

# Assessing the potential of native tree species for carbon sequestration forestry in Northeast China

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## Abstract

Although the native forests of China are exceptionally diverse, only a small number of tree species have been widely utilized in forest plantations and reforestation efforts. We used dendrochronological sampling methods to assess the potential growth and carbon sequestration of native tree species in Jilin Province, Northeast China. Trees were sampled in and near the Changbaishan Biosphere Reserve, with samples encompassing old-growth, disturbed forest, and plantations. To approximate conditions for planted trees, sampling focused on trees with exposed crowns (dominant and co-dominant individuals). A log-linear relationship was found between diameter increment and tree diameter, with a linear decrease in increment with increasing local basal area; no significant differences in these patterns between plantations and natural stands were detected for two commonly planted species (*Pinus koraiensis* and *Larix olgensis*). A growth model that incorporates observed feedbacks with individual tree size and local basal area (in conjunction with allometric models for tree biomass), was used to project stand-level biomass increment. Predicted growth trajectories were then linked to the carbon process model InTEC to provide estimates of carbon sequestration potential. Results indicate substantial differences among species, and suggest that certain native hardwoods (in particular *Fraxinus mandshurica* and *Phellodendron amurense*), have high potential for use in carbon forestry applications. Increased use of native hardwoods in carbon forestry in China is likely to have additional benefits in terms of economic diversification and enhanced provision of “ecosystem services”, including biodiversity protection.

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**Keywords:** Carbon sequestration; China; *Fraxinus mandshurica*; Growth modelling; *Phellodendron amurense*; Species selection

## 1. Introduction

China has experienced dramatic changes in land use and forest cover, and thus forest carbon sequestration, over the last century. During the early-mid-1900s, China underwent a period of widespread deforestation, without strong reforestation practices; only 5.8% of the country remained forested in 1958 (FAO, 2004). China's relative lack of reforestation during and after this period left the country exposed to severe environmental problems, including widespread soil erosion, flooding and dust storms (Wang, 2004). The need to implement large-scale reforestation was politically acknowledged in the early 1980s, and China

embarked on an ambitious strategy that has resulted in an average of  $\sim 10^6$  ha of trees planted per year from the 1980s to the present (FAO, 2004). An estimated 16.5% of the country was under forest cover by 1998. Management goals for the vast areas of planted forests in China are primarily focused on wood supply and mitigation of erosion and flooding risks. Although great numbers of trees have been planted, this has not translated into a high diversity of species. China's industrial plantations are primarily made up of a few species of larch, pine, Chinese fir, and hybrid poplar. In 1998 a ban on logging in natural forests was introduced (Zhao and Shao, 2002). Although the logging ban was a critically important environmental step, one consequence has been that the vast majority of China's  $\sim 2500$  native tree species (FAO 2004) are little utilized at present.

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As planted forests have reached harvestable age, the bulk of wood supply in China has shifted from natural forests to intensively managed plantations. However, current projections suggest that existing plantations will not meet even short-term demand for wood fiber (Barr and Cossalter, 2004). The 1998 ban on logging in natural forests, coupled with increasing urban development, has also led to a smaller land area being available for intensive forest management. In Northeast China (Manchuria), plantation forests consist almost entirely of *Larix olgensis*, *Larix dahurica*, *Pinus sylvestris* var. *mongolica*, and *Pinus koraiensis*. Some serious problems with disease and pests are prevalent. *P. koraiensis* (Korean pine) has a tendency to fork when grown in direct light (likely due to shoot-attacking beetles), and is prone to white pine blister rust, *Cronartium ribicola* (J.C. Fisher), and pine yellow-spotted weevil, *Pissodes nitidus* (Roelofs) (Xu et al., 1987; Richardson, 1990). *L. olgensis* can be planted over a wider range of sites than Korean pine; however it is susceptible to heart rot after 50 years when grown in plantations, and its wood has a limited range of use (Richardson, 1990). Of native hardwoods in Northeast China, small-scale plantings of *Phellodendron amurense* (Rupr.) and *Fraxinus mandshurica* (Rupr.) are occasionally encountered. Although not as species-rich as the sub-tropical and tropical forests of southern China, the provinces of Northeast China have some extensive areas of natural forest containing a wide range of species which represent an untapped source of native trees potentially suitable for use in plantation projects.

Large-scale reforestation and afforestation have significant implications for carbon sequestration, classified as land use, land use change and forestry (LULUCF) activities under the Kyoto Protocol. With a large area planned for plantation establishment, the Chinese government is considering applying the carbon sequestered by these forests towards Kyoto emission credits. The Kyoto Protocol entered into force on February 16, 2005, 90 days after it was ratified by Russia. In the first commitment period (2008–12), afforestation and reforestation activities can be classified as certified emission reductions that can be applied under the Clean Development Mechanism (CDM, defined under Article 12), intended as a cost-effective means to help nations achieve emissions targets (UNFCCC Conference of the Parties 1997). With the entry of the Kyoto Protocol into force, plantation projects planned for the first commitment period, and prior to 2008, can be certified as CDMs. Under the Forestry Action Plan for China's Agenda 21, 6.39 million hectares of timber yield plantations will be established between 2001 and 2010, of which northeast China will host a significant proportion. In this same period 12.73 million hectares will be planted under the Three-North, Upper and Middle Reaches of the Yangtze River, Coastal, and Taihang Mountains Shelterbelt and Afforestation projects (FAO Asia-Pacific Forestry Sector Outlook Study: China's Country Report on Forestry, 2004). In the absence of initiatives promoting

use of under-utilized native species in China, it seems certain that any forests planted for carbon sequestration purposes will consist of the small set of conventional plantation species.

The mixed hardwood forests of Northeast China have long been regarded as a natural treasure of biological resources, and the widely planted *L. olgensis* and *P. koraiensis* may not be the optimal species for carbon sequestration purposes (Zhao, 1999). In particular, certain hardwood species may be faster growing, have a higher wood carbon content per volume, may be better suited to management under long rotations, may result in increased carbon sequestration in belowground pools, or may result in wood products with a longer half-life prior to the release of carbon to the atmosphere. Certain native tree species may also have greater economic value than the coniferous species planted. For example, *F. mandshurica*, *P. amurense* and *Juglans mandshurica* (Rupr.), known historically as the "three precious trees" of Manchuria (Richardson, 1990), have a long history of use in high value finished wood products. Shade-tolerant hardwood species may also be suitable for under-planting in existing *L. olgensis* or *P. koraiensis* plantations as a means to increase plantation diversity, extend rotation length, and increase the accumulation of leaf litter, thus reducing soil erosion and accelerating mineral nutrient cycling (Chen et al., 2003).

The present project was undertaken to provide an empirical assessment of the growth and carbon sequestration potential of a variety of native tree species occurring in Jilin Province in Northeast China. The native forests of this region have received considerable prior research attention, including forest growth and modeling studies (e.g., Shao et al., 1994, 2001; He et al., 2002, Yan and Shugart, 2005). However, evaluations of carbon sequestration potential for most native tree species have not been attempted. Here we link a species-specific empirical model of growth and yield to the ecosystem model InTEC to provide estimates of carbon sequestration potential for 14 native tree species common in the region. We address the following specific questions: (1) Which native tree species show the highest growth rates in natural ecosystems in Northeast China, and under simulated reforestation scenarios? and (2) What rates for net on-site carbon sequestration can be predicted for native hardwood species?

## 2. Methods

### 2.1. Study sites

The majority of sampling was conducted in the *P. koraiensis*-mixed hardwood forest of the Changbaishan Reserve (41°41'–42°51'N; 127°43'–128°16'E), in southern Jilin Province (Barnes et al., 1992). The reserve was established in 1960, and was designated as a UNESCO biosphere reserve in 1980 (Yang and Xu, 2003). It has an area of 216,716 ha, making it the largest forest reserve in China. The study area is located on the northwest side of

Changbai Mountain, an active volcano that last erupted in 1702 (Barnes et al., 1992). The region is characterized by a temperate continental mountainous climate, with an annual rainfall of 600–1000 mm and a growing season of 120–130 days (Richardson, 1990; Yang and Xu, 2003). Most precipitation occurs during this time. The majority of sampling took place in natural undisturbed and disturbed forests near the Changbai Mountain Research Station of Forest Ecosystems, of the Chinese Academy of Sciences, which is located at an elevation of 735 m. Twenty sites at Changbaishan were established—17 in natural forests and three in *L. olgensis* plantations. Additional data were collected from five plantations near the Jilin Province city of Dunhua. Plantations sampled ranged from 18 to 33 yr old. The landscape near Dunhua is primarily agricultural, with tobacco, corn, sunflower and soybean plantations dominating the area.

## 2.2. Field sampling

At each sampling site, a transect was established at least 50 m from the forest edge from the nearest path or access point. Eight plots were placed along each transect, 20 m apart from each other along a straight line. Locations of each plot were recorded using GPS. At each plot, relascope sampling was performed with a basal area factor 2 prism, allowing for calculation of local stand basal area (Bitterlich, 1984). For each tree that fell within the prism sweep, data were recorded on species, diameter at breast height (DBH), canopy class and damage class of bole and crown. Canopy class was recorded as Intermediate, Suppressed, Codominant, Dominant or Emergent, based on the Kraft classification system (Smith et al., 1997). Damage classes were determined with a combination of qualitative and quantitative data (Van Dyke, 1999). Damage class for both the bole and crown was recorded as none/minimal, moderate or severe. For the bole, minimal was defined as no visible damage, or scars/seams on bark that had completely seamed with no evidence of rot or fungal infection. Moderate damage was classified as scars less than 50 cm<sup>2</sup> in area that did not show evidence of significant rot or fungal infection. Severe bole damage was classified as scars greater than 50 cm<sup>2</sup> in area, or scars with rotting wood or fungal infection. Crown damage class was determined primarily in terms of large branch loss and overall crown dieback. Trees with no visible dieback and intact branches were classified as having no/minimal crown damage. Crowns that showed evidence of dieback affecting less than 1/3 of the crown were classified as having moderate damage, and crowns showing more than 1/3 crown dieback and/or the loss of major branches were classified as severely damaged. These classifications are somewhat arbitrary, as is often the case with tree damage class distributions (Van Dyke, 1999). To maintain consistency, damage class was always determined by the same individual.

In order to approximate growth rates in plantation conditions, a sub-sample of trees within prism sweeps were examined. Trees with DBH that fell between 10 and 50 cm, with no/minimal damage to both bole and crown, and with canopy classifications of codominant or dominant were sampled for increment cores. Cores were taken at breast height (1.3 m) and, when possible, from bark to pith of the tree. Cores were glued to plywood boards and transported to a laboratory for analysis. Height of sampled trees was also recorded using a clinometer. A target of at least 20 and preferably 40 cores was set for each species. Species sampled for growth and wood properties include *Acer mono* Maxim., *Betula platyphylla* Suk., *F. mandshurica* Rupr., *J. mandshurica* Rupr., *Maackia amurensis* Rupr. et Max., *P. amurense* Rupr., *P. koraiensis* Sieb. et Zucc., *Populus davidiana* Dode, *Populus ussuriensis* Kom., *Quercus mongolica* Fisch. ex Turcz., *Tilia amurensis* Rupr., *Tilia mandshurica* Rupr. et Maxim, and *Ulmus davidiana* var *japonica* Redhder. The rich understory tree flora of the region, including some 6 species of *Acer*, was not included in analyses as these species are unlikely to be of silvicultural importance.

Increment core samples were analyzed in Toronto using WinDENDRO, an image analysis system specifically designed for tree-ring measurement and analysis. Tree cores were sanded using a belt sander, then a fine hand sander, and were finally wet-sanded with a hand sander. After sanding, tree cores were scanned into WinDENDRO, and annual ring widths were recorded for each tree core to the nearest 0.001 mm. The range of data was unique for each tree core, starting with 2004 growth and including anywhere from 15 to 150 yr growth depending on the core. The final 20 yr of growth only were used in analyses.

## 2.3. Tree growth models and estimates of NPP and carbon sequestration

Data on stand structure and individual tree growth allowed estimation of empirical functions relating tree diameter increment to diameter and local basal area. The relationship between diameter increment and diameter was linear on a semi-log scale (log diameter increment vs. diameter), and that between diameter increment and local basal area linear. Together, these patterns can be expressed as

$$\frac{dD}{dt} = a e^{-bD} - cB, \quad (1)$$

where  $D$  is the tree diameter,  $t$  is time,  $B$  is local basal area, and  $a$ ,  $b$ , and  $c$  are constants. Relationships were fitted using a maximum likelihood estimator in which residual variation was assumed to follow an exponential distribution. Relationships were fitted using the `optim()` function in R (R Core Development Team, 2004). In fitting Eq. (1), the 5 yr average growth increment for the previous 20 yr of growth (where available) was the dependent variable, and the average diameter pertaining to each range of years for a

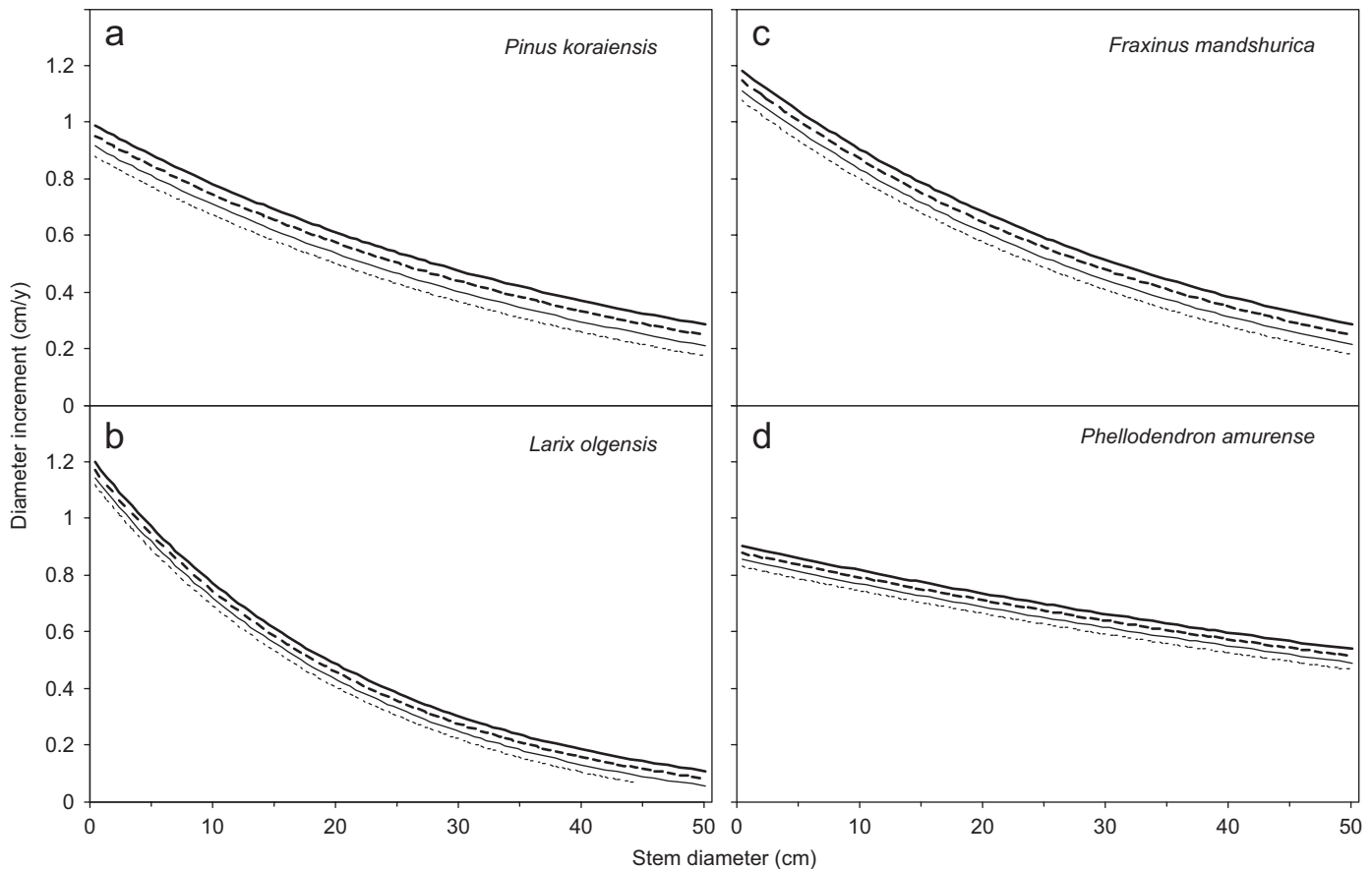


Fig. 1. Examples of relationships between diameter increment, tree diameter, and local basal area used in projecting species-specific growth curves: (a) *Pinus koraiensis*; (b) *Larix olgensis*; (c) *Fraxinus mandshurica*; and (d) *Phellodendron amurense*. Relationships were fit using maximal likelihood estimation for Eq. (1) (see Methods), and are plotted for the following local basal area categories: 10 m<sup>2</sup>/ha (solid, thick line); 20 m<sup>2</sup>/ha (dashed, thick); 30 m<sup>2</sup>/ha (solid, thin); 40 m<sup>2</sup>/ha (dashed, thin).

given tree, and local basal area at the end of this period were the independent variables. We specifically sampled dominant and co-dominant trees spanning a wide range of sizes; however, it was not possible to also sample across a wide range of basal areas in all cases. For this reason we initially examined species-specific relationships between diameter increment, diameter, and local basal area, but where no significant basal area effect was found, we substituted a species-pooled estimate for the constant  $c$  in Eq. (1), based on the subset of species for which a significant partial effect of basal area on diameter increment was detected.

Estimates of aboveground net primary productivity utilized assumptions for tree mortality utilized in the FAREAST gap model previously parameterized for sites in Manchuria, including Changbaishan (Yan and Shugart, 2005). Mortality rate is described by the following function:

$$p_{\text{mort}} = 1 - e^{(-\text{Death}/\text{AGE}_{\text{max}})}, \quad (2)$$

where  $P_{\text{mort}}$  is the annual probability of mortality,  $\text{Death}$  is constant,  $\text{AGE}_{\text{max}}$  is species-specific maximum age, and  $e$  is the base of natural logarithms (see Yan and Shugart, 2005).

Previously published allometric equations were used to estimate individual tree biomass as a function of stem diameter and height (Yan and Shugart, 2005). Empirical height–diameter relationships for each species were utilized to estimate height, described using an exponential generalization of the allometric equation:

$$H = H_{\text{max}} \left[ 1 - e^{(-\alpha D^{\beta})} \right], \quad (3)$$

where  $H$  is tree height,  $H_{\text{max}}$  is maximum tree height,  $D$  is stem diameter (here taken as DBH measured at 1.3 m), and  $\alpha$  and  $\beta$  are constants (Thomas, 1996).

In order to estimate total ecosystem carbon as a function of stand age, we utilized the Integrated Terrestrial Ecosystem C-budget model (InTEC) (Chen et al., 2000). This model requires as input a functional relationship between net primary productivity (NPP) and stand age. The growth and mortality functions (1 and 2), together with species-specific allometric functions, were used to generate NPP curve estimates for theoretical monoculture stands. The following function was then fit to the estimated NPP–age relationships as input to the InTEC model:

$$\text{NPP} = a_1(1 + (a_2(t/a_3)^{a_4}))/e^{(t/a_3)}, \quad (4)$$

where  $a_1$ ,  $a_2$ ,  $a_3$ , and  $a_4$  are arbitrary constants, and  $e$  is the base of natural logarithms.

The InTEC model was run as a “point” estimate of total ecosystem carbon content, and was in addition constrained in several respects. First, constant environmental conditions were assumed (i.e., 20th century average climate conditions for the site, and no CO<sub>2</sub> or nitrogen deposition effects). Second, we assumed default (deciduous angiosperm) values for leaf chemistry and all other physiological parameters. Third, we assumed a “reforestation” scenario with zero loss of soil organic carbon at harvest. Finally, although the InTEC model generally predicts declines in total forest carbon for older stands, we assumed that older stands would be in approximate carbon balance.

### 3. Results

A total of 678 stems of 14 canopy tree species were sampled and included in allometric and dendrochronological analyses. Estimated asymptotic maximum heights of species sampled ranged from 22.2 m (*A. mono*) to 42.0 m (*J. mandshurica*) (Table 1). Relationships between tree diameter increment, stem diameter, and local basal area were described reasonably well by Eq. (1) for most of the species examined (Fig. 1, Table 2). Most species showed parameter values indicating a negative feedback between tree size and diameter increment ( $b > 0$ ) and local basal area and diameter increment ( $c > 0$ ); however, three species showed (non-significant) slightly negative values for  $c$ , and *A. mono* showed a (non-significant) negative value for  $b$ . Substituting the pooled average values for parameters for these species had negligible effects on simulated growth curves.

Simulated biomass accumulation curves showed very wide variation among species (Fig. 2). Two native hard-

woods, *F. mandshurica* and *P. amurense*, showed both the highest estimated peak biomass and peak NPP values, exceeding those for other species, including the widely

Table 2

Estimated parameters of a function describing diameter increment of dominant and co-dominant trees as a function of tree diameter and local basal for 14 tree species sampled at Changbaishan nature reserve, Jilin Province, China

Species	Parameter		
	$a$	$b$	$c$
Gymnosperms			
<i>Abies nephrolepis</i>	0.494	0.00793	0.00582
<i>Larix olgensis</i>	1.280	0.04523	0.00268
<i>Pinus koraiensis</i>	1.048	0.02376	0.00369
Angiosperms			
<i>Acer mono</i>	0.195	−0.00821	0.00192
<i>Betula platyphylla</i>	0.896	0.04127	−0.00014
<i>Fraxinus mandshurica</i>	1.251	0.02733	0.00355
<i>Juglans mandshurica</i>	0.568	0.01806	−0.01532
<i>Phellodendron amurense</i>	0.934	0.01024	0.00239
<i>Populus davidiana</i>	0.712	0.01822	−0.00046
<i>Populus ussuriensis</i>	0.814	0.00983	0.00014
<i>Quercus mongolica</i>	0.292	0.00218	0.00252
<i>Tilia amurenensis</i>	0.365	0.00353	0.00087
<i>Tilia mandshurica</i>	0.272	0.00058	0.00034
<i>Ulmus davidiana</i>	0.809	0.03057	−0.00096

Parameters in Eq. (1) ( $dD/dt = ae^{-bD} - cB$ ) are interpreted as follows: Parameter  $a$  describes the maximum growth of small trees in the absence of neighbors; Parameter  $b$  describes the negative feedback between stem diameter and diameter growth; and parameter  $c$  describes the feedback between local basal area and diameter growth. Equations were fit using a maximum likelihood procedure as described in Methods; the average 95% confidence limit (where estimable) for  $a$  is  $\pm 0.377$ ; for  $b$  is  $\pm 0.02746$ ; and for  $c$  is  $\pm 0.01136$ .

Table 1

Estimated parameters of asymptotic height–diameter relationships for tree species sampled at Changbaishan nature reserve, Jilin Province, China

Species	$N$	$H_{\max}$	$\alpha$	$\beta$	$r^2$	$H_{\max}(\text{FE})$
Gymnosperms						
<i>Abies nephrolepis</i>	49	29.1	0.081	0.841	0.512	30
<i>Larix olgensis</i>	149	37.2	0.013	1.271	0.850	30
<i>Pinus koraiensis</i>	86	30.0	0.003	1.715	0.841	33
Angiosperms						
<i>Acer mono</i>	38	22.2	0.083	0.951	0.181	25
<i>Betula platyphylla</i>	36	34.7	0.042	0.976	0.369	27
<i>Fraxinus mandshurica</i>	39	39.7	0.109	0.562	0.486	33
<i>Juglans mandshurica</i>	37	42.0	0.160	0.386	0.265	29
<i>Phellodendron amurense</i>	20	20.9	0.063	1.027	0.618	32
<i>Populus davidiana</i>	39	33.4	0.234	0.457	0.259	25
<i>Populus ussuriensis</i>	34	27.7	0.003	1.983	0.760	
<i>Quercus mongolica</i>	37	33.9	0.312	0.309	0.067	30
<i>Tilia amurenensis</i>	39	31.5	0.028	1.041	0.675	30
<i>Tilia mandshurica</i>	37	29.2	0.082	0.798	0.376	
<i>Ulmus davidiana</i>	38	30.1	0.060	0.773	0.378	

Trees included in analyses were undamaged dominant or co-dominant in natural stands (with the exception of *Larix olgensis* and *Pinus koraiensis*, which include plantation grown and natural stands).  $N$  is sample size;  $H_{\max}$ ,  $\alpha$ , and  $\beta$  are parameters in the asymptotic allometric equation (Eq. (3)),  $r^2$  is the coefficient of determination for fitted equations, and  $H_{\max}(\text{FE})$  values are previously published estimates of maximum tree height used in the FAREAST gap model (Yan and Shugart, 2005).

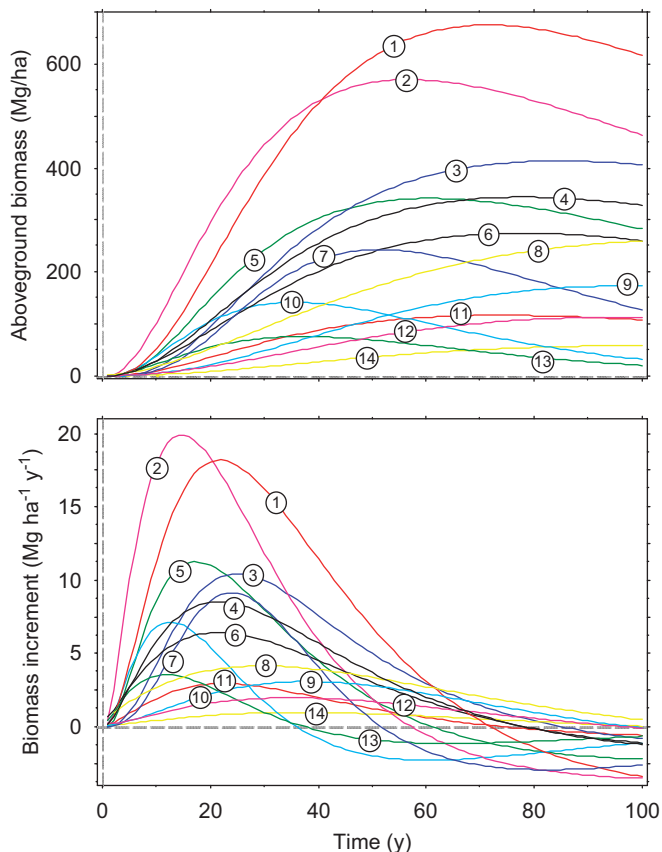


Fig. 2. Estimated growth curves for 14 common native canopy tree species in northeast China, expressed in terms of: (a) aboveground biomass, and (b) biomass increment. Values are projected on the basis of species-specific relationships describing diameter increment as a function of individual tree diameter and local stand basal area (Eq. (1)), and species-specific mortality functions (Eq. (2)). Initial planting conditions are assumed to be 1 cm DBH stock at  $3 \times 3$  m spacing. Numbers indicate species in decreasing order of predicted peak biomass: (1) *Phellodendron amurense*; (2) *Fraxinus mandshurica*; (3) *Pinus koraiensis*; (4) *Ulmus davidiana*; (5) *Larix olgensis*; (6) *Juglans mandshurica*; (7) *Quercus mongolica*; (8) *Populus ussuriensis*; (9) *Tilia amurense*; (10) *Betula platyphylla*; (11) *Abies nephrolepis*; (12) *Tilia mandshurica*; (13) *Populus davidiana*; and (14) *Acer mono*.

planted *L. olgensis* and *P. koraiensis* by 50% or more. Although some common native hardwoods, such as *A. mono*, showed slow growth and low estimated biomass accumulation, roughly half of the hardwood species were comparable in growth to currently planted conifers. Of the two poplar species examined, *P. ussuriensis* was among the pool of species showing relatively high growth, but *P. davidiana* was among the slower growing species. Estimated peak values for NPP were obtained at 13–25 yr for the faster-growing species examined, but not until 30–40 yr for many slow-growing species.

The two widely planted conifer species, *L. olgensis* and *P. koraiensis*, were sampled in both plantation and natural forest conditions. This permitted a test of the assumption that dominant and co-dominant trees in natural stands showed similar growth patterns to trees under plantation conditions (of a similar size and local basal area). Analysis

Table 3

Estimated NPP curve parameter values for 14 tree species sampled at Changbaishan Nature Reserve, Jilin Province, China

Species	Parameter			
	$a_1$	$a_2$	$a_3$	$a_4$
<b>Gymnosperms</b>				
<i>Abies nephrolepis</i>	0.00002148	237,065	10.0576	2.1466
<i>Larix olgensis</i>	0.00002325	683,572	7.4243	2.3806
<i>Pinus koraiensis</i>	0.00069077	15,055	9.3348	2.7293
<b>Angiosperms</b>				
<i>Acer mono</i>	0.00001489	128,397	17.1663	1.9522
<i>Betula platyphylla</i>	0.00001462	442,836	4.2271	2.8770
<i>Fraxinus mandshurica</i>	0.00019927	203,106	7.7434	1.9062
<i>Juglans mandshurica</i>	0.00019716	75,340	12.0685	1.7107
<i>Phellodendron amurense</i>	0.00017262	148,070	9.0099	2.3939
<i>Populus davidiana</i>	0.00003233	192,611	5.1594	2.1674
<i>Populus ussuriensis</i>	0.00001619	13,301	4.1986	5.4557
<i>Quercus mongolica</i>	0.00023408	48,160	22.8536	1.1963
<i>Tilia amurense</i>	0.00010865	34,830	13.8829	2.5395
<i>Tilia mandshurica</i>	0.00009335	38,937	15.532	2.0596
<i>Ulmus davidiana</i>	0.00019478	86,002	10.865	1.9566

Parameters are defined in Eq. (4) ( $NPP = a_1(1 + (a_2(t/a_3)^{a_4}))/e^{(t/a_3)}$ ), used as input for the InTEC model (Chen et al., 2000). Parameters were obtained as least-squares fits of estimated NPP curves projected from the empirical growth model (see Methods).

of covariance of log (increment) as a function of DBH did not reveal any significant difference in slope or intercept for either species ( $P > 0.1$  for all tests). Nor were significant differences found for relationships between diameter increment and local basal area for the two stand types.

Simulations of carbon accumulation made use of fitted equation to input NPP–age relationships for each species (Table 3). Results for species with relatively high growth rates reveal very large differences among species (Fig. 3). Two native hardwoods, *F. mandshurica* and *P. amurense*, showed by far the highest potential carbon sequestration potential. By 50–60 yr, these two species had simulated total forest carbon values 2–3 times that of the two widely planted conifer species *L. olgensis* and *P. koraiensis* (Fig. 3). A number of other native hardwood species, including *Ulmus davidiana*, *J. mandshurica*, and *Q. mongolica* also showed potential for total ecosystem carbon accumulation similar to that of the two widely planted conifers. All species showed small reductions in total ecosystem carbon at young stand ages (<10 yr), but these reductions only amounted to a few Mg C/ha at most.

#### 4. Discussion

In Northeast China, as in other parts of the country, a relatively small number of tree species are currently planted on a large scale. By far the most frequently cultivated are species of *Larix* (*L. olgensis* in the Changbaishan region), and *P. koraiensis*. Our results suggest that although these two species do have relatively fast growth and high potential for carbon sequestration, they are not necessarily the best among the pool of native tree species. Parameter  $a$

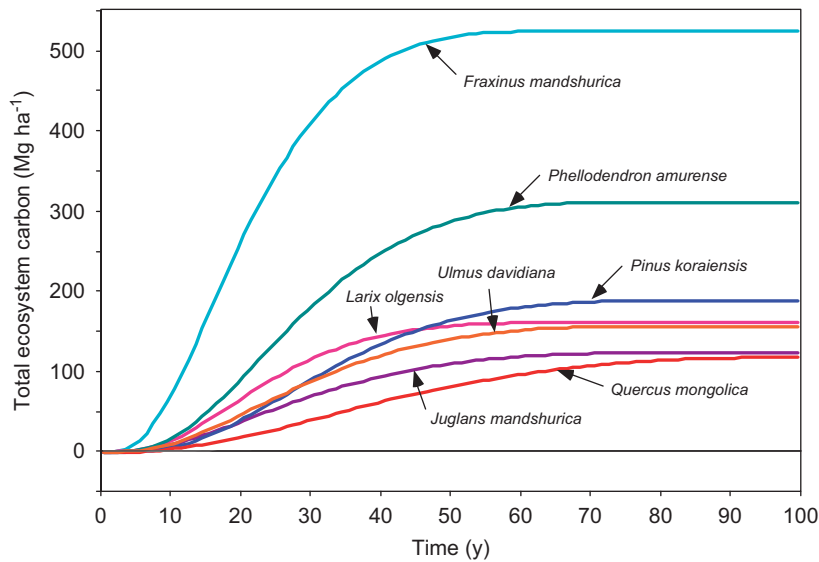


Fig. 3. Projected accumulation curves for total ecosystem carbon (including live trees, dead wood, and soil) for 7 species with high carbon sequestration potential in northeast China. Estimates utilize net primary productivity curves as input to the InTEC model (Chen et al., 2000), and assume steady-state total carbon for older stands. Note that a very small decrease (1–3 Mg C/ha) in carbon is predicted very early in stand development.

in Eq. (1) describes the estimated maximum growth rate of a dominant (“released”) tree of a given species at 1.0 cm DBH. By this measure, species with a “high” growth potential ( $a > 0.5$  cm/yr) do include the two widely planted conifers ( $a = 1.28$  and  $1.05$  for *L. olgensis* and *P. koraiensis*, respectively); however, comparable growth rates are also found for *F. mandshurica* ( $a = 1.25$  cm/yr), *P. amurense* ( $a = 0.93$  cm/yr), *B. platyphylla* ( $a = 0.90$  cm/yr), *Ulmus davidiana* ( $a = 0.81$  cm/yr), *P. ussuriensis* ( $a = 0.81$  cm/yr), *P. davidiana* ( $a = 0.71$  cm/yr), and *J. mandshurica* ( $a = 0.57$  cm/yr).

The potential for a tree species’ utility in carbon forestry obviously depends on a wide variety of properties in addition to the maximum potential growth rate of released trees early in ontogeny. Two essential properties are how fast growth rates decrease through tree ontogeny (i.e., the feedback between size and growth), and the response of the species to competition, particularly from conspecifics. The growth analysis presented here quantifies these two aspects as single parameters (Table 2), and provides the basis for projections of growth trajectories for monocultures of a given species (Fig. 2). These analyses indicate high variation among species in both parameters, which contribute to variation in predicted growth curves (Fig. 2), and ultimately in predicted trends for total system carbon sequestration (Fig. 3). The resulting growth curves suggest that a number of native hardwood species, in particular *F. mandshurica* and *P. amurense*, are likely to outperform the widely planted conifers in terms of both biomass production and carbon sequestration.

Projections based on growth patterns in natural stands represent an unusual approach to estimating potential growth patterns for plantation silviculture: the traditional approach would be to undertake direct silvicultural trials. While there is ultimately no substitute for direct

experimentation, there are two major advantages of the approach used here of particular relevance from a carbon sequestration perspective. First, our approach encompasses long time scales and provides direct data on the growth performance of larger trees that are major stores of carbon in natural ecosystems. Second, it permits a rapid evaluation of growth potential for many species, thus providing an early indication of which species are most worthy of future trials.

The most critical assumption of our projections is that growth patterns observed under natural conditions will predict growth of plantation forests. For the two widely planted conifer species, we found no detectable differences in patterns of tree diameter increment relative to tree size and local competition for natural stands vs. plantations. However, there are biological reasons to suspect that such differences might be manifest for other species. For example, a native pest of *F. mandshurica*, the Emerald Ash Borer (*Agrilus planipennis*), has recently had devastating impacts on ash plantations in North America, and could have similar effects within its native range. On the other hand, growth rates well above those found in natural stands are commonly obtained in well-managed plantations, particularly if efforts are made at genetic improvement or intensive silvicultural methods are employed. The model projections made here may thus be underestimates of the growth and carbon sequestration potential of native hardwoods in the region. The InTEC model used also assumes that tree biomass has a constant carbon content of 0.5. Although actual values deviate from this and tend to favor conifers over hardwoods (Thomas and Malczewski, 2006, this issue), these differences are very small compared to the differences in projected tree growth. Elsewhere in China, timber yields and carbon sequestration have been found to decrease through successive rotations of conifers,

particularly when intensive soil preparation measures are used (Zhang et al., 2004). This phenomenon also argues for increased use of native hardwood species, and perhaps for planned rotations of species, as well as for more careful management of soil carbon during post-felling operations.

In order for carbon sequestration to be an effective tool in the reduction of atmospheric greenhouse gases, provisions must be made to utilize timber upon harvest in a manner that does not result in high emissions of CO<sub>2</sub> into the atmosphere. Information on the economic values of different Chinese timber species, as well as their potential for use in long-lasting wood products, are thus important considerations. Current market values for many of the hardwoods considered here are considerably higher than the commonly grown conifers (Peng Shi-Kui, personal communication). *F. mandshurica*, *Quercus mongolica*, and *Tilia* species, in particular, stand out as having very high market value. These species are also very commonly used in long-lived products such as furniture and cabinetry. Thus, wood utilization and market considerations are also likely to strongly favor planting of native hardwoods in carbon sequestration forestry in Northeast China.

Non-economic forest values, in particular biodiversity conservation, have been commonly overlooked in recent carbon sequestration initiatives. The legal text of the Marrakesh Accord (negotiated under the Kyoto Protocol framework) makes only passing references to biodiversity, and has been strongly criticized for ignoring this issue (Noss, 2001; Shulze et al., 2002). However, it is stated in the Marrakesh Accord that one of the principles that should govern any LULUCF project is its environmental impact on biodiversity and natural systems. As in other regions, increased use of a wider range of native species in Northeast China undoubtedly would have a positive effect on biodiversity (Lugo, 1997; Lamb, 1998; Hartley, 2002), particularly if some forests were later allowed to undergo natural succession. Such forests would also likely positively affect other “ecosystem functions” such as erosion control and flood mitigation. It should also be noted that in terms of combating climate change, it is not the rate at which trees and forests sequester carbon, but the absolute amount of carbon sequestered that is of primary importance. One option for using carbon sequestration to fulfill Kyoto requirements without sacrificing biodiversity would be to modify management practices to better conserve old-growth forests, which have high values for both biodiversity and carbon content (e.g., Suchanek et al., 2004).

## 5. Conclusions

In the broader literature on restoration forestry, it is considered axiomatic that species choice is the single most important decision determining future success (Haggard et al., 1998; McLeod, 2000; Stanturf and Madsen, 2002). The present study reinforces this conclusion for the special case of carbon sequestration forestry. Among the species selected, the highest estimated rates of biomass accumula-

tion and carbon sequestration were obtained for two native hardwood species that are very infrequently planted at present, and several other native hardwoods show potential similar to that of the two widely planted conifers. In addition, incorporation of native hardwood plantations in carbon sequestration forestry in China is likely to contribute substantively to economic diversification and increased provision of “ecosystem services”, including biodiversity conservation.

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