario parameters as closely as possible, such that the total area harvested was within 1% of the planned harvest area for all harvest scenarios for the entire model period. In the RM1 and RM2a scenarios, the post-harvest area replanted for each species was roughly equal to the CM scenario. The RM2b scenario differed from RM2a only in that it increased the area replanted in jack pine and black spruce, by simulating replanting in all of the largest clearcuts (Appendix C).

3.3.3. Contemporary (CND) and historical (ND) natural disturbance scenarios

To determine the effects of fire suppression policies in the absence of timber-harvest, the CND scenario excluded logging and simulated fire using the same fire regime as the CM scenario. To explore the potential effects of a return to pre-EuroAmerican fire regimes on forest composition and landscape structure, the ND scenario simulated fire parameters across the entire region similar to those used for park and wilderness areas in the RM scenarios (Appendix B). To better reflect historical fire size distributions over this larger area, the stand-replacing fire regime included a maximum fire size of 30,000 ha, and low-severity fire regimes were extended 100 m beyond extant red and white pine patches into adjacent forests (excluding wet-forest types).

3.4. Analysis methods

We compared each scenario's landscape condition at model year 200 (reported as the mean of five replicates unless otherwise noted) to six estimated benchmarks of the RNV (Table 1). Three

RNV benchmarks were derived from research on pre-EuroAmerican settlement forest conditions (Swain, 1980; Friedman and Reich, 2005; Schulte et al., 2007), including an estimate that roughly two-thirds of the BLR was shaped by a stand-replacing crown fire regime and the rest primarily by low- to moderate-severity fire regimes, creating a shifting mosaic between fire-dependent forest types over time (Heinselman, 1973, 1981, 1996). From this, the following ranges of forest landscape area proportions were used as benchmarks for restoration: 20–40% for jack pine, 15–30% for red and white pine, and 15–30% for aspen–birch. A single benchmark of a negative exponential age–class distribution was used, based on unmanaged southern boreal forests (Van Wagner, 1978; Bergeron et al., 2002). Two patch size benchmarks were based on a modest assumption that patch size distributions created by historical fire regimes (Heinselman, 1973, 1981) should result in ≥10% of the landscape within large (>1000 ha) fire-created patches of jack pine and ≥5% within medium or larger (>100 ha) fire-maintained patches of red and white pine.

We then assessed four measures of forest landscape bifurcation derived from forest cover type composition, age–class distribution, and two measures of landscape structure (mean patch size and landscape diversity) among major management areas. Cover type and age–class bifurcations were assessed as a function of the difference between two primary land management areas: (1) the Quetico-BWCAW-Voyageurs region (hereafter, "wilderness"); and (2) the rest of the landscape, which is primarily managed for timber resources (hereafter "timber-managed"). Bifurcation of age–class distributions was qualitatively compared among the two management areas. Forest cover type bifurcation (BV) was calculated as,

$$BV = \sum_{i=1}^{n} (|Wi - 0.5|) \times Li$$

where W_i and L_i represent the proportion of forest type i in wilderness areas and the entire forest landscape, respectively, and subtracting 0.5 represents a deviation from an even proportion of each forest type in wilderness and timber-managed areas. The sums were rescaled from 0 to 1, where 0 indicates proportionally even distribution of all forest types among wilderness and timber-managed areas and 1 represents complete bifurcation (i.e., every forest types is either completely in wilderness or completely in timber-managed areas). In order to assess the potential for deviation from the current level of bifurcation, which has been caused by human imposed boundaries and divergent management histories, we considered cover type composition bifurcation values for each scenario at model year 200 in relation to the value for year 0.

Forest landscape structural bifurcation was also assessed as a function of the difference between wilderness and timber-managed areas, but the latter was further divided into U.S. and Canadian management areas to highlight structural effects of distinctive timber harvest practices. Landscape structure indices were calculated using APACK (Mladenoff and DeZonia, 2004) and included mean patch area by forest type and the Shannon–Weaver diversity (SWD) index, a combined measure of richness and evenness of patch types. These two metrics were selected as indicators of landscape structural differences among management areas, and were not meant to characterize overall landscape pattern. Each was calculated for two differently defined landscape mosaics: one delineated by eight major forest types and recently burned forest patches, and one delineated by a combination of eight forest types/burned patches and five age–classes (0–40, 40–80, 80–120, 120–160, 160+).

4. Results

4.1. Forest composition and comparison of scenarios to RNV benchmarks

Temporal and spatial patterns of landscape-level forest composition varied substantially among scenarios, as indicated by the relative proportion of forest types over model time (Fig. 2), and by forest type area values at model year 200 (Fig. 3 and Table 4). Both the RM1 and RM2b scenarios created temporal patterns of forest composition in which jack pine increased initially, and then declined and nearly lost dominance to aspen–birch by the end of the model period, while boreal spruce remained relatively constant. The RM2b scenario, with greater levels of post-harvest planting of conifers, produced a higher peak proportion of jack pine (∼37%) than any other management scenario. In contrast, the RM2a scenario resulted in a gradual decline in jack pine and an increase in aspen–birch that eventually exceeded 40% of the forested area. The CM scenario also resulted in declining area in jack pine, but it caused a substantial increase in spruce-dominated forest, surpassed by fir and aspen–birch toward the end of the model period. The ND scenario exhibited temporal compositional patterns similar to RM1 and RM2b, with jack pine forest dominating almost immediately and peaking at ∼40% of the forested landscape by model year 90. The CND scenario exhibited compositional changes similar to the CM scenario, though much

Table 4
Area (ha) of major forest type and recently burned for model year 0 and for all scenarios at model year 200, followed by the percent area of each forest type in parks and wilderness for year 0 and at model year 200.

Scenario	Boreal hardwoods	Red/white pine	Boreal spruce	Jack pine	Aspen–birch	Lowland conifers	Balsam fir	Oak/pine	Recent burn
Total area (ha)									
Year 0	46,690	197,374	273,515	397,317	481,555	125,995	–	3323	–
CM	16,425	170,805	218,044	183,438	390,082	87,842	305,383	1316	152,433
RM1	13,353	140,640	225,834	321,835	495,885	78,262	137,065	548	112,348
RM2a	14,545	162,879	249,345	210,850	567,821	80,429	116,104	484	123,311
RM2b	12,691	161,860	225,321	361,822	449,029	80,298	112,769	496	121,483
ND	6,649	154,549	291,188	380,906	363,030	50,995	92,026	525	185,902
CND	12,917	191,024	259,823	26,802	108,418	92,614	614,503	1274	218,394
Percentage in parks and wilderness									
Year 0	68.1	54.8	71.1	52.6	29.2	43.3	0.0	93.4	–
CM	46.9	57.7	72.0	7.4	20.6	40.4	76.7	93.1	74.4
RM1	34.4	47.0	72.1	47.6	38.5	30.4	44.9	82.5	70.0
RM2a	33.8	42.1	70.2	61.6	32.7	32.1	58.6	83.8	67.8
RM2b	39.3	42.2	78.0	36.9	41.9	31.7	54.9	88.3	68.4
ND	59.1	49.9	49.4	51.4	50.6	40.4	43.8	96.6	40.7
CND	59.4	54.2	59.5	60.1	61.0	38.7	40.2	97.2	50.3

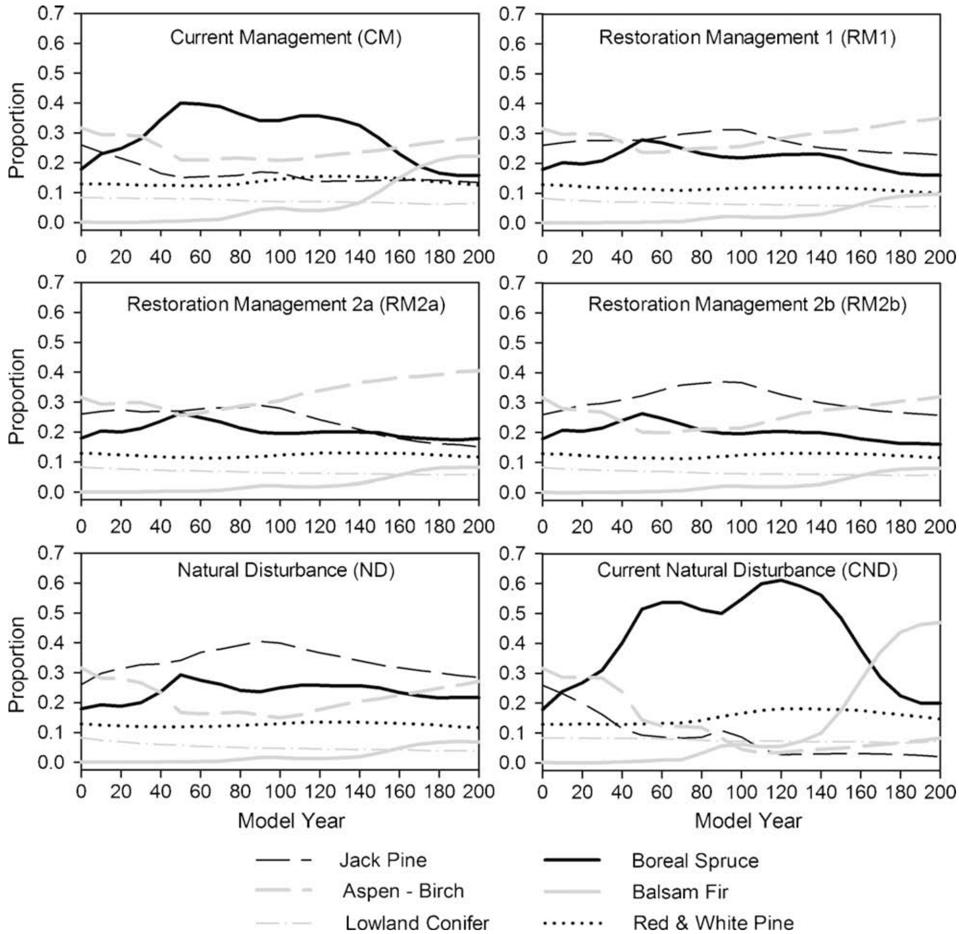

Fig. 2. Major forest type change over 200 years for each of the six scenarios, presented as a proportion of total forest area. Recently burned (not yet regenerated) forest area was not included in these proportions. Boreal hardwood and oak forest types are not shown due to low proportional values throughout model time.

more enhanced, with spruce peaking at ~60% of the landscape before declining and being replaced by fir-dominated forest, while jack pine declined by 93% in total area from year 0 (Table 4). Red and white pine, boreal hardwoods, lowland conifer, and oak–pine forests declined in total area and as a proportion of the forest landscape in all scenarios from year 0 to 200.

The proportion of the landscape within age–classes and large patch sizes by forest type also revealed important structural and compositional differences among scenarios. Nearly 40% of the CND forest landscape at year 200 was composed of large (>1000 ha) balsam fir and spruce patches, compared to ~16% for the CM scenario, and <4% for the ND and restoration scenarios (Fig. 4). The ND, RM1, and RM2b scenarios had the largest proportion (~11–13%) of the forest landscape in large (>1000 ha) jack pine patches, and the RM2a (~23%) in large aspen–birch patches. Large (>100 ha) red and white pine patches comprised 6–8% of the forest landscape in the ND, RM2a, and RM2b scenarios, compared to 4–5% for the other scenarios (Fig. 4). Variable rates and patterns of fire and timber harvest resulted in substantially different forest age–class distributions among and within scenarios by major land management areas. The ND scenario and the restoration scenarios generally approached or achieved negative exponential distributions across the entire BLR landscape, while the CND and CM scenarios did not (Fig. 5).

Differences in compositional and structural forest conditions among the scenarios affected their ability to meet the six RNV benchmarks (Table 5). None of the scenarios achieved restoration goals for red and white pine proportion of the forest landscape area. The ND and RM2b scenarios each met five RNV benchmarks, including target proportions of forest in jack pine and aspen–birch, the target area within large patch sizes for the two pine forest types, and age–class distribution. The RM1 scenario met four

Table 5
Outcomes for each scenario at model year 200 related to each of six range of natural variability benchmarks (a plus sign indicates that a scenario met the RNV benchmark).

Target RNV[a]	Scenarios					
	CM	RM1	RM2a	RM2b	ND	CND
Jack pine forest area (20–40%)		+		+	+	
Red/white pine forest area (15–30%)						
Aspen–birch forest area (20–30%)	+			+	+	+
Negative exponential age–class distribution		+	+	+	+	
Jack pine: 10% in >1000 s ha patches		+		+	+	
Red/white pine: 5% in >100 ha patches		+	+	+	+	

[a] Percent values were calculated based on the total forested (including recently burned) land area.

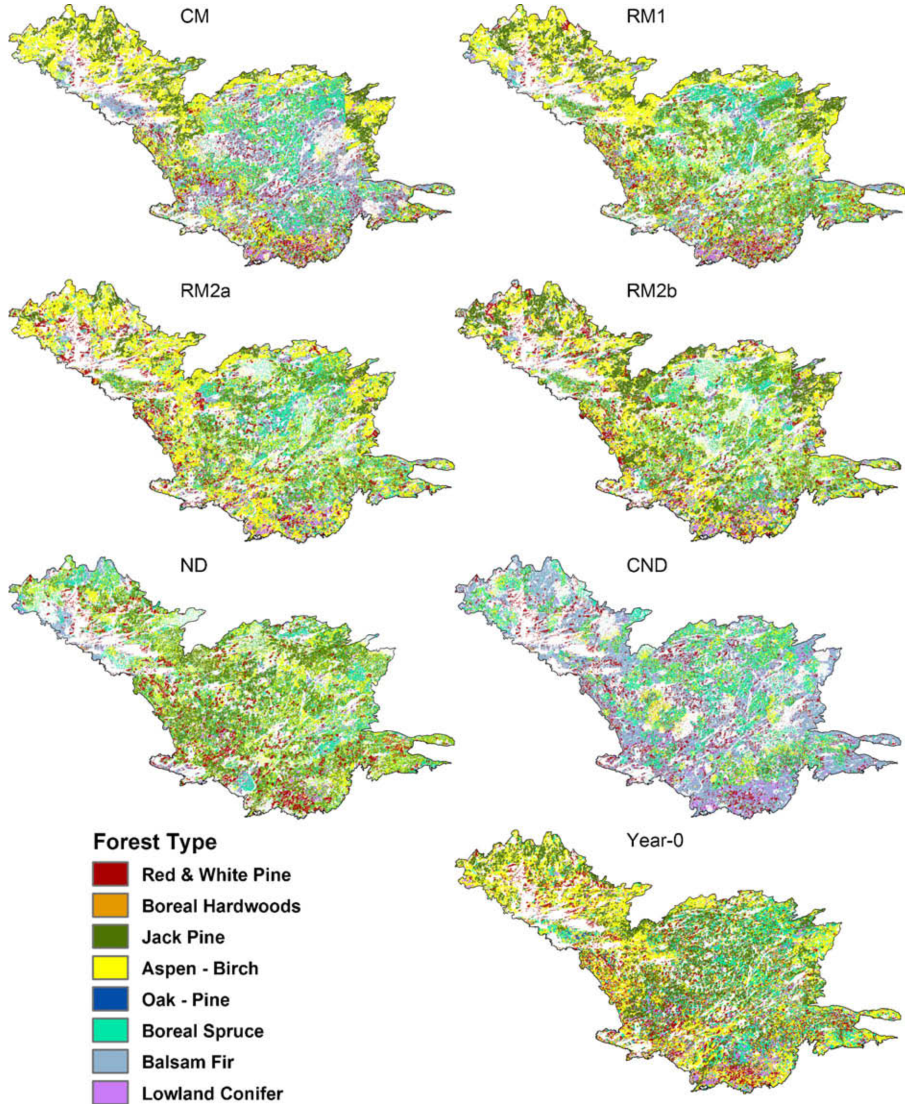

Fig. 3. Major forest types for initial forest conditions at model year 0 and for each of the six scenarios at model year 200. White areas represent lakes, non-forest cover types, and recently burned areas. The variability in white patches among scenarios represents differences in amount of recently burned forest that has not yet regenerated.

benchmarks, including jack pine area, area in large patches for the two pine types, and negative exponential age–class distribution. The RM2a met two benchmarks: area in large patches for red and white pine and negative exponential age–class distribution. The CND and CM scenarios each met one benchmark: proportion of forest dominated by aspen–birch, which largely transitioned to late-successional spruce and fir (Fig. 2).

4.2. Landscape structure and bifurcation

The ND, CND, and RM1 scenarios produced forest composition bifurcation values <0.25, via a convergence of forest type composition among major management areas relative to year 0 (Fig. 6). The ND scenario produced the lowest bifurcation value (0.03) among scenarios, with major forest types generally evenly distributed between timber-managed and wilderness areas. Coalescing patches of late-successional, conifer-dominated forests in the CND scenario resulted in the second lowest compositional bifurcation value by year 200. The RM2b scenario bifurcation value (0.26) was equal to year 0, but might trend lower in future decades as large burned areas (Fig. 3) regenerate to jack pine and aspen. Both the RM2a and CM scenarios exceeded year 0 bifurcation values, with the CM scenario generating the highest value (0.5) among all scenarios, revealing the effects of spatially disparate management that increased spruce and fir within wilderness, and maintained aspen–birch dominance elsewhere. Bifurcation in the CM scenario is also revealed by the negative exponential age–class distribution in timber-managed areas versus more area in older age–classes in wilderness. In contrast, the ND and RM scenarios had similar age–class distributions across management areas that resembled negative exponential distributions of pre-EuroAmerican boreal forests (Fig. 5).

The ND scenario achieved a coefficient of variation $<10\%$ for mean patch size among management areas, while the coefficient of variation for other scenarios generally exceeded 30%, indicating substantially different patch structures across major management regions. Mean patch area was smallest in the ND scenario when measured for landscapes both defined by forest type and by forest type-age class (Fig. 7). Over the entire BLR, the CND and CM scenarios produced the largest mean patch sizes, although the difference was not large (within 2 ha) among scenarios for forest type-age class landscapes. In all scenarios with timber harvest, in both landscape types, the Canadian mean patch sizes were larger than either wilderness or U.S. mean patch sizes. However, the CM and RM1 scenarios had the most dissimilar patch sizes across management areas, with Canadian mean patch sizes generally

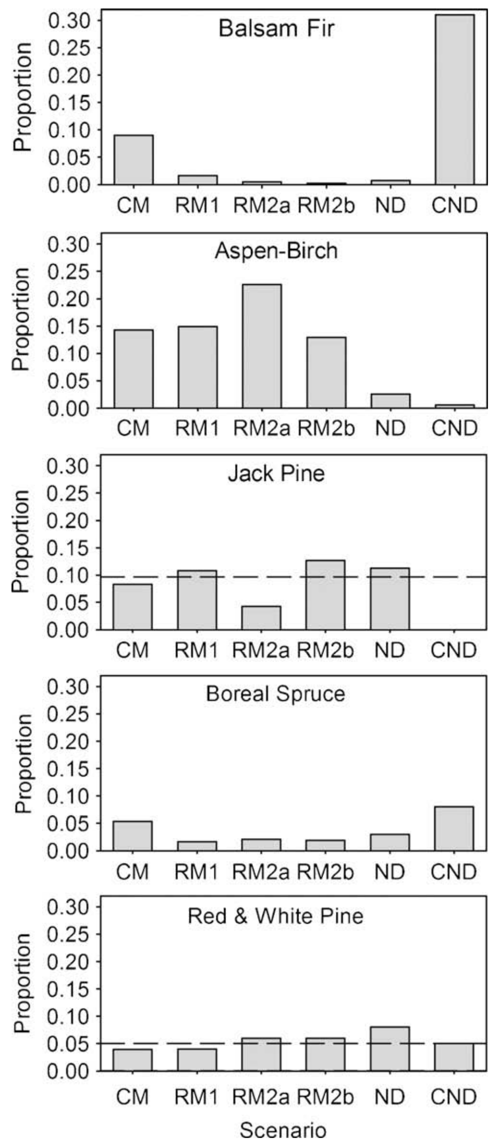

Fig. 4. Proportion of forest area in patch sizes >1000 ha for each of four major forest types, and >100 ha for red/white pine, for each scenario at model year 200. RNV benchmarks for area within large patches is indicated by dashed lines (for jack pine and red/white pine only).

more than twice the size of U.S. and wilderness mean patch sizes (Fig. 7). Landscape diversity was also substantially different among scenarios for the two landscape types (Fig. 7). Over the entire BLR, Shannon-Weaver diversity (SWD) values indicated a less even landscape structure at year 200 for the CND scenario compared to other scenarios, due to more clumping by a few dominant forest types. SWD values had the least within-scenario variation among major management units within the ND scenario, with a coefficient of variation <3% for both landscape types, while SWD values varied the most in the CND, CM, and RM1 scenarios, with the latter two producing less diverse structures in Canada compared to other management areas (Fig. 7).

5. Discussion

5.1. Restoration potential among scenarios

Estimates of RNV derived from historical disturbance regimes are useful for developing regional ecological restoration objectives (Baker, 1989; Shinneman and Baker, 1997; Landres et al., 1999; Tinker et al., 2003). Based on this study and previous studies (e.g., Frelich and Reich, 1995; Friedman and Reich, 2005), extant forest landscapes of the BLR are not within the RNV, due to suppressed fire regimes, reduced pine forest area, and bifurcation caused by over-abundance of younger aspen-dominated forests in timber-managed areas, and increasing dominance of late-successional conifer forests in wilderness areas (Table 4 and Figs. 2, 3 and 5).

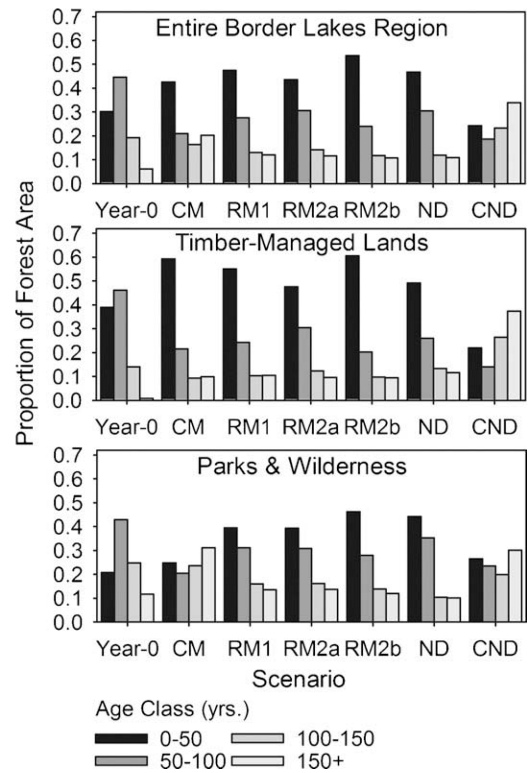

Fig. 5. Proportion of forest within age–classes at model year 200 among scenarios, for entire study area landscape, timber managed lands, and parks and wilderness.

All of the RM (restoration management) scenarios met more RNV benchmarks than the CM (current management) scenario (Table 5), suggesting that wildland fire and coordinated timber harvest across ownerships can be used to meet restoration objectives and minimize compositional bifurcation across administrative boundaries at regional scales. The RM1 scenario, with shorter rotation for both low- and high-severity fires in wilderness areas, and a contemporary timber harvest regime, met four of six RNV benchmarks and effectively reduced bifurcation among management areas (Fig. 6) by producing age–class distributions

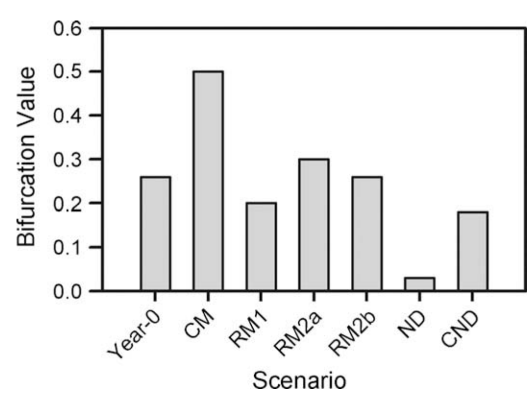

Fig. 6. The calculated bifurcation value (BV) for all scenarios at model year 200 and for year 0.

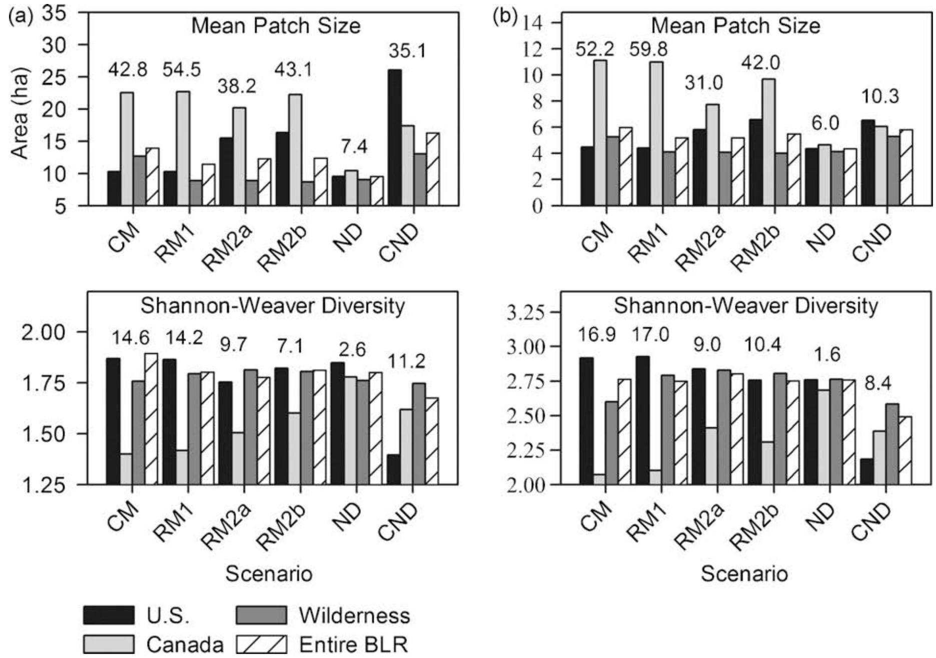

Fig. 7. Landscape diversity and structure indices at model year 200 for the entire study area landscape and by major land management regions, calculated for patches defined by (column a) forest type only and (column b) forest and age–classes combined. Coefficient of variation among major management areas is listed above each bar group as a percent.

similar to historical conditions (Fig. 5) and by creating a shifting balance between aspen–birch and jack pine over time at the regional level (Fig. 2). In the RM2b scenario, which met five RNV benchmarks, simulation of shorter fire rotation in wilderness areas and cross-boundary forest harvest that emulated natural disturbance not only reduced spruce and fir dominance (Fig. 2), but also restored age–class distributions, minimized divergence in landscape structure across management boundaries (Fig. 7) and, with additional planting, increased jack pine forest area and patch sizes (Figs. 2 and 4). Other modeling studies that simulated timber harvest across boundaries or emulated natural disturbance patterns have achieved similar results, including gaining area within desirable age–classes (Thompson et al., 2006) and achieving larger patch sizes (Mehta et al., 2004). However, differences in aspen–birch and jack pine dominance between the RM2a and RM2b scenarios suggest that larger clearcuts will require more extensive post-harvest conifer planting to prevent excessive conversion to aspen. Silvicultural approaches for regenerating jack pine typically include direct seeding, planting, or seed tree techniques with prescribed fire (Benzie, 1977).

The CM scenario, with disparate management among land owners, met only one RNV benchmark (Table 5). It also created the most bifurcated landscape (Fig. 6), as fire suppression in wilderness led to older forests dominated by spruce and fir, while logging elsewhere generally maintained younger forests of aspen–birch (Figs. 2, 3 and 5). The CM scenario also created the greatest divergence in mean patch area and landscape diversity among major land management units (Fig. 7), in part because Canadian timber harvest patch sizes were much larger on average than those in the U.S. (Table 3). Sharply contrasting landscape structures (e.g., with substantially different patch size distributions and edge densities) have also been documented between fire-prone wilderness and fire-excluded, timber-managed landscapes in the Greater Yellowstone Region (Tinker et al., 2003) and between pre-EuroAmerican settlement and timber-managed landscapes in the southern Rocky Mountains (Reed et al., 1996).

Results from the ND (historical natural disturbance) scenario suggest that natural disturbance regimes operating at regional scales can effectively create relatively similar compositional and structural patterns among major land management areas, despite spatially disparate forest conditions caused by management legacies in the initial landscape. The ND scenario produced the lowest compositional bifurcation value, with nearly even distribution of major forest types among management regions (Fig. 6 and Table 4), as well as the most consistent patterns of forest landscape structure (Fig. 7) and negative exponential age–class distributions (Fig. 5) across management areas. The short fire rotations in the ND scenario increased jack pine over time while suppressing spruce and fir (Fig. 2 and Table 4). The simulated gains in jack pine are not improbable, as high-severity fire in mixed-wood boreal forests can produce dense jack pine stands within three years, by opening serotinous jack pine cones and producing ideal seed-bed conditions (Heinselman, 1973). In contrast, the CND (contemporary natural disturbance) scenario suggests that contemporary fire management policies will not restore ecosystems, as the coalescing trends in forest composition, age–class, and landscape structure across boundaries represented movement away from the RNV, including a 93% decline in jack pine by model year 200, a regional transition to spruce and fir dominance (Fig. 2 and Table 4), and an age–class distribution that was nearly the inverse of historic conditions (Fig. 5). Similar trends caused by fire suppression have been observed within the BWCAW (Frelich and Reich, 1995).

Prior disturbance modeling in the BLR assessed only the effects of fire regimes within the BWCAW. Baker (1992) determined that distinct, age-defined, forest landscape structures were created during three historical fire periods, including pre-EuroAmerican settlement, early settlement with increased fire occurrence, and post-settlement with effective fire exclusion. Scheller et al. (2005) found that longer fire rotations and fire exclusion led to increased spruce- and fir-dominance, while short-rotation fire regimes generally increased or maintained jack pine and aspen–birch. Both Baker (1992) and Scheller et al. (2005) found that fire exclusion created a more even and more diverse landscape structure compared to short-rotation fire regimes. In contrast, fire suppression in our model generated larger mean patch sizes and a less-even, less-diverse landscape structure relative to landscape structures created by short-rotation, natural disturbance regimes (Fig. 7). The likely reason for this contrast between studies is that our landscape structures were defined by forest type and forest type-age classes rather than age alone as in the previous studies. Indeed, a post-hoc analysis of the entire landscape using an age–class only defined landscape structure, shows that the CND scenario created a slightly more heterogeneous, more even, and less clumpy landscape with smaller patch sizes compared to the ND scenario (data not shown). Landscape structures created by the ND, RM, and CM scenarios also differed due to dynamic and complex patch shapes created by fire, versus static and less-complex, human-delineated stand boundaries used for timber harvest.

Our analysis across ownership and management boundaries also highlighted the potential for forest landscape bifurcation (Figs. 5–7), and underscores the need for coordinated restoration strategies at regional scales, as an often unintended consequence of disparate management practices within multi-ownership landscapes is the creation of spatially contrasting forest structures and compositions. This bifurcation can undermine restoration efforts (Nonaka and Spies, 2005) and negatively affect landscape-level ecological processes, including biogeochemical cycles and metapopulation dynamics (Turner et al., 2001; Hansen and DeFries, 2007). For instance, woodland caribou (*Rangifer tarandus caribou*) are largely extirpated in northern temperate-southern boreal forests of Ontario. Although favorable habitat conditions exist within wilderness parks such as Quetico, these areas may not be sufficiently large to support viable caribou populations if incompatible management occurs on surrounding lands (Vors et al., 2007). Restoration strategies that consider relationships between nature reserves and surrounding lands at regional scales may be most effective at sustaining ecological processes and biodiversity (Poiani et al., 2000; Lindenmayer and Franklin, 2002; Bengtsson et al., 2003).

Despite modeled potential to meet RNV-based restoration objectives at regional scales, whether it is actually possible to achieve restoration over the long term is uncertain. For instance, although jack pine initially increased in the ND, RM1, and RM2b scenarios, it then gradually declined while aspen–birch simultaneously increased (Fig. 2). This is likely because fire, logging, and planting regenerated jack pine on portions of the landscape, while some older jack pine stands senesced in the absence of disturbance and were eventually replaced by other forest types. Also, despite specific efforts to maintain red and white pine, these forest types declined over time in all scenarios (Fig. 2 and Table 4) as the small, spatially scattered stands not perpetuated via restoration-harvest or low-severity fire in the model either senesced, were destroyed by high-severity fire, or were lost to indiscriminant timber harvest. Similar dynamics occurred for lowland conifer, boreal hardwood, and oak–pine forest types (Table 4) although, in contrast to red and white pine, these types lacked a restoration focus in the model. Similar trends also occurred historically, after early logging of red and white pine stands was followed by large slash fires, leaving isolated seed sources that were inadequate for regional regeneration (Friedman and Reich, 2005). The RM2 scenarios did reveal a potential benefit of a regional effort to restore red and white pine stands using low-severity fire and restoration timber harvest, by producing a greater proportion of the landscape within large patches of red and white pine (Fig. 4) more evenly distributed across the landscape (visually apparent in Fig. 3), as compared to the CM and CND scenarios. Larger and more evenly distributed habitat patches can substantially improve species viability and dispersal opportunities (Wiens, 1997; Wei and Hoganson, 2006).

5.2. Model strengths and weaknesses

FLSM models are not predictive tools, and the output must be interpreted cautiously if used to guide forest and fire management, especially given simplified model assumptions, inherent model limitations, and issues of scale. For instance, management regions and harvest parameters remained constant throughout model time, due to unknowable future changes in climate and forest management. Other disturbance agents were not modeled, including spruce budworm outbreaks, which would likely have decreased balsam fir dominance over time (Maclean and Ostaff, 1989). Limitations in the LANDIS-II harvest extension prevented leaving remnant patches of uncut forests within larger clearcuts, an intended strategy for most management plans. This likely exaggerated structural and compositional differences among the ND, CM, and restoration scenarios. Fire behavior is also limited, as simulated fire regimes in each fire region did not reflect fine-scale fuel type conditions or more abundant small fires. However, realistic simulation of large fires may be more important, given their relative influence on landscape structure and composition in boreal forests (Johnson et al., 1998). Finally, although the 1 ha resolution used here was appropriate for the large study area, model behavior and landscape analysis results can be highly scale-dependent (Scheller et al., 2005; Ravenscroft et al., in press).

5.3. Management implications

The simulation of a pre-settlement fire regime in the ND scenario and in the wilderness portions of the restoration scenarios suggests that fire alone could be a highly effective tool to substantially restore fire-dependent forests, reduce bifurcation, and produce more consistent landscape structures among major land management areas (Figs. 6 and 7). However, restoring fire and fire-prone jack pine forests irrespective of ownership and management will not likely be economically or socially feasible, given potential loss of timber resources and safety issues in developed areas. Thus, any increase in wildfire use for restoration purposes would likely be restricted to parks and wilderness areas. Potential drawbacks to a parks and wilderness area focus include possible fire spread onto surrounding developed landscapes and unintended loss of other forest types in need of restoration. However, a continued policy of fire suppression would likely also be counterproductive, as demonstrated by the increase in fire-prone, late-successional spruce and fir forests in the CND and CM scenarios that would pose similar fire risks (Fig. 3). In all scenarios, the decline of forest types initially comprising small portions (<10%) of the landscape in scattered distributions (Table 4) suggests that a greater focus will be required to maintain such forest components. For red and white pine, low- to moderate-severity prescribed fire is a potentially effective tool, because it can increase regeneration and decrease balsam fir and other ladder fuels in the understory that facilitate crown fires (Beverly and Martell, 2003; Woodman, 2005).

The restoration scenarios suggest that coordinated timber harvest that emulates natural disturbance patterns across ownership boundaries can create larger pine forest patches and more consistent regional landscape structures. Using ecologically sustainable harvest rates, large clearcuts may also limit the impact of timber harvest on the landscape, by concentrating activities in fewer areas over a given planning period. Harvest patterns and rotations that mimic fire regimes may serve the goal of restoring natural landscape structure, composition, and age–class distributions (Cissel et al., 1999; Bergeron et al., 2002; Drever et al., 2006; Didion et al., 2007). Moreover, the ability to restore large patches of key forests types, such as red and white pine, may require cross-boundary coordination given current spatial distributions.

Additional factors must be considered to achieve social and ecological objectives using timber harvest. Ensuring that the burdens and benefits of ecologically driven timber harvest are distributed fairly among land owners is one key factor (Thompson et al., 2006). Our restoration scenarios did not necessarily do this because, although restoration harvest rates were equivalent to contemporary rates at the regional level, rates were permitted to diverge from contemporary harvest among individual land owners. Also, timber harvest cannot completely mimic the effects of fire on biogeochemical cycles or post-fire patterns of mortality, remnant biomass, and coarse-woody debris, and logging roads and logging-caused soil disturbance can negatively affect forest function and biodiversity (McRae et al., 2001; Lindenmayer et al., 2006).

6. Conclusions

A continuation of fire suppression and uncoordinated timber harvest will move the BLR further from the RNV by: (1) increasing regional bifurcation between older, conifer-dominated forests in wilderness and younger, aspen-dominated forests in timber-managed areas; (2) creating distinctly different landscape structure and diversity patterns among major land management areas; and (3) causing a continued decline of key forest types, such as jack pine, red pine, and white pine. In contrast, the restoration scenarios suggest that greater use of wildland fire in wilderness areas and cross-boundary timber harvest mimicking natural disturbance elsewhere in the landscape may help to achieve some regional restoration objectives. Despite this, strategies to simultaneously achieve regional-level forest restoration, timber harvest, and fire-risk reduction objectives across large, multi-ownership landscapes will be challenging, especially if there are incompatible socio-economic influences, including increased demands for timber resources and fire exclusion. Moreover, adaptive management strategies will likely be required, as dynamic climate change effects (Millar et al., 2007), including increases in fire-prone weather conditions (Flannigan et al., 2008), alter future boreal forests. Adaptive restoration may promote ecosystem resiliency in the face of climate change and associated changes in natural disturbance regimes (Fulé, 2008), while degraded ecosystems may be particularly vulnerable to climate change via rapid, non-linear ecological responses (Burkett et al., 2005). Thus, exploring options for restoration of forest landscapes is highly relevant to long-term objectives for sustainability.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at doi:10.1016/j.foreco.2009.10.042.

References

Anderson, J.L., Grigal, D.F., 1984. Soils and Landscapes of Minnesota. University of Minnesota Extension Service, AG-FO-2331, St. Paul, MN.
Aplet, G.H., Keeton, W.S., 1999. Application of historical range of variability concepts to biodiversity conservation. In: Baydeck, R.K., Campa, III, H., Haufler, J.B. (Eds.), Practical Approaches to the Conservation of Biological Diversity. Island Press, Washington, DC, pp. 71–86.
Baker, W.L., 1989. Landscape ecology and nature reserve design in the Boundary Waters Canoe Area, Minnesota. Ecology 70, 23–25.
Baker, W.L., 1992. Effects of settlement and fire suppression on landscape structure. Ecology 73, 1879–1887.
Baker, W.L., 1994. Restoration of landscape structure altered by fire suppression. Conserv. Biol. 8, 763–769.
Bauer, M.E., Loeffelholz, B., Shinneman D.J. Border Lakes Land Cover Classification. Forest Service Research Map Series RMAP-NRS-01. USDA Forest Service Northern Research Station, Newtown Square, PA, in press.
Bengtsson, J., Angelstam, P., Elmqvist, T., Emanuelsson, U., Folke, C., Ihse, M., Moberg, F., Nyström, M., 2003. Reserves, resilience and dynamic landscapes. Ambio 32, 389–396.
Benzie, J.W., 1977. Manager's handbook for jack pine in the north central states. General Technical Report NC-32. USDA Forest Service, St. Paul, MN.
Bergeron, Y., Leduc, A., Harvey, B.D., Gauthier, S., 2002. Natural fire regime: a guide for sustainable management of the Canadian boreal forest. Silva Fenn. 36, 81–95.
Beverly, J.L., Martell, D.L., 2003. Modeling *Pinus strobus* mortality following prescribed fire in Quetico Provincial Park, northwestern Ontario. Can. J. For. Res. 33, 740–751.
Burkett, V.R., Wilcox, D.A., Stottlemyer, R., Barrow, W., Fagre, D., Baron, J., Price, J., Nielsen, J.L., Allen, C.D., Peterson, D.L., Ruggerone, G., Doyle, T., 2005. Nonlinear dynamics in ecosystem response to climatic change: case studies and policy implications. Ecol. Complex. 2, 357–394.
Cardille, J.A., Ventura, S.J., 2001. Occurrence of wildfire in the northern Great Lakes region: effects of land cover and land ownership assessed at multiple scales. Int. J. Wildland Fire 10, 145–154.
Christensen, N.L., Bartuska, A.M., Brown, J.H., Carpenter, S., D'Antonio, C., Francis, R., Franklin, J.F., MacMahon, J.A., Noss, R.J., Parsons, D.J., Peterson, C.H., Turner, M.G., Woodmansee, R.G., 1996. The report of the Ecological Society of America Committee on the scientific basis for ecosystem management. Ecol. Appl. 6, 665–691.
Cissel, J.H., Swanson, F.J., Weisberg, P.J., 1999. Landscape management using historical fire regimes: Blue River, Oregon. Ecol. Appl. 9, 1217–1231.
Didion, M., Fortin, M.J., Fall, A., 2007. Forest age structure as indicator of boreal forest sustainability under alternative management and fire regimes: a landscape level sensitivity analysis. Ecol. Modell. 200, 45–58.
Drever, C.R., Peterson, G., Messier, C., Bergeron, Y., Flannigan, M., 2006. Can forest management based on natural disturbances maintain ecological resilience? Can. J. For. Res. 36, 2285–2299.
Ecological Stratification Working Group, 1995. A National Ecological Framework for Canada: Report and National Map at 1:7,500,000 Scale. Agriculture and Agri-Food Canada, Research Branch, Centre for Land and Biological Resources Research and Environment Canada, State of the Environment Directorate, Ecozone Analysis Branch, Ottawa/Hull.
Flannigan, M., Stocks, B., Turetsky, M., Wotton, M., 2008. Impacts of climate change on fire activity and fire management in the circumboreal forest. Global Change Biol. 15, 549–560.
Franklin, J.F., Forman, R.T.T., 1987. Creating landscape patterns by forest cutting: ecological consequences and principles. Landscape Ecol. 1, 5–18.
Frelich, L.E., 2002. Forest Dynamics and Disturbance Regimes: Studies from Temperate Evergreen–Deciduous Forests. Cambridge University Press, Cambridge, UK.
Frelich, L.E., Reich, P.B., 1995. Spatial patterns and succession in a Minnesota southern-boreal forest. Ecol. Monogr. 65, 325–346.
Friedman, S.K., Reich, P.B., 2005. Regional legacies of logging: departure from presettlement forest conditions in northern Minnesota. Ecol. Appl. 2, 726–744.
Fulé, P.Z., 2008. Does it make sense to restore wildland fire in changing climate? Restor. Ecol. 16, 526–531.
Gustafson, E.J., Lytle, D.E., Swaty, R., Loehle, C., 2007. Simulating the cumulative effects of multiple forest management strategies on landscape measures of forest sustainability. Landscape Ecol. 22, 141–156.
Gustafson, E.J., Shifley, S.R., Mladenoff, D.J., Nimerfro, K.K., He, H.S., 2000. Spatial simulation of forest succession and timber harvesting using LANDIS. Can. J. For. Res. 30, 32–43.
Gustafson, E.J., Zollner, P.A., Sturtevant, B.R., He, H.S., Mladenoff, D.J., 2004. Influence of forest management alternatives and land type on susceptibility to fire in northern Wisconsin, USA. Landscape Ecol. 19, 327–341.
Hansen, A.J., DeFries, R., 2007. Ecological mechanisms linking protected areas to surrounding lands. Ecol. Appl. 17, 974–988.
Hanson, D.S., 2002. Development of landtype associations in Minnesota. In: Smith, M. (Ed.), Land Type Associations Conference: Development and Use in Natural Resources Management, Planning and Research. USDA Forest Service, Newtown Square, PA, p. 98.
He, H.S., Mladenoff, D.J., 1999. Spatially explicit and stochastic simulation of forest landscape fire disturbance and succession. Ecology 80, 81–99.
He, H.S., Mladenoff, D.J., Boeder, J., 1999a. An object-oriented forest landscape model and its representation of tree species. Ecol. Modell. 119, 1–9.
He, H.S., Mladenoff, D.J., Crow, T.R., 1999b. Linking an ecosystem model and a landscape model to study forest species response to climate warming. Ecol. Modell. 114, 213–233.
Heinselman, M.L., 1973. Fire in the virgin forests of the Boundary Waters Canoe Area, Minnesota. Quaternary Res. 3, 329–382.
Heinselman, M.L., 1981. Fire intensity and frequency as factors in the distribution and structure of northern ecosystems. In: Mooney, H.A., Bonnicksen, T.M., Christensen, N.L., Lotan, J.E., Reiners, W.A. (Eds.), Fire Regimes and Ecosystem

Properties. Forest Service, General Technical Report WO-26. USDA Forest Service, Washington, DC, pp. 7–57.
Heinselman, M.L., 1996. The Boundary Waters Wilderness Ecosystem. University of Minnesota Press, Minneapolis, MN.
Hobbs, R.J., Norton, D.A., 1996. Toward a conceptual framework for restoration ecology. Restor. Ecol. 4, 93–110.
Johnson, E.A., Miyanishi, K., Weir, J.M.H., 1998. Wildfires in the western Canadian boreal forest: landscape patterns and ecosystem management. J. Veg. Sci. 9, 603–610.
Kneeshaw, D., Gauthier, S., 2003. Old growth in the boreal forest: a dynamic perspective at the stand and landscape level. Environ. Rev. 11 (Suppl. 1), S99–S114.
Landres, P.B., Morgan, P., Swanson, F.J., 1999. Overview of the use of natural variability concepts in managing ecological systems. Ecol. Appl. 9, 1179–1188.
Lindenmayer, D.B., Franklin, J.F., 2002. Conserving forest biodiversity: a comprehensive multiscaled approach. Island Press, Washington, DC.
Lindenmayer, D.B., Franklin, J.F., Fischer, J., 2006. General management principles and a checklist of strategies to guide forest biodiversity conservation. Biol. Conserv. 131, 433–445.
Lytle, D.E., Cornett, M.W., Harkness, M.S., 2006. Transferring landscape ecological knowledge in a multipartner landscape: the Border Lakes region of Minnesota and Ontario. In: Perera, A.H., Buse, L.J., Crow, T.R. (Eds.), Forest Landscape Ecology: Transferring Knowledge to Practice. Springer, New York, pp. 97–128.
Maclean, D.A., Ostaff, D.P., 1989. Patterns of balsam fir mortality caused by an uncontrolled spruce budworm outbreak. Can. J. For. Res. 19, 1087–1095.
McRae, D.J., Duchesne, L.C., Freedman, B., Lynham, T.J., Woodley, S., 2001. Comparisons between wildfire and forest harvesting and their implications in forest management. Environ. Rev. 9, 223–260.
Mehta, S., Frelich, L.E., Jones, M.T., Manolis, J., 2004. Examining the effects of alternative management strategies on landscape-scale forest patterns in northeastern Minnesota using LANDIS. Ecol. Modell. 180, 73–87.
Millar, C.I., Stephenson, N.L., Stephens, S.L., 2007. Climate change and forests of the future: managing in the face of uncertainty. Ecol. Appl. 17, 2145–2151.
Minnesota Forest Resources Council, 2003. Recommended desired outcomes, goals and strategies: northeast landscape region. Minnesota Forest Resources Council Landscape Program. http://www.frc.state.mn.us/Info/MFRCdocs.html (last accessed 29 May 2009).
Mitchell, R.J., Palik, B.J., Hunter, M.L., 2002. Natural disturbance as a guide to silviculture. Forest Ecol. Manage. 155, 315–317.
Mladenoff, D.J., 2005. The promises of landscape modeling: successes, failures, and evolution. In: Wiens, J.A., Moss, M.R. (Eds.), Issues and Perspectives in Landscape Ecology. Cambridge University Press, Cambridge, UK, pp. 90–100.
Mladenoff, D.J., DeZonia, B., 2004. APACK 2.23 Analysis Software: User's Guide. http://landscape.forest.wisc.edu/projects/apack/ (last accessed 29 May 2009).
Mladenoff, D.J., He, H.S., 1999. Design, behavior and applications of LANDIS, an object-oriented model of forest landscape disturbance and succession. In: Mladenoff, D.J., Baker, W.L. (Eds.), Spatial Modeling of Forest Landscape Change. Cambridge University Press, Cambridge, UK., pp. 125–162.
Mladenoff, D.J., Host, G.E., Boeder, J., Crow, T.R., 1996. LANDIS: a spatial model of forest landscape disturbance, succession, and management. In: Goodchild, M.R., Steyaert, L.T., Parks, B.O. (Eds.), GIS and Environmental Modeling: Progress and Research Issues. GIS World Books, Fort Collins, CO, pp. 175–180.
Mladenoff, D.J., White, M.A., Pastor, J., Crow, T.R., 1993. Comparing spatial pattern in unaltered old-growth and disturbed forest landscapes. Ecol. Appl. 3, 293–305.
Nonaka, E., Spies, T.A., 2005. Historical range of variability in landscape structure: a simulation study in Oregon, USA. Ecol. Appl. 15, 1727–1746.
Ontario Ministry of Natural Resources, 2001. Forest management guide for natural disturbance pattern emulation, Version 3.1. Queen's Printer for Ontario, Toronto.
Pastor, J., Post, W.M., 1986. Influence of climate, soil moisture, and succession on forest carbon and nitrogen cycles. Biogeochemistry 2, 3–27.
Poiani, K.A., Richter, B.D., Anderson, M.G., Richter, H.E., 2000. Biodiversity conservation at multiple scales: functional sites, landscapes, and networks. BioScience 50, 133–146.
Ravenscroft, C., Scheller, R.M., Mladenoff, D.J., White, M.A. Simulating forest restoration in a mixed ownership landscape under climate change. Ecol. Appl., in press.
Reed, R.A., Johnson-Barnard, J., Baker, W.L., 1996. Contribution of roads to forest fragmentation in the Rocky Mountains. Conserv. Biol. 10, 1098–1106.
Sample, V.A., 1994. Building partnerships for ecosystem management on mixed ownership landscapes. J. Forest. 92, 41–44.
Scheller, R.M., Mladenoff, D.J., 2004. A forest growth and biomass module for a landscape simulation model, LANDIS: design, validation, and application. Ecol. Modell. 180, 211–229.
Scheller, R.M., Domingo, J.B., Sturtevant, B.R., Williams, J.S., Rudy, A., Gustafson, E.J., Mladenoff, D.J., 2007. Design, development, and application of LANDIS-II, a spatial landscape simulation model with flexible temporal and spatial resolution. Ecol. Modell. 201, 409–419.
Scheller, R.M., Mladenoff, D.J., Crow, T.R., Sickley, T.A., 2005. Simulating the effects of fire reintroduction versus continued fire absence on forest composition and landscape structure in the Boundary Waters Canoe Area, northern Minnesota, USA. Ecosystems 8, 396–411.
Schulte, L.A., Mladenoff, D.M., Crow, T.R., Merrick, L.C., Cleland, D.T., 2007. Homogenization of northern U.S. Great Lakes forests due to land use. Landscape Ecol. 22, 1089–1103.
Shifley, S.R., Thompson III, F.R., Dijak, W.D., Larson, M.A., Millspaugh, J.J., 2006. Simulated effects of forest management alternatives on landscape structure and habitat suitability in the Midwestern United States. Forest Ecol. Manage. 229, 361–377.
Shinneman, D.J., Baker, W.L., 1997. Nonequilibrium dynamics between catastrophic disturbances and old-growth forests in ponderosa pine landscapes of the Black Hills. Conserv. Biol. 11, 1276–1288.
Superior Mixed Forest Ecoregional Planning Team, 2002. The Superior Mixed Forest Ecoregion: A Conservation Plan. The Nature Conservancy, Madison, WI..
Swain, A.M., 1980. Landscape patterns and forest history in the Boundary Waters Canoe Area, Minnesota: a pollen study from Hug Lake. Ecology 61, 747–754.
Swetnam, T.W., Allen, C.D., Betancourt, J.L., 1999. Applied historical ecology: using the past to manage for the future. Ecol. Appl. 9, 1189–1206.
Syphard, A.D., Clarke, K.C., Franklin, J., 2007. Simulating fire frequency and urban growth in southern California coastal shrublands, USA. Landscape Ecol. 22, 431–445.
Thompson, J.R., Johnson, K.N., Lennette, M., Spies, T.A., Bettinger, P., 2006. Historical disturbance regimes as a reference for forest policy in a multiowner province: a simulation experiment. Can. J. For. Res. 36, 401–417.
Tinker, D., Romme, W., Despain, D., 2003. Historic range of variability in landscape structure in subalpine forests of the Greater Yellowstone Area, USA. Landscape Ecol. 18, 427–439.
Turner, M.G., Gardner, R.H., O'Neill, R.V., 2001. Landscape Ecology in Theory and Practice. Springer-Verlag, New York.
USDA Forest Service, 2001. Boundary Waters Canoe Area Wilderness Fuel Treatment—Record of Decision and Final Environmental Impact Statement. USDA Forest Service, Milwaukee, WI.
USDA Forest Service, 2004. Superior National Forest Land and Resource Management Plan. USDA Forest Service, Milwaukee, WI.
USDI National Park Service, 2002. Fire Management Plan for Voyageurs National Park. United States Department of Interior National Park Service, International Falls, MN.
Valett, H.M., Crenshaw, C.L., Wagner, P.F., 2002. Stream nutrient uptake, forest succession, and biogeochemical theory. Ecology 83, 2888–2901.
Van Wagner, C.E., 1978. Age–class distribution and the fire cycle. Can. J. For. Res. 8, 220–227.
Vors, L.S., Schaefer, J.A., Pond, B.A., Rodgers, A.R., Patterson, B.R., 2007. Woodland caribou extirpation and anthropogenic landscape disturbance in Ontario. J. Wildlife Manage. 71, 1249–1256.
Ward, B.C., Mladenoff, D.J., Scheller, R.M., 2005. Simulating landscape-level effects of the interaction between residential development and public forest management in northern Wisconsin, USA. For. Sci. 51, 616–632.
Wei, Y., Hoganson, H., 2006. Spatial information for scheduling core area production in forest planning. Can. J. For. Res. 36, 23–33.
White, M.A., Host, G.E., 2008. Forest disturbance frequency and patch structure from pre-European settlement to present in the Mixed Forest Province of Minnesota, USA. Can. J. For. Res. 38, 2212–2226.
Wiens, J.A., 1997. Wildlife in patchy environments: metapopulations, mosaics, and management. In: McCullough, D. (Ed.), Metapopulations and Wildlife Conservation Management. Island Press, Washington, DC, pp. 53–84.
Wimberly, M.C., 2002. Spatial simulation of historical landscape patterns in coastal forests of the Pacific Northwest. Can. J. For. Res. 32, 1316–1328.
Woodman, B.D., 2005. The effect of season of fire on post-fire legacies in northwestern Ontario red pine (*Pinus resinosa*) mixedwoods. Master's Thesis. University of Waterloo, Waterloo, Ontario.
Woods, G.T., Day, R.J., 1977. A fire history study of Quetico Provincial Park. Fire Ecology Study Report No. 4. Ontario Ministry of Natural Resources, Atikokan, Ontario.
Xi, W., Coulson, R.N., Waldron, J.D., Tchakerian, M.D., Lafon, C.W., Cairns, D.M., Birt, A.G., Klepzig, K.D., 2008. Landscape modeling for forest restoration planning and assessment: lessons from the Southern Appalachian Mountains. J. Forest. 106, 191–197.