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Soil organic carbon stocks and CO_2 effluxes of native and exotic pine plantations in subtropical China

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ABSTRACT

The selection of tree species is critical for soil organic carbon (SOC) stock and dynamics under afforestation. Exotic tree species were widely introduced and have been considered as a form of human-induced global change. However, how exotic tree species, as compared with native ones, affect SOC storage and soil respiration (R_s) has not been sufficiently investigated. Here we studied this issue in an even-aged (22 years) plantation composed by native Masson pine (*Pinus massoniana* Lamb.) (MP) and exotic slash pine (*Pinus elliottii* Engelm.) (SP) in subtropical China. A trenching method was used to partition R_s into heterotrophic (R_h) and autotrophic respiration (R_a). R_s and R_h were measured approximately biweekly during August 2007 to December 2009. Our results showed that seasonal patterns of R_s and R_h in both pine forests were mainly affected by soil temperature and moisture ($R^2 = 0.56$–0.78, $P < 0.001$). There was no difference in SOC stock (forest floor + mineral soil of 0–40 cm), R_h and turnover rate (R_h/SOC stock) between MP and SP forests. However, annual R_s, R_a and R_a/R_s of MP were significantly 24%, 119% and 74% higher than those of SP, respectively. In contrast, even though MP and SP had similar fine and coarse root biomass, MP had significant higher ratios of R_a/fine root biomass and R_a/coarse root biomass than SP, suggesting a higher respiration rate in native pine species on a root biomass basis. We concluded that the exotic SP, as compared with indigenous MP, did not alter SOC stock, R_h and turnover rate but depressed R_a and R_s. Therefore, both pine species were recommended in case of reforestation in terms of SOC sequestration.

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1. Introduction

Globally, forests cover 30% of the land area (FAO, 2006) and contain approximately 1146 Pg of carbon (C), of which over two thirds of the C is stored in the soils (Dixon et al., 1994). Subtropical forests are expected to be large potential C sinks due to their large area and high C sequestrations (Wen et al., 2010; Chen et al., 2011). In southern China, subtropical forests are widely distributed and currently encompass a total area of approximately 53 million ha (Y. Wang et al., 2011). Prior to this, the majority of natural evergreen broad-leaved forests were heavily destroyed by the late 1970s. Hereafter, reforestation campaigns were extensively launched to prevent environmental degradation. These forest plantations were widely established by conifers due to their characteristics of fast growth and pioneer in natural succession (Wang et al., 2012a).

Masson pine (*Pinus massoniana* Lamb.) (MP) is a dominant native species and a pioneer identity common in secondary forests. The MP plantations cover approximately 5.8 million ha (SFA, 2005). Even though the MP plantations had important ecosystem services, problems were also found, e.g. crooked stems, *Bursaphelenchus xylophilus* and

other insect pests (Wilson, 1993). Therefore, alternative tree species were also employed during the reforestation efforts. Slash pine (*Pinus elliottii* Engelm.) (SP), introduced from southeastern United States, has been extensively planted due to its high growth rate and resin production. The SP plantations covered more than 1 million ha in subtropical China. These exotic SP plantations should be concerned as a human-induced regional change. It is important to understand how the alien pine species influences soil ecosystems in subtropical China.

Tree species were known to greatly shape soil structure, processes and function (e.g. Binkley and Giardina, 1998; Hobbie et al., 2007). Soil organic C (SOC) storage in forest ecosystems was reported to be closely related to tree species identities (e.g. Ladegaard-Pedersen et al., 2005; Russell et al., 2007; Schulp et al., 2008). Even species within a functional group can differ in their short-term effects on SOC (Fisher, 1995; Russell et al., 2004). First, tree species generally have inherent characteristics in litterfall input, chemistry and decomposition (Hobbie et al., 2006; Russell et al., 2007; Vesterdal et al., 2012), which determines accumulation of forest floor C pool. Second, organic C depositions and re-formations in rhizosphere as well as their vertical distributions are also dependent on tree species (Binkley and Giardina, 1998; Valverde-Barrantes, 2007). As compared with native species, the exotic species may have much more potential to alter SOC stock and dynamics due to their intrinsic features such as growing rate,

biomass, C accumulation, resistance (Ma et al., 2014) and associated microorganism such as mycorrhizal fungi (Chapela et al., 2001). In support of this concept, several exotic tree species, as compared with native species, changed SOC stocks (Chapela et al., 2001; Zinn et al., 2002; Ehrenfeld, 2003; Ladegaard-Pedersen et al., 2005; Russell et al., 2007). However, no significant differences in SOC storage between the native and exotic species were also documented (Ehrenfeld, 2003; Ayres et al., 2009; Ma et al., 2014). Therefore, it is an important but unresolved issue that how exotic tree species affect SOC storage as compared with the native ones.

Soil respiration (R_s) is the main terminal output of SOC mineralization. According to the location of CO_2 production and the age of utilized C, R_s can be partitioned into heterotrophic (R_h) and autotrophic respiration (R_a), of which R_h is microbial respiration of root-free soil and R_a is the respiration of recent C from roots and microbes in rhizosphere (Kuzyakov, 2006; Subke et al., 2006). R_s and its components were controlled by a set of biotic and abiotic factors such as species-specific respiration (Rakonczay et al., 1997; Yang et al., 2012), substrate quality (Hobbie et al., 2006) and supply (e.g. Kuzyakov and Gavrichkova, 2010), soil temperature (T_s) and soil water content (SWC) (e.g. Y. Wang et al., 2011). Due to the differences in the biotic and abiotic factors among tree species identities, R_s and its components were found to differ among tree species (e.g. Metcalfe et al., 2011; Fender et al., 2013; Zeng et al., 2014). In contrast, some studies (Valverde-Barrantes, 2007; Vesterdal et al., 2012) demonstrated little difference in R_s among several tree species.

In subtropical China, exotic tree species were widely introduced and are increasingly becoming the focus of research. However, it is not clear how introduced species, as compared with native ones, influence SOC storage, CO_2 efflux and SOC turnover rate. The objective of this study was to compare (i) SOC stocks and turnover rates (R_h/SOC stock); (ii) temporal patterns of R_s and its components as well as their controls; and (iii) annual R_s and its components in relation to their sources (SOC and root) between the native MP and exotic SP plantations in subtropical China.

2. Materials and methods

2.1. Study site

This study was conducted in an even-aged pine plantation in the Qianyanzhou Ecological Research Station (26°44′N, 115°03′E) in subtropical China. The pine plantation was mainly established by a native MP (*P. massoniana*) and an exotic SP (*P. elliottii*) around 1985. The initial planting density of MP and SP was similar, with a value of 7500 stem ha⁻¹. Prior to this pine plantation, the vegetation was mainly shrubs and grasses. At present, the main midstory and understory species of MP and SP were *Quercus fabric* Hance, *Loropetalum chinense* (R. Br.) Oliver, *Dicranopteris dichotoma* (Thunb.) Bernh. and *Woodwardia japonica* (L. f.) Sm. Mean altitude of the study area is about 100 m, with a range of ±50 m. Slope angle is generally of 10–30°. The study region was characterized by a humid monsoon climate. The annual mean temperature is 17.9 °C, with a minimum daily mean temperature of 6.4 °C in January and a maximum of 28.8 °C in July (1985–2008). Annual total precipitation is 1469 mm year⁻¹ (1985–2008). Even though annual rainfall is high, wet (January–June) and dry (July–December) seasons alternate over an annual cycle (Y. Wang et al., 2013). Annual evapotranspiration was 747 mm year⁻¹ during the period of 2003–2007 (Wen et al., 2010). About 30% of rainfall and 54% of evapotranspiration occurred in the dry season during 2003–2010. The soil, weathered from red sandstone and mud stone, is classified to Typic Dystrudepts using the U.S. soil taxonomy. Further details can be found in Wang et al. (2014) and Xu et al. (2014).

2.2. Experimental design

There were three duplicate experimental plots in each tree species type (MP and SP). The locations of the six experimental plots are shown in Fig. 1. The background informations are documented in Table 1. The size of each experimental plot is 10 m × 10 m. In each experimental plot of MP and SP, 9 soil collars separated by approximately 2.4 m were placed on two diagonals of the each experimental plot for R_s measurements.

2.3. Measurement of SOC stock and other soil properties

In each experimental plot, soil bulk density was measured at 0–10, 10–20 and 20–40 cm using the conventional core method with a volume of 100 cm³. Mineral soils of 0–10, 10–20 and 20–40 cm depth were sampled at five points and then mixed into one sample in each plot in September 2007. The soil samples were air-dried and sieved with a 2 mm mesh screen to remove the roots and coarse gravel before chemical analysis. SOC content was determined by potassium dichromate reduction of organic C. Forest floor (organic layer) C pool was measured using a harvest method. SOC stock was calculated using Eq. (1). Soil total nitrogen was determined by the wet oxidation of soil organic matter using standard Kjeldahl procedure (UDK140 Automatic Steam Distilling Unit, Automatic Titroline 96, Italy) with sulphuric acid and digestion catalyst. Soil pH was measured with a pH meter (PHS-3C, Shanghai Dapu Instrument Co. Ltd., Shanghai) using a soil:water ratio of 1:5 (g g⁻¹).

$$SOC_{stock} = P_f + C_{SOC} \times B_d \times D \times (1-G) \times 10 \qquad (1)$$

where SOC_{stock} is the SOC stock (g C m⁻²), P_f indicates forest floor C pool (g C m⁻²), C_{SOC} refers to the SOC content of mineral soil (g kg⁻¹), B_d represents bulk density (g cm⁻³), D is the soil sampling depth (cm), and G is gravel content (%).
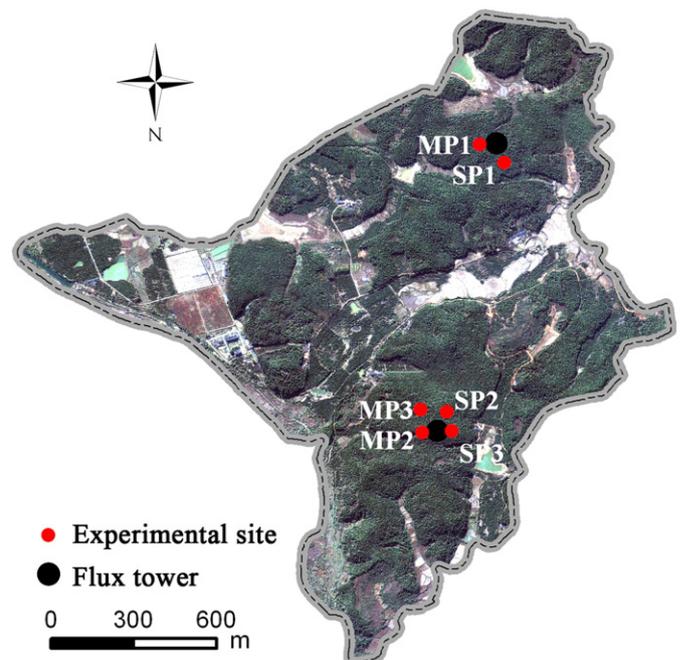
Fig. 1. The locations of experimental plots of the native Masson pine (MP) and exotic slash pine (SP) plantations.

Table 1
Background informations of the native and exotic pine plantations.

Forest type	Plot name	Slope position	Altitude (m)	Slope direction[a]	Basal area[b] (m^2 100 m^{-2})	Dominant species
MP	MP1	Lower	101	315°	0.35	*Pinus massoniana*
	MP2	Middle	110	118°	0.32	Lamb.
	MP3	Upper	111	10°	0.30	
SP	SP1	Upper	92	335°	0.38	*Pinus elliottii*
	SP2	Upper	114	270°	0.32	Engelm.
	SP3	Lower	109	180°	0.35	

Note: SP, slash pine; MP, Masson pine.
[a] The north is defined as 0° and the angle deasil increases.
[b] Total area of stem at breast height. Little trees and shrubs with diameter at breast height below 5 cm are not included.

2.4. Partition of soil respiration

A trenching method was employed to partition R_s into heterotrophic (R_h) and autotrophic respiration (R_a) (Kuzyakov, 2006) in MP1, MP2, SP1 and SP2 on March 19–20, 2008. Trenching treatment was carried out at 6 sampling points for R_h measurements in each experimental plot. To reduce trenching-induced impacts of soil disturbance and dead root decomposition, the measurements of R_h before August 1, 2008 (more than 4 months after trenching) were abandoned. Double thick plastic films were used to prevent root growth into the root-free soil cores in trenching treatments. The root-free soil cores are square areas with side of 0.8 m. The trench is 0.5 m in deep and 0.2 m in wide. After trenching, soil was backfilled into the trench.

2.5. Measurement of soil respiration and heterotrophic soil respiration

Soil collar, with a diameter of 10.6 cm and a height of approximately 6 cm, was inserted into the mineral soil to approximately 2 cm for CO_2 efflux measurement. We measured R_s and R_h using an automated soil CO_2 flux system (LI-8100, LI-COR Biosciences Inc., Lincoln, Nebraska, U.S.) equipped with a diameter of 10 cm survey chamber (Model 8100-102, LI-COR Biosciences Inc.). First flux measurement started one week after soil collar insertion in order to avoid disturbance-induced CO_2 emissions. Subsequently, CO_2 efflux measurements were carried out biweekly from August 2007 to December 2009. Each soil collar was measured 3 times at an interval of 1 min, with an observation time length of 2 min and a dead band of 30 s for each time. After a month, we found small standard errors of the triple measurements, and subsequently each soil collar was measured twice. Unfortunately, we could not measure the CO_2 flux of all experimental plots simultaneously due to the substantial field work. The measurements of CO_2 efflux of the 6 experimental plots were arranged at 8:30–11:30 a.m. and 14:00–17:00 p.m. Considered the diurnal variation of R_s, the measurement order of the 6 experimental plots changed in turn during the research period. There were no measurements taken during or immediately following a rainfall event (>1.5 mm) which can affect surface soil moisture (Y. Wang et al., 2011).

2.6. Measurement of soil temperature and moisture

Soil temperature at 5 cm depth near each soil collar was measured simultaneously with flux measurement using a portable thermocouple (JM624, Jinming Instruments Co. Ltd., Tianjin, China) in all experimental plots over the research period. Volumetric SWC (0–5 cm) was monitored simultaneously with flux measurement using a portable time domain reflectometer (Model TSC-I, China Agricultural University, Beijing, China) in all experimental plots during the study period. Hourly continuous T_s at 5 cm depth was recorded using a temperature sensor (HOBO UTBI-001 TidbiT v2, Onset Computer Corporation, U.S.) located in the center of each experimental plot over the study period. Hourly continuous SWC at 5 cm depth was recorded using moisture sensors (Model

CS616-L, Campbell Scientific Inc. and AV-EC5, Avalon Scientific Inc., U.S.) at two flux-tower sites near the 6 experimental plots (Fig. 1). Relationships of SWC between the experimental plots and the two flux-tower sites were developed using linear regression according to proximity principle (Fig. 1). Afterward, hourly continuous SWC of each experimental plot was predicted using the linear relationships and the continuous SWC dataset of the two flux-towers.

2.7. Observation of fine and coarse root biomass and litterfall input

Fine (<2 mm diameter) and coarse (≥ 2 and < 5 mm diameter) roots were sampled at 0–40 cm depth using a harvest method in August 2007. Roots were washed off soils and then oven-dried at 65 °C until constant weights were obtained. Litterfall inputs were measured monthly using litter traps (1 m × 1 m). There were three duplicate litter traps in each experimental plot. Litterfalls were collected and then separated into leaves, branches, flowers and fruits. All litterfall samples were oven-dried at 65 °C until constant weights were obtained. Litterfall C input of each experimental plot was calculated according to the C concentration described by Ma (2007).

2.8. Data analysis

In each experimental plot, Eq. (2) (Wang et al., 2012b), derived from the Arrhenius equation, was used to simulate the responses of R_s and R_h to T_s and SWC. Due to the effect of trenching treatment on T_s and SWC, measured R_h was further reproduced using its fitted model, generated by Eq. (2), and T_s and SWC of the no-trenching site. Annual R_s and R_h of each experimental plot were estimated using the fitted models and continuous T_s and SWC datasets of the non-trenching sites. In this study, the SOC turnover rate was the ratio of R_h to SOC stock. All linear and nonlinear regressions were performed using SPSS 13.0 (SPSS Inc., Chicago, Illinois). Comparisons of annual R_s, annual R_h, T_s and SWC between MP and SP were analyzed using one-way ANOVA of SPSS 13.0. Figures were drawn by ORIGIN 8.0 (OriginLab Corporation, Northampton, MA) and CorelDraw 9 (Corel Corporation, Canada).

$$R_s = R_{ref}e^{(E_a/R)(1/T_{ref}-1/T_K)}SWC^\beta \qquad (2)$$

where R_s is the measured soil respiration (µmol m^{-2} s^{-1}), R_{ref} represents the soil respiration at a reference temperature (T_{ref}) (µmol m^{-2} s^{-1}), E_a is a fitted site-specific parameter which refers to the required activation energy (J mol^{-1}), R indicates the gas constant (8.314 J K^{-1} mol^{-1}), T_{ref} is the reference soil temperature in Kelvin (K), T_K is the soil temperature in Kelvin (K), SWC is volumetric soil moisture (cm^3 cm^{-3}), and β is a fitted parameter.

3. Results

3.1. SOC stocks of native MP and exotic SP forests

Compared with the native MP, the exotic SP had no significant difference in forest floor C pool (Table 2). Similarly, there was no significant difference in mineral SOC density and bulk density at three horizons (0–10, 10–20 and 20–40 cm) between MP and SP (Table 2). Thus, the exotic SP, as compared with the native MP, did not alter the total SOC reservoir (forest floor + mineral soil of 0–40 cm).

3.2. R_s, R_h, R_a, T_s, SWC, litterfall input and mineral soil properties

During the study period, R_s and R_h of MP and SP all exhibited seasonal variations basically following the fluctuation of T_s (Fig. 2a, b, c, e, f and g). The annual mean R_s of native MP (2.82 ± 0.12 µmol m^{-2} s^{-1}) was evidently higher than that of exotic SP (2.23 ± 0.19 µmol m^{-2} s^{-1}). However, the annual mean R_h of native MP (1.74 ± 0.31 µmol m^{-2} s^{-1})

Table 2
Soil organic carbon stocks (organic layer + mineral soil of 0–40 cm) of the native MP and exotic SP plantations.

Forest type	Organic layer carbon pool (g C m^{-2})	Mineral SOC density (g kg^{-1})			Soil bulk density (g cm^{-3})			Total SOC stock (g C m^{-2})
		0–10 cm	10–20 cm	20–40 cm	0–10 cm	10–20 cm	20–40 cm	
MP	619 ± 310	13.40 ± 0.53	7.27 ± 0.83	4.77 ± 0.64	1.42 ± 0.07	1.53 ± 0.03	1.59 ± 0.04	5151 ± 102
SP	924 ± 271	14.67 ± 2.31	7.77 ± 1.76	4.53 ± 0.32	1.40 ± 0.01	1.56 ± 0.05	1.60 ± 0.05	5640 ± 498

1) was similar as introduced SP (1.71 ± 0.16 μmol m^{-2} s^{-1}). The SWC of the control sites in MP and SP also had basically seasonal patterns, with high values during January–June and low values during July–December (Fig. 2d and h). Furthermore, there was no difference in mean T_s and SWC between the two plantations (Table 3). The native MP and exotic SP also had similar characteristics in annual litterfall input, mineral soil total nitrogen, C/N ratio, and pH (0–40 cm) (Table 3).

After trenching treatment, mean T_s did not significantly change in both MP and SP (Table 4). In contrast, mean SWC notably increased 32% and 18% under trenching treatment in the native MP and exotic SP, respectively (Table 4).

Seasonal patterns of R_s were mainly affected by T_S and SWC in all experimental plots of the MP and SP (R^2 = 0.64–0.71, P < 0.001) (Table 5). Similarly, seasonal patterns of R_h were also mainly explained by T_S and SWC in the experimental plots of the MP and SP (R^2 = 0.56–0.78, P < 0.001) (Table 5).

Annual R_s, R_a, and R_a/R_s of the native MP were significantly 24%, 119% and 74% higher than that of exotic SP (Table 6), respectively. However, there was no difference in R_h, and R_h/R_s between the MP and SP (Table 6).

3.3. SOC turnover rate (R_h/SOC stock), R_a/root biomass

The SOC turnover rates of native MP and exotic SP were 8.6% ± 1.9% year^{-1} and 9.3% ± 1.6% year^{-1}, respectively. The native MP and exotic SP had similar SOC turnover rate (Fig. 3a). Moreover, the native MP and exotic SP had similar biomass in fine and coarse root (0–40 cm) (Table 3). However, the exotic SP, as compared with native MP, had remarkably lower ratios of R_a/fine root biomass and R_a/coarse root biomass (Fig. 3b and c).

4. Discussion

4.1. SOC stock and turnover rate of native and exotic tree species

The literature on plant–soil interactions strongly suggested that an exotic species had the potential to change SOC dynamics (Ehrenfeld, 2003). In support of this concept, several exotic tree species, as compared with native species, altered SOC stocks (Chapela et al., 2001; Zinn et al., 2002; Ladegaard-Pedersen et al., 2005), even they were both belong to same functional group such as conifer (Ehrenfeld, 2003; Ladegaard-Pedersen et al., 2005) and broad-leaved evergreen species (Russell et al., 2007). In contrast, our results showed that exotic SP, as compared with native MP, did not change the SOC stock as well as its two main components (forest floor and mineral soil of 0–40 cm). Thus, both pine species were suitable for reforestation in terms of SOC storage management. This result is in agreement with that in several forests (Ehrenfeld, 2003; Ayres et al., 2009; Trum et al., 2011). The inconsistent effects of exotic tree species on SOC stocks may be explained by the inherent species-specific characteristics and the stand age. First, tree species influence SOC stocks by the species-specific organic C input through litterfall and root activity (Binkley and Giardina, 1998; Valverde-Barrantes, 2007; Vesterdal et al., 2012). In our study, the amounts of litterfall input and fine and coarse roots of the native MP and exotic SP were similar (Table 3). The cumulative decomposition rates of coniferous litterfall of the two species were also comparable due to the similar lignin content (Li et al., 2007), with rates of approximately 71% over three years (Wang et al., 2009). Thus, it is reasonable that there was no significant difference in forest floor C pool between the two species (Table 2). Second, mineral SOC stocks of the native MP and exotic SP decreased fast until they reached approximately 80% of the prior shrubs and grasses level during the initial 7–8 years of forest
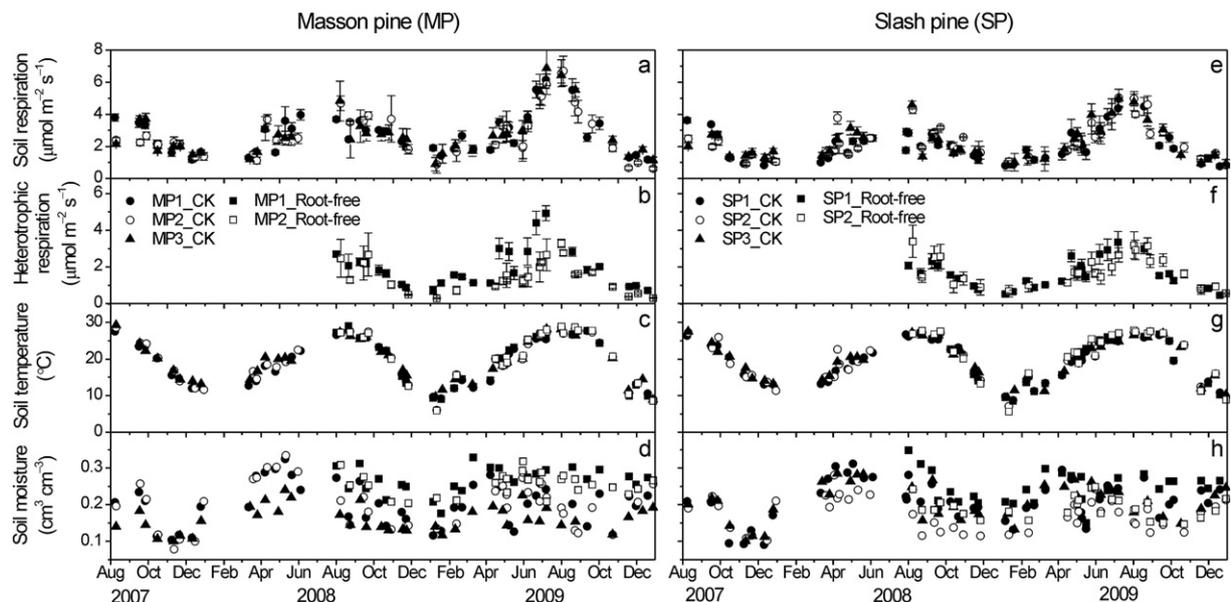
Fig. 2. Temporal patterns of R_s, R_h, T_s and SWC of the native Masson pine and exotic slash pine plantations during August 2007 to December 2009. Control (CK), n = 9. Trenching treatment (root-free), n = 6.

Table 3
Comparisons of environmental factors (T_s and SWC), mineral soil properties (0–40 cm), litterfall input, fine and coarse root biomass (0–40 cm) between the MP and SP forests (n = 3).

Forest type	Environmental variables		Mineral soil properties			Annual litterfall input (g C m^{-2})	Fine root biomass (g m^{-2})	Coarse root biomass (g m^{-2})
	T_s (°C)	SWC (cm^3 cm^{-3})	Total nitrogen (g kg^{-1})	C/N ratio	pH			
MP	18.14 ± 0.08	0.20 ± 0.03	0.51 ± 0.05	14.09 ± 1.21	4.55 ± 0.09	242 ± 6.7	444 ± 146	303 ± 66
SP	18.60 ± 0.33	0.21 ± 0.02	0.56 ± 0.09	12.27 ± 1.17	4.55 ± 0.07	264 ± 15	395 ± 196	323 ± 163

succession and increased slightly thereafter in subtropical China (Huang et al., 2007). In our study, the forest succession periods of the native MP and exotic SP were relative short (22 years). Thus, this relative short period may further hamper the influence of the exotic SP on SOC stock.

In this study, the ratio of R_h/SOC stock was used to represent the current SOC turnover rate. We found no difference in SOC turnover rate between the native MP and exotic SP (Fig. 3a). The reason may be that the SOC stability and dynamics have been demonstrated to be predominated by environmental and biological controls by recent analytical and experimental advances (Schmidt et al., 2011). And the native MP and exotic SP forests had similar T_s, SWC, fine and coarse root biomass, litterfall input (Table 3) and basal area (Table 1). In our study, the turnover time of SOC of the native MP and exotic SP was estimated to be about 10.8–11.6 years. The SOC turnover rates here were much higher than that of several temperate forests (<5.7% year^{-1}) (Gaudinski et al., 2000; Vesterdal et al., 2012), which may due to the warmer climate. According to our previous study, annual forest floor decomposition (D_f) accounted for 33% of R_s in the exotic SP (Wang et al., 2009), with a predicted value of 260.4 g C m^{-2} year^{-1} based on the current R_s. Thus, the turnover rate (D_f/P_f) and time (P_f/D_f) of forest floor of the exotic SP were estimated to be 28.2% year^{-1} and 3.5 years, respectively. Furthermore, our result suggested that the turnover rate (($R_h - D_f$) / mineral SOC stock) and time (mineral SOC stock / ($R_{r-f} - D_f$)) of mineral SOC of the exotic SP was estimated to be 5.6% year^{-1} and 17.8 years, respectively. This turnover rate of mineral SOC was much faster than that of temperate forests (Gaudinski et al., 2000), which may be also attributed to the warmer climate.

4.2. Soil CO$_2$ efflux and components of native and exotic tree species

Large variations of R_s among tree species were reported in various biomes: boreal forests (Laganière et al., 2012), temperate forests in European (Vesterdal et al., 2012), temperate forests in northern Wisconsin (Martin et al., 2009), the Serengeti ecosystem in Tanzania (Ruess and Seagle, 1994), temperate forests in northern China (Liu et al., 2014), subtropical forests in southern China (B. Wang et al., 2011; H. Wang et al., 2013), and tropical rainforests (Bréchet et al., 2009). In our study, despite similar T_s and SWC, R_s of the native MP was 24% higher than the exotic SP. The higher R_s of indigenous MP, compared with exotic SP, was attributed to its higher R_a (Table 6). Previous studies showed that R_s and R_a among tree species were correlated with root biomass (e.g. Epron et al., 2006; Martin et al., 2009). Furthermore, in forest ecosystems, a major part of gross primary production is allocated below-ground and 60% outcome of below-ground net primary production is R_a (Litton et al., 2007). In our study, the native MP and

exotic SP were both established in 1985 and had similar fine and coarse root biomass. However, they had significantly differences in R_s and R_a (Table 6). The ratios of R_a/fine root biomass and R_a/coarse root biomass of the native MP were significantly higher than that of the exotic SP (Fig. 3). This result indicated that native MP had higher root biomass-based respiration rate than exotic SP, which may due to its indigenous characteristic such as root anatomy, physiology (Yohannes et al., 2011), lower growing and C accumulation rate, stronger resistance (Ma et al., 2014) and more mycorrhizal associations (Tam, 1994). In contrast, there was no difference in R_h between MP and SP in our study. This result may be explained by the similar SOC stock (Table 2), mineral soil properties (total nitrogen, C/N and pH), T_s, SWC (Table 3) and PLFA-based microbial biomass (unpublished data) between the two species. In addition, R_h of the two species accounted for 45–68% of R_s (Table 6), which was comparable to some boreal (Comstedt et al., 2011), temperate (Gaudinski et al., 2000; Díaz-Pinés et al., 2010) and subtropical forests (Yi et al., 2007; Q. Wang et al., 2013).

4.3. Trenching- and topography-induced artifacts and uncertainty

Trenching is recognized as a useful and widely used methodology for partitioning R_s into two components (R_h and R_a) in field researches (Kuzyakov, 2006; Subke et al., 2006; Laganière et al., 2012). However, trenching is a problematic method with three main known artifacts: 1) change in soil moisture mainly caused by the absence of transpiration (Comstedt et al., 2011), 2) residual root decomposition (Díaz-Pinés et al., 2010), and 3) potential decrease in heterotrophic activity due to the absence of rhizodeposition (Drake et al., 2012) in trenched plots. In our study, trenching-induced artifacts also arose. First, we found that SWC was strongly increased in the trenched plots (Table 4), making it difficult to directly attribute observed CO$_2$ emission in trenched plots to potential R_h in control soils. However, we think that the response mechanism of measured R_h to the wide-ranged SWC is little influenced by trenching. Trenching-induced artifact of SWC can be amended by using Eq. (2) and the T_s and SWC of the control soils. Therefore, we further reproduced R_h using the fitted model, generated by Eq. (1), and the T_s and SWC of the control soils. The reproduced R_h could represent the inherent R_h under the non-trenched soil conditions. Thus, the reproduced R_h can be used for estimating annual R_h and its contribution to R_s. Furthermore, in our study, the trenched plots were arranged at stand gaps in order to avoid cutting thick roots and indeed we did not find thick roots were cut. However, simply waiting 4.3 months and then assuming no longer change in root decomposition might be not an adequate way to eliminate the above artifact of residual root

Table 4
Effect of trenching treatment on mean T_s and SWC during August 2008 to December 2009.

Plot name	Mean T_s (°C)		Mean SWC (cm^3 cm^{-3})	
	Non-trenching[a]	Trenching[b]	Non-trenching[a]	Trenching[b]
MP1	19.5	19.6	0.197	0.267*
MP2	21.6	21.7	0.199	0.257*
SP1	19.4	19.5	0.211	0.248*
SP2	21.8	21.9	0.162	0.192*

[a] n = 9.
[b] n = 6.
* Significant difference between the trenching and non-trenching treatments at $P < 0.05$ level.

Table 5
Responses of R_s and R_h to T_s and SWC using Eq. (2). All $P < 0.001$.

Plot name	Respiration	$R_{s,10}$ (μmol m^{-2} s^{-1})	E_a (kJ mol^{-1})	β	R^2
MP1	R_s	3.19	40.75	0.46	0.66
	R_h	5.27	39.07	1.22	0.56
MP2	R_s	1.23	56.30	0.05	0.66
	R_h	0.92	63.18	0.51	0.76
MP3	R_s	2.46	51.18	0.36	0.64
SP1	R_s	2.92	45.64	0.67	0.71
	R_h	1.03	46.58	0.19	0.63
SP2	R_s	2.01	57.49	0.45	0.69
	R_h	1.56	60.96	0.61	0.78
SP3	R_s	4.04	48.68	0.84	0.65

Table 6
Comparisons of R_s, R_h, R_a and their ratios between the MP and SP forests.

Forest type	Annual respiration (g C m^{-2} year^{-1})			Ratios (%)	
	R_s [a]	R_h [b]	R_a [b]	R_h/R_s	R_a/R_s
MP	975.3 ± 32.2[*]	441.3 ± 89.7	537.9 ± 45.2[*]	44.9 ± 7.1	55.1 ± 7.1[*]
SP	789.1 ± 45.9	525.3 ± 32.5	245.8 ± 79.9	68.4 ± 8.4	31.6 ± 8.4

[a] n = 3.
[b] n = 2.
[*] Significant difference between the MP and SP forests at $P < 0.05$ level.

decomposition (Díaz-Pinés et al., 2010). Therefore, it is necessary to account for trenching-induced artifacts of the residual root decomposition and the absence of rhizodeposition in the future study. In addition, even though the terrain of our study site is gentle with a relative elevation <50 m and a slope gradient <30°, topography-induced uncertainty may also exist and should be considered in the future research.

5. Conclusions

Compared with the native species (*P. massoniana*), the introduced species (*P. elliottii*) had no impact on SOC stock (forest floor + mineral soil of 0–40 cm), R_h and turnover rate (R_h/SOC stock) over 22-years of development in the subtropical pine plantations of China. However, the exotic species hampered annual R_s through
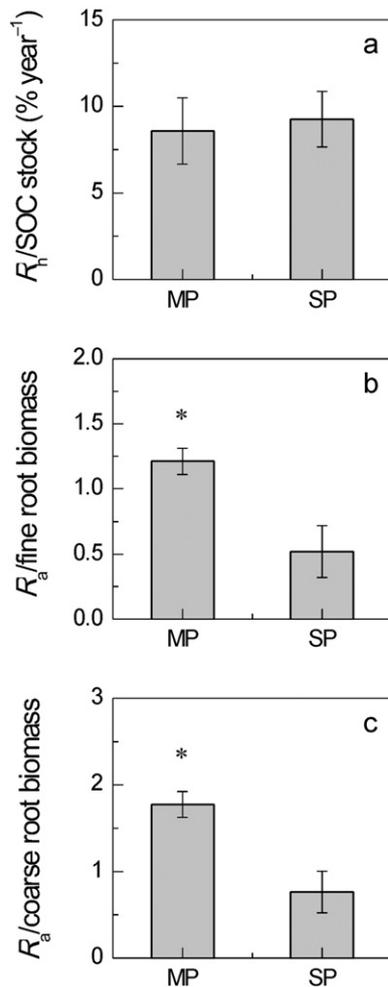
Fig. 3. Comparisons of R_h/SOC stock (organic layer + mineral soil of 0–40 cm) (a), R_a/fine root biomass (b), R_a/coarse root biomass (c). R_h and R_a, g C m^{-2} year^{-1}. SOC stock, g C m^{-2}. Fine and coarse root biomass, g m^{-2}. Each value refers to an average of two duplicate experimental plots and error bar represents one standard deviation (±SD). *Significant difference between the native and exotic pine forests at $P < 0.05$ level.

decreasing R_a as compared with the indigenous species. Furthermore, the exotic species had lower root biomass-based R_a rate than the native species. Consequently, both pine species were recommended in case of reforestation for SOC sequestration.

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