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Evaluating simulated effects of succession, fire, and harvest for LANDIS PRO forest landscape model

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ABSTRACT

Forest landscape models are effective tools for exploring the effects of long-term and large-scale landscape processes such as seed dispersal, fire, and timber harvest. These models have been widely used for about a decade, and although significant advances in theory and technology have been incorporated into their development, evaluating the veracity of simulated results from forest landscape models remains challenging. In this study, we evaluated simulated forest succession and the effects of simulated fire and harvest by a spatially explicit forest landscape model (LANDIS PRO), initialized using forest inventory data (second and third tier data from years 2000 and 2010). Our results suggested that the initialized forest landscape constructed from the year 2000 forest inventory data adequately represented the forest composition and structure from that year. The simulated density and basal area from year 2010 adequately represented the forest inventory data from that year at landscape scales. Our results indicated that the simulated fire and harvest effects were comparable to the field data (measured density and basal area). Results in this study quantified the near-term reliability and confidence of the model as well as prediction uncertainties.

1. Introduction

Forest landscape models (FLMs) have been a primary tool for investigating the effects of forest succession and forest landscape processes such as fire and harvest on species composition, age structure, and spatial pattern (Baker, 1992; Foster et al., 1998; Wallin et al., 1996; He et al., 2011) because controlled field experiments designed to investigate these long-term, large-scale effects are difficult (He et al., 2011; Shifley et al., 2006). FLMs are increasingly used to address actual problems in forest management planning, and thus, evaluating predictions of FLMs becomes not only important but necessary. Evaluation of model results ensures the model is acceptable for its intended use and meets the user requirements (Rykiel, 1996). Evaluating prediction results provides an opportunity to better understand prediction uncertainty and model strengths and deficiencies (Bellassen et al., 2011; Shifley et al., 2009). Many studies have suggested that evaluating the results of FLMs can reveal the level of reality and accuracy in the simulation of real forest ecosystems (Scheller and Mladenoff, 2004 Wang et al., 2013a).

Evaluating model results in the traditional sense involves comparing model predictions to independent datasets, a difficult task for FLMs because predicted results of FLMs are spatial and temporal, and independent spatiotemporal data do not often exist (He, 2008). Consequently, most previous evaluations of ecological model results have compared the simulated results with other independently developed model simulations, sporadic field sample data (e.g., flux tower data), or empirical knowledge (Chen et al., 2002; Bugmann, 2001; Gustafson et al., 2000; He et al., 2005; Scheller and Mladenoff, 2004; Thompson et al., 2011). For example, Liu et al. (2012) conducted their model evaluation by comparing their simulated results with the statistics from independent historical empirical studies in the Great Xing'an Mountains. Bu et al. (2008) evaluated their landscape model simulation results by comparing them with other model simulations and found that their results were consistent with other studies conducted using gap models in the same region. Chang et al. (2007) and Liu et al. (2009) both conducted their simulated fire verifications by examining whether significant difference (one-simple *t*-test) exists between the derived and the simulated fire parameters. Scheller et al. (2011) evaluated their model results by using flux

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tower NEE (net ecosystem exchange) data and found that the model successfully captured the timing and magnitude of NEE for 2005 and 2006 at the stand scale.

FLM predictions involve multiple spatial scales from pixels to landscape. Evaluating pixel-level predictions should be logically conducted by comparing to corresponding stand/plot data whereas evaluating landscape-level predictions involves comparing multiple sets of stand/plot data stratified throughout the landscape. Most prior studies evaluated FLM predictions using field data at the stand scale (Scheller and Mladenoff, 2004; Seidl et al., 2012). Few have evaluated FLM predictions at the landscape scale, and studies that evaluated the effects of the simulated forest landscape processes (succession, regeneration, fire, and harvest) are even more scarce. Evaluating the simulated effects of forest landscape processes may improve simulation credibility and thus reduce prediction uncertainties (Purves and Pacala, 2008).

The objective of this research was to evaluate the FLM predictions using forest inventory data at landscape scales. Specifically, we (a) initialized and calibrated a forest landscape model LANDIS PRO based on the forest inventory data, (b) evaluated the vegetation succession dynamic at the landscape scale, and (c) evaluated the simulated fire and harvest against the observed field data. This study can improve the reliability and confidence of FLM results and provide a sound scientific basis for the application of FLMs.

2. Materials and methods

2.1. Study area

Our study landscape is located in the Great Xing'an Mountain region in northeastern China (51°35′–53°25′N, 122°25′–125°35′E), including the Huzhong, Tahe, and Xinlin Forestry Bureaus (2.8×10^6 ha) (Fig. 1). The region has a temperate, continental climate, with long and severe winters but short and warm summers. The study area has an average elevation of 849 m, with primarily hilly mountains ranging from 173 m to 1511 m above sea level. Average annual precipitation and temperature are 428 mm and −2.8 °C, respectively, and the growing season is ∼110 days. The canopy species composition is relatively simple in this region. The main tree species include larch (*Larix gmelinii*), Mongolian Scots pine (*Pinus sylvestris* var. *mongolica*), Korean spruce (*Picea koraiensis*), white birch (*Betula platyphylla*), and aspen (*Populus davidiana*). Minor species occur in specific habitats, such as dwarf pine (*Pinus pumila*), mostly at elevations >800 m, and willow (*Chosenia arbutifolia*), commonly found along riversides.

Fire and timber harvest are two major forest landscape processes in the Great Xing'an Mountains. Historically, fire regulated species composition and forest successional stage. For more than a half century, the Chinese government has implemented a fire suppression policy (Chang et al., 2008) that has profoundly altered the fire regime in this region, significantly changing the mean fire return interval from 30 years in the past to about 270 years at present. Fires have changed from historically frequent but low intensity to currently infrequent but high intensity (sometime catastrophic).

Timber harvest has also altered the forest composition and structure of this region. Based on historical forest inventory data, continuous timber harvest activities have resulted in forest landscapes that are dominated by mid-seral, secondary forests, excepting for these forests in the natural reserves. To maintain long-term, sustainable forest development, timber harvest has been restricted to its current level (the total annual harvest volume in our study area was approximately 4.5×10^5 m^3, annual 0.3% of

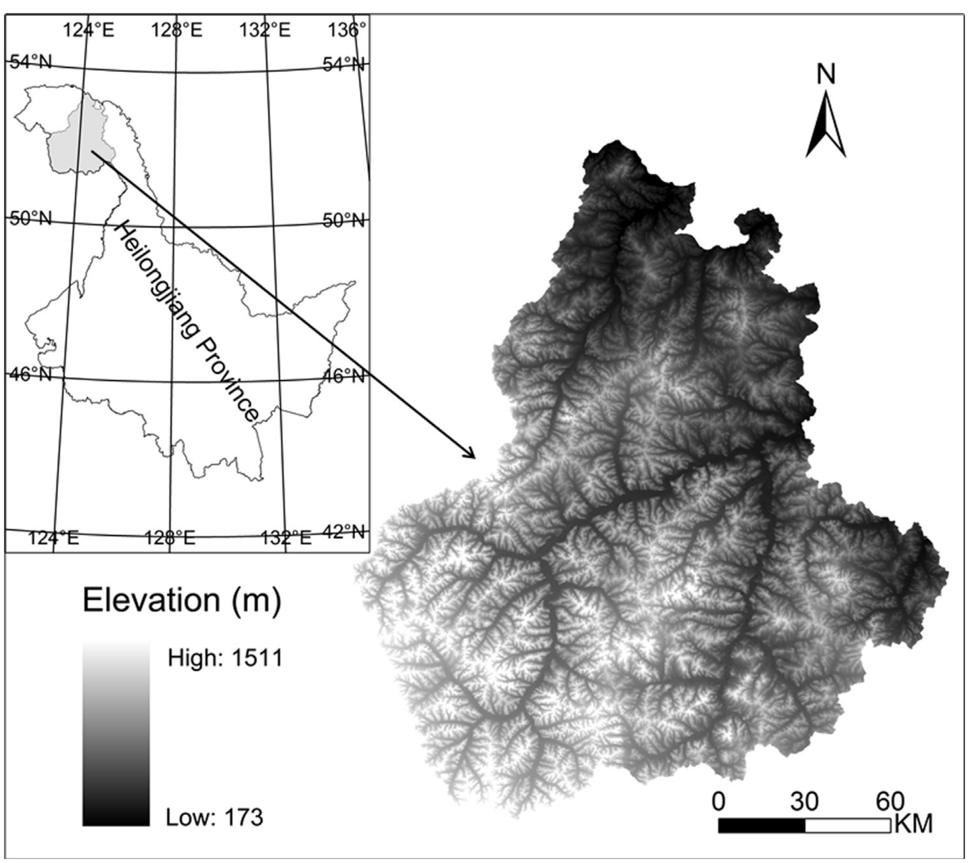

Fig. 1. The geographic location and elevation of the study area.

total volume removed) by the government of China since 1999 (Li et al., 2013; Hu and Liu, 2006).

2.2. Description of forest landscape model

We used a spatially explicit forest landscape model, LANDIS PRO (v. 7.0, http://landis.missouri.edu) to evaluate FLM predictions with forest inventory data at landscape scales. LANDIS PRO is a grid-based FLM designed to explore succession under natural (e.g., fires) and anthropogenic (e.g., timber harvesting) disturbances (Wang et al., 2013b, 2014; Fraser et al., 2013). LANDIS can be used to simulate forest landscape change over large spatial (10^3–10^8 ha) and temporal (10^1–10^3 years) scales. Differing from most FLMs, LANDIS PRO simulates forest landscape processes in combination with the simulation of succession dynamics at a tree species level. In LANDIS, a landscape is modeled as a grid of cells with vegetation information stored as attributes for every grid cell. At each grid, the model tracks numbers of trees and diameter at breast height (DBH) by species age cohort. The model uses forest inventory data for model parameterization, calibration, and validation.

In LANDIS PRO, the model simplifies individual tree information and within-stand processes in which large-scale questions such as spatial pattern and disturbances can be adequately addressed. Species succession is a site-level, competitive process driven by species vital life history attributes (e.g., longevity, age of sexual maturity, shade tolerance class, fire tolerance class). The LANDIS PRO succession and dispersal module adds number of tree for each species age cohort present at each cell. The succession module uses stand density index (SDI) to determine growing space, which regulates seedling establishment and self-thinning (Reineke, 1933). Stand density index (SDI) is used for growing space which refers to the surface area available for nutrients, water, and light accumulations for tree growth (Oliver and Larson, 1996). Minimum growing space is further derived for each standard tree based on maximum SDI. The competition intensity is quantified by growing space occupied (GSO), estimated by the percentage of the total minimum growing space required by all trees presently in a grid cell and estimated competition intensity is used to model the regeneration process. With the quantitative output information (e.g., basal area, stand density, age cohort), the model results can be evaluated by directly comparing the simulated results with the forest inventory data. The added quantitative stand attributes improve the realism of the simulated succession dynamics (Wang et al., 2013b).

Fire is an important forest landscape process. The fire module of LANDIS PRO includes three major components: fire occurrence, fire spread, and fire effects. It divides a fire occurrence into two consecutive processes: fire ignition and fire initiation. Fire ignition agents are either natural (e.g., lightning) or anthropogenic (e.g., accidental). The module generates the number of ignitions based on the Poisson distribution with the given ignition density (i.e., average fire ignitions per decade per hectare) (Yang et al., 2004, 2008). Whether a fire event can be initiated depends on the fuel loading, arrangement, and moisture content. For each initiation, LANDIS will randomly select a fire size from a log-normal distribution with fire parameters (mean fire size, standard deviation of fire size) to simulate fire spread. The fire module can simulate fire spread behavior with respect to fuel configuration, topography, and prevailing wind, with the fire spreading out differentially to neighboring sites along eight directional (N, NE, E, SE, S, SW, W, NW) gradients. Fire intensity is determined by the quantity and quality of fuel. Fires of different intensities affect different age cohorts (e.g., fire is a bottom-up disturbance and fires of increasing severity affect younger age classes first). To determine the percent mortality for each age-cohort of each species, a logistic regression model is used (He and Mladenoff, 1999).

In the harvest module, harvest is conducted with stand and management area maps, which provide harvest boundaries. Harvest events can occur within a certain management area at any time step and have the option of a specified interval. The stand map is also an integer-based raster that details stand boundaries. Stands comprise the smaller contiguous units within a management area. Clearcut, thinning, and group selection can be selected for the harvest module (Fraser et al., 2013).

2.3. Model initialization and parameterization

The five most common tree species in the study region were modeled in this study. Species' vital attributes (Table 1) were estimated based on previous studies of this region (He et al., 2002b; Xu et al., 2004a). We used China National Forest Inventory second and third tier data (2172 plots) and an extant stand map of year 2000 to construct the initial forest composition map. China National Forest Inventory second and third tier data (website: http://www.cfsdc.org) contained number of trees and age class information by species. The forest stand maps recorded boundaries of stands and species composition in each stand polygon. We integrated the stand map and the second and third tier data (point) to derive number of trees by age class for each species in the initial forest composition map (raster). To reduce computational loads during landscape simulation, the forest composition map was gridded to a 90 m cell size (0.81 ha), which yielded 2609 rows × 2217 columns.

The heterogeneous landscape was stratified into relatively homogeneous landtypes. The landtype map was constructed based on the land use data, classified TM imagery, and the digital elevation model (DEM). In our study, we derived six landtypes (non-forest, terrace, south-facing slope, north-facing slope, ridge top, water body), and each landtype was homogeneous in terms of resource availability represented by maximum growing space available and species assemblage represented by species establishment probability (SEP). SEP is a value ranging from 0 to 1 that quantifies how different environmental conditions favor a particular species in terms of its seedling establishment and subsequent species succession. The SEP values used as input to LANDIS were derived from the available literature of previous studies in similar regions (He et al., 2002a; Li et al., 2013).

The current fire regime for our model simulation was parameterized based on historical fire records from 1967 to 2005. Based on these records, we calculated the mean fire return interval, mean fire size, and fire ignition density of six land types (Table 2). Our study area was divided into three management areas: harvesting was permitted, restricted, and banned. All these management area boundaries were obtained from the stand map. The stand map is an integer-based raster that details stand

Table 1
Species life history attributes for the study area.

Species	LON[a]	MTR[b]	SHD[c]	FIR[d]	MAXD[e]	MDBH[f]	MSDI[g]	NPGS[h]
Larch	300	20	2	4	150	55	600	10
Pine	250	40	2	3	200	60	560	20
Spruce	300	30	4	1	150	60	520	10
Birch	150	15	1	3	2000	30	690	30
Aspen	120	10	1	2	2000	50	680	30

[a] Species longevity (years).
[b] Age of maturity (years).
[c] Shade tolerance class (1–5, 1 = least tolerant, 5 = most tolerant).
[d] Fire tolerant class (1–5, 1 = least tolerant, 5 = most tolerant).
[e] Maximum seedling distance (m).
[f] Maximum diameter of species in centimeters.
[g] Maximum stand density (trees/ha).
[h] Number of potential germination seeds.

Table 2
Parameters for the fire scenario and SEPs by species for each land type in the study area.

Land type	TAO[a]	MRI[b]	FI[c]	MF[d]	MRD[e]	SEP of larch	SEP of pine	SEP of spruce	SEP of birch	SEP of aspen
Non-forest	17080	1500	0.0020	0	0.6500	0	0	0	0	0
Terrace	319200	500	0.0018	90	0.7500	0.2000	0.0500	0.0500	0.0300	0.0700
South	1064000	150	0.0033	200	0.7500	0.3500	0.3500	0.0050	0.3500	0.0300
North	1170400	160	0.0029	210	0.7500	0.4000	0.0100	0.0300	0.1500	0.0050
Top	187600	140	0.0081	238	0.6000	0.2000	0.0100	0	0.0700	0.0200
Water	41720	0	0	0	0.7500	0	0	0	0	0

[a] Total area occupied (ha).
[b] Mean fire return interval (yr).
[c] Fire ignition density (number of fires/decade/ha).
[d] Mean fire size (ha).
[e] Maximum relative density.

boundaries. Stands are smaller contiguous units within a management area. The stand map of our study area was a GIS layer which consisted of 113,778 survey units (polygons, each averaged 10 ha) that included boundaries of survey units and species composition within each unit. In the past 30 years, pine and spruce were extracted extremely because of their high economic value, and now relatively few of pine and spruce occupied in our study region. Thus to fulfill the current management policy, the harvest species were larch, birch, and aspen, whereas pine and spruce were protected from cut in this study. We obtained the annual harvest amount from the calculation of Chinese National Forest Inventory third tier data. We added harvest adjacency constraints to ensure adjacent stands are not harvested for at least 5 years to prevent creation of large openings in the boreal forests. We assumed that management area, divided sub-areas, and harvest module parameters remain unchanged in all simulation periods (Table 3).

2.4. Simulation scenarios and data analysis

To evaluate FLM results with forest inventory data, we designed three simulation scenarios: (1) succession only scenario (fire and harvest were not simulated), (2) fire only scenario (fire and succession were simulated with fire regime approximating the current fire suppression practiced in recent decades), and (3) harvest only scenario (harvest and succession were simulated with harvest regime reflecting the current forest harvest activities). The entire study was simulated for 300 years (up to year 2300) at a 5-year time step with 5 replicates to assess model stochasticity.

To evaluate model simulation results, the forest age classes of the study area were calculated statistically based on five classes: seedling (0–40 years), middle-aged (41–100 years), quasi-matured (101–140 years), matured (141–180 years), and over-matured cohorts (>180 years) (Xu et al., 2004b; Gustafson et al., 2010; Li et al., 2013). To assess the simulated results with the forest inventory data, paired *t*-tests were conducted to test the difference between the simulated results and forest inventory data in both tree density and basal area aspects. These paired *t*-tests indicated the extent to which the simulated landscape coincided with the real landscape to determine whether the simulated results could be accepted for future study. All statistical analyses were performed using SPSS 16.0 software.

2.5. Model calibration

We initialized the model and calibrated it for different landtypes. We used 70% of forest inventory plot data (1520 plots), which usually consist of the number of trees and DBH information of different species and age cohorts, and the extant stand map of year 2000, which provides boundaries to initialize the forest composition map of year 2000. We then ran the model for 10 years and selected about 30% of the forest inventory data (652 plots) to evaluate the initialized map for year 2000. We iteratively adjusted the age-DBH relationship in the model to make the initialized species density and basal area match the remaining 30% of the forest inventory data until no significant differences were found between the simulated data and the remaining forest inventory data.

2.6. Evaluating simulated succession, fire, and harvest results

We divided the study region into six landtypes, but only three (terrace, south-facing slope, and north-facing slope) were statistically evaluated because they covered more than 90% of the total landscape. We used the forest landscape of year 2000 (as described above) as a starting point to conduct the model simulation for 10 years (to 2010) for the succession only scenario. For each landtype, we then iteratively adjusted the number of potential germination seeds (NPGS, parameters in LANDIS PRO that influences species density and basal area) for each simulated species until the simulated density and basal area of year 2010 matched the forest inventory data of year 2010 at the landscape scale and different landtypes until no significant differences in both tree density and basal area were found between the predictions and observations. After the model result verification was passed, we conducted the succession only scenario to verify whether the long-term simulated results were consistent with the ecological principles of forest dynamics and species succession.

To evaluate the effects of fire on the simulated forest landscape, we compared the simulated results with field data at different successional periods. First, we ran the module with the fire module switched on for 300 years and selected 40 fires with different years and locations in which no fires recurred in the next simulated 25 years from the simulated landscape based on the fire only scenario outputs. We selected only fires with low intensities

Table 3
Parameters for harvest scenario.

Management Area	Harvest type	Removal species/order	Repeat Interval	Entry year	Proportion treated	Minimum harvest BA (m^2/ha)
Harvest restricted area	Thinning	Larch, birch and aspen	5	5	0.13	17.2
Harvest permitted area	Thinning	Larch, birch and aspen	5	5	0.15	14.9

because more than 90% fires in the region occurred at this level. Second, we investigated 40 fires (low intensity fires in coniferous and broadleaf forests) 5, 10, 15, 20, and 25 years ago in the field. Third, we recorded the coniferous (larch, pine, and spruce) and broadleaf (birch and aspen) species tree number and DBH for each burned site at both simulated landscapes and sampled plots. Finally, we statistically compared the density and basal area of the simulated fires of coniferous and broadleaf species at 5, 10, 15, 20, and 25 post-fire years with corresponding fires in the field, respectively.

To evaluate the simulated forest response to harvest, we first conducted the model for 300 years (from 2000 to 2300) with the harvest module switched on (harvest only scenario). We then statistically compared the density and basal area in two harvest areas (harvest restricted area and harvest permitted area) at year 2010 with the forest inventory data. We iteratively adjusted the harvest module's parameters to ensure the simulated density and basal area of the two management areas matched the mean forest inventory data surveyed in the two corresponding harvest areas at the landscape scale.

3. Results

3.1. Model calibration

Our simulated results indicated that the initialized forest landscape constructed from the forest inventory data of year 2000 reasonably represented the forest composition of year 2000 in both density and basal area for the entire landscape and for each landtype (landscape: $p = 0.59$, south-facing slope: $p = 0.56$, north-facing slope: $p = 0.49$, terrace: $p = 0.78$). There were no significant differences (paired t-tests, df = 4, $p > 0.05$) in density and basal area between simulated results and forest inventory data for the entire landscape and for each landtype in year 2000 (Fig. 2). Thus, we accepted the calibrated model for further use.

3.2. Evaluating simulated succession results at landscape scale

Our simulated results showed that the simulated forest landscape in year 2010 closely represented the observed forest composition in both density and basal area at terrace landtype (Fig. 3a: $p = 0.61$, Fig. 3b: $p = 0.88$), south-facing slope landtype (Fig. 3c: $p = 0.64$, Fig. 3d: $p = 0.60$), and north-facing slope landtype (Fig. 3e: $p = 0.84$, Fig. 3f: $p = 0.90$), respectively. No significant differences (paired t-tests, df = 4, $p > 0.05$) were found in density and basal area between simulated results and observed data at year 2010.

In the terrace landtype, coniferous species were the dominant species; only half of coniferous areas were occupied by broadleaf species. This landtype was covered mainly by seedling (0–40 years) and middle-aged (41–100 years) cohort class trees, with relative few quasi-aged (101–140 years), matured (141–180 years), and over-matured (>180 years) individual trees (Fig. 3a). The mean basal area of coniferous and broadleaf species simulated in 2010 were 9.7 m^2/ha and 10.5 m^2/ha, respectively. The basal area distributed unevenly in all five age cohort stages; the matured stage (141–180 years) had more basal area than other stages, followed by middle-aged (41–100 years), quasi-matured (101–140

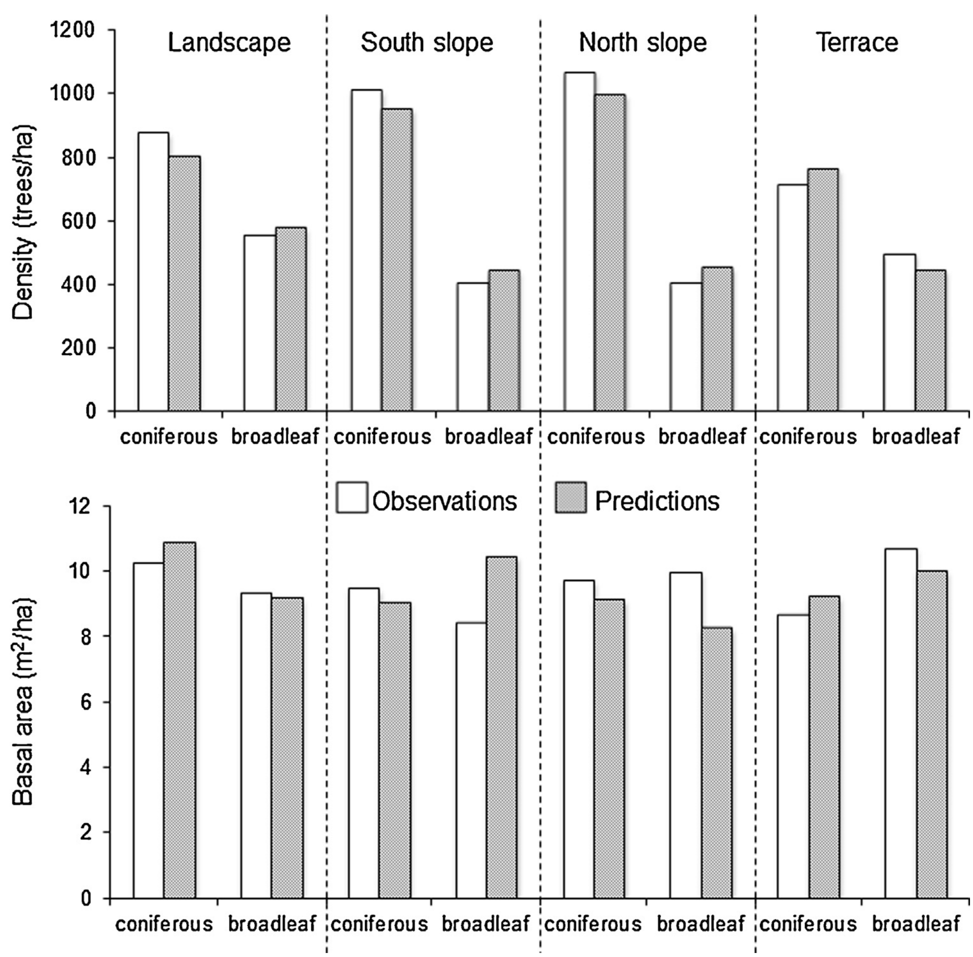

Fig. 2. Species density and basal area by coniferous and broadleaf species of landscape and landtypes for the inventory data and predictions at year 2000.

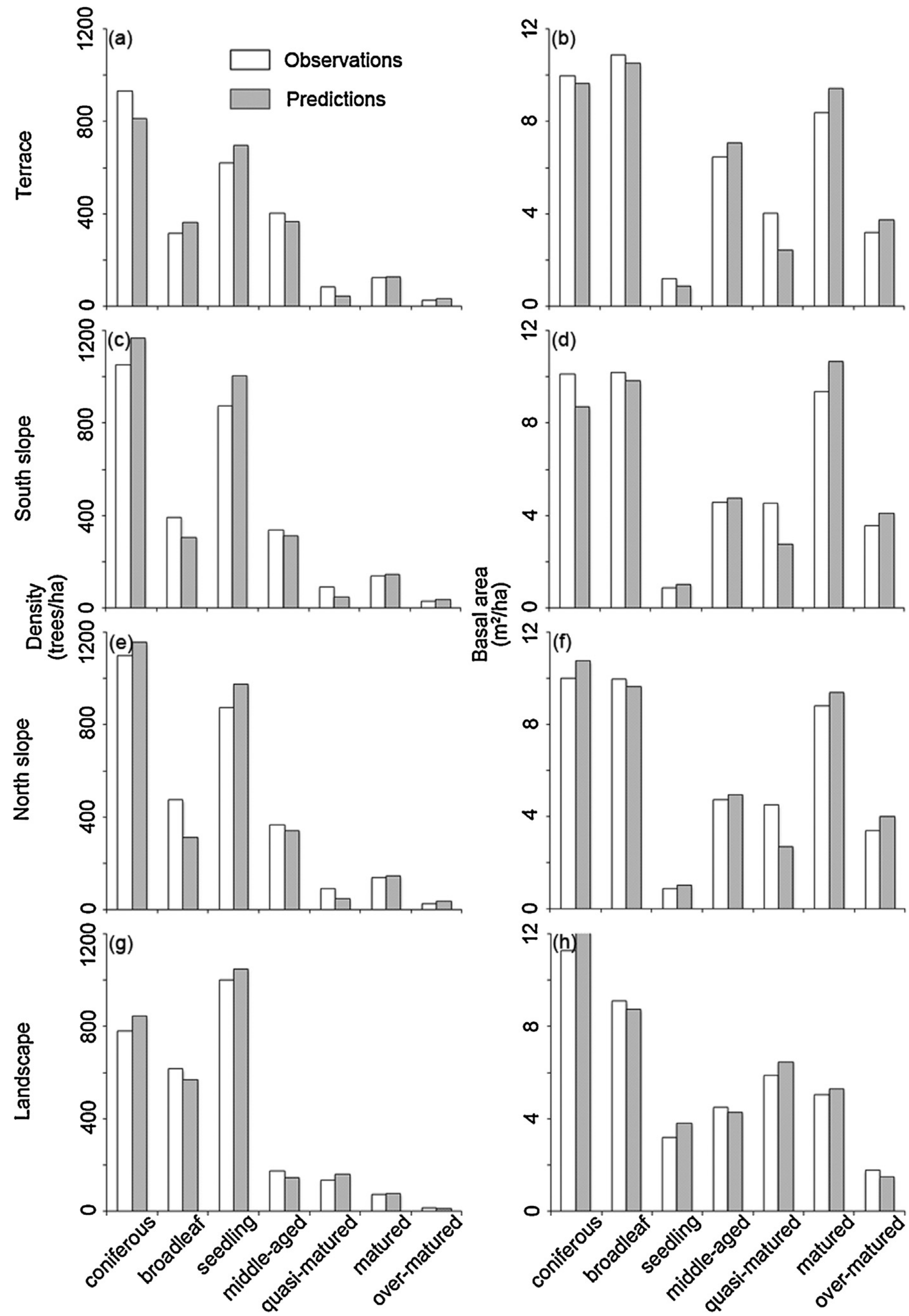

Fig. 3. Landscape and landtype species density and basal area by coniferous and broadleaf trees and different age cohorts for the inventory data and predictions at year 2010. Seedling: 0–40 years, middle-aged: 41–100 years, quasi-matured: 101–140 years, matured: 141–180 years, and over-matured cohorts: >180 years.

years), over-matured (>180 years), and seedling (0–40 years), respectively (Fig. 3b).

In the south-facing slope landtype, coniferous trees densities were obviously more abundant than broadleaf species. This landtype was also more abundant than other landtypes such as terrace and north-facing slope. Middle-aged (41–100 years) individual trees occupied a stand density of 370 trees/ha in this landtype, a higher abundance than quasi-matured (101–140 years) and matured age class (141–180 years) trees (Fig. 3c). Coniferous and broadleaf species occupied an equivalent basal area in the south-facing slope landtype, which each comprising 50% in total basal area. Matured trees had about 9.5 m^2/ha in total basal area, followed by middle-aged, quasi-matured, over-matured, and seedling trees (Fig. 3d).

Coniferous species were consistently abundant across the north-facing slope landtype, comprising 80% of the total density. The predicted stand densities were larger in seedling, middle-aged, and matured stages, but smaller in quasi-matured and over-matured stages compared to the observed data at year 2010 (Fig. 3e). The simulated results showed that coniferous species were relative more abundant than broadleaf species in total basal area. The matured age cohort trees had about 8.8 m^2/ha of basal area, comprising 41% of the total basal area, followed by middle-aged, over-matured, and quasi-matured age cohorts, respectively (Fig. 3f).

Our results showed that the simulated forest landscape of year 2010 reasonably represented the forest inventory data surveyed at year 2010 in both density and basal area across the landscape

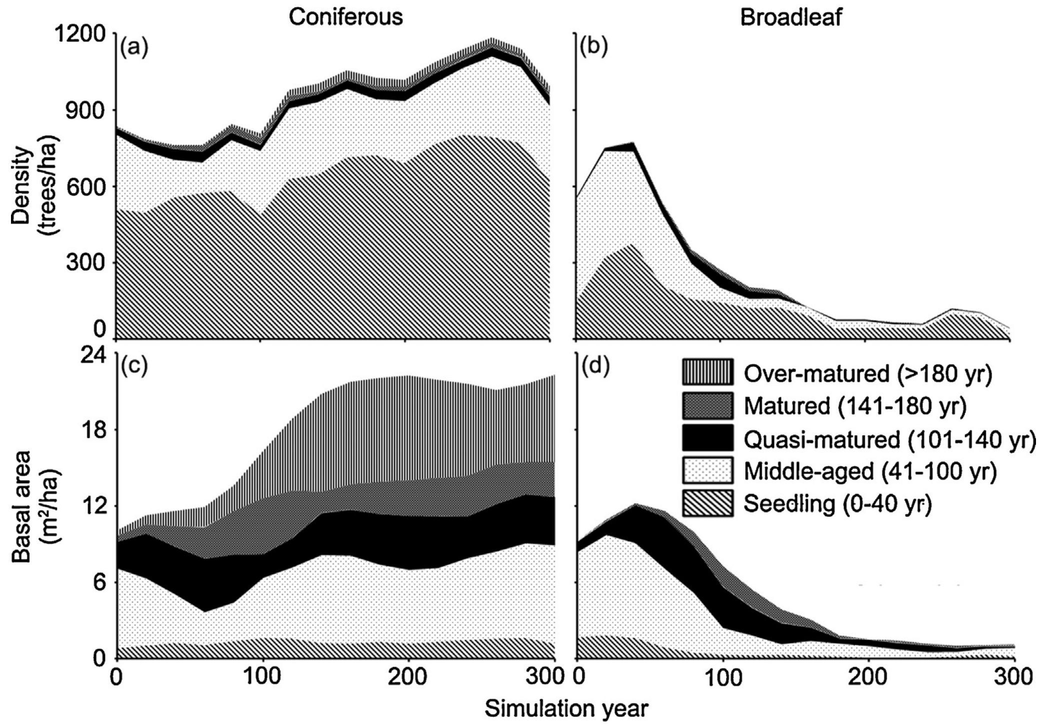

Fig. 4. Predicted density (a) and (b), basal area (c) and (d) by forest age distribution and species composition at landscape scales over 300 years in northeastern China.

(Fig. 3g: $p = 0.23$, Fig. 3h: $p = 0.28$) (Fig. 3g and h). About 60% of study areas were occupied by small trees in early-seral stage (seedling stage), and relative few trees appeared in middle-aged and quasi-matured stages (Fig. 3g). The basal area of broadleaf trees (8.8 m^2/ha) accounted for 40% of total basal area in the landscape at year 2010. Total basal area distributed relative evenly among seedling, middle, quasi-matured, and matured age groups (Fig. 3h).

Our results showed that the curves of coniferous species were significantly different from the broadleaf ones performed in long-term simulation (Fig. 4). Coniferous species densities of different age classes were increased slightly over time (Fig. 4a). The predicted density of broadleaf species increased distinctly in the first 40 years because of the lack of disturbance collaboration and available growing space to occupy, and then the density decreased

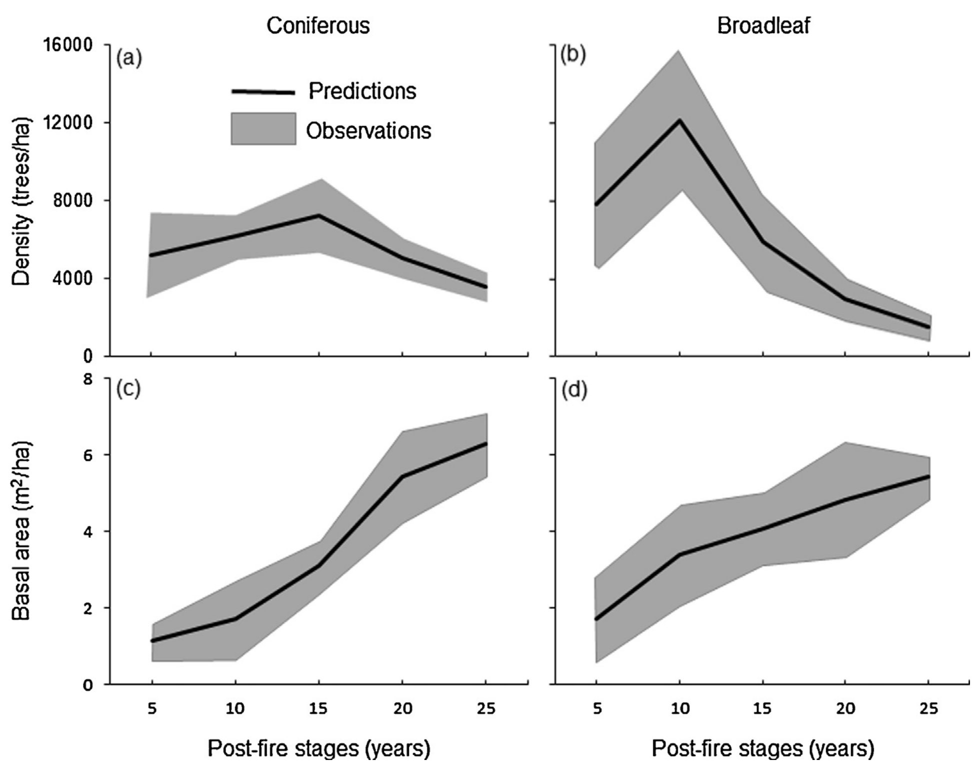

Fig. 5. Changes in predicted stand density of coniferous (a) and broadleaf (b), and basal area of coniferous (c) and broadleaf (d) in burned areas in relation to post-fire year.

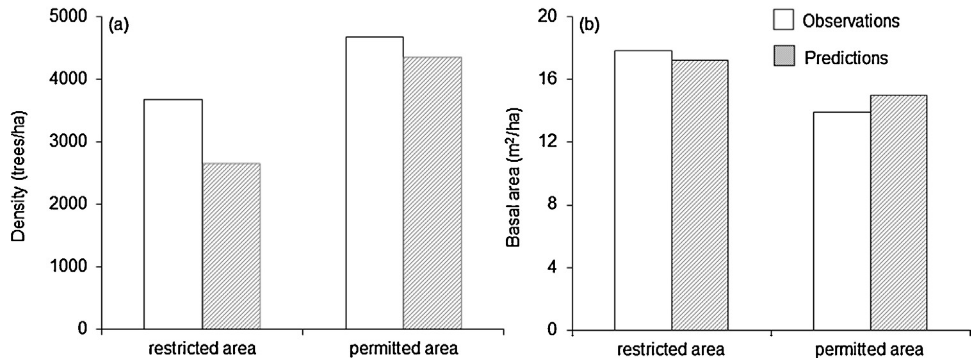

Fig. 6. Mean values of density (a) and basal area (b) by harvest restricted and permitted area for the observations and predictions at year 2010.

significantly over the later 260 simulation years as a result of mortality from self-thinning (Fig. 4b). The basal area of coniferous species increased to a peak at 180 years, followed by slight declines in the next 50 years (Fig. 4c). The predicted basal area of broadleaf species increased from 2000 to 2040, followed subsequently by gradual declines over time (Fig. 4d).

3.3. Evaluating simulated fire and harvest

The results showed that the post-fire density of coniferous species increased in the first 15 years (up to 7219 trees/ha) and then decreased to 3580 trees/ha after 25 years (Fig. 5a). Compared to coniferous species, the broadleaf species more sharply increased in density in the first 10 years and then decreased in the next 15 years (Fig. 5b). The post-fire basal areas of coniferous and broadleaf species both showed increasing trends throughout the observed 25 years (Fig. 5c and d). The predicted density and basal area were within the observed ranges of the field sample data (gray area).

Our simulated results showed that the simulated density in harvest restricted area (2653 trees/ha) was lower than the harvest permitted area (4340 trees/ha) (Fig. 6a), but basal area in harvest restricted area (17.2 m^2/ha) was higher than harvest permitted area (15.0 m^2/ha) (Fig. 6b). Trees that appeared in the harvest permitted area were extracted more frequently than those in the harvest restricted area, and more growing space was thus allocated for pioneer species (birch and aspen) to establish in the harvest permitted area. Similar results were found between these two management areas in forest inventory data. Compared with field inventory data and simulated results, the harvest module reasonably reflected the harvest events.

4. Discussion

Evaluating FLMs simulated results is critical in quantifying the reliability and confidence of model predictions (Clark et al., 2001; Shifley et al., 2008). In this study, simulated succession and fire and harvest effects were directly compared with forest inventory data at landscape scales and indicated that the simulated succession and fire and harvest results were comparable with the inventory data. The results showed that the predicted accuracy (with small error) in density and basal area at the landscape level (Fig. 3g and h) were higher than those at these landtypes (terrace, south-facing slope, and north-facing slope), possibly because the forest inventory data investigated in year 2010 have lower variance at the landscape level than at these landtypes (Guisan et al., 2007).

Our results showed that the density and basal area of coniferous species were higher than broadleaf species. Coniferous species (larch was the dominant species) had a wider distribution than broadleaf species in this area (Wang et al., 2006; Liang et al., 2011, 2014), and dominant species tended to be modeled more accurately than minor species because of relatively abundant field data available for parameter calibration (Mcpherson et al., 2004; Guisan et al., 2007). Broadleaf species (mainly birch and aspen) were less abundant than coniferous species. Broadleaf species such as birch and aspen are early successional species, and their abundance declined as forest aged and succession progressed continued (Luo et al., 2014).

Results showed that the coniferous species density in the north-facing slope landtype was the highest, followed by south-facing slope landtype and terrace landtype. There were no significant differences in basal area among these three landtypes (basal area ranging from 8.7 to 10.7 m^2/ha), consistent with previous studies that described coniferous species located mainly in the cold north-facing slope landtype (Leng et al., 2008). The distribution of solar energy and available water were the primary drivers of the current coniferous species distribution (Zhou, 1997). The coniferous species density was low whereas basal area was high in all three landtypes, possibly because the natural life cycle of tree mortality and large broadleaf trees occupied most basal area values.

Our simulated results showed that most trees species occurred in seedling and mid-matured age stages, similar to other boreal forest regions in China (Li et al., 2013; Liu et al., 2012), because initial forest landscapes were influenced by natural and anthropogenic disturbances that occurred in the past and persisted. Likewise, low tree densities with high basal area values were found in the matured age stage group, a finding also evident in the forest inventory data of 2000 and 2010, which contained some old larch, pine, and spruce trees historically serving as the seed trees for natural generation.

The simulated fire and harvest results were comparable with forest inventory data. The densities of burned area changed significantly with succession. The densities of broadleaf species increased sharply in the first 10 years and decreased in the next 15 years because fires removed most trees and released growing space for pioneer species such as birch and aspen to establish. After the released growing space was fully occupied by these pioneer species in 10 to 15 years, self-thinning started to cause mortality and reduced the number of tree in the following years (Wang et al., 2014). The simulated harvest results also captured the dynamics of actual harvest events. In addition, we only evaluated the effects of the simulated fire and harvest at the landscape scale because fire and harvest events ignore landtype boundaries and are best evaluated across landtypes.

In our study, the forest inventory data had a relatively short time span (2000–2010 for the study area), which prohibited it from being used to evaluate landscape change for long simulation periods. Nevertheless, our forest inventory data provided a spatial series of 3103 plots that randomly distributed in our study area, which may mediate the relatively short survey time span (Araújo et al., 2005). We believe our study provides a convincing case of evaluating FLMs predictions and the effects of the simulated forest landscape processes.

5. Conclusions

In this study, we evaluated the prediction results of a spatially explicit forest landscape model (LANDIS PRO) at the landscape scale. The LANDIS PRO model provides quantitative information (e.g., basal area, stand density), which can be applied to address critical questions regarding forest composition and structure. Based on the number of trees and DBH by species age cohort, the model can more realistically simulate natural disturbance and forest management scenarios. In summary, we concluded that (1) the LANDIS PRO model can be successfully implemented in such a large area in northeastern China, (2) the model results were evaluated at the landscape scale, (3) predicted forest successional results were consistent with the expected successional patterns in boreal forests of the study region, (4) compared with field inventory data, the simulated fire, and harvest performed well, indicating a satisfactory reflection of the fire and harvest effects on post regeneration and recovery, and (5) We have used the currently available data for model evaluation. If possible, we will conduct longer term model verification when remeasured forest data become available. These study results provided useful information for FLMs evaluation.

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