

# Effects of afforestation on soil carbon turnover in China's subtropical region

\*WANG Shaoqiang<sup>1</sup>, LIU Jiyuan<sup>1</sup>, ZHANG Cheng<sup>1</sup>, YI Chuixiang<sup>2</sup>, WU Weixing<sup>1</sup>

1. Key Lab of Ecosystem Network and Modelling, Institute of Geographic Sciences and Natural Resources Research, CAS, Beijing 100101, China;

2. Queens College, City University of New York, New York, USA

**Abstract:** Afforestation in China's subtropics plays an important role in sequestering CO<sub>2</sub> from the atmosphere and in storage of soil carbon (C). Compared with natural forests, plantation forests have lower soil organic carbon (SOC) content and great potential to store more C. To better evaluate the effects of afforestation on soil C turnover, we investigated SOC and its stable C isotope ( $\delta^{13}\text{C}$ ) composition in three planted forests at Qianyanzhou Ecological Experimental Station in southern China. Litter and soil samples were collected and analyzed for total organic C,  $\delta^{13}\text{C}$  and total nitrogen. Similarly to the vertical distribution of SOC in natural forests, SOC concentrations decrease exponentially with depth. The land cover type (grassland) before plantation had a significant influence on the vertical distribution of SOC. The SOC  $\delta^{13}\text{C}$  composition of the upper soil layer of two plantation forests has been mainly affected by the grass biomass  $^{13}\text{C}$  composition. Soil profiles with a change in photosynthetic pathway had a more complex  $^{13}\text{C}$  isotope composition distribution. During the 20 years after plantation establishment, the soil organic matter sources influenced both the  $\delta^{13}\text{C}$  distribution with depth, and C replacement. The upper soil layer SOC turnover in masson pine (a mean 34% of replacement in the 10 cm after 20 years) was more than twice as fast as that of slash pine (16% of replacement) under subtropical conditions. The results demonstrate that masson pine and slash pine plantations cannot rapidly sequester SOC into long-term storage pools in subtropical China.

**Keywords:** soil organic carbon; stable carbon isotope; plantation forest

## 1 Introduction

Soils play an important role in the carbon (C) cycle and afforestation has been proposed as an effective approach to sequestering more CO<sub>2</sub> in the short term and mitigating rising atmospheric CO<sub>2</sub> concentration over longer periods (Silver *et al.*, 2004; de Alcântara *et al.*, 2004; Richards *et al.*, 2007). Tropical and subtropical forest soils contain about 30% of the global soil organic matter (SOM) (Dalal and Carter, 2000). Recently, forests have been

**Received:** 2010-05-06 **Accepted:** 2010-07-21

**Foundation:** National Natural Science Foundation of China, No.30670312; The Key Project of Knowledge Innovation Program of the Chinese Academy of Sciences, No.KZCX2-YW-305-3

**Author:** Wang Shaoqiang (1972–), Ph.D and Professor, specialized in global change ecology research.

E-mail: sqwang@igsnr.ac.cn

studied for their potential for C sequestration (Canadell *et al.*, 2008). However, little is known about rates of soil C turnover and sequestration into SOM in subtropical and tropical plantations (Richards *et al.*, 2007). Therefore, more knowledge of C stocks and SOM fractions is necessary to elucidate C storage in subtropical and tropical ecosystems, especially when there is conversion from C<sub>3</sub> to C<sub>4</sub> vegetation or vice versa (Paul *et al.*, 2002; Silver *et al.*, 2004; de Alcântara *et al.*, 2004; Richards *et al.*, 2007).

Vegetation and soil C dynamics after land use change are often studied by using differences in the natural abundance of <sup>13</sup>C ( $\delta^{13}\text{C}$ ) in plants with C<sub>3</sub> (trees) and C<sub>4</sub> (many tropical and subtropical grasses) photosynthesis pathways, or by using differences in  $\delta^{13}\text{C}$  ratios between older original soil C and newer inputs of C (Farquhar *et al.*, 1982; Ehleringer *et al.*, 2000). The results can help understand the ecological consequences of land-use change (Bernoux *et al.*, 1998; Hobbie *et al.*, 2004; Cheng *et al.*, 2006). Plants with a C<sub>3</sub> pathway have  $\delta^{13}\text{C}$  values in the range of  $-35\text{‰}$  to  $-20\text{‰}$ , while those with a C<sub>4</sub> pathway have higher  $\delta^{13}\text{C}$  values, ranging from  $-19\text{‰}$  to  $-9\text{‰}$  (Bernoux *et al.*, 1998; Staddon, 2004). Because values of soil  $\delta^{13}\text{C}$  are close to that of adjacent vegetation, a number of studies have used the changes or differences in soil  $\delta^{13}\text{C}$  to estimate turnover times of soil organic carbon (SOC) during transitions from C<sub>3</sub> to C<sub>4</sub> vegetation (or vice versa) and to calculate the amount of C derived from each vegetation type (Balesdent and Mariotti, 1996; Bernoux *et al.*, 1998). Several studies have focused on small variations in  $\delta^{13}\text{C}$  values of mature forest soils to infer soil C dynamics across regional or latitudinal gradients (Bird *et al.*, 1996; Garten *et al.*, 2000). Therefore, the isotopic signature of soil C reflects the relative contribution of C<sub>3</sub> and C<sub>4</sub> vegetation to soil C accumulation, and integrates the species' contributions over long time periods (Boutton, 1996; Diels *et al.*, 2001; Biedenbender *et al.*, 2004).

Vegetation change that involves a different dominant photosynthetic pathway will gradually change the isotopic composition. For example, the introduction of C<sub>3</sub> plants ( $\delta^{13}\text{C}\approx-27\text{‰}$ ) to an ecosystem previously developed under C<sub>4</sub> vegetation ( $\delta^{13}\text{C}\approx-13\text{‰}$ ), or vice versa, provides an *in situ* labeling of new organic matter with a different isotopic composition incorporated into the soil (Lefroy *et al.*, 1993). Basically, two models have been used to estimate the sources of SOC. One assumes a similar <sup>13</sup>C fractionation during decomposition of litter from both new and old vegetation, whereas the other does not account for any fractionation upon the humification of the newly introduced materials (Balesdent and Mariotti, 1996; de Alcântara *et al.*, 2004). Soil C from previous plant communities will gradually decay out of the SOC pool and be replaced by new C derived from the subsequent plant communities. The isotopic discrepancy created by land use change would therefore persist for some time (Boutton *et al.*, 1998). The time that it takes for C from the older plant community to be depleted from the SOC pool is determined by, and can be used as a direct measure of, SOM turnover rate (Cerri *et al.*, 1985; Bernoux *et al.*, 1998). Therefore, the impacts of land use change, such as afforestation, on SOC dynamics and turnover rate can be explored using stable C isotopes.

In China, the isotopic composition method has been used to reconstruct past vegetation changes and better understand SOC dynamics after such changes (Piao *et al.*, 2001). Plantations were first established in southern China to prevent serious soil erosion in the early 1980s. Many studies have evaluated these afforestation projects in subtropical regions of China but most of them did not focus on SOC dynamics (Ma *et al.*, 2007, 2008). In this pa-

per, we use C isotopic analysis to evaluate the impact of conversion from grasslands to planted pine forests (masson pine and slash pine, which are two exotic species in subtropical China) on SOC dynamics in a Chinese red earth (a Udic Ferrisol).

The main purpose of this study was to use stable C isotopes ( $\delta^{13}\text{C}$ ) and trends of  $\delta^{13}\text{C}$  with depth in soil profiles to evaluate the potential effects of afforestation on SOC dynamics after conversion of soils from grassland to pine forests.

## 2 Methods and materials

### 2.1 Site description

The study was carried out at the Qianyanzhou Forest Experimental Station (QFES), located in Jiangxi Province in South China (26°44'48", 115°04'13"E). The land area of QFES is 204.17 ha. The topography is gently undulating with an average slope of 10–30°. The elevation is approximately 60 m above sea level. The soil parent material consists of red sandstone and mudstone, and soils are mainly red soils (Udic Ferrisols), paddy soils (Anthrosols), and fluvio-aquic soils. The area of red soils is the largest, being about 140.5 ha. The annual mean air temperature is 17.9°C, while the highest and lowest extreme temperatures are 39.5 and -5.8°C, respectively. The annual mean precipitation is around 1489 mm, with a dry period from July to August. The landscape of the QFES area is typical of subtropical red soils hill land, occupying low hill, terrace, and floodplain positions in the landscape. The region lies in the middle subtropical evergreen broad-leaved zone, but the original forests have almost disappeared due to deforestation and land-use conversion to agricultural production prior to the 1980s. The land cover types of QFES currently include planted conifer and broad-leaved forests, orchard, cropland, shrub, and other secondary vegetation.

Before 1983, Qianyanzhou was a small village of 31 people with a stagnant economy. There were no natural forests left in the region of QFES and it was a degraded area covered with grass and shrubs due to overharvesting. The perennial gramineous grasses were community constructive species at QFES and included *Arundinella setosa* Trin, *Miscanthus floridulus* Warb., *Dicranopteris dichotoma* (Thunb.) Benth., *Imperata cylindrical* var. *major*, and *Isachne globosa* (Thunb.) Kuntze. Most shrubs were secondary shrubs or small coppiced shrubs such as *Quercus fabric* Hance, *Lespedeza formosa* (Vog.) Koehne, *Vitex negundo* L. var. *cannabifolia* (Sieb. et. Zucc.) Hand.-Mazz., and *Photinia davidsoniae* Rehd.

A conservation plan for red soils resources was implemented in 1983 and subsequently the Chinese Academy of Sciences (CAS) established the Qianyanzhou Forest Experimental Station. Different plantation and land use patterns were designed and implemented to restore land resources and reduce soil and water erosion. Land statistics for the experimental area are available for the following years: 1983, 1990 and 1997 (Wang *et al.*, 2004). Land cover types include residential land, vegetable plots, paddy land, dryland, plantations, and land used for transport with plantations occupying more than 70% of the QFES area.

### 2.2 Soil and litter sampling

The reference site was maintained under a native grassland with scattered shrubs (canopy density <5% of the total area). In 2003 the most common grasses (grassy C<sub>4</sub>-dominated) in

this ecosystem were *Imperata cylindrica* var. *major*, *Scleria levis*, *Carex gibba*, *Arundinella hirta* var. *depauperata*, and *Potentilla freyniana*, and the main shrubs were *Solanum grandiflorum*, *Prunella vulgaris* L. and *Dimorphandra mollis*. The plantation area was converted into forest in 1984, after clearing the native grass vegetation. It was planted with a mixture of tree species, the most common of which were the C<sub>3</sub> species masson pine (*Pinus massoniana*), slash pine (*Pinus elliottii engelm*), China fir (*Cunninghamia lanceolata*), and broad-leaved coniferous mixed forest species.

Soil sampling was carried out in August 2003. Three major planted forests (slash pine, masson pine and China fir) were selected and five soil profiles sampled from each forest type from shady and sunny aspects at the ridge-slope, mid-slope, and toe-slope respectively. Soils samples were collected from each pedogenetic horizon and bulk density was measured using a cutting ring. The sampling depth extended to reach the parent material layer. Soil samples from each horizon were stored in cloth-bags until they could be processed for measuring C content. The soil moisture content was determined by TDR at each site.

Soil samples were air-dried (40°C) and hand-sieved through a 2-mm screen to remove roots, stone and other debris. Five hundred gram samples of each soil horizon were weighed to determine SOC, total nitrogen (TN), pH value and stable C isotope content. Part of sieved samples were ground and sieved to 1mm, so that sieved samples were used to determine pH by the electrometric method. Then, 50g samples for each soil horizon were fine-ground to pass a 0.28 mm sieve and used to determine SOC and TN concentrations using the sulfuric acid-potassium dichromate oxidation method, and to measure <sup>13</sup>C composition. The SOC concentration of each soil horizon was summed for the entire soil profile to estimate total SOC concentration using the method of Wang *et al.* (2004b). Table 1 shows the SOC concentrations for each sampled profile.

Litter sampling was conducted in September 2005. The litter samples were collected with 100×100 cm frames adjacent to the soil sampling sites. The aboveground litter samples were oven-dried (80°C) for at least 48h to constant weight, and ground to pass a 0.28 mm sieve. Sieved litter samples of each vegetation type were mixed to yield one composite sample, resulting in five composite samples. Then, 10g composite samples of each vegetation type were used for stable C isotope analyses.

### 2.3 <sup>13</sup>C analytical method and C-origin estimations

The δ<sup>13</sup>C of organic materials (litter and soil samples) were measured after conversion of total C to CO<sub>2</sub> on an isotope ratio mass spectrometer (MAT 253) at the Centre for Isotope Research at the Institute of Geographic Sciences and Natural Resources Research (IGSNRR, Beijing, P.R.China). By convention, isotope ratios are expressed as δ<sup>13</sup>C (‰):

$$\delta^{13}C = \left( \frac{R_{sample}}{R_{standard}} - 1 \right) \times 1000 \quad (1)$$

where *R* is the molar ratio of the heavy to light isotope of the sample or standard. The standard is Vienna-Pee Dee Belemnite (V-PDB). Pee Dee Belemnite (PDB) is the belemnite carbonate standard of the Pee Dee Formation in South Carolina, USA (Chiang *et al.*, 2004; Cheng *et al.*, 2006). The systematic error for δ<sup>13</sup>C measurement was no larger than 0.3‰ based on repeated measurements (IGSNRR, Chinese Academy of Sciences, Beijing, China).

In the planted forest, we estimated the proportion of SOC that originated from the forest litter (newly introduced material;  $F_{FC}$ ), and the proportion that remained from the previous grass vegetation ( $F_{GC}$ ). According to Balesdent and Mariotti (1996), a modified two-compartment linear mixing model equation applied to estimate plantation ( $C_3$ ) and grass ( $C_4$ )-derived SOC proportions is given by:

$$F_{FC} = \left( \frac{\delta_T - \delta_{grasssoil}}{\delta_{forestlitter} - \delta_{grasssoil}} \right) \quad (2)$$

and

$$F_{GC} = 1 - F_{FC} \quad (3)$$

where  $F_{FC}$  is the fraction of SOC originating from the plantation litter (forest-derived C),  $\delta_T$  is the  $\delta^{13}C$  of the SOC in the surface soil horizon of the planted forest,  $\delta_{grasssoil}$  is the  $\delta^{13}C$  of the SOC in grass in the surface soil horizon, and  $\delta_{forestlitter}$  is the  $\delta^{13}C$  of the forest litter in that horizon. Equation (3) gives  $F_{GC}$ , which is the fraction of grass-derived C remaining. We assumed a similar isotopic fractionation during decay between grasslands and planted forest according to de Alcântara *et al.* (2004). In our study, the mean value of  $\delta^{13}C$  from grass litter was  $-19.19\%$ .  $\delta_{forestlitter}$  values of masson pine and slash pine litters were  $-28.03\%$  and  $-29.49\%$ , respectively. The proportion of grass-derived C in the soil was calculated by subtracting the amount of  $C_3$ -derived C from the total SOC content of the forest site. To estimate C dynamics, we calculated the percentage of replacement of grass-derived C by the introduced forest-derived C during the study period for the surface soil (0–10 cm).

The variance range ( $\pm$ ) of SOC, TN,  $\delta^{13}C$ , pH value and soil thickness for land use types was computed by using a *t*-test (Wilding *et al.*, 2001; Wang *et al.*, 2004a). SPSS version 10.0 was employed for significant difference analysis and regression analysis using  $\delta^{13}C$  and SOC measurements.

### 3 Results

#### 3.1 Whole soil profile organic C and total N storage

No significant differences were found in mean SOC ( $P=0.112>0.05$ , sig. (2-tailed), Independent Samples Test) and TN ( $P=0.101>0.05$ , sig. (2-tailed), Independent Samples Test) concentrations of soil profiles for plantations and grassland (Tables 1 and 2). The fir forest soil was slightly richer in SOC and TN and had the highest C:N ratio compared with other plantations and grassland (Table 2), while masson pine had the lowest SOC and TN concentrations, C:N ratio and pH value, and was least variable among plantations. Grassland had the largest variability of SOC, TN content, C:N ratio and pH value among the four vegetation types. The plantations produced more aboveground litter than grassland but this did not lead to higher C and N contents in the forest soils over the 20 years. However, plantation establishment could make soil to be more acidification, with the pH values of plantations being lower than that of grassland at QFES (18 soil profiles,  $P=0.013<0.05$ , sig. (2-tailed), Mann-Whitney Test, Independent Samples Test, Nonparametric Test).

Concentrations in the top 80 cm of soils showed that 20-year-old masson pine plantations had lower C and N than slash pine and fir forest (Table 1), and this difference was also ob-

served when comparing C concentrations through 100 cm depth. Although aboveground biomass was higher under the different forest types than grasslands, soil C and TN in the top 30 cm soil layer in grasslands ( $0.799\% \pm 0.11\%$ ,  $0.068\% \pm 0.01\%$ ) was greater than in masson pine ( $0.749\% \pm 0.11\%$ ,  $0.065\% \pm 0.01\%$ ) and slash pine ( $0.727\% \pm 0.09\%$ ,  $0.060\% \pm 0.01\%$ ) plantation sites (Figure 1). Amongst the forest sites, the total C sequestered by fir-forest soils was greater than for the other two forest soils (Tables 1 and 2).

**Table 1** SOC and TN concentrations of individual soil profiles at QFES

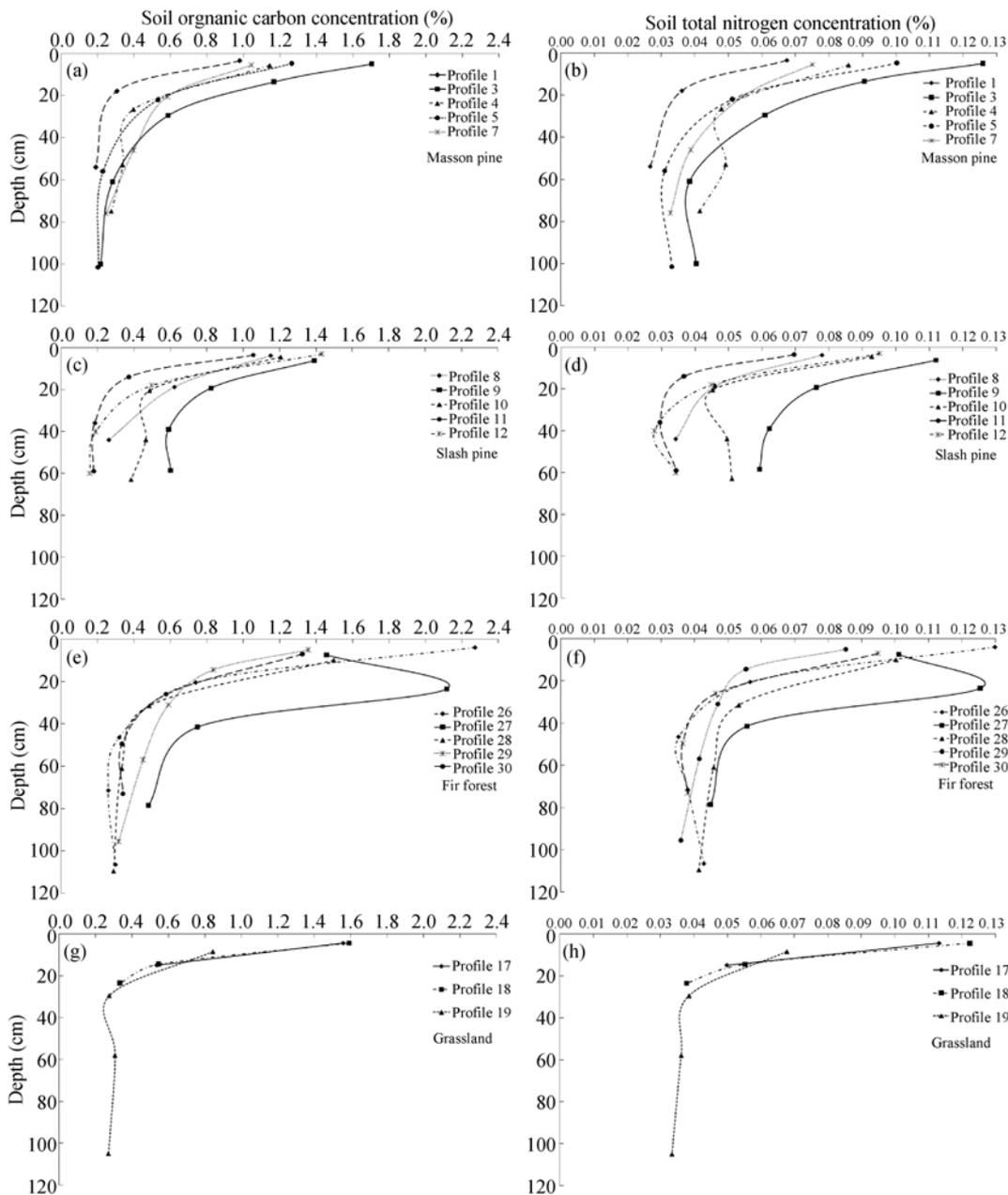
	Profile No.	Depth (cm)	SOC (%)	Total N (%)	C/N	pH	Transitions	Slope
Masson pine	1	79	0.291	0.033	8.14	4.99	C <sub>4</sub> -C <sub>3</sub>	Ridge-slope
	3	120	0.493	0.054	7.98	4.99	C <sub>4</sub> -C <sub>3</sub>	Mid-slope
	4	85.5	0.451	0.052	8.17	5.02	C <sub>4</sub> -C <sub>3</sub>	Ridge-slope
	5	125.5	0.357	0.041	7.87	4.98	C <sub>4</sub> -C <sub>3</sub>	Mid-slope
	7	90	0.468	0.045	9.94	4.86	C <sub>4</sub> -C <sub>3</sub>	Toe slope
	8	58	0.517	0.044	10.87	5.20	C <sub>4</sub> -C <sub>3</sub>	Ridge-slope
Slash pine	9	65	0.795	0.074	10.46	5.21	C <sub>4</sub> -C <sub>3</sub>	Mid-slope
	10	70	0.551	0.054	9.91	5.13	C <sub>4</sub> -C <sub>3</sub>	Toe slope
	11	67	0.314	0.036	7.76	5.25	C <sub>3</sub> -C <sub>3</sub>	Mid-slope
	12	90	0.392	0.041	8.34	5.41	C <sub>3</sub> -C <sub>3</sub>	Toe slope
	26	130	0.502	0.048	9.17	5.20	C <sub>3</sub> -C <sub>3</sub>	Toe slope
Fir forest	27	106	0.928	0.068	12.70	4.85	C <sub>3</sub> -C <sub>3</sub>	Mid-slope
	28	140	0.504	0.053	8.53	5.19	C <sub>3</sub> -C <sub>3</sub>	Ridge-slope
	29	120	0.527	0.045	11.07	5.24	C <sub>3</sub> -C <sub>3</sub>	Mid-slope
	30	85	0.567	0.049	10.80	5.12	C <sub>3</sub> -C <sub>3</sub>	Toe slope
Grassland	17	21	0.974	0.077	12.04	5.41	C <sub>4</sub> -C <sub>4</sub>	Ridge-slope
	18	26	0.826	0.073	10.62	5.39	C <sub>4</sub> -C <sub>4</sub>	Mid-slope
	19	136	0.351	0.039	8.54	6.04	C <sub>4</sub> -C <sub>4</sub>	Toe slope

**Table 2** SOC and TN concentrations of land use types at QFES

	Profile number	Depth		SOC		TN		C/N		pH	
		cm	S.D	%	S.D	%	S.D	S.D	S.D	S.D	
Masson pine	5	100	21.2	0.415	0.085	0.045	0.009	8.36	0.86	4.97	0.06
Slash pine	5	70	12.0	0.504	0.184	0.049	0.015	9.36	1.35	5.25	0.10
Fir forest	5	116	21.5	0.595	0.182	0.052	0.009	10.29	1.65	5.13	0.16
Grassland	3	61	65.0	0.490	0.325	0.048	0.021	9.24	1.76	5.88	0.37

SOC and TN concentrations of all soil profiles decreased with depth (Figure 1). The fir forest soils had significantly higher SOC ( $P < 0.001$ , sig. (2-tailed)) and TN ( $P = 0.015 < 0.05$ , sig. (2-tailed)) concentrations in the surface 0 to 20 cm than the other vegetation types based on Independent Samples Test. The SOC and TN concentrations decreased exponentially from 0 to 50 cm and remained stable from 50 to 100 cm for most soil profiles (Figure 1).

The vertical distributions of SOC concentrations were well described by a power-law function with depth except for the profile 27 (Figure 1 and Table 3). We speculate that this profile has been affected by anthropogenic disturbance in the A1 and A2 horizons. The ver-



**Figure 1** Vertical distribution of SOC and TN content of soil profiles at the QEFS

**Table 3** Regression power functions of the variation of SOC concentration with depth

Land use type	The change of photosynthetic pathway	b	a(exponent)	R <sup>2</sup>	n
Masson pine	C <sub>4</sub> -C <sub>3</sub>	3.067	-0.585	0.847	20
Slash pine	C <sub>4</sub> -C <sub>3</sub>	2.327	-0.428	0.725	11
Slash pine	C <sub>3</sub> -C <sub>3</sub>	2.844	-0.710	0.962	8
Fir forest	C <sub>3</sub> -C <sub>3</sub>	4.543	-0.617	0.937	18

tical distribution of SOC and TN concentrations was different under the different various land use types. Above 50 cm, the SOC concentration of the fir forest soil varied between 0.32% and 2.27%, while the SOC concentration of masson pine and slash pine forest soils varied from 0.19% to 1.70%. Below 50 cm, SOC concentrations ranged from 0.48% to 0.62% in the fir forest soil (average 0.35%) and from 0.19% to 0.34% in the masson pine soil (average 0.25%). This shows that the fir forest soils had higher C storage in the top soil layer (0–50 cm) than did masson pine and slash pine ( $P=0.011<0.05$ , sig. (2-tailed), Independent Samples Test).

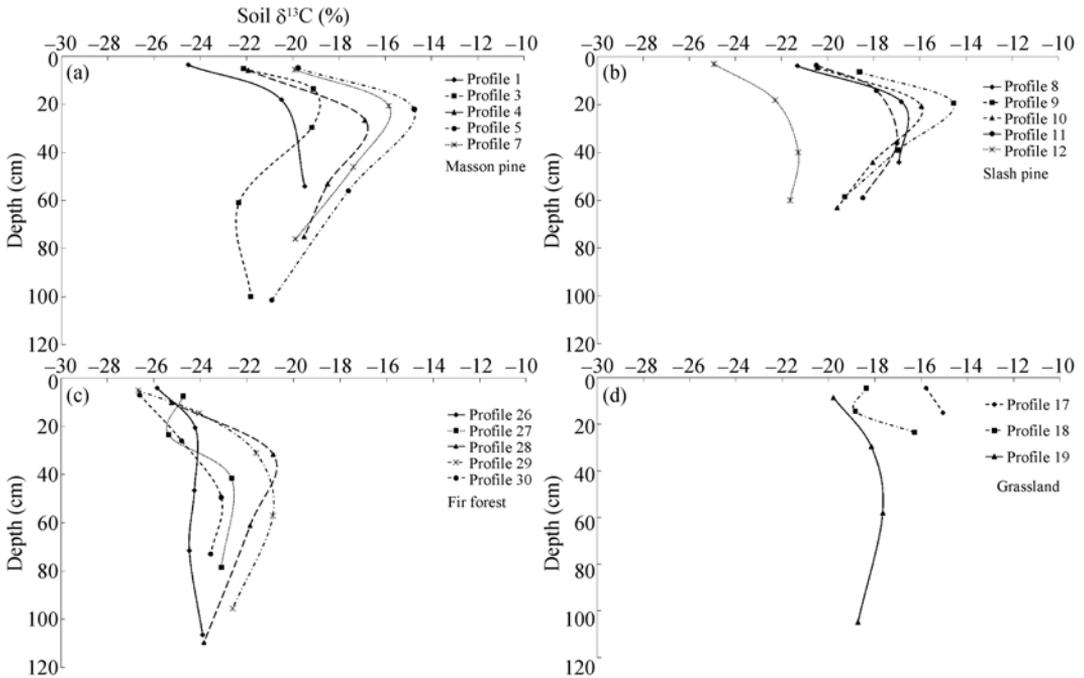
### 3.2 $\delta^{13}\text{C}$ values of litter and soil

The  $\delta^{13}\text{C}$  values derived from litter and soil samples are presented in Table 4. They were significantly different between grassland and plantations. Conversion from grassland to forest plantation ( $C_4 \rightarrow C_3$ ) caused a slight decrease in the soil  $\delta^{13}\text{C}$  values of masson pine and slash pine forests. Within the three planted forests,  $\delta^{13}\text{C}$  values increased from litter to the topsoil (0 to 20 cm), indicating  $^{13}\text{C}$  enrichment from litter to SOM.

Figure 2 shows the difference among  $\delta^{13}\text{C}$  profiles for the different vegetation types. Soil  $\delta^{13}\text{C}$  profiles of masson pine and fir forests showed distinct  $C_3$  vegetation  $^{13}\text{C}$  values at 0–5 and 5–10 cm depths (mean  $-24\text{‰}$  and  $-26\text{‰}$ ) (Figure 2). Soil  $^{13}\text{C}$  at these depths was more enriched than litter (Table 4). Below 20–25 cm,  $\delta^{13}\text{C}$  values of masson pine and slash pine soils excluding profiles 11 and 12 decreased quickly and were more depleted in  $^{13}\text{C}$ . Below 30 cm depth, patterns of variation in soil  $\delta^{13}\text{C}$  values were similar between masson pine and

**Table 4**  $\delta^{13}\text{C}$  distribution of soil and litter at QFES (‰)

Land use type	Soil profiles	Depth (cm)	Soil $\delta^{13}\text{C}$	S.D	Mean litter $\delta^{13}\text{C}$	Mean soil $\delta^{13}\text{C}$ of land use
Masson pine	1	79	-20.21	2.67	-28.03	
	3	120	-21.30	1.63		
	4	85.5	-18.64	2.12		-19.39±1.55
	5	125.5	-18.46	2.71		
	7	90	-18.15	2.00		
	8	58	-17.44	2.58	-29.49	
Slash pine	9	65	-17.24	2.10		-17.57±0.70
	10	70	-17.97	1.97		
	11	67	-17.91	1.48		
	12	90	-22.03	1.66		-20.27±8.88
	26	130	-24.27	0.76	-28.28	
Fir forest	27	106	-23.62	1.29		
	28	140	-23.05	1.97		-23.49±0.91
	29	120	-22.46	2.29		
	30	85	-24.30	1.57		
Grassland	17	21	-15.38	0.51	-19.19	
	18	26	-18.21	1.36		-18.11±3.18
	19	136	-18.51	0.91		
Shrubland					-29.01	



**Figure 2**  $\delta^{13}\text{C}$  profiles at each site

slash pine. With increasing depth,  $\delta^{13}\text{C}$  values of fir forest soils increased in the surface layer (0–35 cm), but decreased slightly below 40 cm except for profiles 28 and 29 (Figure 2). Soil  $\delta^{13}\text{C}$  under fir forests ranged from  $-26.7\text{‰}$  to  $-20.9\text{‰}$  and was lower than that in other plantations.

In our study, although  $\delta^{13}\text{C}$  values of grassland litter were enriched by 9.8‰ to 10.3‰ more than plantation forest litters, soil  $\delta^{13}\text{C}$  values of grassland were only enriched 5.38‰, 1.28‰, and 2.16‰ more than fir forest, masson pine, and slash pine (without photosynthetic pathway change) respectively (Table 4). This means that large amounts of grass-derived SOC remained in the masson pine and slash pine soils over and above the  $^{13}\text{C}$  fractionation effect during the litter decomposition processes.

Based on the change of photosynthetic pathway, all soil profiles were divided into two types: (1) a changed photosynthetic pathway ( $\text{C}_4 \rightarrow \text{C}_3$ ); and (2) an unchanged photosynthetic pathway ( $\text{C}_3 \rightarrow \text{C}_3$ ). Profiles that have changed their photosynthetic pathway from  $\text{C}_4$  to  $\text{C}_3$  for 20 years, showed a similar tendency in that soil  $\delta^{13}\text{C}$  values increased quickly with increasing depth and reached their maximum value around 25 cm before decreasing rapidly below 25 cm. The decreasing soil  $\delta^{13}\text{C}$  values with depth became slow below 50 cm. Among these soil profiles ( $\text{C}_4 \rightarrow \text{C}_3$ ), profile 1 was different with soil  $\delta^{13}\text{C}$  values increasing with increasing depth. For the soil profiles ( $\text{C}_3 \rightarrow \text{C}_3$ ), soil  $\delta^{13}\text{C}$  values increased slowly with depth reaching a maximum at about 40–50 cm. At depths below 50 cm, soil  $\delta^{13}\text{C}$  values decreased slightly with depth. Profiles 11 and 27 performed differently from the others. Soil  $\delta^{13}\text{C}$  values in profile 27 decreased in the topsoil and increased from 20 cm to 40 cm (Figure 2c). Profile 11 not only had similar soil  $\delta^{13}\text{C}$  values, but also a similar profile trend with depth to the masson pine profile under the same photosynthetic pathway change.

### 3.3 Forest and grass contribution to SOC

Table 5 shows that  $\delta^{13}\text{C}$  values of soil surface layers and litter in masson pine and slash pine with a  $\text{C}_4 \rightarrow \text{C}_3$  transition were different reaching 6.37‰ and 8.32‰, respectively. It shows that  $^{13}\text{C}$  enrichment of the soil surface layer in masson pine and slash pine was partly due to fractionation during the process of organic matter decomposition. However, the difference in  $\delta^{13}\text{C}$  values between the soil surface layer and litter in the fir forest was only 2.45‰. During ecological restoration, around 1984, masson pine and slash pine ( $\text{C}_4 \rightarrow \text{C}_3$  transition) were planted in previous grassland. In contrast, fir forest was planted in previous shrub land without a change in photosynthetic pathway. Hence, the SOM with a different mixed  $^{13}\text{C}$  composition after plantation establishment was the reason for the considerable difference between the litter and soil surface layer under masson pine and slash pine, because of some remaining SOC derived from the previous vegetation since plantation establishment.

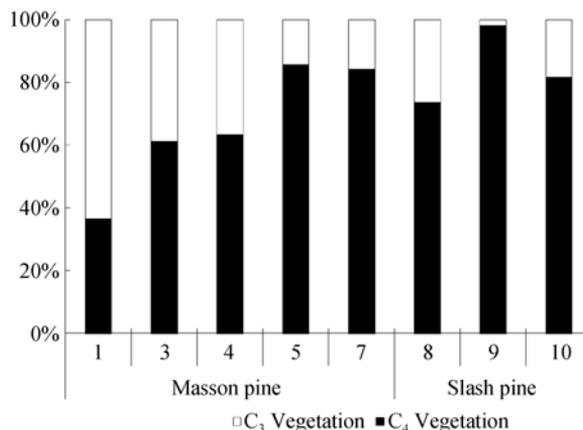
**Table 5**  $\delta^{13}\text{C}$  distribution of soil surface layer and litter of masson pine and slash pine with a photosynthetic pathway transition at QFES (‰)

Land use type	Profile No.	Surface layer Depth (A) (cm)	Soil $\delta^{13}\text{C}$	Mean litter $\delta^{13}\text{C}$	Mean soil $\delta^{13}\text{C}$ of surface layer
Masson pine	1	0–7.0	–24.53	–28.03	
	3	0–10	–22.14		
	4	0–11.5	–21.94		–21.66±0.87
	5	0–9.5	–19.77		
	7	0–11	–19.92		
Slash pine	8	0–7.5	–21.33	–29.49	
	9	0–12.5	–18.62		–20.13±0.80
	10	0–9	–20.44		

\* The  $\delta^{13}\text{C}$  distribution of soil surface layers has a significant difference with  $\delta^{13}\text{C}$  of litter in masson pine ( $P=0.040<0.05$ , sig. (2-tailed)) and slash pine ( $P=0.028<0.05$ , sig. (2-tailed)) based on Independent Samples Test, respectively.

Equation (2) was used to estimate the proportion of grass-derived and planted forest-derived C in the soil surface layer (A horizon) under masson pine and slash pine. Figure 3 shows that grass-derived SOC is a very high proportion of the total SOC in the surface layer (0–10 cm) of eight soil profiles. The lowest proportion of grass-derived SOC (35%) was in profile 1 under masson pine and the highest proportion (94.5%) in profile 9 under slash pine. Overall, the proportion of grass-derived SOC for the 8 profiles varied from 35% to 80%. Surface layer SOC mainly originated from former grass- and forest-derived SOC was generally much less in masson pine and slash pine 20 years after planting in previous grassland.

The highest proportion of forest-derived C occurred in the soil surface layer (0–10 cm) of profile 1 under masson pine. Figure 3 shows that the total proportion of forest-derived C for the top 10 cm represents 14%–64% and 2%–26% of the total SOC stock found under masson pine and slash pine, respectively. It also suggests that masson pine litter decomposes more quickly than that of slash pine, assuming the same decomposition rate for grass-derived SOC in the two plantations.



**Figure 3** Proportion of SOC derived from different photosynthetic pathways in surface soil

## 4 Discussion

### 4.1 $\delta^{13}\text{C}$ values of litter and soil

SOC originates from the litter of aboveground vegetation, so the litter from these vegetation species may affect soil  $\delta^{13}\text{C}$  variation. Logically, there should be no difference among litter and soil  $\delta^{13}\text{C}$  values of the three planted forests if the predominant vegetation was the same. However, the original vegetations were grass and shrubs before masson pine and fir forest establishment, respectively, so differences in litter  $\delta^{13}\text{C}$  values from the original vegetation were the main reason for the differences in soil surface  $\delta^{13}\text{C}$  values of the plantations. However, for profile 27 under fir forest, the unexpected distribution of soil  $\delta^{13}\text{C}$  values may be caused by disturbance during site preparation/plantation. For profile 11 of slash pine where the original vegetation was shrubs, soil  $\delta^{13}\text{C}$  values ranged from  $-20.5\%$  to  $-17.0\%$ , which was similar to profiles of masson pine. The results indicate that anthropogenic factors have affected the stable C isotope composition of profiles 11 and 27.

The  $\delta^{13}\text{C}$  signature of litter from masson pine, slash pine, fir forest and shrubs clearly shows a pure C<sub>3</sub> material with values ranging from  $-28\%$  to  $-29\%$  for these vegetation types (See Table 4). The addition of C<sub>3</sub> material to the soil led to a decrease in the soil  $\delta^{13}\text{C}$  values in the top 10 cm ( $\approx 4\%$ ) for the plantations. A similar trend was observed by de Alcântara *et al.* (2004) in a Typic Acric Red–Yellow Latosol where land use was transitional over 8 years from a previously savannah vegetation (predominantly C<sub>4</sub>) to riparian forests (predominantly C<sub>3</sub>). Although some studies have found  $^{13}\text{C}$  enrichment with soil depth (Nadelhoffer and Fry, 1988; Balesdent and Mariotti, 1996; Ehleringer *et al.*, 2000; de Alcântara *et al.*, 2004), this did not occur in deeper layers below 20 cm for masson pine and slash pine at QFES where the difference in subsurface soil  $\delta^{13}\text{C}$  values was very small between grassland and planted forests. Moreover, soil  $\delta^{13}\text{C}$  values of the planted forests decreased rapidly by 6–8‰ below 20 cm. Microbial fractionation and the Suess effect could not explain this trend of decreasing soil  $\delta^{13}\text{C}$  values in deeper layers, suggesting that historical changes in land use may be the main cause, and further indicating a strong influence of old C from C<sub>3</sub> vegetation.

In forests, the main cause of differences in the  $\delta^{13}\text{C}$  signature of litter and the underlying

soil is the persistence of  $C_4$  SOM in soil horizons (de Alcântara *et al.*, 2004). Compared to grassland, tree plantations are responsible for the decreases in soil  $\delta^{13}C$  in the upper 20 cm (mainly 0–10 cm) under masson pine and slash pine due to the addition of new  $C_3$  material to the soils. Low  $\delta^{13}C$  values were found between 20 and 120 cm in the plantation sites, showing the presence at depth of forest-derived humus from forest that preceded the grass as de Alcântara *et al.* (2004) mentioned. This earlier predominance of  $C_3$ -vegetation may have affected soil  $^{13}C$  values below 20 cm arising from deforestation and cultivation incorporating old  $C_3$  from native subtropical forest. Grasses would have had a minor influence in the deeper soil below 20 cm because their belowground contribution is small in comparison to that of the grass vegetation in the topsoil at QFES. Our results confirm the influence of the C source (old or new) on soil  $\delta^{13}C$  values. We believe that the present data reflect a historical forest–grass iterative transition at QFES.

## 4.2 Proportion of grass and forest derived SOC

SOM roughly inherits the isotope signature of the contributing vegetation, and therefore the C isotopic composition of SOM is a function of the relative contribution of  $C_3$  and/or  $C_4$  plants to the SOC pool (Midwood and Boutton, 1998; Boutton *et al.*, 1998). Under the planted forests in this study, input of  $C_3$  material had increased while input of  $C_4$  material had decreased leading to the decrease in  $^{13}C$  composition in the soil surface layer. In the upper layer (0–10 cm) of the soil of the masson pine and the slash pine converted from grassland, a mean 40%–80% of total C was derived from grassland soils after 20 years of plantation. Carbon replacement in the surface horizon in the masson pine and the slash pine ( $C_4$ - $C_3$  transition) ranged from 14% to 64% (mean 34%) and from 2% to 26% (mean 16%), respectively, suggesting a relatively slow turnover rate and slow forest litter decomposition. This indicates a low replacement of SOC after conversion of grass (predominantly  $C_4$ ) into masson pine and slash pine. Some studies have also found that the SOC stock increased rapidly after 30–80 years of plantation (Jenkinson, 1991; Lugo *et al.*, 1986). It means that it will take 40–100 years at QFES for the soil to reach a new equilibrium with the current tree species according to the mean forest-derived C replacement ratios in the soil surface layer calculated from the last 20 years. These results demonstrate that masson pine and slash pine plantations do not rapidly sequester SOC into long-term storage pools in subtropical China. This is consistent with a previous study of Richards *et al.* (2007).

In this study, forest-derived SOC of surface layer in profile 1 was a higher proportion than in the other soil profiles, while forest-derived SOC of surface layer in profile 9 was a lower proportion. The main reason for the difference may be the different decomposition rates of fresh litter for the different forest species.

## 4.3 Relationships between SOC concentrations and soil $\delta^{13}C$ values

The relationship between SOC contents and  $^{13}C$  composition is important to understand SOC accumulation and decomposition, but few studies have focused on quantifying this relationship (Michael, 2004). Piao *et al.* (2001), Harden *et al.* (2002) and Wynn *et al.* (2006) have used different approaches to analyze the relationship between SOC concentrations and  $\delta^{13}C$  values, in order to eliminate the influence of  $^{13}C$  component fractionation during SOM decomposition. Wynn *et al.* (2006) presented a strong relationship between the natural loga-

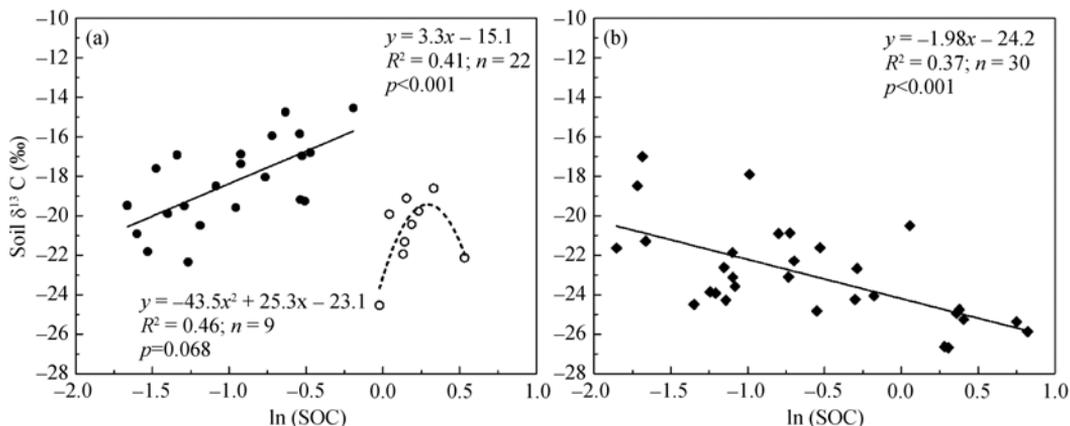
rhythm of SOC concentration and its isotopic composition, consistent with a model Rayleigh distillation of  $^{13}\text{C}$  in decomposing soil due to kinetic fractionation during decomposition. In this study, because of the mixture and selective decomposition of SOM with different  $^{13}\text{C}$  composition, and C isotope fractionation during SOM decomposition processes, depth profiles of soil  $^{13}\text{C}$  composition also represent to a certain extent the distribution characteristics under the different vegetation covers at QFES. This study attempted to analyze the relationship between SOC concentration and  $^{13}\text{C}$  composition and to understand the SOC turnover characteristics under the different vegetation covers.

Piao *et al.* (2001) indicated that there was a linear relationship between  $\delta^{13}\text{C}$  values and the reciprocal of SOC concentration in regions with a change of photosynthesis pathway and they used this relationship to correct SOC  $\delta^{13}\text{C}$  values in  $\text{C}_3$  vegetation. However, in this study, this linear relationship was not found in soil profiles undergoing the  $\text{C}_4 \rightarrow \text{C}_3$  transition. For soil profiles without a change in photosynthesis pathway, there was a conic relationship between soil  $\delta^{13}\text{C}$  values and SOC concentration for slash pine and fir forest. This means that soil  $\delta^{13}\text{C}$  values increased with the increasing reciprocal of the SOC concentration to a certain depth at first and then decreased with increasing reciprocal values of SOC. However, it was difficult to estimate the sources and turnover rate of SOC by using this relationship with the change of photosynthesis pathway.

Harden *et al.* (2002) found a linear relationship between soil  $\delta^{13}\text{C}$  values and the natural logarithm of SOC in tropical rainforests, in which the slope ranged from 1.39 to 3.80. If the slope of a particular profile is too large, it indicates that the soil profile contains a history of vegetation photosynthesis pathway conversion. In our study, for the soils of masson pine and slash pine ( $\text{C}_4 \rightarrow \text{C}_3$  transition),  $\ln(\text{SOC})$  and soil  $\delta^{13}\text{C}$  values presented a significant linear relationship when  $\ln(\text{SOC})$  increased from  $-2.0$  to  $-0.15$  ( $R=0.6$ , Figure 4a) and a parabolic relationship when  $\ln(\text{SOC})$  increased from  $-0.03$  to  $0.5$  ( $R=0.244$ , Figure 4a), in which soil  $\delta^{13}\text{C}$  values increased with increasing  $\ln(\text{SOC})$  until  $\ln(\text{SOC})$  reached  $-0.75$ , after which  $\delta^{13}\text{C}$  values began to decrease (Figure 4a). The linear regression function of  $\ln(\text{SOC})$  with soil  $\delta^{13}\text{C}$  was significant for fir forest and slash pine ( $\text{C}_3 \rightarrow \text{C}_3$ ), with a slope of 1.98 ( $R=-0.629$ , Figure 4b). It indicated that the model could be used to simulate the relationship between SOC concentration and soil  $\delta^{13}\text{C}$  values in regions without a change in photosynthesis pathway (Figure 4b). If there was a change of photosynthesis pathway, it would be difficult to estimate vegetation change history by using a slope change of a parabolic regression relationship. Furthermore, using this model also fails to explain the effect of vegetation photosynthesis pathway change on SOC dynamics.

#### 4.4 Biomass and SOC of planted forests in QFES

Before afforestation, the total area of forests at QFES was about 0.82 ha, 0.4% of the whole station. In 2003, the total forested area was 145.4 ha, 76.3% of the whole station. Planted slash pine covered the largest area and its aboveground biomass was  $75.4 \text{ t ha}^{-1}$ , but masson pine had the largest aboveground biomass per unit area at nearly  $87.7 \text{ t ha}^{-1}$  (Ma *et al.*, 2008). SOC contents in the masson pine and slash pine forests were not significantly different (Tables 1 and 2) and C isotope analysis indicated that the turnover rate of SOC under masson pine forests was slightly faster than that in slash pine forests. Therefore, we suggest that planting masson pine might sequester more C in subtropical regions of China than planting



**Figure 4** Relationship between the logarithmic value of SOC content and soil  $\delta^{13}\text{C}$  value

slash pine. Some studies have also illustrated that, in southern China, planted slash pine had smaller net primary productivity than natural forests, while masson pine was the opposite (Ma *et al.*, 2007; Tu *et al.*, 2008). This may partly confirm that masson pine has higher productivity in this region than slash pine.

When compared to the local natural vegetation, planted needle-leaved forests in this region are generally as productive as the original forests. A recent study showed that the mean values of biomass in evergreen broad-leaved forest, coniferous and broad-leaved mixed forest, pine forest, and Chinese fir forest of ecological service forests in Zhejiang Province, which is also located in China's eastern subtropical zone, were 89.19, 70.06, 51.25, and 54.15 t ha<sup>-1</sup> respectively (Zhang *et al.*, 2007). These values are mostly in the range of reported biomass values in QFES (Ma *et al.*, 2007), indicating that afforestation projects are practicable in this region.

More than 45 million ha of tree plantations in the subtropics have been established in southern China from the 1980s. However, large areas of slash pine and masson pine have suffered from low productivity, fragility and vulnerability of pure planted forests. In this region particularly, stocking volume, carbon density and structural stability of forests are not optimal, which is one reason why recent snowstorms in January 2008 destroyed an area of about 18 million ha of forests in the region with an estimated loss of 16 billion yuan (about €2 billion) (Stone, 2008). Hence, our study indicates that alternative plantation systems and strategies may need to be investigated to achieve greater soil C sequestration and to produce timber and other benefits in the future, as suggested by Richards *et al.* (2007).

#### 4.5 Uncertainties

Previous soil  $^{13}\text{C}$  studies have usually selected sites with good drainage, on gentle slopes and with even forest cover. In order to fully understand the soil  $^{13}\text{C}$  composition of the three planted forest types at QFES, most of the soil profiles were located on different slope grades and directions. Also, the time interval between sampling the soil and litter was overlong. Moreover, the litter  $\delta^{13}\text{C}$  value was the mean of mixed litter samples for each plantation, which partially eliminated the variability of litter  $\delta^{13}\text{C}$  value and reduced the precision of sampled litter  $\delta^{13}\text{C}$  values in situ. These conditions could lead to differences in the origin of

SOC in the plantations. It also may have contributed to the weak linear relationship between SOC concentrations and soil  $\delta^{13}\text{C}$  values.

In this study, the influence of clay content on soil  $\delta^{13}\text{C}$  was ignored because of a lack of sufficient soil texture data. The proportions of grass and forest derived C in the deeper layers were not calculated due to different depth partitioning of the soil samples. Meanwhile, we took SOC  $\delta^{13}\text{C}$  values in grassland as a reference value to calculate the proportion of different vegetation-derived SOC in soil surface layers, so it was difficult to remove the possible influence of fractionation during SOM decomposition processes. These factors limited further analysis on the characteristics of soil C turnover.

## 5 Conclusions

In this study we used stable isotopes ( $\delta^{13}\text{C}$ ) to study the impacts of afforestation at QFES on SOC dynamics in red soils and its implications for current SOC status and trends in subtropical regions in China. Historical land use had a greater effect on SOC turnover in previous subtropical grassland than current plantations. Differences in the  $\delta^{13}\text{C}$  composition of vegetation before plantation establishment were the main reason for the differences in soil  $\delta^{13}\text{C}$  composition in three planted forests at QFES. Conversion of grassland into pine forests has caused significant changes in SOM dynamics, such as an increase in C and N stocks (Tables 1 and 2). The addition of forest litter ( $\delta^{13}\text{C}$ :  $-29.49\%$  to  $-28.03\%$ ) to a  $\text{C}_4$ -predominant ecosystem decreased the  $\delta^{13}\text{C}$  signature in the soil profile (from 20–30 cm). After changing from a predominantly belowground (grass) into a largely aboveground C input (forest), although  $\delta^{13}\text{C}$  values decreased in the topsoil due to the deposition of  $\text{C}_3$  litter on the soil surface, the presence of residual grass C led to higher  $\delta^{13}\text{C}$  values in the topsoil under masson pine and slash pine. Below 20 cm,  $\delta^{13}\text{C}$  values greatly decreased with soil depth except for fir forest. Land use history before grassland had a strong influence on the behavior of the  $^{13}\text{C}/^{12}\text{C}$  ratio with soil depth. Results from soil profiles which had had no change of photosynthesis pathway, suggested that fractionation of the stable C isotope existed during SOM decomposition. From comparisons of soil C:N ratios, the low proportion of forest-derived SOC was mainly caused by slow decomposition of forest litter rather than by small quantities of forest litter. Further analysis of the relationship between SOC and soil  $\delta^{13}\text{C}$  values by adopting different models is recommended.

The SOC turnover in masson pine (with a mean 34% of replacement in the top 10 cm soil layer after 20 years) was relatively faster than that in slash pine (16% of replacement) in subtropical conditions. It indicates that SOC in the surface layer was mainly derived from grass before plantation establishment. It also means that replacement of C in the soil  $^{13}\text{C}$  signature may occur over shorter periods of time for masson pine than slash pine. This finding will provide insights into the potential of masson pine and slash pine to sequester C in order to offset atmospheric  $\text{CO}_2$  emissions in subtropical regions of China. Density fractionation allows a better understanding of SOM dynamics (Alcântara *et al.*, 2004; Richards *et al.*, 2007), so we may adopt a density fractionation approach to further elucidate forest SOC dynamics in subtropical China. Also, combining the techniques of stable and radioactive C isotopes may quantify the turnover rate of SOC and allow exploration of the mechanism of  $^{13}\text{C}$  and  $^{14}\text{C}$  enrichment in SOC, and further evaluate C sequestration potential in

forest ecosystems.

## Acknowledgments

The authors would like to thank B. Chen (School of Geography and Earth Sciences, McMaster University, Canada) and X. Xu (Queen College, City University of New York) for their invariably thoughtful and helpful comments on papers.

## References

- Aerts R, 1997. Climate, leaf litter chemistry and leaf litter decomposition in terrestrial ecosystems, a triangular relationship. *Oikos*, 79: 439–449.
- Balesdent J, Mariotti A, 1996. Measurement of soil organic matter turnover using  $^{13}\text{C}$  natural abundance. In: Boutton T W, Yamasaki S (eds.), *Mass Spectrometry of Soil*. New York: Marcel Dekker, 83–111.
- Bernoux M, Cerri C C, Neill C *et al.*, 1998. The use of stable carbon isotopes for estimating soil organic matter turnover rates. *Geoderma*, 82: 43–58.
- Biedenbender S H, McClaran M P, Quade J *et al.*, 2004. Landscape patterns of vegetation change indicated by soil carbon isotope composition. *Geoderma*, 119: 69–83.
- Bird M I, Chivas A R, Head J, 1996. A latitudinal gradient in carbon turnover times in forest soils. *Nature*, 381: 143–145.
- Boutton T W, Archer S R, Midwood A J *et al.*, 1998.  $\delta^{13}\text{C}$  values of soil organic carbon and their use in discriminating vegetation change in a subtropical savanna ecosystem. *Geoderma*, 82: 5–41.
- Canadell J G, Raupach M R, 2008. Managing forests for climate change mitigation. *Science*, 320: 1456–1457.
- Cerri C C, Feller C, Balesdent J *et al.*, 1985. Application du tracage isotopique naturel en  $\delta^{13}\text{C}$  à l'étude de la dynamique de la matière organique dans les sols. *Comptes Rendus de L'Académie des Sciences. Paris*, 300: 423–428.
- Cheng X, Luo Y, Chen J *et al.*, 2006. Short-term C4 plant *Spartina alterniflora* invasions change the soil carbon in C3 plant-dominated tidal wetlands on a growing estuarine Island. *Soil Biol. Biochem.*, 38: 3380–3386
- Chiang P N, Wang M K, Chiu C Y *et al.*, 2004. Change in the grassland-forest boundary at Ta-Ta-Chia long term ecological research (LTER) site detected by stable isotope ratio of soil organic matter. *Chemosphere*, 54: 217–224.
- Dalal R C, Carter J O, 2000. Soil Organic matter dynamics and carbon sequestration in Australian tropical soils. In: Lal R, Kimble, J M, Stewart B A (eds.), *Global Climate Change and Tropical Ecosystems*. CRC Press LLC, Boca Raton, FL, 285–314.
- de Alcântara F A, Buurman P, Furtini A E *et al.*, 2004. Conversion of grassy cerrado into riparian forest and its impact on soil organic matter dynamics in an Oxisol from southeast Brazil. *Geoderma*, 123(3/4): 305–317.
- Diels J, Vanlauwe B, Sanging N *et al.*, 2001. Temporal variation in plant  $\delta^{13}\text{C}$  values and implications for using the  $^{13}\text{C}$  technique in long-term soil organic matter studies. *Soil Biol. Biochem.*, 33: 1245–1251.
- Ehleringer J R, Buchmann N, Flanagan L B, 2000. Carbon isotope ratios in belowground carbon cycle processes. *Ecol. Appl.*, 10: 412–422.
- Farquhar G D, O'Leary M H, Berry J A, 1982. On the relationship between carbon isotope discrimination and the intercellular carbon dioxide concentration in leaves. *Aust. J. Plant Physiol.*, 9: 121–137.
- Feigl B J, Mellilo J, Cerri C, 1995. Changes in the origin and quality of soil organic matter after pasture introduction in Rondônia (Brazil). *Plant and Soil*, 175: 21–29.
- Garten Jr C T, Cooper L W, Post W M *et al.*, 2000. Climate controls on forest soil C isotope ratios in the southern Appalachian mountains. *Ecology*, 81: 1108–1119.
- Harden J W, Fries T, Pavich M, 2002. Cycling of beryllium and carbon through hillslope soils in Iowa. *Biogeochemistry*, 60(3): 317.

- Hobbie E A, Johnson M G, Rygielwicz P T *et al.*, 2004. Isotopic estimates of new carbon inputs into litter and soils in a four-year climate change experiment with Douglas-fir. *Plant and Soil*, 259: 331–343.
- Jenkinson D S, 1991. The Rothamsted long-term experiments: Are they still of use? *Agron. J.*, 83(1): 2–10.
- Lefroy R D B, Graeme J B, Strong W M, 1993. Changes in soil organic matter with cropping as measured by organic carbon fractions and  $^{13}\text{C}$  natural isotope abundance. *Plant and Soil*, 155/156: 399–402.
- Lugo A E, Sanchez A J, Brown S, 1986. Land use and organic carbon content of some subtropical soils. *Plant and Soil*, 96: 185–196.
- Ma Z, Liu Q, Wang H *et al.*, 2008. Observation and modeling of NPP for *Pinus elliottii* plantation in subtropical China. *Science in China (Series D)*, 51(7): 955–965.
- Ma Z Q, Liu Q J, Xu W J *et al.*, 2007. Carbon storage of artificial forest in Qianyanzhou, Jiangxi province. *Scientia Silvae Sinicae*, 43(11): 1–7. (in Chinese)
- Michael A P, 2004. A theoretical analysis of steady state  $\delta^{13}\text{C}$  profiles of soil organic matter. *Global Biogeochem. Cycles*, 18: 1–13.
- Midwood A J, Boutton T W, 1998. Soil carbonate decomposition by acid has little effect on  $\delta^{13}\text{C}$  of organic matter. *Soil Biology and Biochemistry*, 30: 1301–1307.
- Nadelhoffer K J, Fry B, 1988. Controls on natural nitrogen-15 and carbon-13 abundance in forest soil organic matter. *Soil Sci. Soc. Am. J.*, 52: 1633–1640.
- Paul K I, Polglase P J, Nyakuengama J G *et al.*, 2002. Change in soil carbon following afforestation. *For. Ecol. Manage.*, 168: 241–257.
- Piao H, Liu Q, Yu D *et al.*, 2001. Origins of soil organic carbon with the method of natural  $^{13}\text{C}$  abundance in maize fields. *Acta Ecologica Sinica*, 21(3): 434–439. (in Chinese)
- Stone R, 2008. Ecologists report huge storm losses in China's forests. *Science*, 319: 1318–1319.
- Richards A E, Dalal R C, Schmidt S, 2007. Soil carbon turnover and sequestration in native subtropical tree plantations. *Soil Biol. Biochem.*, 39: 2078–2090.
- Roscoe R, Buurman P, Velthorst E J *et al.*, 2000. Effects of fire on soil organic matter in a “Cerrado sensu stricto” from southeast Brazil as revealed by changes in  $\delta^{13}\text{C}$ . *Geoderma*, 95: 141–160.
- Silver W L, Kueppers L M, Lugo A E *et al.*, 2004. Carbon sequestration and plant community dynamics following reforestation of tropical pasture. *Ecol. Appl.*, 14: 1115–1127.
- Smith B N, 1971. Carbon isotope ratios of respired  $\text{CO}_2$  from castor bean, peanut, pea, radish, squash, sunflower and wheat seedlings. *Plant Cell Physiol.*, 12: 451–455.
- Staddon P L, 2004. Carbon isotopes in functional soil ecology. *Trends Ecol. Evol.*, 19(3): 148–154
- Tu J, Liu Q J, 2008. Benefits of  $\text{CO}_2$  fixation and  $\text{O}_2$  production of artificial *Pinus elliottii* forest in the red soil and hilly region of subtropical China. *Resources and Environment in the Yangtze Basin*, 17(1): 30–35. (in Chinese)
- Wang S, Huang M, Shao X *et al.*, 2004a. Vertical distribution of soil organic carbon in China. *Environment Management*, 33: S201–S209.
- Wang S, Liu J, Yu G *et al.*, 2004b. Effects of land use change on the storage of soil organic carbon: A case study of Qianyanzhou Forest Experimental Station in China. *Clim. Change*, 67(2/3): 247–255.
- Wilding L P, Drees L R, Nordt L C, 2001. Spatial variability: enhancing the mean estimate of organic and inorganic carbon in a sampling unit. In: Lal R, Kimble J M, Stewart B A eds. *Advances in Soil Science: Assessment Methods for Soil Carbon*. Boca Raton, FL, USA: CRC Press, 69–104.
- Wynn J G, Harden J W, Fries T L, 2006. Stable carbon isotope depth profiles and soil organic carbon dynamics in the lower Mississippi Basin. *Geoderma*, 131: 89–109.
- Zhang J, Ge Y, Chang J *et al.*, 2007. Carbon storage by ecological service forests in Zhejiang Province, subtropical China. *For. Ecol. Manage.*, 245(1–3): 64–75.