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Impacts of fire severity and post-fire reforestation on carbon pools in boreal larch forests in Northeast China

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Abstract

Aims

Boreal larch (*Larix gmelinii*) forests in Northeast China have been widely disturbed since the 1987 conflagration; however, its long-term effects on the forest carbon (C) cycling have not been explored. The objective of this study thus was to quantify the effects of fire severity and post-fire reforestation on C pools and the changes of these forests.

Methods

Sixteen permanent plots have been set in two types of larch stands (*L. gmelinii*-grass, LG; and *L. gmelinii-Rhododendron dahurica,* LR) with three levels of fire severity (unburned, low-severity and high-severity but replanted), at 1987 burned sites in Daxing'anling, northeastern China, to repeatedly measure ecosystem C pools in 1998 and 2014. C components were partitioned into vegetation (foliage, branch, stem and roots), soil and detritus (standing and fallen woody debris and litter). The fire effects on post-fire C dynamics were examined by comparing the differences of C pools and changes between the two field investigations caused by fire severity.

Important Findings

During the study period, unburned mature stands were C sinks (105 g C m^{-2} $year^{-1}$ for LG, and 190 g C m^{-2} $year^{-1}$ for LR), whereas the low-severity stands were C-neutral (−4 and 15 g C m^{-2} $year^{-1}$ for LG and LR, respectively). The high-severity burned but reforested stands were C sinks, among which, however, magnitudes (88 and 16 g C m^{-2} $year^{-1}$ for LG and LR, respectively) were smaller than those of the two unburned stands. Detritus C pools decreased significantly (with a loss ranging from 26 to 38 g C m^{-2} $year^{-1}$) in the burned stands during recent restoration. Soil organic C pools increased slightly in the unmanaged stands (unburned and low-severity, with accumulation rates ranging from 4 to 35 g C m^{-2} $year^{-1}$), but decreased for the high-severity replanted stands (loss rates of 28 and 36 g C m^{-2} $year^{-1}$ for LG and LR, respectively). These results indicate that fire severity has a dynamic post-fire effect on both C pools and distributions of the boreal larch forests, and that effective reforestation practice accelerates forest C sequestration.

Keywords: carbon budget, wildfire, fire severity, *Larix gmelinii*, reforestation

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INTRODUCTION

As a major part of global forest ecosystems, boreal forests occupy around 34% of forest area (1.5×10^9 ha; Bonan 2008), with a carbon (C) stock of 272 ± 26 Pg (1 Pg = 10^{15} g; Pan *et al.* 2011). This large amount of C stored in the cold biome may be extremely sensitive to ongoing climate change (Gorham 1991; Harden *et al.* 2000). Boreal forests have experienced frequent wildfire disturbances during the recent decades (Amiro *et al.* 2001; Lü *et al.* 2006). A catastrophic fire not only causes a sudden emission of C from the forest ecosystem to the atmosphere, but also has long-term effects on post-fire forest structure and function (Amiro *et al.* 1999; Law *et al.* 2001). However, long-term effects of stand-replacing fire on forest C budgets are still uncertain, including those of vegetation recovery (Amiro *et al.* 2003; Kashian *et al.* 2006), C fluxes of soil (Goulden *et al.* 2011; Hagemann *et al.* 2010), and energy and nutrient exchange of forest detritus (Bond-Lamberty *et al.* 2003; Kauffman *et al.* 1988; Nalder and Wein 1999).

The role of fire in the terrestrial C cycle has been studied extensively, especially in boreal forests (e.g. Drobyshev *et al.* 2014; Randerson *et al.* 2006; Rouvinen and Kuuluvainen 2001), but long-term effects of wildfires on ecosystem C budgets are still poorly understood. A stand-replacing fire destroys vegetation structure and alters the distribution of live and dead C pools (Bormann *et al.* 2008; Kashian *et al.* 2013). In addition, C emission from post-fire decomposition of woody debris and litter over a long period may exceed sudden emission from combustion (Dixon and Krankina 1993; Hicke *et al.* 2003; Wirth *et al.* 2002). Most studies have investigated the effects of fire and post-fire management on C dynamics via a space-for-time substitution approach (Brassard and Chen 2008; Goulden *et al.* 2011). This may have serious flaws, especially for the soil C budget (cf. Wang *et al.* 2003). Generally, both biometric (Fang *et al.* 2007) and year-round eddy covariance methods (Grace *et al.* 1995; Curtis *et al.* 2002) have been used to estimate C balance of forests. Goulden *et al.* (2011) combined the two methods to estimate C balance along a fire chronosequence in a boreal forest of Canada, and found the recovery of C balance from fire was rapid, but it still based on a space-for-time substitution approach. Ideally, long-term repeated measurements in permanent plots would substantially reduce the uncertainty, but there have been few such efforts so far.

Wang *et al.* (2008) synthesized chronosequence studies in global forest ecosystems. They reported that the ecosystem was a net C source right after vegetation removal, gradually changed to a net C sink when relative stand age (defined as the ratio of actual stand age to the stand rotation age) approached 0.3, and maximized C sequestration capacity in pre-mature or mature stands. However, the tipping point for switching from C source to C sink typically depends on disturbance intensity and post-event managements (Ma *et al.* 2013; Meigs *et al.* 2009). An intensively replanted forest after fire may quickly shift from CO_2 source to sink in evergreen coniferous forests (Thornton *et al.* 2002), but this transition requires decades for boreal conifers (Fredeen *et al.* 2007). Nevertheless, all the previous studies used a space-for-time substitution approach. Thus, we need a more critical validation, such as through long-term monitoring of permanent forest stands.

A notorious conflagration on 6 May 1987 in northeastern China lasted about a month, burning more than 1.14 million hectares (Mha) of boreal forest (Cahoon *et al.* 1991; Guo *et al.* 2015). This, one of the largest forest fires in Chinese history, caused huge economic losses and rapid release of 25–49 Pg C into the atmosphere (Wang *et al.* 2001). Following up the study of Wang *et al.* and using three levels of fire severity (unburned mature, low-severity and high-severity) in the Daxing'anling region of Northeast China during 1998 (Wang *et al.* 2001), we measured C pools of all components in the same permanent plots (16 permanent plots in total; see Materials and Methods section) of two types of boreal forest stands (*Larix*-grass, LG and *Larix-Rhododendron*, LR) in 2014. The objective of the present study was to quantify the effects of fire severity and post-fire reforestation on C pools and changes of this boreal forest ecosystem.

MATERIALS AND METHODS

Study sites and experimental design

The study was conducted in the Daxing'anling region (52°44′N, 123°50′E, 520 m a.s.l.) in 1998 (Wang *et al.* 2001) and 2014. Mean annual temperature is −3°C, and mean annual precipitation is between 350 and 500 mm, of which 85–90% falls in summer (Xu 1998). The soil is dark brown forest soil (Table 1).

The forests were dominated by *Larix gmelinii*, accompanied by *Pinus sylvestri*, *Betula platyphylla* and *Populus davidiana*. The variation of the understory species is used to define different *Larix* forest ecosystems. We included two dominant larch

Table 1: Changes in site characteristics for the two types of *Larix gmelinii* forests (*Larix*-grass (LG) and *Larix-Rhododendron* (LR) stands) between the two investigations (1998 vs. 2014)

Forest type	Fire severity	*N*	Plot size (m^2)	Age (year)	Density (trees ha^{-1})		Mean DBH (cm)		Mean height (m)	
					1998	2014	1998	2014	1998	2014
LG	Unburned	3	20 × 30	130	920	917	18.6	16.2	N.D.	16.7
	Low severity	3	20 × 30	110	489	817	21.6	18.1	19.0	16.8
	High severity	3	20 × 20	17	14	2600	17.0	9.3	8.0	11.4
LR	Unburned	1	25 × 40	130	1340	1340	15.5	18.1	10.2	18.2
	Low severity	3	25 × 40	110	486	410	20.8	18.5	N.D.	16.5
	High severity	3	20 × 20	17	41	1900	17.0	7.4	17.0	8.1

Abbreviations: Age = stand age measuring from tree cores at breast height (Schweingruber 1988), DBH = diameter at breast height, 1.37 m, *N* = numbers of each plot; N.D. = not detected.

forest types, *L. gmelinii*-grass (LG) and *L. gmelinii-Rhododendron dahurica* (LR). LG stands were on gentle slopes, where the understory was dominated by *Pyrola incarnate* and *Carex* sp. LR stands were on steeper slopes, where the common understory species include *R. dahurica*, *Vaccinium uliginosum* and *Vaccinium vitis-idaea*.

The experimental design included the aforementioned forest types (LG and LR) and fire severity levels. Most of the trees in the high-severity stands were killed by the 1987 conflagration, whereas trees in the low-severity stands suffered minor damage. The unburned mature stands survived the fire, owing to physical barriers preventing fire spread. Three replicate plots were established in each stand, except for the unburned mature LR (only one plot established in August 1998 remained in 2014). A total of 16 plots (detailed information on which is given in Table 1) were re-investigated in August 2014 (Fig. 1). In 2000, the high-severity burned plots were planted with 3-year-old seedlings of *L. gmelinii* with a density of ~3000 trees ha^{-1} in the LG and LR stands. All C components were measured using the same method in the two field investigations (Zhu *et al.* 2010).

Measurement of plant C pools

At each plot, we measured diameter at breast height (DBH, in centimeters) of all trees taller than 1.3 m. Tree height (H, in meters) was respectively estimated with the following DBH-H relationships for *L. gmelinii* and *B. platyphylla*:

$$\textit{L. gmelinii}: \mathrm{H} = 1.18 + 0.958\ \mathrm{DBH}\ \left(R^2 = 0.95,\ n = 30\right) \quad (1)$$

$$\textit{B. platyphylla}: \mathrm{H} = -0.57 + 1.150\ \mathrm{DBH}\ \left(R^2 = 0.99,\ n = 15\right) \quad (2)$$

We applied the allometric equations (Wang *et al.* 2001) with the DBH and H data to estimate biomass components (stem, branch, leaf, root and total biomass) of each individual tree and calculated these for each plot. Biomass of understory and ground vegetation were measured using three 0.5×0.5 m^2 subplots in each plot. All understory and ground vegetation were harvested, dried at 70°C for more than 48 h to a constant mass, and weighed to the nearest 0.01 g. We applied generic C concentrations (45% for leaf and fine roots and 50% for other woody tissues; Gower *et al.* 2001) to convert biomass to C density. Meanwhile, we measured the age of 10 trees with the largest DBH in each forest type by using the tree-ring analysis (Worbes *et al.* 2003). The age of the oldest tree was defined as the age of the forest, although such forest ages might be lower than their real ages (Schweingruber 1988).

Soil and detritus investigation

Three soil profiles down to the C horizon were randomly dug in each plot. Paired samples of soil were collected at 10-cm intervals to 40–50 cm depth using a 100-cm^3 soil sampler. One soil core sample was dried at 105°C for 48 h to a constant mass, and weighed to the nearest 0.01 g to calculate bulk density. The other sample was air-dried for 2 weeks to measure C concentration by the Walkley-Black method (Black 1965). With bulk density and soil C concentration, we calculated C density of each depth, then summed all the layers to estimate the entire soil organic C pool along 40–50 cm depth per hectare.

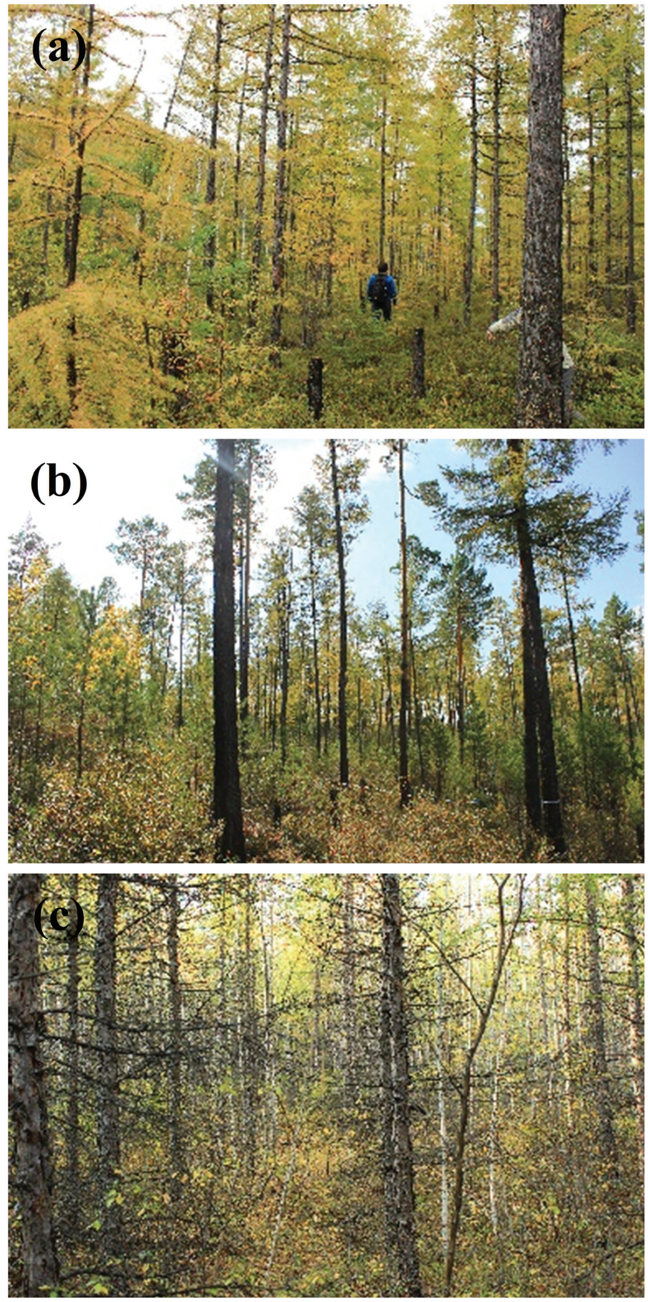

Figure 1: Site photos taken in August of 2014 showing (a) unburned mature stand, (b) low-severity stand, and (c) high-severity stand in this study. Note that the high-severity stands were replanted with 3-year-old seedling of *Larix gmelinii* intensively in 2000.

Detritus consisted of coarse woody debris (CWD, including snag and downed log), fine woody debris (FWD) and litter. CWD was defined as wood of ≥10 cm in diameter at the largest part, FWD as that with 2 cm ≤ diameter < 10 cm, and litter as that with diameter <2 cm (Harmon *et al.* 1986). Woody debris at each plot was collected and weighed. If dead wood could not be weighed directly (e.g. wood ≥25 cm diameter), we weighed three of its parts (middle part and both end cylinders of ~15 cm length) after oven drying at 85°C to a constant mass (Grove 2001). We constructed a linear regression model between volume and weight across the three cylinders, and then used this model to calculate the mass of over-weighted wood debris dependent on its volume. The model was extrapolated to other large woody debris of the same decay status and species for its biomass estimation. We defined CWD decay into four classes: (1) recently downed log with wood and bark intact; (2) sapwood beginning to decay, bark lost or easily removed, and heartwood sound; (3) bark loss, heartwood decay and large branch loss but stubs present and (4) heartwood rotten (modified from Sollins 1982 and Marra and Edmonds 1994). Five subplots (2 × 2 m^2) were set up in each plot to measure the litter C density. All leaf litter (including fresh and semi-decomposed), reproductive organs, fallen bark, small dead wood <2 cm diameter (at the largest part) and other dead plant materials in the surface litter layer were collected, oven dried at 65°C to a constant mass and weighed.

Statistical analyses

Each plot was used as the experimental unit for all calculations and statistical analyses. A paired *t*-test was used to determine significant differences in each C component between the two field investigations (1998 vs. 2014). One-way analysis of variance (ANOVA) was used to detect significant differences in C pools and their changes among three fire severities. All statistics were determined using R 2.15.1 software (R Core Team 2012).

RESULTS

Carbon stocks and changes in vegetation

Vegetation C densities were 48.5 ± 8.7 Mg C ha^{-1} (LG) and 133.7 Mg C ha^{-1} (LR) for the two unburned forest stands in 2014 (Fig. 2). Vegetation C densities were lower in the low- and high-severity stands than in the unburned stands. Twenty-seven years after the fire disturbance, vegetation C densities were 38.0 Mg C ha^{-1} in the low-severity stands (both LG and LR), and 32.0 ± 1.7 and 21.0 ± 0.2 Mg C ha^{-1} for the LG and LR in high-severity stands, respectively. For the two forest stands, vegetation C density was 22–72% lower in the low-severity stands than that in the unburned mature larch stands in 2014 ($P < 0.01$; Table 2), which are significantly higher than corresponding decreases (5–67%) measured in 1998. ($P < 0.05$). However, because of replanting in 2000, vegetation C densities in 2014 were only 34% (LG) and 84% (LR) lower in the high-severity stands than in the unburned forest stands, which were lower than those (76 and 94%, respectively) in 1998 (P from 0.054 to < 0.001).

Compared with biomass C densities obtained in 1998 (36.8 ± 7.4 Mg C ha^{-1} in LG and 108.4 Mg C ha^{-1} in LR), changes in biomass C densities in the two unburned forest plots were 11.7 ± 2.2 (LG) and 25.3 (LR) Mg C ha^{-1}, indicating substantial biomass C uptakes of 73 ± 14 (LG) and 158 (LR) g C m^{-2} $year^{-1}$ in these boreal forests during the last two decades ($P < 0.05$). For low-severity stands, C densities in vegetation slightly increased from 35.1 ± 7.4 Mg C ha^{-1} in 1998 to 38.0 ± 8.7 Mg C ha^{-1} in 2014 in the LG plot, and 35.3 ± 3.6 Mg C ha^{-1} to 38.0 ± 2.8 Mg C ha^{-1} in the LR plot. The changes

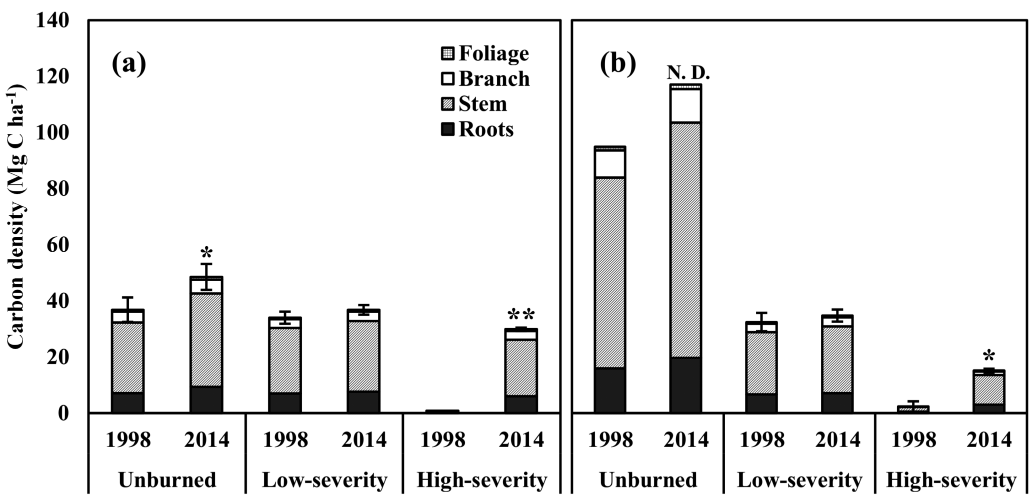

Figure 2: Comparison of vegetation carbon density (mean ± SD, n = 3) and its distribution between the two investigations (1998 vs. 2014) for the *Larix*-grass (a) and *Larix-Rhododendron* stands (b) by biomass components and fire severities. Asterisks denote significant differences between the two investigations: *$P < 0.05$ and **$P < 0.01$. N.D. = not detected.

in vegetation C densities (18 and 17 g C m^{-2} $year^{-1}$ for LG and LR, respectively) were considerably smaller than those in the unburned forest plots. However, for the replanted high-severity plots, the vegetation C pool showed a large C sink, comparable to the unburned forest stands (145 vs. 73 ± 14 g C m^{-2} $year^{-1}$ in LG stands and 89 ± 38 vs. 158 g C m^{-2} $year^{-1}$ in LR stands).

Carbon stocks and changes in soil and detritus

Fire affected soil C density of the two forests differently (Fig. 3). Soil C densities of high-severity stands were slightly higher than unburned stands for both LG and LR stands in 1998. Sixteen years after the first field investigation, although soil organic C density in the unburned stand of LR was 8.5 and 46.5 Mg C ha^{-1}, which was less than low- and high-severity stands, respectively ($P < 0.001$), soil C density was similar between LG plots in 2014. Increasing soil C densities of unburned and low-severity forests were observed in the second field investigation. However, compared with the first investigation, the soil C densities in the high-severity stands decreased by 4.4 (LG) and 5.7 (LR) Mg C ha^{-1} ($P < 0.01$, Table 2).

The total detritus C pool (including fallen log, standing snag and litter) varied from 8.9 to 19.1 Mg C ha^{-1} across all forest stands (Fig. 4). Twenty-seven years after the fire, C densities of fallen log and standing snag (CWD) at the burned sites were still 3–10 times greater than those at the unburned sites, and constituted 36–70% of the total detritus C of the burned sites. Correspondingly, CWD accounted for only 14% of the total detritus C pool in the unburned mature forest stands. In contrast, C densities in litter at the burned sites (5.3–9.1 Mg C ha^{-1}) were slightly lower than those at the unburned sites (7.7–10.4 Mg C ha^{-1}). Fire affected the constitution of forest detritus in this boreal forest. Even 27 years after the catastrophic fire, we determined a considerably smaller contribution of litter C pool to the total detritus pool than that of the unburned site.

Table 2: Results of the one-way ANOVA for effects of fire severity on the carbon density of forest ecosystem carbon component (Mg C ha^{-1}) and its change rate from 1998 to 2014 (g C m^{-2} $year^{-1}$) for the *Larix*-grass (LG) and *Larix-Rhododendron* (LR) stands

Forest types	C components	df_1	df_2	1998		2014		Change rates	
				F	*P*	*F*	*P*	*F*	*P*
LG	Vegetation	2	6	90.9	**<0.001**	20.0	**0.002**	15.8	**0.004**
	Soil	2	6	1.4	0.31	14.0	**0.005**	1.2	0.371
	Detritus	2	6	138.5	**<0.001**	0.6	0.558	21.2	**0.002**
	Ecosystem total	2	6	7.9	**0.021**	2.2	0.191	1.0	0.414
LR	Vegetation	2	4	414.4	**<0.001**	874.5	**<0.001**	87.8	**<0.001**
	Soil	2	4	21.3	**0.007**	198.4	**<0.001**	0.2	0.860
	Detritus	2	4	12.9	**0.018**	10.0	**0.028**	1.1	0.41
	Ecosystem total	2	4	19.0	**0.009**	437.1	**<0.001**	8.4	**0.037**

Bold *P*-values indicate significant effect ($P < 0.05$).

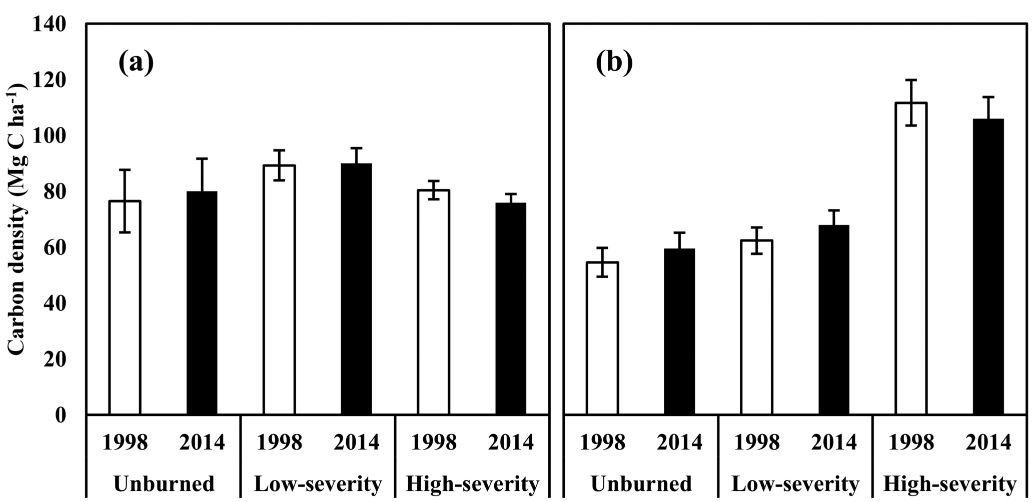

Figure 3: Comparison of soil organic carbon density (mean ± SD, *n* = 3) between the two investigations for the *Larix*-grass (a) and *Larix-Rhododendron* (b) stands by fire severities.

There were no statistically significant differences between the detritus C densities in 1998 and 2014 at the unburned mature forest sites (<6 g C m^{-2} year^{-1}). However, these densities in the low- and high-severity stands were substantially altered. For example, the fallen log C density declined from 15.5 Mg C ha^{-1} in 1998 to 10.7 Mg C ha^{-1} in 2014 in the low-severity LR plot (loss rate 30 g C m^{-2} year^{-1}), and 11.5–7.6 Mg C ha^{-1} (loss rate 24 g C m^{-2} year^{-1}) in the high-severity LG plot. Litter lost C density at 7 g C m^{-2} year^{-1} in the high-severity LR plot and at 58 g C m^{-2} year^{-1} in the low-severity LG plot. Overall, detritus C pools increased slightly at unburned mature forest sites, but decreased at ~30 g C m^{-2} year^{-1} in all the burned forest stands (P < 0.05).

Changes in ecosystem carbon

Annual changes of total ecosystem C pools at the unburned sites were 105 g C m^{-2} (LG) and 190 g C m^{-2} (LR) over the 1998–2014 periods (Fig. 5). The boreal mature forests have

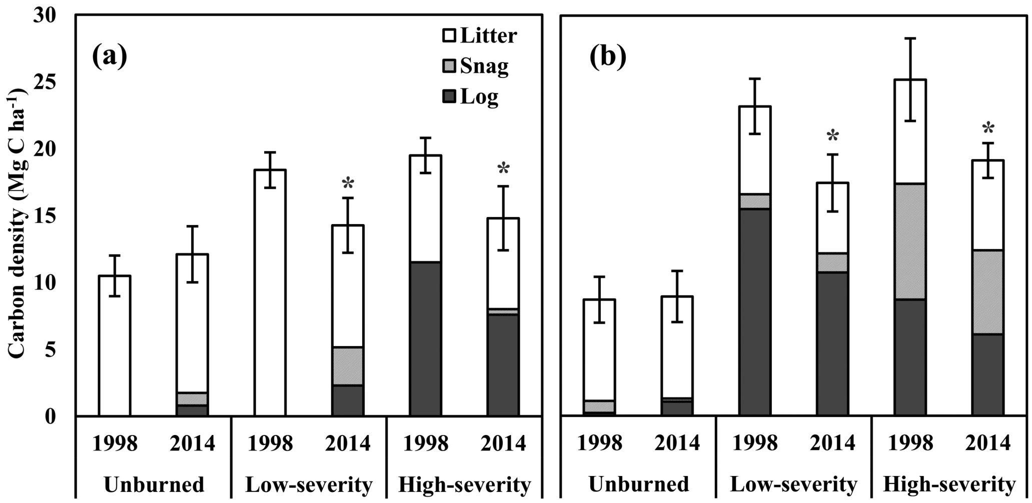

Figure 4: Comparison of detritus carbon density (mean ± SD, n = 3) and its distribution into litter, snag and log for the *Larix*-grass (a) and *Larix-Rhododendron* (b) stands by fire severities. Asterisks denote significant differences between the two investigations at P < 0.05.

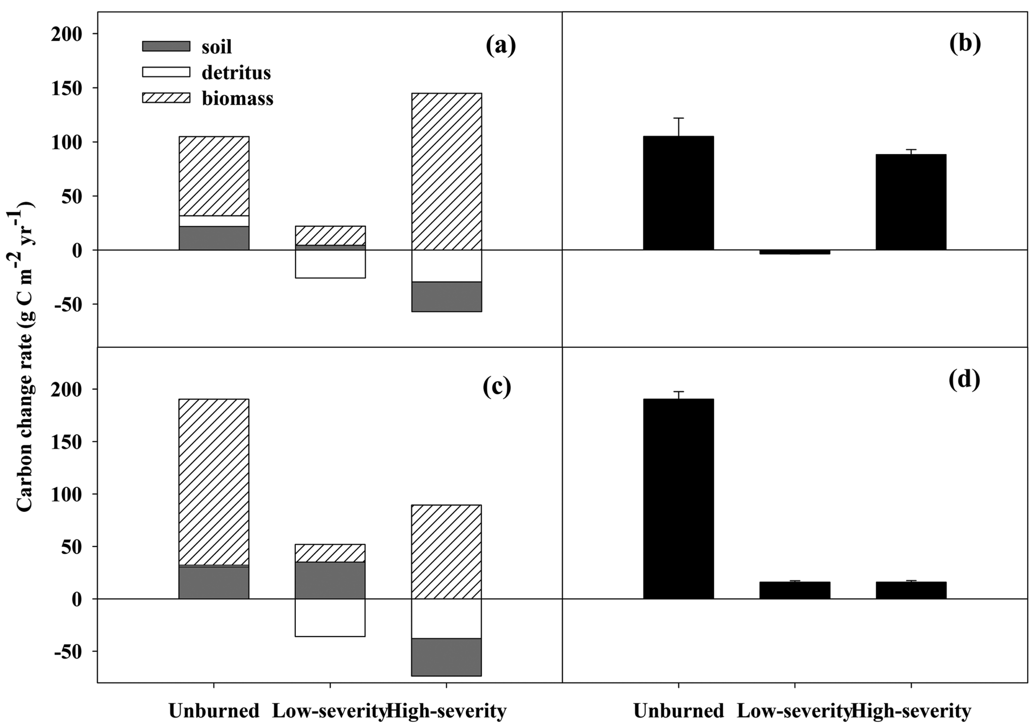

Figure 5: Change rates of component and total ecosystem carbon stocks (mean ± SD, n = 3) for the *Larix*-grass stands (a, b), and the *Larix-Rhododendron* stands (c, d) by fire severities.

accumulated 73 g C m^{-2} $year^{-1}$ (70%, LG) and 158 g C m^{-2} $year^{-1}$ (83%, LR) in their vegetation; 22 g C m^{-2} $year^{-1}$ (21%, LG) and 31 g C m^{-2} $year^{-1}$ (16%, LR) in soil organic matter and 10 g C m^{-2} $year^{-1}$ (10%, LG) and 2 g C m^{-2} $year^{-1}$ (1%, LR) in detritus. In contrast, 27 years after fire, the low-severity forests are C-neutral or a slight C sink. Even the replanted stands (high-severity stands) accumulated only 88 g C m^{-2} $year^{-1}$ (LG) and 16 g C m^{-2} $year^{-1}$ (LR), which is lower than the accumulation rate at their corresponding unburned mature forest stands.

DISCUSSION

Carbon budget of the mature larch forests

The ecosystem C stock (141 Mg C ha^{-1} for LG and 202 Mg C ha^{-1} for LR) in the unburned *L. gmelinii* forests was smaller than that of another mature *L. gmelinii* forest stand in a similar mature forest (275 Mg C ha^{-1}) (Mu *et al.* 2013). Soil organic C density (80.0 Mg C ha^{-1} for LG and 59.5 Mg C ha^{-1} for LR stands) was also considerably lower than that (174.9 Mg C ha^{-1}) estimated by Mu *et al.* (2013). In our study, vegetation C density varied greatly between LG and LR stands (48.5 vs. 133.7 Mg C ha^{-1}). These results suggest considerable differences in C density of each ecosystem component, even across similar forest stands (Noh *et al.* 2013). Numerous studies have estimated C dynamics of special forest ecosystems using different age categories (Brown and Lugo 1990; Hughes *et al.* 1999; Schulze *et al.* 1999), but long-term observation of the C changes in permanent forest stands or plots is rare (He *et al.* 2013). Based on our repeated investigation of permanent forest plots at a 16-year interval, the boreal mature *Larix* forest ecosystem was a C sink.

The annual rate of C accumulation in vegetation was estimated at 73 (LG) and 158 (LR) g C m^{-2} $year^{-1}$, which is substantially greater than that (14 g C m^{-2} $year^{-1}$) in boreal forests of the world (Pan *et al.* 2011), but it is comparable to that of Mu *et al.* (2013). C densities of soil organic matter and detritus increased slightly with time (Figs 3 and 4). In the unburned mature stands, the total increment of detritus and soil C densities accounted for 30% (LG) and 17% (LR) of the total ecosystem C accumulation rate. Soil organic C accumulation rate was 22 (LG) and 31 (LR) g C m^{-2} $year^{-1}$, comparable to 21–40 g C m^{-2} $year^{-1}$ at two Scots pine ecosystems in southern Germany (Prietzel *et al.* 2006).

Effects of fire severity and afforestation on post-fire carbon budget

The wildfire destroyed the boreal forest ecosystem structure and function by killing plants, altering energy and elementary budgets (Christensen 1987), changing species composition (Gilliam and Platt 1999; de Paz and Raffaele 2013; Zhang *et al.* 2014), and thus influencing C stocks and fluxes (Gough *et al.* 2007; Irvine *et al.* 2007; Ouyang *et al.* 2014). Dore *et al.* (2008) reported that the high-severity burned *Pinus ponderosa* forest remained a major source of CO_2 to the atmosphere after 10 years of restoration. In our study, the unmanaged low-severity stands were C-neutral during the 11–27 years after the fire (Fig. 5). Transition (shift from C source to sink) required 10–30 years in temperate forests of the Pacific Northwest United States and even longer in boreal forest. But Bond-Lamberty *et al.* (2004) found the boreal *Picea mariana* forest in Canada was a carbon moderate C source (100 g C m^{-2} $year^{-1}$) immediately after fire, and the middle-aged stands (12–20 years after fire) was a strong sink (100–300 g C m^{-2} $year^{-1}$). The replanted high-severity stands represented an obvious C sink (88 and 16 g C m^{-2} $year^{-1}$) during our 1998–2014 study period. These results suggest that the effects of fire on the ecosystem C balance persist for a certain period, typically depending on fire intensity and forest recovery (Wang *et al.* 2003). However, an effective management after a stand-replacing fire could accelerate post-fire restoration of the C balance from C source to sink.

Many efforts have been made to examine the effects of fire on C budgets by quantifying changes in C pools of variously aged forest stands originating from fires (Cahoon *et al.* 1994; Goulden *et al.* 2011; Wang *et al.* 2003). Fire reduces total C stock and shifts ecosystem C allocation from vegetation to detritus (Brassard and Chen 2008; Meigs *et al.* 2009; Wang *et al.* 2001). In contrast with the alternative method of space-for-time substitution, we directly measured C stocks of each ecosystem component at six permanent forest plots in 1998 (11 years after the 1987 fire) and 2014. We found that vegetation growth rates were slower in the unmanaged low-severity stands than rates in the unburned stands. In contrast, the high-severity but replanted stands had a rapid vegetation growth rate between 1998 and 2014. Meanwhile, detritus C pools decreased at all burned sites during the field investigations. Release of these C pools was possibly attributable to a higher decomposition rate of dead organic matter at the burned sites relative to that at the unburned site (Dore *et al.*).

The temporal dynamic of soil organic C pool can be more complicated than that of biomass, with a result of greater uncertainty. Soil organic C pools increased in the unburned and low-severity stands but decreased in the replanted high-severity stands during the two field investigations. The decline in soil organic C storage might have been caused by the low C input from the young stands and comparably high output of surface soil in the high-severity stands (McHugh *et al.* 2003), which concurs with several studies of mineral soil C dynamics in coniferous plantations (Guo and Gifford 2002; Wang *et al.* 2006). However, the slight decrease in surface soil organic C pool over the first 14 years of replantation is just a transient phenomenon. With increasing biomass input and potential input of inherent high detritus C in the post-fire forests in the region, C accumulation of soil will occur (Guo and Gifford 2002; Post and Kwon 2000; Scott *et al.* 1999).

CONCLUSION

The studied boreal *L. gmelinii* forest in northeastern China was a moderate C sink (105 and 190 g C m^{-2} $year^{-1}$ for LG and LR, respectively) in the last two decades. The vegetation growth contributed 70% of this C sink at the LG site and 83% at the LR site. Soil and detritus represented more than 30 g C m^{-2} $year^{-1}$ of the

C sink. The forest ecosystem after a low-severity fire disturbance changed to C-neutral during a relatively short restoration. Fire changed ecosystem C allocation but increased vegetation and decreased detritus C density during the two investigations, suggesting that the post-fire forest ecosystem is restoring its C balance. Effective reforestation practice accelerates forest C sequestration.

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