

Should we respect the historical reference as basis for the objective of forest restoration? A case study from Northeastern China

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Abstract Under climate change, the adoption of historical reference as the objective of forest restoration is being questioned. In this study, the spatially explicit forest landscape model LANDIS was utilized to analyze how the forest landscape in the upper Hun River area of Liaoning province in northeast China would be affected under current climate trends and future climate change; and to explore whether the historical reference should be the objective of restoration efforts. The results showed that (1) the area percentage (AP) of *Quercus mongolica* under climate change is always higher than that under the current climate regime, while the AP of *Pinus koraiensis* is lower than that under current climate; and (2) the competitive ability of *Q. mongolica* and *Populus davidiana* increases, while that of other species decreases under climate change. As interspecies competition shifts under climate change, the historical reference appears in appropriate to serve as the objective of forest restoration. In addition, although *Q. mongolica* would likely benefit from a warmer and drier climate, use of this species for forest restoration under climate change still requires further research.

Keywords Forest restoration · Climate change · LANDIS · Forest succession · Species competition

Introduction

During the last decades, the rapidly growing population and socio-economic development have contributed to the over-exploitation and subsequent degradation of forests in many

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parts of the world. As a consequence, forest restoration has become one of the important foci of restoration ecology (Griscom and Ashton 2011; Holl 2011). While forests restoration is itself a complex and challenging issue (Dobson et al. 1997; Shinneman et al. 2010; Griscom and Ashton 2011; Rodrigues et al. 2011), it becomes even more complicated under climate change (Parmesan and Yohe 2003; Hulme 2005; Harris et al. 2006).

The science of restoration ecology and the practice of ecological restoration have developed rapidly over the past several decades (Harris et al. 2006). At the same time, a cohesive body of restoration theory is emerging, which is linked to increasingly sophisticated restoration practices (Falk et al. 2006; Van Andel and Aronson 2006). Historical conditions have always served as the basis of explicit objectives for restoration in many of these theories and practices, and degraded forests are always promoted toward pre-disturbance conditions (Yu et al. 2011; Shinneman et al. 2012). However, considering the trajectory of contemporary climate change, it remains unclear if and where the shifts in climate will affect forest restoration efforts (Parmesan and Yohe 2003), given the pronounced changes that are being observed beyond historical ranges of variability. Harris et al. (2006) pointed out that as the biological envelope has been altered geographically, habitats have no longer been able to support a vast amount of historical species. Studies on plant species distribution have found that some species moved out of their previous habitats and migrated toward more northern areas due to global warming in the Northern Hemisphere (He et al. 1999; Leng et al. 2008). In effect, under rapid climate change new quasi-stable states may be achieved although the original states disappeared (Harris et al. 2006). Accordingly, traditional conservation management that excludes all non-native species could conceivably lead to a catastrophic failure (Yao et al. 2012), because the historical systems will not be able to respond to the changed environment. In addition, the random use of non-native species would lead to a failure of restoration efforts if such species are not well adapted to the entire combination of environmental factors. In contrast, if the historical species that are conserved can adapt to the rapidly changing selective pressures (Rice and Emery 2003), their conservation should be sustainable. Therefore, it is necessary to demonstrate the better over-all adaptive potential of non-native species versus native species before replacing the latter.

The choice of restoration objectives is essential but is not an easy decision. It is expected that restoration will eventually result in forests with their community composition broadly reflecting the potential natural composition for the region and environment type (Mason et al. 2012). Unfortunately, under climate change scenarios, the potential natural composition for a particular region is unclear due to possible species gains and losses. As a result, understanding the responses of individual species and forest composition to climate change is the key to determining restoration objectives. Such responses are complicated at the individual species level and can be further confounded when driven by intra- and inter-species competition (Shao 2003; Heikkila et al. 2009). Modelssuch as LANDIS (He and Mladenoff 1999b) and LANDCLIM (Schumacher et al. 2004) incorporate parameters of life history, intra- and inter-specific competition, climate variability, and other environmental variables, and thus can predict the responses of species under global climate change and help improve our understanding of how species respond to environmental change (Rice and Emery 2003; Schumacher et al. 2006).

Particularly over the past decade, forest degradation in China has been receiving more and more attention (Zhang et al. 2010; Liu et al. 2011; Meng et al. 2011; Yu et al. 2011; Huang et al. 2012). In Northeastern China, one of the three major forest regions in the country, most of the original forests have been degraded due to anthropogenic disturbance. The problem is particularly severe in Eastern Liaoning province (Fu 2009), where forests

have been degraded from the mixed broad-leaved *P. koraiensis* forest (He et al. 2002) to secondary forests dominated by *Q. mongolica*, *Betula* spp., *Populus* spp. and other early or mid-successional species. To resolve the problem of forest degradation, a number of research efforts has emerged to explore methods of forest restoration, and most such efforts have taken the historical forest as the reference for restoration. Objectives have usually been to restore the degraded forest to the climax forest—i.e., the mixed broad-leaved *P. koraiensis* forest (Chen et al. 2003; Bu et al. 2008). It is reasonable to question whether such direction of forest restoration is conducive to the sustainability of the forest ecosystem in the face of global change (Yao et al. 2012).

In this study, LANDIS 6.0 was utilized to examine shifts in community composition of forests in Liaoning province under different climate scenarios. Three specific questions are addressed: (1) How would community composition of forests in the upper Hun River region change under different climate scenarios?; (2) How does the competitiveness of the climax species, *P. koraiensis*, and other species respond to different climate scenarios?; and (3) Should historical forest composition be used as the objective of restoration?

Methods

Study area

The Hun River is one of the main tributaries of the Liao River, the latter being the most important river in the southern part of Northeastern China. The Hun flows through the Dahuofang Reservoir and the central city of the region, Shenyang (Fig. 1). The upper Hun River area (41°47'52"–42°28'25"N, 124°20'06"–125°28'58"E) in Qingyuan county of eastern Liaoning Province encompasses 2.5×10^5 ha (Fig. 1). The forests dominated by *P. koraiensis* in Upper Hun River area belong to the Changbai Mountain flora and are important for the ecological integrity and water security of the downstream area. Unfortunately, these forests have become degraded to secondary mixed forests of lower productivity, which include *Pinus koraiensis*, *Quercus* spp., *Larix* spp., *Pinus tabulaeformis*, *Pinus densiflora*, *Fraxinus rhynchophylla*, *Fraxinus chinensis*, *Juglans mandshurica*, *Betula* spp. and *Populus* spp.

Description of LANDIS 6.0

LANDIS is a spatially explicit, cell-based and stochastic landscape model. Since actual species-level ecological processes are too complex to be simulated with current computational capabilities, ecological processes across different spatial and temporal scales are integrated through choosing and simplifying fine-scale mechanistic components (He and Mladenoff 1999b). LANDIS can simulate species-level forest succession dynamics and different kinds of disturbance, such as wind, fire, insect disease, harvesting and fuelwood cutting, and the interaction between them in large areas over long time spans.

In LANDIS, the heterogeneous landscape is composed of various homogeneous forest land types for which environmental conditions such as soils, slope, aspect, climate and so on are assumed to be same. The species establishment coefficients, which quantify how different land types favor or inhibit the establishment of a particular species, are same for a single land type. Each cell of a land type map contains a matrix that consists of a species list and 10-year age cohorts. At the cell scale, LANDIS simulates species birth, growth, mortality, and vegetative reproduction based on species vital attributes and species

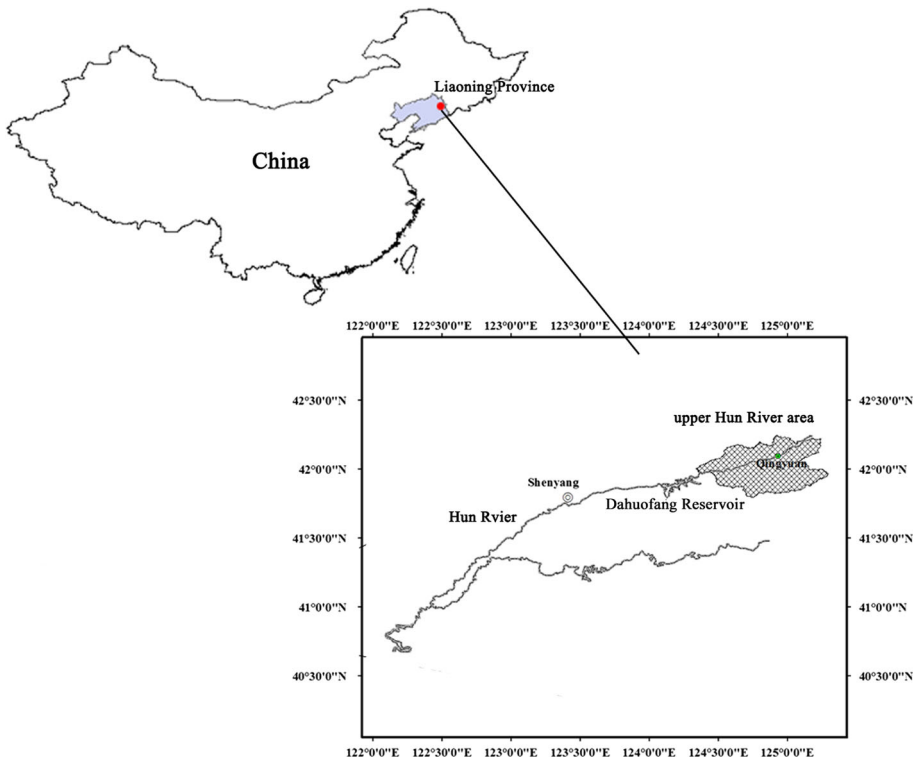


Fig. 1 Location of the study area

establishment coefficients for each cell on the grid. At the landscape scale, spatial processes such as seed dispersal are simulated (He and Mladenoff 1999b).

The capability of LANDIS 6.0 has been revolutionized to conduct simulation using the hash-table-based approach. The variable time steps in LANDIS 6.0 range from 1 to 10 years as defined by the user. The model can also operate on very large maps (e.g. 20,000 × 20,000 pixels).

Parameterization of LANDIS 6.0

Species attributes and species composition maps

Vital attributes of sixteen common tree species in the upper Hun River area (Table 1) were estimated based primarily on literature on species characteristics in this region (Li et al. 2005; Zhang et al. 2007; Zhu et al. 2008). Information about species characteristics was also obtained through consultations with local experts. In addition, the parameterization of species characteristics from other research in northeastern China was also referred to in completing the species attribute table (He et al. 2005; Chang et al. 2007; Bu et al. 2008).

Forest inventory data always included a stand map and its corresponding stand attribute database. Inventory data used in this study was gathered in 2006 and provided by the Qingyuan County Forestry Bureau. Distribution of species was obtained from stand maps

Table 1 Species' vital attributes derived for forests in the upper Hun River area of Northeastern China

Species	LONG	MTR	ED	MD	ST	FT	VP	MVP
<i>Quercusmongolica</i>	350	40	20	200	3	5	0.9	60
<i>Populusdavidiana</i>	100	8	−1	−1	1	2	1	10
<i>Betulaplatyphylla</i>	150	15	20	400	1	1	0.8	50
<i>Ulmuspumila</i>	250	10	30	100	2	4	0.3	60
<i>Acer pictum subsp. mono</i>	250	10	12	350	4	2	0.3	50
<i>Juglansmandshurica</i>	250	15	50	150	3	4	0.9	60
<i>Fraxinuschinensis</i>	250	30	50	150	3	3	0.3	80
<i>Fraxinus rhynchophylla</i>	250	30	50	150	3	3	0.3	80
<i>Tiliaamuresis</i>	300	30	50	100	4	4	0.9	30
<i>Pinuskoraiensis</i>	400	40	50	200	5	1	0	0
<i>Larixolgensis</i>	300	30	10	400	1	5	0	0
<i>Pinustabulaeformis</i>	200	30	10	500	2	1	0	0
<i>Pinusdensiflora</i>	200	30	10	500	2	1	0	0
<i>Piceaasperata</i>	300	30	80	150	5	3	0	0
<i>Abiesnephrolepis</i>	250	40	80	150	5	3	0	0
<i>Pinussylvestris var. mongolica</i>	250	40	30	100	2	2	0	0

Long-longevity (years); *MTR* age of maturity (years); *ED* effective seeding distance (m); *MD* maximum seeding distance (m); *ST* shade tolerance class (1–5: 1 = least shade tolerant; 5 = most shade tolerant); *FT* fire tolerance class (1–5: 1 = least tolerant; 5 = most tolerant); *VP* vegetative reproduction probability; *MVP* minimum age of vegetative reproduction (years)

of forest inventory data, and information on dominant species and age groups for each species was acquired from stand attribute data bases corresponding to stand maps. Finally, the forest composition map was produced in ARCGIS and resampled at a resolution of 60 m × 60 m, yielding 1,320 rows × 836 columns.

Land type map

A land type in LANDIS is a set of relatively homogeneous units in the heterogeneous landscape. The chances for species establishment (species establishment coefficients) within each land typewere assumed to be similar (He and Mladenoff 1999b). The land type map for the study area was derived based on terrain attributes given in forest inventory data taken in 2006 and the Digital Elevation Model (DEM) of Qingyuan County taken in 1992 (<http://www1.csd.cn/>). Non-active areas such as water bodies and cities were excluded in advance. The land type map included seven land types: North Ridge (NR), South Ridge (SR), North Slope (NL), South Slope (SL), North Slope of valley (NV), South Slope of valley (SV) and terrace (T) (Fig. 2).

Species establishment coefficients and simulation scenarios

The species establishment coefficients in LANDIS estimate the probability of a species becoming successfully established in a land type under different environmental conditions—i.e., land type and climate. The status of the forests in the study area in 2006 was represented as the initial forest composition including species/age classes. Two simulated climate scenarios were employed in this study –current climate and climate change. The

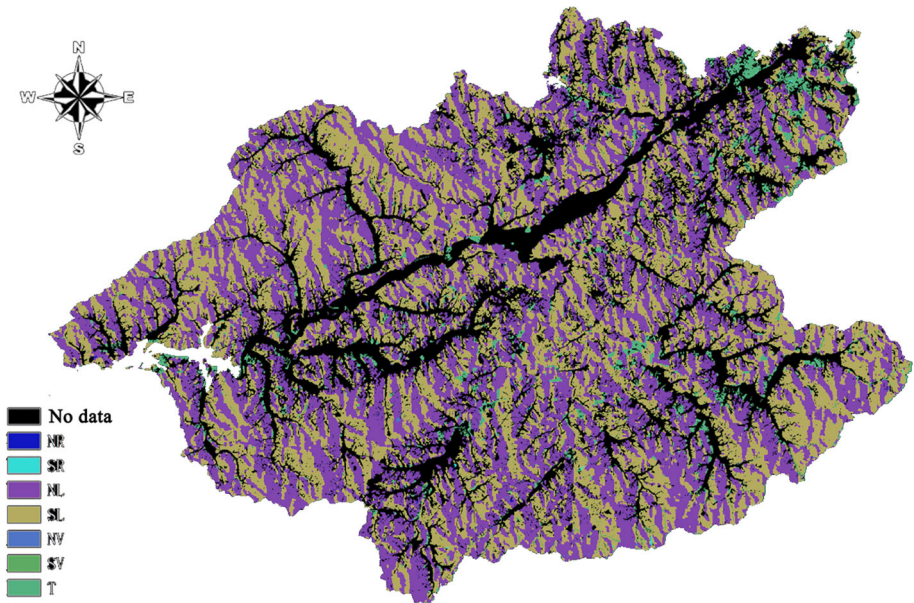


Fig. 2 Landtype map of the Upper Hun River area (*NR* north ridge, *SR* south ridge, *NL* north slope; *SL* south slope, *NV* north slope of valley, *SV* south slope of valley, *T* terrace)

species establishment coefficients that represent the responses of individual species to climate in the two scenarios were derived from LINKAGES by He et al. (2005). LINKAGES is a derivative of the JABOWA/FORET class of gap models, which can simulate the physiological response of individual species to current and warming climate conditions (Bugmann et al. 1996). The climate variables acquired in it are 12-month mean temperature and precipitation. The study by He et al. (2005) was conducted in the Changbaishan Nature Reserve, which is also characterized by Changbai Mountain flora. The current climate data were derived from four weather stations distributed across Changbai Mountain, and the future climate predictions were generated via use of the second version of the Canadian Global Coupled Model (CGCM2, IPCC B2) for the Changbai Mountain area (4.6 °C average annual temperature increase and little precipitation change) (He et al. 2005). The predicted temperature and precipitation changes between 1990 and 2090 is linear and indicates that warming will occur gradually over the next 100 years. The resulting warmed conditions will persist for the simulation years after 2090.

Analysis methods

The cell numbers of each species for each 0-year step were obtained from the output map in the LANDIS output statistical program for the two scenarios. The 300-year period was initially divided into three shorter periods—short-term (year 1–year 50); middle-term (year 51–year 100); and long-term (year 101–year 300). For each scenario, the area percentage (AP) of each species in the last year of each period was calculated. Then, for both

scenarios, the AP of each species for every year over the 300 simulated years was calculated and AP trends for all species were depicted in SigmaPlot.

Results

Species dynamics in different simulation periods

Under both scenarios, *Q. mongolica* was the most dominant species, with the AP being close to 50 % in the short- and middle-term periods (Fig. 3, “Appendix”). Although *Q. mongolica* was still the most dominant species in the long-term period, its AP was much lower than in the two other periods (Fig. 3, “Appendix”). At year 300, *P. koraiensis* would be one of the two most dominant species, displaying a much higher AP than in the other two periods (Fig. 3, “Appendix”). The forest in the upper Hun River area appears to be slowly moving toward a climax forest. However, regardless of length of simulation period—i.e., short-, middle-, or long-term, the AP of *Q. mongolica* under climate change is higher than under the current climate scenario; while that of *P. koraiensis* lower than under the current scenario. Thus the increase of AP for *Q. mongolica* would be promoted by climate change, whereas AP for *P. koraiensis* would be depressed.

The variation of area percentage trend of species

With the exception of *Q. mongolica*, the AP of every species in the upper Hun River area showed similar trends under the two different scenarios (Fig. 4). The APs of late succession species (*P. koraiensis*, *P. asperata*, *A. nephrolepi*, *U. pumila*, *A. pictum subsp. mono* and *T. amuresis*) increased; that of mid-succession species (*P. tabulaeformis*, *P. densiflora*, *F. chinensis*, *F. rhynchophylla* and *J. mandshurica*) first increased and then decreased; while AP for early succession species (*Betula platyphylla*, *Pinus sylvestris var. mongolica*) decreased under both climate scenarios. *P. davidiana* is an early succession species, but its AP remained stable until dramatically increasing around year 250. The AP of *Q. mongolica* under the two scenarios displayed opposite trends, decreasing under the current climate scenario while increasing under climate change (Fig. 4).

Although most species displayed similar trends under both scenarios, it was still evident that the trends of some species were enhanced or weakened by climate change. The increase in AP for *P. davidiana* and *U. pumila* was enhanced, as was the decrease in AP for *Betula platyphylla*, *Pinus sylvestris var. mongolica*, *P. tabulaeformis*, *P. densiflora*, *F. chinensis*, *F. rhynchophylla* and *J. mandshurica*. In contrast, AP increases were weakened for *P. koraiensis*, *P. asperata*, *A. nephrolepi*, *A. pictum subsp. mono*, *T. amuresis* and mid-succession species.

Discussion

One of the important issues of forest restoration is the choice of the objective of restoration (Brown and Johnson 1998; Norton 2009; Mason et al. 2012; Critchley et al. 2013); or more specifically, the choice of species for restoration, since planting is thought to be the most effective method to promote forest succession (Lamb 1998; Brancalion et al. 2012; Johnston and Zedler 2012; Munro et al. 2012; Bannister et al. 2013). Species are always

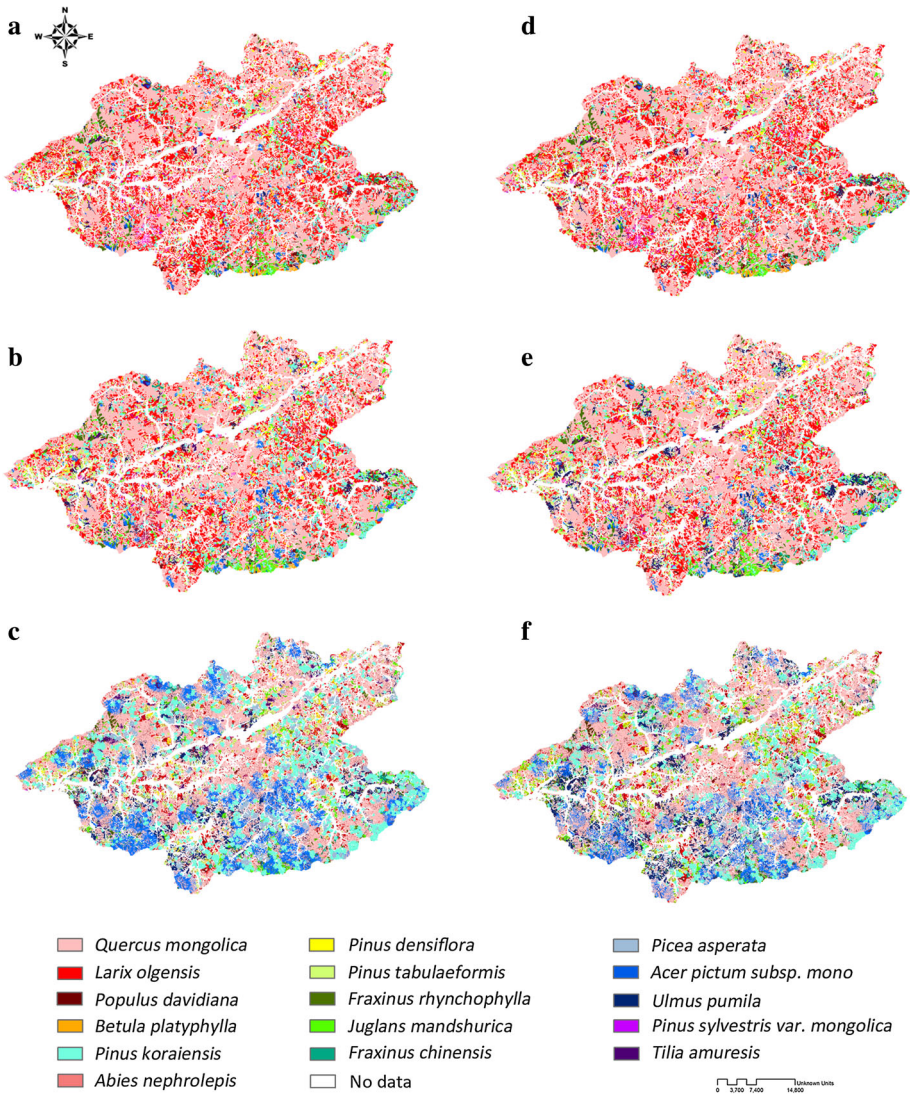


Fig. 3 Distribution of tree species in forests of the upper Hun River area (**a, b, c** Distribution of tree species under the current climate scenario at year 50, 100, 300 respectively; and **d, e, f** Distribution of tree species under the climate change scenario at year 50, 100, 300 respectively)

selected in order to promote the degraded forest toward the historical climax forest (Lamb 1998; Wang et al. 2006; Meng et al. 2011). However, because of a changed environment with respect to temperature, precipitation, soil, fire return cycle and so on, forests do not necessarily follow their historical trajectory (Godefroid et al. 2007; Liu et al. 2011; Singh et al. 2012). It has been noted that forests can achieve a new stable state other than their historical composition under a new environment (Stromayer and Warren 1997; Jasinski and Payette 2005; DeLonge et al. 2008). This casts doubt on the necessity of promoting degraded forests to historical climax forests under a changed environment. Moreover, some

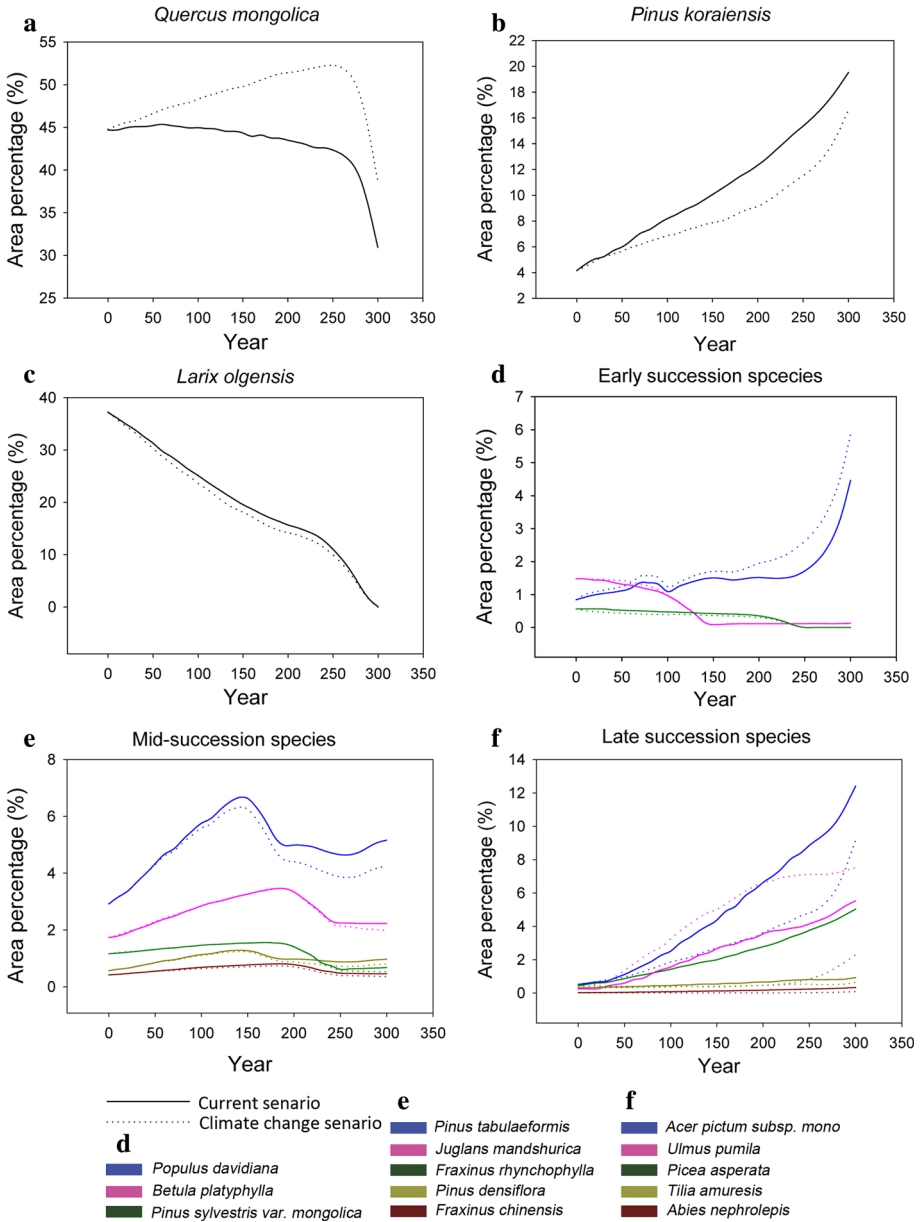


Fig. 4 Trajectory of area percentage (AP) of different species under two climate scenarios

dominant species of climax forests are deemed to be less adaptive than other species under an environment altered by climate change or other factors (Chrysopolitou et al. 2013; Katona et al. 2013; Ponce-Reyes et al. 2013).

This study found that although the upper Hun River forests in both scenarios slowly moved toward the historical climax forest dominated by *P. koraiensis*, the variation in AP

trends of *Q. mongolica* and *P. koraiensis* (Fig. 4) indicate that climate warming promotes the increase of AP for *Q. mongolica*, while depressing that of *P. koraiensis*. This suggests that the relative increase in competitive ability of *Q. mongolica* is likely more pronounced in response to climate warming. Oaks are commonly considered to be drought- and heat-tolerant trees not only in the Changbai Mountain region (He et al. 2005), but also in other forest ecosystems (Wyckoff and Bowers 2010; Michelot et al. 2012). Rigling et al. (2013) found that in the Swiss Rhone valley the contemporary trend from Scots pine to oak forests is obvious at lower altitudes and drier sites despite the dominance of Scots pine. In contrast, the dominant species, *P. koraiensis* is reported to be more suitable to remain established in the high elevation areas (Aizawa et al. 2012; Wang et al. 2013), suggesting that *P. koraiensis* may favor a cool and moist climate (Zhang et al. 2012; Zhou et al. 2012). Moreover, in studying the physiological responses of *P. koraiensis*, Yan et al. (2013) found that the nitrogen and carbon levels of *P. koraiensis* decreases under increased air temperature and decreased relative humidity, which implies that the competitive ability, species productivity and functioning of *P. koraiensis* is affected by global warming, with or without concurrent increasing precipitation. With this in mind areas that supported *P. koraiensis* in the past under conditions similar to those projected for the future might be considered as restoration sites for “neo-native” stands of *Q. mongolica* (Millar et al. 2007). The forest dominated by *Q. mongolica* is probably the “neo-native” forest in the future of the study area, so planting *P. koraiensis* as the primary restoration measure should be reconsidered cautiously and the spread of *Q. mongolica* should not be suppressed (Millar 1998).

Some researchers admit that forest succession would probably deviate from its historical trajectory; but they still insist that there should be a reference for restoration and that the historical reference could serve as a basis for such an objective of restoration (Hohensinner et al. 2005; Maloney et al. 2011; Hanberry et al. 2012), because historical reference is deemed to reflect a perfectly self-sustaining state without anthropogenic disturbance. In their opinion, although a degraded forest never returns exactly back to its historical condition, it can be improved to close to historical forest composition and maintained in a self-sustaining state as historical forest (Woelfle-Erskine et al. 2012; Trueman et al. 2013). However, we assume that choosing a proper species and improving forest succession based on species' adaptation to climate change is a better choice, since planting is a labor-intensive and expensive activity, and planting improper species would both increase mortality rates and increase costs by reducing planting efficiency (Gebrekirstos et al. 2011; Liu and Guo 2012; Craven et al. 2013).

The results of this study suggest that *Q. mongolica* is a species which could be chosen for forest restoration under climate change in the Upper Hun River area of Northeast China, a conclusion that is also supported by other recent research (He et al. 2005; Wyckoff and Bowers 2010; Michelot et al. 2012; Rigling et al. 2013). At the same time, there remain alternative opinions that detrimental effects on oaks can be expected when elevated temperature occurs simultaneously with drought (Arend et al. 2013). Adapting silvicultural rules and practices to maintain optimum species-site relationships is one important restoration strategy (IPCC 2000). Therefore, species suitability tests for the area are necessary (Bolte et al. 2009). Choice of a proper species for forest restoration is clearly be one of the basic elements of effective operational management (Ogden and Innes 2007).

In addition to the variation in species adaptation, the variation of relative competitiveness of species is also contributing to changes in forest composition under global warming (Saxe and Kerstiens 2005; Bolte et al. 2010). In the present study, the adaptation of species is reflected by establishment coefficients (not shown in this paper) which are

summarized from the results of LINKAGES (He et al. 1999, 2005). The establishment coefficients of *Q. mongolica*, *P. davidiana*, *B. platyphylla*, *F. chinensis*, *F. rhynchophylla*, *J. mandshurica*, *A. pictum subsp. mono*, and *U. pumila* are higher under climate change; however, not all of the AP trends of these species increase (Fig. 4). We infer that the spread of species is not only determined by the adaptation of species themselves, but also the competition between them (Reyer et al. 2010; Yu and Gao 2011; Xu et al. 2012). As an example, consider the AP of early successional species such as *P. davidiana* (Fig. 4). There are likely two reasons for the increasing AP of *P. davidiana*. The first involves characteristics of R strategy species (Gillson and Willis 2004; Milad et al. 2011), including: (1) strong seed dispersal ability (He and Mladenoff 1999a; Imbert and Lefevre 2003); (2) a large seed crop every year (Gonzalez et al. 2010; Lee et al. 2011); and (3) strong adaptation of seeds to various environments (Zhang et al. 2004, 2005; Gonzalez et al. 2010; Li et al. 2010). As a result, *P. davidiana* would probably occupy the gaps created by mortality of other species. A second reason is that the relative competitive ability of *P. koraiensis*, *P. tabulaeformis*, *P. asperata*, *A. nephrolepioides* and *A. pictum subsp. monoc* clearly decrease under the climate change scenario (Fig. 4). The tree line of *P. koraiensis*, *P. tabulaeformis*, *P. asperata*, *A. nephrolep* migrates toward higher altitudes and latitudes under warmer climate conditions (Shao 2003; He et al. 2005). The phenomenon that the AP of *A. pictum subsp. Mono* displays a weaker increasing trend under the climate change scenario than the current climate scenario has also been reported by He et al. (2005). Climate change would favor some species not only because of their species-specific characteristics, but also due to the delayed immigration of other species (Dukes and Mooney 1999; Kirschbaum 2000).

Some model assumptions of LANDIS may affect the simulation results. For example, the establishment coefficients which reflect the adaptation of species are constant. However, the adaptation of species likely changes with time (Rice and Emery 2003). The fossil record indicates that evolutionary change occurs slowly, and even Darwin (1859) concluded that natural selective pressures are too weak for evolutionary influences in the wild, so some would think that it is not necessary to consider evolutionary influences in forest development simulation for time spans that are not excessively long. However, in the past few decades, some evolutionary biologists have claimed that selection in natural populations is strong enough to cause evolutionary shifts within a few generations (Rodrigues et al. 2009; Meli et al. 2013). Therefore, in LANDIS the adaptation of some species could possibly be underestimated.

Conclusion

The adoption of historical community composition as the objective of forest restoration should be considered carefully, even though important lessons may be learned from the past. Because forest succession and its response to climate change are occurring at a slow pace, and sometimes responses of plant species to climate change may diverge, species sustainability assessment should be an important focus of research for forest restoration management, as should the interaction among species change. The latter could also strongly affect changes in species' dominance and community composition, and even possibly alter the pathway of forest succession.

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Appendix

See Table 2.

Table 2 The area percentage of each species at the final year of each period under different climate scenarios

Species	Area percentages (%)					
	Current climate scenario			Climate change scenario		
	Y50	Y100	Y300	Y50	Y100	Y300
<i>Pinus koraiensis</i>	5.99	8.19	19.53	5.67	6.87	16.63
<i>Pinus tabulaeformis</i>	4.33	5.76	5.16	4.30	5.59	4.26
<i>Pinus densiflora</i>	0.85	1.14	0.97	0.84	1.11	0.80
<i>Pinus sylvestris</i> var. <i>mongolica</i>	0.52	0.47	0.00	0.43	0.39	0.00
<i>Larix olgensis</i>	31.30	25.12	0.00	30.29	23.65	0.00
<i>Picea asperata</i>	0.87	1.43	5.04	0.41	0.40	2.30
<i>Abies nephrolepis</i>	0.05	0.09	0.34	0.02	0.02	0.09
<i>Populus davidiana</i>	1.12	1.09	4.47	1.24	1.24	5.83
<i>Betula platyphylla</i>	1.32	0.97	0.13	1.43	0.98	0.12
<i>Ulmus pumila</i>	0.60	1.57	5.54	1.36	3.26	7.53
<i>Fraxinus chinensis</i>	0.55	0.69	0.46	0.54	0.65	0.36
<i>Fraxinus rhynchophylla</i>	1.32	1.46	0.68	1.32	1.46	0.52
<i>Juglans mandshurica</i>	2.27	2.85	2.23	2.30	2.85	1.99
<i>Quercus mongolica</i>	45.22	44.97	30.94	46.63	48.32	38.74
<i>Acer pictum</i> subsp. <i>mono</i>	1.12	2.52	12.42	0.96	1.87	9.17
<i>Tilia amuresis</i>	0.38	0.46	0.94	0.33	0.37	0.65

Y Year

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