



Effect of prescribed burning on the small-scale spatial heterogeneity of soil microbial biomass in *Pinus koraiensis* and *Quercus mongolica* forests of China

Xu Dou¹ · Hongzhou Yu¹ · Jianyu Wang¹ · Fei Li¹ · Qi Liu¹ · Long Sun¹ · Tongxin Hu¹

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Abstract Prescribed burning can alter soil microbial activity and spatially redistribute soil nutrient elements. However, no systematic, in-depth studies have investigated the impact of prescribed burning on the spatial patterns of soil microbial biomass in temperate forest ecosystems in Northeast China. The present study investigated the impacts of prescribed burning on the small-scale spatial heterogeneity of microbial biomass carbon (MBC) and microbial biomass nitrogen (MBN) in the upper (0–10 cm) and lower (10–20 cm) soil layers in *Pinus koraiensis* and *Quercus mongolica* forests and explored the factors that influence spatial variations of these variables after prescribed burning. Our results showed that, MBC declined by approximately 30%

in the 10–20 cm soil layer in the *Q. mongolica* forest, where there were no significant effects on the soil MBC and MBN contents of the *P. koraiensis* forest ($p > 0.05$) after prescribed burning. Compared to the MBC of the *Q. mongolica* forest before the prescribed burn, MBC spatial dependence in the upper and lower soil layers was approximately 7% and 2% higher, respectively. After the prescribed burn, MBN spatial dependence in the upper and lower soil layers in the *P. koraiensis* forest was approximately 1% and 13% lower, respectively, than that before the burn, and the MBC spatial variability in the 0–10 cm soil layer in the two forest types was explained by the soil moisture content (SMC), whereas the MBN spatial variability in the 0–10 cm soil layer in the two forests was explained by the soil pH and nitrate nitrogen (NO_3^- -N), respectively. In the lower soil layer (10–20 cm) of the *Q. mongolica* forest, elevation and ammonium nitrogen (NH_4^+ -N) were the main factors affecting the spatial variability of MBC and MBN, respectively. In the 10–20 cm soil layer of the *P. koraiensis* forest, NO_3^- -N and slope were the main factors affecting the spatial variability of MBC and MBN, respectively, after the burn. The spatial distributions of MBC and MBN in the two forests were largely structured with higher spatial autocorrelation (relative structural variance $C/[C_0 + C] > 0.75$). However, the factors influencing the spatial variability of MBC and MBN in the two forest types were not consistent between the upper and lower soil layers with prescribed burning. These findings have important implications for developing sustainable management and conservation policies for forest ecosystems.

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✉ Long Sun
sunlong@365.com

✉ Tongxin Hu
htxhtx@nefu.edu.cn

¹ Key Laboratory of Sustainable Forest Ecosystem Management – Ministry of Education of P. R. China, College of Forestry, Northeast Forestry University, 26 Hexing Road, Harbin 150040, People's Republic of China

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Abbreviations

MBC	Microbial biomass carbon
MBN	Microbial biomass nitrogen
SMC	Soil moisture content
FT	Forest type
SL	Soil layer
FT × SL	Interaction of forest type and soil layer
DOC	Dissolved organic carbon
DON	Dissolved organic nitrogen
ST	Soil temperature
GPS	Global positioning system
BRT	Boosted regression tree

Introduction

In areas prone to forest fires, prescribed burning is a common forest tool (Burrows and McCaw 2013) and is widely used worldwide to reduce the load of combustibles. Prescribed burning may affect forest soil nutrient elements and soil microorganisms through direct and indirect effects, further affecting forest productivity and changes in soil carbon (C) and (N) storage. Generally, frequent burning will reduce the soil N pool (Harden et al. 2002), decrease the rate of N mineralization (Reich et al. 2001), and lead to a decrease in N supply, in turn reducing the rate of decomposition of combustibles in the forest (Hernández and Hobbie 2008), which may lead to a shortening of the fire cycle (Kasischke and Turetsky 2006).

Resource heterogeneity is widely accepted to be important in driving ecosystem productivity and stability (Eldridge et al. 2020). However, less is known about the spatially heterogeneous changes in microbes across environmental gradients and their drivers, thus limiting our understanding of the potential impact of these changes on ecosystem function. Soil microbial biomass carbon (MBC) and microbial biomass nitrogen (MBN) account for only 1.92% of soil organic C pools and 3.43% of soil N pools, respectively (Zhou and Wang 2015), but are important for the decomposition of forest litter, formation and maintenance of soil fertility, and nutrient cycling (Schloter et al. 2003; Green et al. 2004). Forest fires influence soil microbial communities (Choromanska and DeLuca 2001; Mabuhay et al. 2003; Rutigliano et al. 2007) by modifying soil physical, chemical, and biological properties (González-Pérez et al. 2004); however, the effects of prescribed burning on soil microbes under specific environmental conditions remain unclear (Vega et al. 2013; Volkova et al. 2014; Akburak et al. 2018). Soil microorganisms are more sensitive than soil physical and chemical properties to environmental disturbances. High temperatures kill microorganisms by increasing soil temperature (Cairney and Bastias 2007) and alter microbial biomass with long-term effects on the soil microenvironment (Williams et al.

2012; Hu et al. 2021) via changes in temperature (Xu et al. 2013), moisture (González-Pérez et al. 2004), and soil nutrient characteristics (such as inorganic N, dissolved organic carbon (DOC), dissolved organic nitrogen (DON), and pH) (Hu et al. 2019). The availability of organic matter is a major factor influencing microbial activity (Scott-Denton et al. 2006), and prescribed burning could reduce substrate C input in soil by decreasing forest litter (Keith 1991) and affect microbial biomass indirectly by reducing the available nutrients in the short term (Ponder Jr et al. 2009). Some studies have also suggested that prescribed burning alters plant species and soil microbial community structure (Bastias et al. 2006) and might change the rate of mineralization and the activity of nitrifying microbial communities (Long et al. 2014; Hanan et al. 2017).

Spatial heterogeneity is an inherent property of soil, thereby contributing several important functional implications. Considering that the spatial variability of underground soil ecosystems is high and that soil microbes are very sensitive to changes in the external environment, soil microorganisms must be quantified at different depths (Outeiro et al. 2008). Even under similar site conditions, MBC and MBN in forest soils under different tree species can be quite different (Liu and Wang 2010). Spatial variation and heterogeneity of soil MBC and MBN are critical for understanding the structure and function of soil and its productivity. The scales of environmental changes and microbial biomass also vary greatly, from tens to thousands of kilometers to several meters or even microns (Vos et al. 2013). Studies have also described complex forest fire patterns at global and landscape scales (Krawchuk et al. 2009) so that the heterogeneity caused by fire at large scales is largely understood. At small spatial scales, the environmental changes are relatively poorly understood, especially after a prescribed burn. Several studies have reported that prescribed burning has uneven effects on the landscape, with large variations even at small scales (Robichaud and Miller 1999). Therefore, spatial variation in soil microbial biomass at a small-scale should not be overlooked.

In farmland and savanna grassland ecosystems, the spatial dependence of microbial biomass at landscape levels is at scales < 1 m (Robertson et al. 1997). The spatial patterns of microbial biomass in forests rich in organic matter are quite different from the patterns in farmland ecosystems which have lower concentrations of organic matter. For example, the spatial variation pattern of microbial biomass in mixed forests is greater than 8 m (Saetre and Bååth 2000). Jin et al. (2009) demonstrated that microbial biomass had strong spatial heterogeneity over a small scale on karst slopes in southwestern China, which was caused by a combination of historical evolutionary events (distance separation, physical barriers, diffusion history, and past environmental heterogeneity) and contemporary environmental factors (such as vegetation, climate,

topography, soil, and human factors). Although previous studies have focused on the spatial patterns of soil properties (such as inorganic N and cations) after prescribed burning (Outeiro et al. 2008; Rodríguez et al. 2009), no in-depth studies have focused on the spatial patterns of soil microbes after a prescribed burn. Soil microbes drive soil organic matter and soil nutrient transformation and cycling and participate in the decomposition of organic matter, the formation of humus, and the regulation of soil energy and nutrient cycling, which are of great significance to the structure and function of forests (Hicke et al. 2012). Therefore, an in-depth understanding of the spatial distribution patterns of soil microbial biomass and their regulation after prescribed burns in forest ecosystems will advance our understanding of the effects of prescribed burning on these ecosystems.

The aim of the present study was to determine the spatial heterogeneity of microbial biomass in *P. koraiensis* and *Q. mongolica* forests and the factors influencing this spatial heterogeneity after prescribed burning. We expected that prescribed burning would increase the spatial heterogeneity of microbial biomass in forest soils in Northeast China. We also hypothesized that soil physical and chemical properties would be the major factors regulating the spatial variability of soil microbial biomass after prescribed burning. To test our hypotheses, we subjected temperate coniferous and broad-leaved forests in Northeast China to a prescribed burn, then used geostatistical methods to explore the spatial variability of soil microbial biomass.

Materials and methods

Site description

The research area is located in a low hilly area in the Hongqi Forest Farm, Hegang City, northeastern Heilongjiang Province, China (129° 39' 00"–32° 31' 00" E, 47° 03' 30"–48° 21' 00" N). The slope in the industrial zone is gentle, mostly between 7° and 22° range. The area had an average elevation of 220 m a.s.l. The climate is northern temperate continental monsoon, with strong winds in spring, low rainfall, and widespread drought. The annual average temperature is 2.9 °C; the frost-free period is 105 days, and the annual average precipitation is 640 mm. The soils are classified as Alfisols. The dominant tree species include *Pinus koraiensis* Sieb. et Zucc., *Quercus mongolica* Fisch. ex Ledeb., *Larix gmelinii* (Rupr.) Kuzen.,

Pinus sylvestris var. *mongolica* Litv., and *Abies fabri* (Mast.) Craib. The shrubs in the study area are dominated by *Lespedeza bicolor* Turcz. and include *Carex siderosticta* Hance., *Convallaria majalis* Linn., *Thalictrum aquilegifolium* var. *sibiricum* Linnaeus., *Stellaria radicans* Linn., *Viola acuminata* Ledeb., and *Doellingeria scaber* (Thunb.) Nees. The basic information of the survey site is shown in Table 1.

Experimental design

The study plots were selected based on forests with contrasting species, including *P. koraiensis* and *Q. mongolica* forests. In mid-October 2018, a 20 m × 30 m plot was demarcated in each of the two forest types before prescribed burning. Forty-five sample points were arranged to meet requirements for statistical analysis in each plot, with a minimum of 0.5 m and maximum of 36.06 m between the 45 sampling points. In addition, the soils were sampled in the same places before and after the burn (Fig. 1). According to the calculation requirements of the geostatistics method, sample plot distributions were determined (Fig. 2).

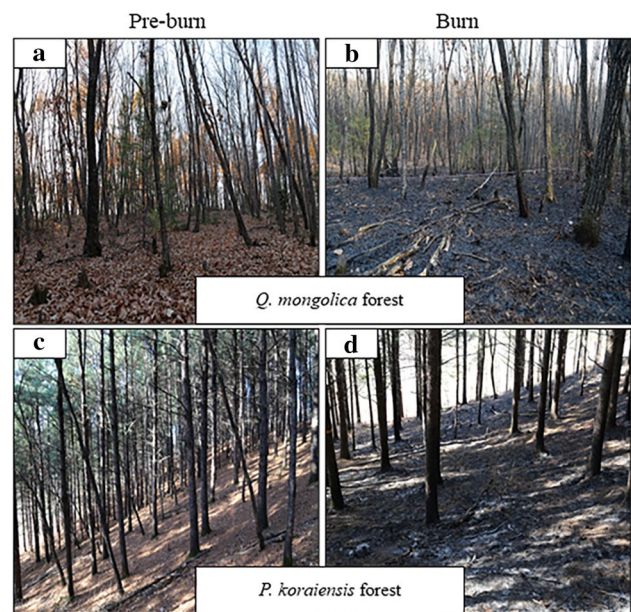


Fig. 1 Plot conditions before and after the prescribed burn in the *Quercus* and *Pinus* forests in Northeast China

Table 1 Basic data for topography and *Quercus* and *Pinus* species at the experimental sites in Northeast China

Forest type	DBH (cm)	MTH (m)	AS (year)	Elevation (m a.s.l.)	Slope (°)	OML (cm)	SD (%)
<i>Q. mongolica</i>	12.3	8.5	34.0	273.6	17	5.3	50.5
<i>P. koraiensis</i>	26.0	11.2	59.0	214.0	21	2.6	75.3

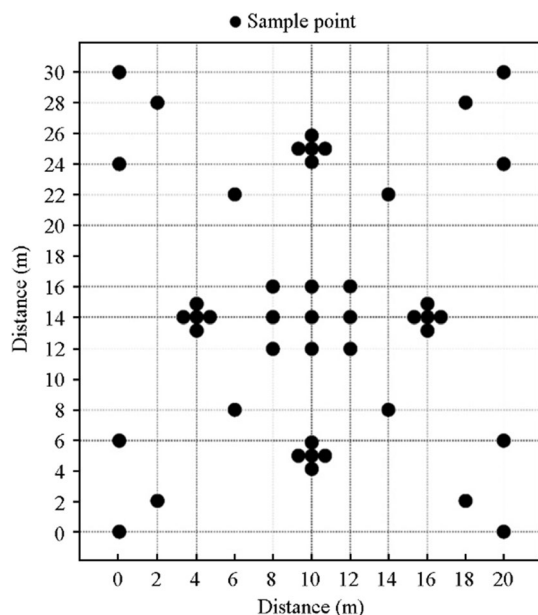


Fig. 2 Soil sampling design

Fire behavior variables

Table 2 lists the weather conditions, headfire rate of spread, fuel load and fire intensity before the two forest types were burned. Before the burn, we harvested the aboveground fuel load in a quadrat (1 m × 1 m) near each sampling point and brought it to the laboratory. Dry mass was determined after oven-drying at 65 °C to a constant mass, and the total aboveground fuel load of the entire sample site was calculated. The headfire rate of spread was directly measured and recorded during the prescribed burn. The fire intensity was calculated using the formula of Byram (1959):

$$I = H\omega R, \quad (1)$$

where I is fire intensity (kW m^{-1}); H is heat of combustion (kJ kg^{-1}); ω is mass of fuel load per unit area (kg m^{-2}), and R is headfire rate of spread (m s^{-1}), standardized to 0.5 kJ kg^{-1} .

Table 2 Weather conditions and fire behavior variables at the time of the prescribed burns of *Pinus koraiensis* and *Quercus mongolica* forests

Forest type	Maximum temperature (°C)	Minimum RH (%)	Wind speed (km h^{-1})	Fuel load (t ha^{-1})	Headfire rate of spread (m h^{-1})	Intensity (kW m^{-1})
<i>Q. mongolica</i>	16	50	1.15	5.47	100	273.50
<i>P. koraiensis</i>	15	54	0.70	8.69	150	651.75

RH relative humidity

Sampling and soil analysis

We used a handheld global positioning system (GPS) receiver on the ground to determine the elevation of each sample point. Slope degree was measured with a clinometer. We used a soil auger (inner diameter 70 mm) to collect 45 pre-burn and 45 after-burn samples at soil depths of 0–10 cm and 10–20 cm in the two forest types. Litter and humus were removed on the soil surface, and the samples were transported to the laboratory in coolers inside polyethylene bags. The soil samples were divided into two parts: one part was stored at 4 °C for later analysis of soil MBC, MBN, inorganic N (ammonium N [$\text{NH}_4^+\text{-N}$] and nitrate N [$\text{NO}_3^-\text{-N}$]) and soil moisture content (SMC). SMC was calculated using the formula:

$$\text{SMC} = (W_1 - W_2) / W_1, \quad (2)$$

where W_1 is initial mass of sample (g); W_2 is mass of dried sample after reaching constant mass (g).

The other sample part was air dried naturally, ground fully, and then passed through a 2 mm sieve to measure soil pH. For inorganic N measurements, burned and unburned soil samples were extracted using 2 mol L^{-1} KCl (Liu et al. 1996). Concentrations of inorganic N ($\text{NH}_4^+\text{-N}$ and $\text{NO}_3^-\text{-N}$) in the filtered extracts were determined using No. 1 filter papers (Whatman International Ltd, Maidston, Kent, UK) and a flow injection autoanalyser (BRAN + LUEBBE-AA3, Germany). Soil pH was determined using the glass electrode method (PHS-3B) in water (water: soil = 2.5:1 volume ratio: mass). SMC was determined by oven drying soil at 105 °C for 24 h to a constant mass. Soil temperature (ST) was measured using a temperature probe (p/n 8100–201) in the field. MBC and MBN were determined using the chloroform-extraction method (Brookes et al. 1985). Fresh soil (5 g) was extracted with 0.5 mol L^{-1} K_2SO_4 (water to soil ratio of 1:4) and shaken at 200 rpm for 1 h. Fresh soil (5 g) was weighed and fumigated in the dark at 25 °C for 24 h. DOC and total dissolved nitrogen (TDN) concentrations were analyzed in both fumigated and unfumigated extracts using a Multi C/N analyser (Multi C/N 3000, Analytik Jena, Germany). Prescribed burning was carried out in early November, and samples were collected again 24 h after prescribed burning. The collection procedure was similar to that applied before prescribed burning.

MBC and MBN were calculated as:

$$\text{MBC} = E_C/0.45, \quad (3)$$

$$\text{MBN} = E_N/0.54, \quad (4)$$

where E_C is the difference between the organic C extracted from the fumigated soil and the organic C extracted from the unfumigated soil, E_N is the difference between the total N extracted from fumigated soil and total N extracted from unfumigated soil, and 0.45 and 0.54 are respective conversion coefficients. Dissolved organic nitrogen (DON) was calculated as:

$$\text{DON} = \text{TDN} - (\text{NH}_4^+ + \text{NO}_3^-) \quad (5)$$

where TDN is total dissolved nitrogen content (mg kg^{-1}); NH_4^+ is ammonium nitrogen content ($\text{NH}_4^+\text{-N}$ [mg kg^{-1}]); NO_3^- is nitrate N content ($\text{NO}_3^-\text{-N}$ [mg kg^{-1}]).

Statistical analyses

Descriptive statistical analyses

We used the pastecs package in R 3.5.3 (R Core Team 2019) to descriptively analyze the data and obtain the coefficient of variation (CV) for each factor before and after the prescribed burn (Grosjean et al. 2014). To avoid any proportional effect and to ensure that the data were normally distributed during the geostatistical analysis, the data were tested for normality using the Shapiro–Wilk test (Butler et al. 2017), based on the statistic W (Shapiro and Wilk 1965):

$$W = \frac{(\sum_{i=1}^n a_i x_{(i)})^2}{\sum_{i=1}^n (X_i - \bar{X})^2}, \quad (6)$$

where $X_{(1)} \leq X_{(2)} \leq \dots \leq X_{(n)}$ are the ordered values of the sample and a_i are tabulated constants. When the p value is significantly greater than 0.05 and the W value was close to 1, the data were considered normally distributed.

Before performing geostatistical analyses, we log-transformed the data to eliminate any proportional effects on the original data that did not satisfy the normal distribution.

Differences in soil and soil properties were determined using a one-way analysis of variance (ANOVA). The least significant difference (LSD) post hoc test was used for comparisons between the means of multiple groups. A two-way ANOVA was performed to examine the effects of forest type and soil layer on soil MBC and MBN before and after the prescribed burn.

Geostatistical analyses

In soil spatial variability research, geostatistical analysis has been demonstrated to be the optimal method for quantifying spatial distribution characteristics and variations in soil properties (Deutsch and Journel 1992; Goovaerts 2001; Wang et al. 2002). Geostatistics aims to reveal and simulate spatial heterogeneity using a range of deterministic and probabilistic tools (Deutsch and Journel 1992). We used GS + V9 from Gamma Design Software in 2008 (Hu et al. 2016), which is a geostatistical analysis software for soil indicator data to perform semivariogram analysis and model fitting optimization. The choice of the model depends on the fitting parameters of the theoretical model of the variogram, and the best-fit model is selected based on the principle of minimum residual sum of squares (RSS) and maximum coefficient of determination (R^2) (Hu et al. 2016).

The semivariogram $\gamma(h)$ describes the spatial structure of the data and is calculated as:

$$\gamma(h) = \frac{1}{2N(h)} \sum_{i=1}^{N(h)} [Z(x_i) - Z(x_i + h)]^2, \quad (7)$$

where $\gamma(h)$ is the total number of sample couples for the lag interval distance h ; $N(h)$ is the number of observation pairs separated by the distance h ; $Z(x_i)$ is the measured sample value at point x_i ; and, $Z(x_i + h)$ is the measured sample value at point $(x_i + h)$.

The spherical, exponential, Gaussian and linear functions were fitted as follows:

$$\text{Spherical function } \gamma(h) = \begin{cases} 0, & h = 0 \\ C_0 + C \left(\frac{3}{2} \frac{h}{a} - \frac{1}{2} \frac{h^3}{a^3} \right), & 0 < h \leq a \\ C_0 + C, & h > a \end{cases} \quad (8)$$

$$\text{Exponential function } \gamma(h) = \begin{cases} 0, & h = 0 \\ C_0 + C \left(1 - e^{-\frac{h}{a}} \right), & h > 0 \end{cases} \quad (9)$$

$$\text{Gaussian function } \gamma(h) = \begin{cases} 0, & h = 0 \\ C_0 + C \left(1 - e^{-\frac{h^2}{a^2}} \right), & h > 0 \end{cases} \quad (10)$$

$$\text{Linear function } \gamma(h) = \begin{cases} C_0, & h = 0 \\ Ah, & h > 0, \end{cases} \quad (11)$$

where h is the average separation distance between observation pairs; C_0 is the value of the nugget; C is the value of structural variance; $C_0 + C$ is the sill value; and, a is the range.

In a semivariogram $\gamma(h)$, when $h=0$, the variogram obtains the value of the nugget constant C_0 , which represents the variation or measurement error caused by the random part. The sill value ($C_0 + C$) indicates the spatial heterogeneity of the observed variables in the system region, that is, the larger the abutment value is, the higher the spatial heterogeneity of the observed variables in the system region (Hu et al. 2016). The relative structural variance $C/(C_0 + C)$ was used to estimate the degree of spatial autocorrelation (magnitude of spatial heterogeneity). Generally, 0–25% indicates weak spatial autocorrelation. In addition, $25\% < C/[C_0 + C] < 75\%$ indicates domination by structural factors and random factors, which are moderately correlated, and $> 75\%$ indicates strong autocorrelation. The range is the spatial distance at which the semivariogram reaches a plateau. When it exceeds the separation distance (h), it will no longer have a spatial correlation.

We used the Surfer 16.0 (Golden Software Inc., Golden, CO, USA) kriging mapping method to show the spatial heterogeneity of the variogram description and the variable spatial distribution patterns from a two-dimensional perspective (Olea 1974).

Boosted regression tree analysis

Boosted regression tree (BRT) analysis is based on classification and regression tree analysis. As a self-learning algorithm, the method generates multiple regression trees through random selection and autonomous learning methods to improve model stability and prediction accuracy (Elith

et al. 2008). The influence of the independent variable on the dependent variable can be reflected in a horizontal bar graph. A higher bar indicates a stronger influence of the dependent variable on the independent variable. Logarithmic transformation of data in fitted BRT, using elevation, slope, ST, SMC, $\text{NH}_4^+\text{-N}$, $\text{NO}_3^-\text{-N}$, DOC, and DON, as independent variables, and MBC and MBN as dependent variables. The BRT analysis was performed using the *bgm* package (Cai et al. 2013) in R 3.5.3 (R Core Team 2019). The parameters were as follows: Gaussian error distribution, complexity 5, learning rate 0.001, and bag fraction 0.5.

Results

Characteristics of soil microbial biomass C and N and soil physical and chemical properties

There were differences in microbial biomass between different forest types before and after the burn (Table 3, Table S1). The MBC values of the lower soil layer in the *Q. mongolica* forest were reduced after the burn by approximately 30% compared to those before burning ($p < 0.05$), but prescribed burning had no significant effect on the upper and lower layer MBN concentrations in either the *Q. mongolica* or *P. koraiensis* forests ($p > 0.05$) (Table 3). After the burn, the contents of ST, $\text{NO}_3^-\text{-N}$ and DON in the upper layer of the *Q. mongolica* forest were lower than the level before prescribed burning. After the burn, the contents of ST, $\text{NH}_4^+\text{-N}$, $\text{NO}_3^-\text{-N}$, DOC and DON in the upper layer of the

Table 3 Descriptive statistics for MBC and MBN before and after the prescribed burn

Forest type	Soil layer (cm)	Treatment	Factors	Mean \pm SE (mg kg ⁻¹)	CV (%)	$C_0 + C$	$C/(C_0 + C)$	Range (m)	Model	R^2
<i>Q. mongolica</i>	0–10	Pre-burn	MBC	414.68 \pm 30.27 a	48.96	0.08	0.88	12.33	Gaussian	0.83
			MBN	43.88 \pm 4.69 a	71.66	0.28	0.94	2.70	Gaussian	0.47
	10–20		MBC	501.86 \pm 34.10 a	45.58	0.09	0.83	11.22	Gaussian	0.40
			MBN	43.83 \pm 4.07 a	62.34	0.22	0.85	3.55	Gaussian	0.59
	0–10	After-burn	MBC	369.60 \pm 33.35 a	60.53	0.26	0.94	2.51	Spherical	0.12
			MBN	52.51 \pm 6.89 a	87.89	0.25	0.76	2.51	Spherical	0.13
10–20		MBC	355.82 \pm 37.95 b	71.54	0.26	0.85	52.71	Exponential	0.51	
		MBN	50.62 \pm 7.64 a	101.29	0.50	0.99	1.99	Gaussian	0.36	
<i>P. koraiensis</i>	0–10	Pre-burn	MBC	308.68 \pm 27.77 a	60.35	0.32	0.96	41.00	Spherical	0.71
			MBN	29.88 \pm 2.48 a	55.61	0.22	1.00	33.69	Exponential	0.81
	10–20		MBC	245.25 \pm 19.80 a	54.17	0.25	0.90	46.29	Exponential	0.84
			MBN	16.07 \pm 1.54 b	64.45	0.27	0.87	30.47	Gaussian	0.80
	0–10	After-burn	MBC	345.73 \pm 25.93 a	50.32	0.62	0.95	71.01	Gaussian	0.67
			MBN	27.72 \pm 1.93 a	46.82	0.02	0.88	6.97	Exponential	0.06
	10–20		MBC	297.93 \pm 28.16 a	63.40	0.34	0.78	123.00	Exponential	0.51
			MBN	24.02 \pm 2.39 a	66.75	0.21	0.99	3.01	Spherical	0.67

SE standard error. Different lowercase letters after values indicate a significant differences in plots before and after the burn for the same forest type and soil layer

P. koraiensis forest were lower than that before the burn (Table S2). The CV value of MBN in the 10–20 cm soil layer of the *Q. mongolica* forest showed strong variability after the burn (CV > 100%). The rest of the CV values of the MBC and MBN of the *Q. mongolica* and *P. koraiensis* forests before and after prescribed burning showed a moderate variation intensity (10–100%) (Table 3).

Effects of prescribed burn on spatial variability and distribution of soil microbial biomass C and N

The spatial patterns of soil microbial biomass and physicochemical properties were significantly different in the *Q. mongolica* and *P. koraiensis* forests (Table 3, Table S2; Figs. 3, 4). MBC spatial dependence in the *Q. mongolica* forest after the burn was higher than that before the burn, and MBC spatial dependence in the *P. koraiensis* forest was lower than that before the burn. The spatial autocorrelation ranges of MBC and MBN of *Q. mongolica* forest before and after the burn were significantly lower than those of *P. koraiensis* forest (Fig. 4, Table 3). Both forest-type SMC and NO_3^- -N showed moderate spatial heterogeneity after the burn but showed strong spatial

heterogeneity before the burn. After the burn, the spatial dependence of DOC and DON in the upper soil of the *Q. mongolica* forest decreased, while the spatial dependence of DOC, DON and pH in the *P. koraiensis* forest increased.

Boosted regression tree (BRT) analysis for effects of prescribed burn on soil microbial biomass C and N

The burn altered the degree of influence of various factors on MBC and MBN (Figs. 5, 6, Figs. S1–S8). In the 0–10 cm soil layer, after the burn, SMC had the greatest impact on the spatial distribution of MBC in the two forest types, with effects of approximately 29% and 47%, respectively, and pH and NO_3^- -N were the main factors influencing the spatial distribution of MBN in both forests. In the 10–20 cm soil layer after the burn, elevation and NH_4^+ -N were the main influencing factors affecting the spatial distribution of MBC in *Q. mongolica* and *P. koraiensis* forests, respectively, and SMC and slope were the main influencing factors affecting the spatial distribution of MBN in *Q. mongolica* and *P. koraiensis* forests respectively.

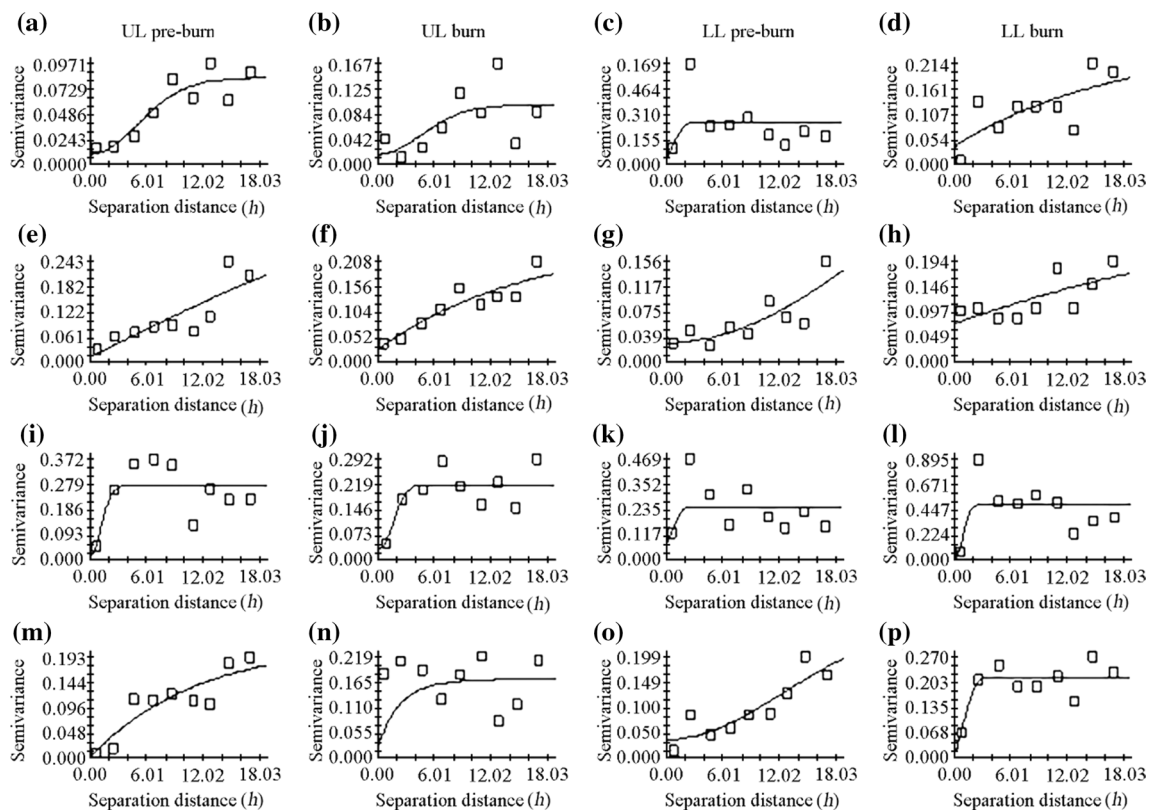


Fig. 3 Semivariance variograms of soil microbial biomass carbon (MBC) and soil microbial biomass nitrogen (MBN) in the upper (UL, 0–10 cm) and lower layers (LL, 10–20 cm) before and after prescribed burning in *Quercus mongolica* and *Pinus koraiensis* forests.

a–d Soil MBC in *Q. mongolica* forest; **e–h** soil MBC in *P. koraiensis* forest; **i–l** soil MBN in *Q. mongolica* forest; **m–p** soil MBN in *P. koraiensis* forest

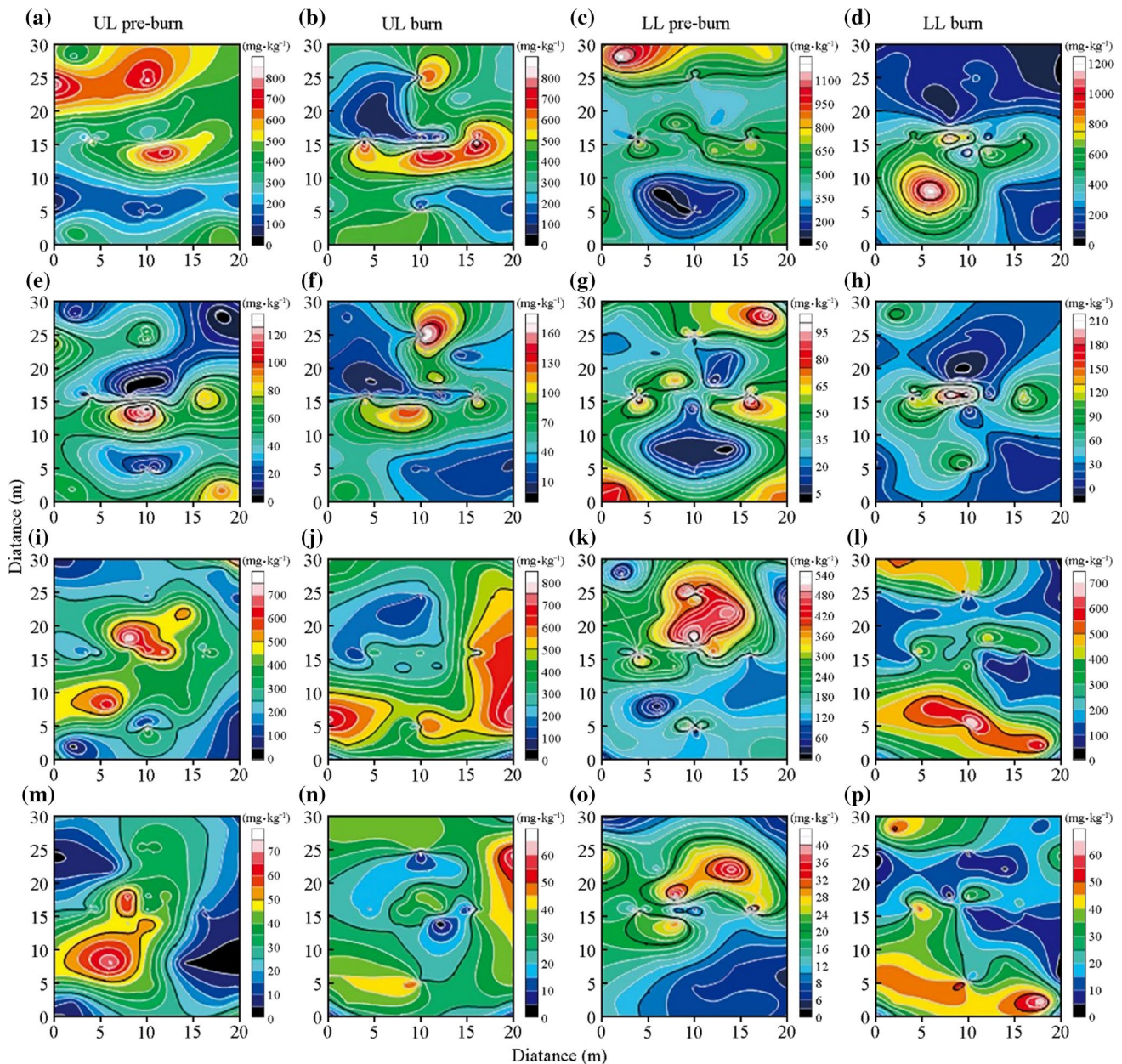


Fig. 4 Spatial distribution of microbial biomass carbon (MBC) and microbial biomass nitrogen (MBN) in the upper (UL, 0–10 cm) and lower soil layers (LL, 10–20 cm) before and after the prescribed burn

in *Quercus mongolica* and *Pinus koraiensis* forests. **a–d** Soil MBC of *Q. mongolica* forest; **e–h** soil MBN of *Q. mongolica* forest; **i–l** soil MBC in *P. koraiensis* forest; **m–p** soil MBN in *P. koraiensis* forest

Discussion

The effect of prescribed burning on soil physical and chemical properties

Although the prescribed burn changed the soil microbial biomass to a certain extent, it had no significant effect on the soil microbial biomass of the two forest types (except for the lower layer soil MBC in the *Q. mongolica* forest) (Table 3). However, there is currently no unanimous conclusion about

the impact of fire disturbance on soil microbial biomass (Grady and Hart 2006; Mabuhay et al. 2006). In a global meta-analysis of forest fires, Dooley and Treseder (2012) found that wildfires decreased microbial biomass by 33%, while prescribed burning had no significant impact on microbial biomass. High-severity forest fires could completely burn out the litter and organic matter of the forestland. The lack of substrate supply after a fire and the death of some mycorrhizal fungi cause the soil microbial biomass to decrease in the short term after a fire (Pietikäinen and

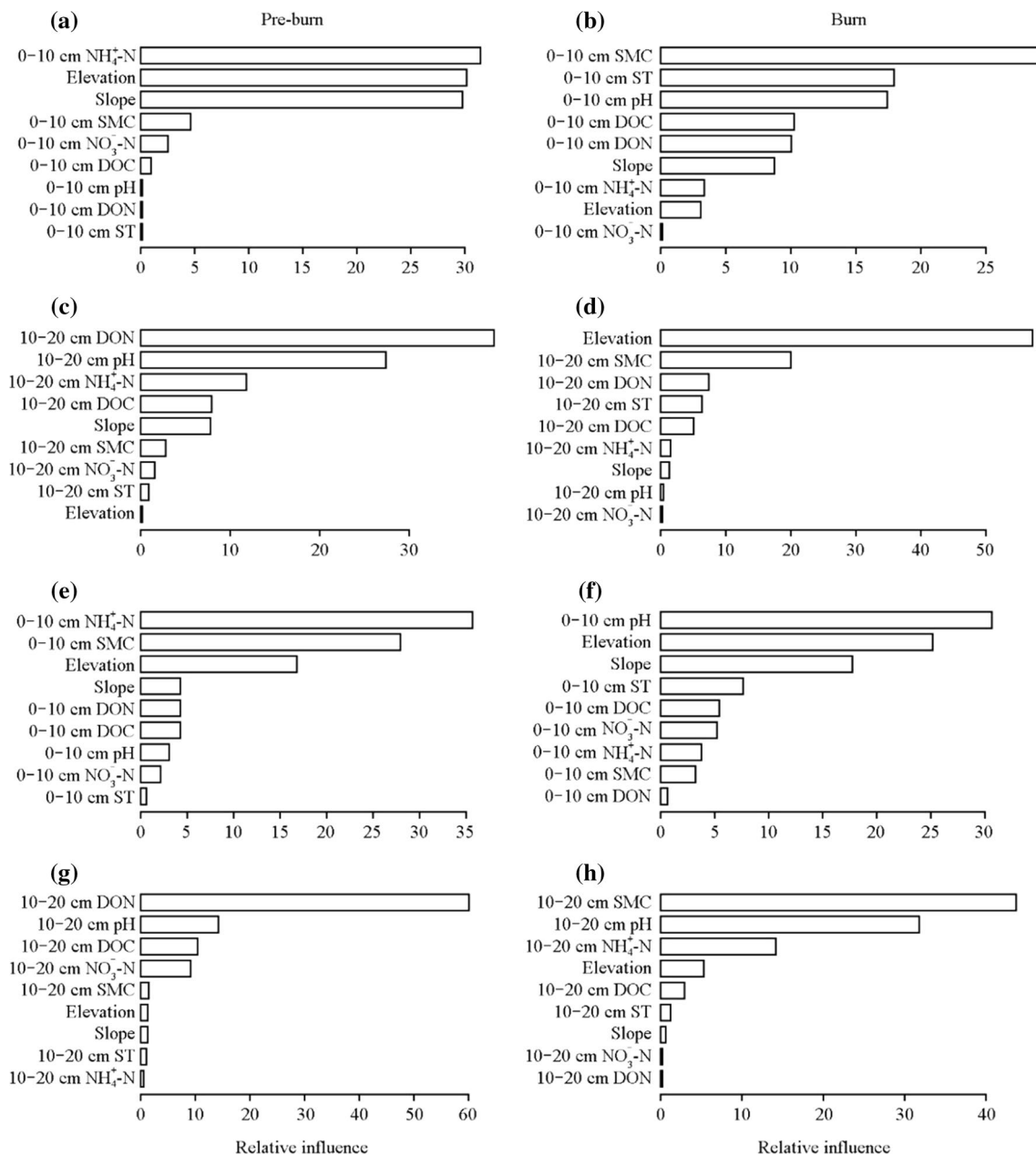


Fig. 5 Boosted regression tree (BRT) analysis of factors influencing the distribution of microbial biomass carbon (MBC) and microbial biomass nitrogen (MBN) in the upper (UL, 0–10 cm) and lower soil layers (LL, 10–20 cm) in the *Quercus mongolica* forest before and after the prescribed burn. **a, b** UL soil MBC of **a** before and **b** after

the burn; **c, d** LL soil MBC **c** before and **d** after the burn; **e, f** UL soil MBN **e** before and **f** after the burn; **g, h** LL soil MBN **g** before and **h** after the burn. DON=dissolved organic nitrogen, DOC=dissolved organic carbon, SMC=soil moisture content, ST=Soil temperature

Fritze 1995). In general, the effect of prescribed burning on microorganisms is mainly manifested in the organic matter layer, whereas no significant effect on the mineral layer is found because the burn does not transfer enough energy to the soil, leading to the direct restructuring of the microbial community (Gundale et al. 2005). In contrast, Zhou et al. (1999) reported that in grassland ecosystems, soil microbial biomass was the same as in the unburned land a short

time after the fire. The inconsistency between these results is mainly related to combustibility (Zhao et al. 2016). Compared with wildfires, the field temperature and surface litter combustion during prescribed burning are heterogeneous (Choromanska and DeLuca 2001), and the response of the microbial community is not obvious (Kara and Bolat 2009). Meanwhile, microorganisms automatically adjust their population balance in response to critical nutrients released after

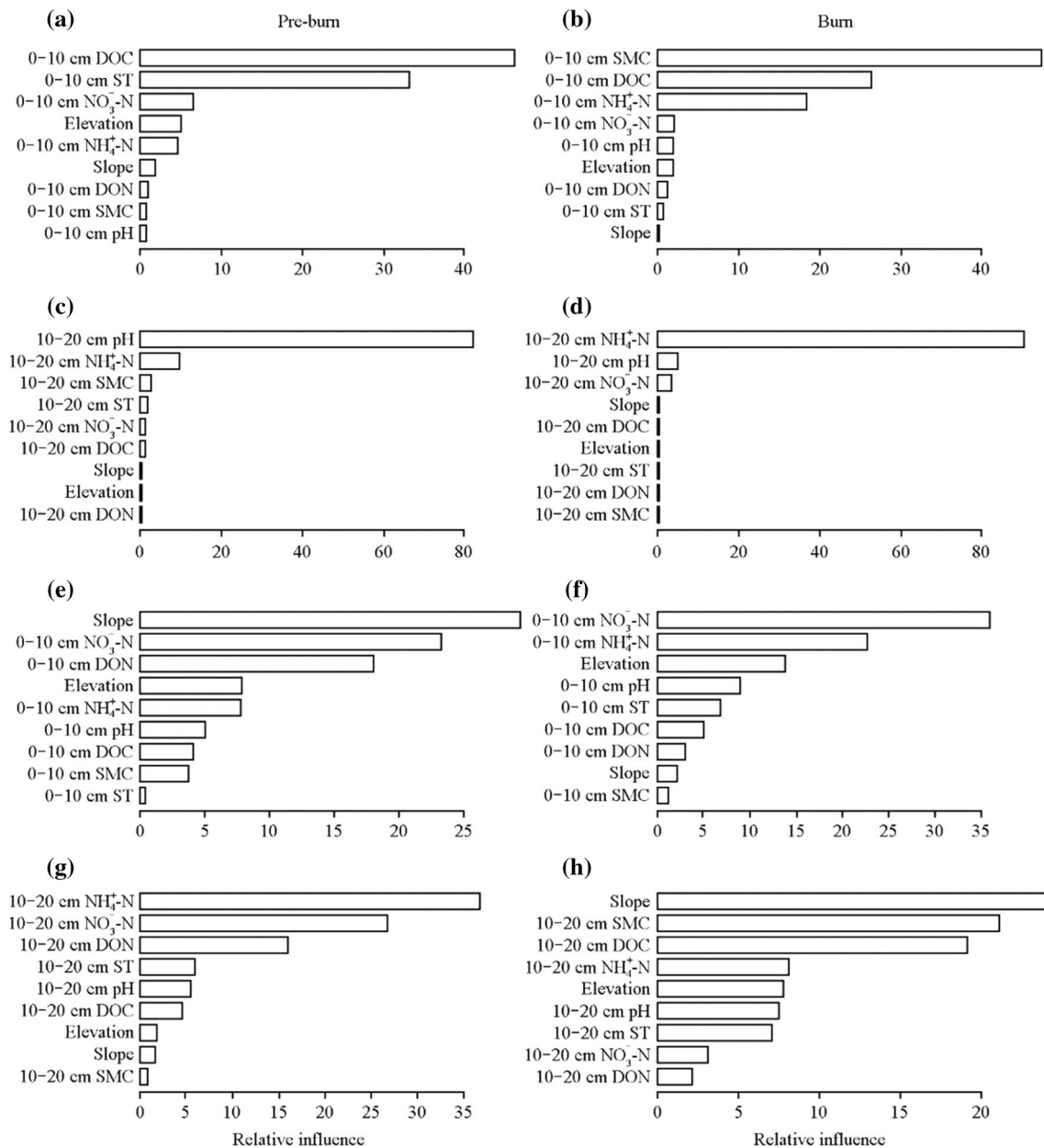


Fig. 6 Boosted regression tree (BRT) analysis of factors influencing microbial biomass C (MBC) and microbial biomass N (MBN) distribution before and after the prescribed burn in the upper (UL, 0–10 cm) and lower soil layers (LL, 10–20 cm) of the *Pinus koraiensis* forest. **a, b** UL soil MBC before and after the burn; **c, d** LL soil

MBC before and after the burn; **e, f** UL soil MBN before and after the burn; **g, h** LL soil MBN before and after the burn. DON=dissolved organic nitrogen, DOC=dissolved organic carbon, SMC=soil moisture content, ST=Soil temperature

the burn through metabolic regulatory mechanisms (Ilstedt et al. 2003).

The ST and SMC of the two forest types changed after the prescribed burn. Our study showed that compared with the conditions before prescribed burning, the ST in different soil layers of the two forest types decreased significantly after the burn, perhaps because the litter was completely burned, exposing the soil. In addition, the prescribed burn was executed in November, when the low air temperature

may have exacerbated the decrease in soil temperature. SMC is a useful indicator of the effect of forest fire on microbial activity disturbance and influences the depth to which heat or temperature is transferred to lower soil layers (Hobbie 1992). According to our results, the SMC of the two forest types increased after prescribed burning, which could be linked to the destruction of the hydrophobic layer in the soil by prescribed burning or to the clearance of plant and ground cover by prescribed burning, which facilitates faster

water penetration into the soil profile (Neary et al. 2005). Although fire will increase soil temperature and reduce vegetation cover (Russell et al. 2001), increasing rates of evaporation, the reduction in vegetation after fire will reduce evapotranspiration and more soil moisture will be retained (Moore and Keeley 2000). In addition, due to the complexity of hydrological effects, the changes in water content after prescribed burning are not consistent (Boisramé et al. 2018).

In the present study, prescribed burning significantly changed some soil chemistry characteristics, especially the upper soil layer (0–10 cm). In a Mediterranean pine plantation, prescribed burning did not cause major changes in specific mineral soil properties, and only changed the amount of litter cover (Meira-Castro et al. 2015). This result might be attributed to the tree species studied, which differed from our study. The average slopes of our two study plots were relatively high. Post-prescribed burning, the soil lost ground-cover protection, and soil nutrients were readily lost. In addition, there were still some differences in the changes in these soil properties in the two plots, which may be due to the differences in tree species in the two forest types, surface litter and load, and organic matter combustion conditions after the prescribed burn (Shibata et al. 2003). With the large-scale planting of plantations in the forest areas of Northeast China, the effect of forest fires in plantation forest ecosystems needs to be further studied. Due to the impact of uneven terrain in the forest area, it is also necessary to monitor the impact of prescribed burning in different forest types on the soil structure and physical and chemical properties of the forest ecosystems.

Effect of prescribed burning on spatial variability of soil biomass carbon and nitrogen

A previous study reported that microbial biomass had extremely high spatial variability at different scales, from point to global scales (Zhao et al. 2009). We compared the spatial effects of MBC and MBN combustion using experimental and theoretical variograms and found that the MBC and MBN contents of the upper and lower soils of the *Q. mongolica* and *P. koraiensis* forest before and after the prescribed burn had strong spatial heterogeneity ($C/[C_0 + C] > 0.75$). The increase in spatial heterogeneity could further influence the long-term succession of forest ecosystems by influencing the abundance of microbial communities in the ecosystem during the early stages of fire (Johnstone et al. 2008). In the boreal forest ecosystems of Alaska, as organic matter accumulates after a fire, the spatial heterogeneity of soil microbial biomass decreases over time (Lavoie and Mack 2012). However, exogenous factors may reduce or promote the spatial dependence of soil microorganisms (Yanai et al. 2003; Rezanezhad et al. 2019). We found that the MBC spatial dependence of the *Q. mongolica* forest increased, and the MBC spatial dependence

of the *P. koraiensis* forest decreased after the prescribed burn, perhaps caused by the different vegetation types and prescribed burning intensity. Previous studies have shown that microbial activity has a strong spatial autocorrelation at a small sampling scale (< 0.25 m) (Stark et al. 2004). However, we found that the minimum spatial autocorrelation distance of the MBC and MBN after prescribed burning was 0.94–123.00 m, which indicated that the unevenness of the severity of the prescribed burning within the forest caused large differences in soil microbial spatial variation.

The distribution of soil microbial biomass is associated with various physical and chemical characteristics of forest soils at different spatial locations (Sun et al. 2011). This relationship causes environmental factors to continuously modify the spatial distribution patterns of microorganisms at small spatial scales (Zhang et al. 2007). Soil moisture is a key factor in determining microbial activity. After prescribed burning, SMC was the most important factor affecting the spatial distribution of MBC in the upper soil layer of the *Q. mongolica* and *P. koraiensis* forests (Figs. 5b, 6b). In China's semiarid grassland ecosystem, affected by soil moisture content, the MBN downhill was 36.7% higher than uphill, and the temporal variability of soil microorganisms increased under fire disturbance (Liu et al. 2010). After the prescribed burn, SMC increased, the influence of spatial changes caused by infiltration and runoff gradually decreased, and the spatial heterogeneity of SMC decreased. The survival of microorganisms depends on the appropriate moisture content. Obviously, changes in SMC can directly or indirectly affect the spatial changes in MBC, which helps explain the mechanism underlying the spatial heterogeneity of MBC in forest ecosystems (Qian et al. 2009).

Soil pH is among the most influential factors of community composition of microbial ecosystems (Shen et al. 2013). Bru et al. (2010) found that soil pH alone explained 17.1% and 21.4% of the spatial variability of the total bacterial community and functional microbial community involved in the N cycle, respectively. We found that the soil pH after the burn was the main factor affecting the spatial distribution pattern of the upper soil layer MBN of the *Q. mongolica* forest. Although there was no significant change in soil pH before and after the burn, spatial heterogeneity increased after the burn, which might strongly alter the spatial distribution pattern of soil MBN after a prescribed burning.

Nitrogen plays an important role in the ecology of the microbial community. We found that NH_4^+ -N had an important effect on the spatial distribution of MBC and MBN in the two forest types (Figs. 5a, b, 6d, g). NH_4^+ -N is an important N source for microorganisms; thus, a change in its content inevitably affects the growth and reproduction of some microorganisms that sensitive to NH_4^+ -N (Ahn et al. 2012). Zhou et al. (2015) found that an increase in microbial abundance was related to an increase in NH_4^+ -N in the

soil. Although our study only measured microbial biomass, our results clearly show the close relationship between soil microbes and $\text{NH}_4^+\text{-N}$ in forest ecosystems.

The identified factors affecting the spatial heterogeneity of soil microbial biomass in the *Q. mongolica* and *P. koraiensis* forests under prescribed burning may support our hypothesis 2, that the part of the physical and chemical properties of soil might be the main factors leading to the spatial variability of microbial biomass. We can thus infer that the changes in soil physical and chemical properties after the prescribed burn were an important reason for the spatial variability of soil microorganisms in the two forest types. We should note that the results of our study cannot be used to show the seasonal variation in soil microbial biomass after a prescribed burn based on the short-term effects of prescribed burning on soil microbial biomass and soil physicochemical properties. Because microorganisms play an important role in global biogeochemical cycles, the characteristics of the microbes in different fire interference backgrounds are receiving increasing attention in efforts to understand global environmental changes and the generation of greenhouse gases. However, the spatial heterogeneity of microbial characteristics is very high and hard to parameterize accurately. Therefore, further investigation of the long-term spatial variability of soil microbial biomass after prescribed burning is required, which would offer a critical scientific basis for elucidating the impact of prescribed burns on temperate forests in Northeast China.

Conclusions

In this study of the spatial heterogeneity of soil MBC and MBN in temperate forests in Northeast China in response to prescribed burning, the spatial heterogeneity of MBC and MBN in the *Q. mongolica* forests differed from that in the *P. koraiensis* forests, but the most important factors affecting its spatial distribution were the physical and chemical properties of the soil. The results indicated that the factors driving for spatial distribution of soil microbial biomass on a small scale are more complex and that the impact of prescribed burns on typical temperate forests in Northeast China should not be ignored. Our findings will also enable a deeper understanding of forest ecosystem functioning and sustainability after such burns and contribute to the development of accurate models for large-scale prescribed burns.

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