Journal of
Plant Ecology

VOLUME 6, NUMBER 3,
PAGES 240–252

JUNE 2013

doi:10.1093/jpe/rts023

Advance Access publication
14 August 2012

available online at
www.jpe.oxfordjournals.org

Changes in soil carbon sequestration and soil respiration following afforestation on paddy fields in north subtropical China

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Abstract

Aims
Although many studies have reported net gains of soil organic carbon (SOC) after afforestation on croplands, this is uncertain for Chinese paddy rice croplands. Here, we aimed to evaluate the effects of afforestation of paddy rice croplands on SOC sequestration and soil respiration (R_s). Such knowledge would improve our understanding of the effectiveness of various land use options on greenhouse gas mitigation in China.

Methods
The investigation was conducted on the Chongming Island, north subtropical China. Field sites were reclaimed from coastal salt marshes in the 1960s, and soils were homogeneous with simple land use histories. SOC stocks and R_s levels were monitored over one year in a paddy rice cropland, an evergreen and a deciduous broad-leaved plantation established on previous paddy fields and a reference fallow land site never cultivated. Laboratory incubation of soil under fast-changing temperatures was used to compare the temperature sensitivity (Q_{10}) of SOC decomposition across land uses.

Important Findings
After 15–20 years of afforestation on paddy fields, SOC concentration only slightly increased at the depth of 0–5 cm but decreased in deeper layers, which resulted in a net loss of SOC stock in the top 40 cm. Seasonal increase of SOC was observed during the rice-growing period in croplands but not in afforested soils, suggesting a stronger SOC sequestration by paddy rice cropping. However, SOC sequestered under cropping was more labile, as indicated by its higher contents of dissolved organic carbon and microbial biomass. Also, paddy soils had higher annual R_s than afforested soils; R_s abruptly increased after paddy fields were drained and plowed and remained distinctively high throughout the dry farming period. Laboratory incubation revealed that paddy soils had a much higher Q_{10} of SOC decomposition than afforested soils. Given that temperature was the primary controller of R_s in this region, it was concluded that despite the stronger SOC sequestration by paddy rice cropping, its SOC was less stable than in afforested systems and might be more easily released into the atmosphere under global warming.

Keywords: land use • soil organic matter (SOC) • temperature sensitivity • Q_{10} • Chongming Island.

Received: 7 March 2012 Revised: 4 June 2012 Accepted: 21 June 2012

INTRODUCTION

Historically, deforestation has been mainly responsible for a net release of approximately 40–90 pentagrams (Pg) of C from cultivated soils into the atmosphere (Cole *et al*. 1997; Houghton 1999), accelerating global climate warming. Afforestation on agricultural soils has been proposed as an effective measure to offset greenhouse gas (GHG) emission by sequestering C back into the soil, as stated by Article 3.3 of the Kyoto Protocol (Jandl *et al*. 2007). The potential of C sequestration in afforested soils is guaranteed by reduced soil disturbances, better protection of soil organic carbon (SOC; Six *et al.*

2002) and increased return of litter into the soil (Jandl *et al.* 2007; Laganière *et al.* 2010).

However, although the immediate effect of afforestation on C sequestration by living biomass is evident, its consequences for SOC are uncertain and not well known. Most studies have found increased SOC stocks following afforestation on croplands (Laganière *et al.* 2010; Morris *et al.* 2007). Paul *et al.* (2002) noticed that during the first 10 years after afforestation on previous cropping soils, SOC increased by 0.87% per year in the top 30 cm or 1.88% per year in the top 10 cm. By a meta-analysis of long-term experimental data, Guo and Gifford (2002) found that afforestation on croplands increased SOC by 18%. In contrast, Degryze *et al.* (2004) observed no difference in SOC stocks within the top 50 cm of soil between a conventionally tilled cropland and an afforested poplar system . The inconsistent results may depend on many factors, such as previous land uses and specific management regimes (Laganière *et al.* 2010; Post and Kwon, 2000).

China has the largest area of tree plantation in the world (Huang *et al.* 2007). Afforestation has created an important C sink for China, and soil is considered to play an equivalent role to vegetation in this process (Houghton and Hackler, 2003). However, previous studies implied that afforestation on croplands would not necessarily lead to net gains of SOC in China. For example, comparable 1-m SOC densities between paddy and forest soils have been found in eastern China (Li and Zhao, 2001) and SOC levels (0–20 cm) even increased after conversion of woodlands to paddy fields in subtropical China (Iqbal *et al.* 2009; Wang *et al.* 2006). Paddy rice croplands cover 26% of China's total cropland area and showed a mean SOC sequestration rate of 0.40 t C ha^{-1} yr^{-1} since the early 1980s (Pan *et al.* 2003), as high as that of afforested soils as estimated by Post and Kwon (2000). However, although it has been widely recognized that paddy soils have higher SOC sequestration than cultivated dryland soils (Pan *et al.* 2003; Xie *et al.* 2007), few studies have explicitly addressed SOC sequestration and associated processes of paddy soils in comparison with afforested soils (Cai 1996; Iqbal *et al.* 2008; Iqbal *et al.* 2009; Wang *et al.* 2006). Such knowledge would contribute to a better assessment of the GHG mitigation potential of afforestation practices in China and provide useful information to policy makers.

Despite the significant SOC sequestration, paddy soils were also reported to exhibit a higher soil C emission (i.e. soil respiration, R_s) than tree plantations (Iqbal *et al.* 2008), suggesting a possibly lower stability of SOC sequestered by rice cropping. Because temperature has been considered the primary determinant of R_s in subtropical China (Iqbal *et al.* 2008; Lou *et al.* 2004; Sheng *et al.* 2010), it is important to evaluate whether the SOC of paddy soils would be more sensitive to temperature changes than that of afforested soils, especially considering current global warming concerns (IPCC 2001).

In this study, we compared SOC sequestration and soil respiration of paddy rice croplands and two tree plantations afforested on previous paddy fields on the Chongming Island, subtropical China. These land uses were established on young homogenous soils (ages ≤40 years) reclaimed from coastal wetlands and had simple land use histories. A fallow land under secondary succession was chosen as the reference because it had never been cultivated since reclamation from wetlands. SOC storage and R_s levels were measured. Also, laboratory incubation under fast-changing temperatures was conducted to compare the temperature sensitivity of SOC decomposition between paddy and afforested soils. The objectives were to (i) compare the characteristics of SOC sequestration among paddy and afforested soils and (ii) to compare CO_2 emission from these soils and the sensitivity of their SOC decomposition to possible changes in temperature.

MATERIALS AND METHODS

Site description

The Chongming Island is located in the Yangtze Estuary, China (31°27′–31°51′N, 121°09′–121°54′E). Formed on estuarine sediments, the island has typical flat plain topography. It has a typical monsoon climate, with a hot and rainy summer and a relatively cold and dry winter. The mean annual temperature is 15.3°C and annual precipitation amounts to 1003.7 mm (Zhou and Ji 1989). The hottest and wettest month is July, with a mean temperature of 27.5°C and precipitation of 166.9 mm; the coldest and driest month is January, with a mean temperature of 2.8°C and precipitation of 42.3 mm. The mean pH measured in water is 8.20 ± 0.15 (Zhou and Ji 1989). Reclamation of coastal wetlands has been very frequent and has doubled the island area in the past century (He and Gu 2003). Traditionally, newly reclaimed lands have been used as fish farms or paddy rice croplands.

Experimental design

Four types of land uses were selected in the eastern part of the Chongming Island, including a paddy rice cropland (CL) rotating between rice and barley cultivation; two plantations on previous paddy fields, one an evergreen broad-leaved *Cinnamomum camphora* (CC) plantation and the other a deciduous broad-leaved *Koelreuteria bipinnata* (KB) plantation; and a fallow land (FL). All sites except FL were located in the Qianshao Farm reclaimed from coastal salt marshes in the 1960s, and they were all within 5 km from each other. CL has persisted since reclamation, whereas CC and KB were converted from paddy rice croplands 15 and 20 years ago, respectively (Table 1). Every year in spring and autumn, CC and KB were fertilized with inorganic compound fertilizers (containing N, P and K). FL was reclaimed from salt marshes in 1998 and had been used for fish farming until 2005. Thereafter, it was abandoned and secondary succession began, with the vegetation being dominated by *Phragmites australis*. FL was chosen as a reference site because it had never been cultivated since reclamation and hence was most close to natural soils in this area. All soils had a similar silty loam texture (Table 1).

Table 1: Selected soil properties (0–20 cm) and soil CO_2 flux under different land uses[a].

Land use (years)[b]	Bulk density ($g\ cm^{-3}$)	SOC ($t\ C\ ha^{-1}$)	TN ($t\ C\ ha^{-1}$)	DOC ($mg\ kg^{-1}$)	SMBC ($kg\ C\ ha^{-1}$)	SMBC/SOC (%)	Annual mean R_s ($\mu mol\ m^{-2}\ s^{-1}$)[c]	Soil texture (%)[d]		
								Sand	Silt	Clay
CC (20)	1.35 AB	25.53 B	2.68 A	14.89 B	292.48 B	1.14 B	2.97 (1.39) A	4.60	76.50	18.90
KB (15)	1.32 B	25.26 B	2.75 A	18.98 AB	313.54 B	1.24 B	1.74 (1.12) B	4.60	72.70	22.70
CL (40)	1.26 B	31.78 A	3.13 A	24.38 A	566.58 A	1.79 A	2.67 (4.83) A	2.20	73.60	24.20
FL (3)	1.27 B	18.01 C	1.77 B	15.64 B	135.43 C	0.75 C	2.66 (2.78) A	5.90	68.30	25.80

[a]In each column, treatments followed by the same uppercase letter were not significantly different from each other. [b]Numbers in parentheses were the persisting years of each land use. [c]Data in parentheses were the mean R_s averaged over January–May, 2008. [d]One composite sample per land use was used in the analysis of texture.

Three replicate plots, *ca.* 30×30 m^2 in area and about 50–100 m apart, were set up under each land use. Soil sampling and measurement of soil respiration were conducted repeatedly in each plot. For measurement of soil respiration, six polyvinyl chloride collars (20 cm in diameter and 12 cm in height) per plot were inserted 9 cm deep into soil. All collars were fixed *in situ* throughout the experiment, except in CL, where due to the interferences by agricultural practices we had to insert collars 1 day before measurement.

Soil sampling and analysis

To determine the 1-m SOC stocks under each land use, six soil cores were collected from depths of 0–5, 5–10, 10–20, 20–40, 40–60, 60–80 and 80–100 cm with a self-made steel corer (5.4 cm inner diameter) in September 2007. The sampling was conducted at three places in every 30×30 m^2 plot under each land use. Soil cores from the same plots were homogenized and this resulted in three composite samples per depth per land use. Meanwhile, three intact soil cores were collected from each plot to measure soil bulk densities. To investigate the seasonal dynamics of SOC and soil microbial biomass carbon (SMBC), topsoil samples at 0–5, 5–10 and 10–20 cm were collected seasonally in December 2007 and in March, June and September of 2008.

Soil bulk density was measured by oven-drying intact soil cores at 105°C to constant weight. SOC concentration was determined by wet oxidation with $K_2Cr_2O_7$ and titration with $FeSO_4$ (Lu 1999). Total nitrogen (TN) was only analyzed for soils collected in December 2007, with a FlashEA 1112 NC analyzer (Thermo Fisher Scientific Inc., Italy). Before analyses of SOC and TN, samples were air-dried, handpicked to remove roots, litters or soil animals and then ground to pass a 0.15-mm sieve. SMBC was determined by the chloroform fumigation–extraction method with K_2SO_4, using triplicate 2-mm-sieved samples (25 g) that had been adjusted to 50% water-holding capacity (Wu *et al.* 2006). Carbon extracted with 0.5 M $KaSO_4$ prior to fumigation for determining SMBC was taken as the dissolved organic carbon (DOC). DOC was also determined by the $K_2Cr_2O_7$ wet oxidation method.

Measurement of soil respiration

From November 2007 to November 2008, soil respiration rates (R_s) under all land uses were measured monthly using an LI-8100 automated soil CO_2 flux system (Li-COR Corporate, USA). Field measurement for each land use was carried out once every month between 11:00 and 14:00 hrs on a sunny day. For each collar fixed in the field, measurements of R_s were repeated three times. Meanwhile, soil temperature and moisture at 5 cm were measured using a thermocouple right beside the collars.

Considering that there might be dramatic changes in soil respiration when paddy fields were drained and plowed for dry farming, we monitored 24-h respiration in CL on 2 days that had similar weather, one shortly after the soil was drained but not plowed (17 November 2008) and the other immediately after the soil was plowed (3 December 2008). In each plot, soil respiration was measured with LI-8100 for every collar once every 2 hours for 24 h, with soil temperature and moisture recorded at the same time.

Laboratory incubation of soil

To estimate the temperature sensitivity of SOC decomposition, laboratory incubation was conducted under fast-changing temperatures following the procedure of Chen *et al.* (2010). For incubation, we used only surface soils (0–20 cm) collected from all land uses in December 2008, i.e. shortly after rice harvesting. Soils were sampled at that time to see whether SOC sequestered under rice cropping was more sensitive to temperature than that in afforested soils, which had just undergone the growing season. Before incubation, soils were handpicked to remove roots and sieved using a 2-mm mesh. Soil moisture was adjusted to the field-holding capacity by adding deionized water. Then, soils equivalent to a dry weight of 15 g were put in 120-ml jars, loosely capped with soft porous plugs and preincubated at 25°C for 1 week. When incubation began, a cryogenic water bath was used to control jar temperatures. The water bath was first kept at 30°C for 1 hour, and then 5 ml of gas was sampled using a 5-ml syringe. The jars were immediately refilled with 5 ml of CO_2-free gas to maintain the original air pressure and incubated at the same temperature for another hour, before another 5 ml of gas was sampled. After the second gas sampling was completed, the temperature was decreased to 25°C within 1 hour and the procedures above were repeated. The incubation temperature was decreased in steps of 5°C to an ultimate temperature of 5°C. Throughout the incubation, there was always sufficient

O_2 in the jars. CO_2 concentration of gas samples were analyzed with a gas chromatograph (Agilent 6890, Agilent Corp., USA).

Data analysis

The van't Hoff equation was used to fit the relationship between R_s and temperature (Davidson *et al.* 2006):

$$R_s = R_0 \exp(bt) \quad (1)$$

where R_s is soil respiration rate (μmol m^{-2} s^{-1}), t is soil temperature (°C) at 5 cm and R_0 and b are fitted parameters. Respiration rates under laboratory incubation (i.e. heterotrophic respiration, R_h) were expressed either as CO_2-C emission per unit soil weight per unit time (μg CO_2-C g $soil^{-1}$ h^{-1}), or as the "specific R_h", i.e. emission per unit SOC per unit time (μg CO_2-C g C^{-1} h^{-1}). The temperature sensitivity of SOC decomposition was represented by Q_{10} values, i.e. the factor by which respiration is multiplied when temperature increases by 10°C (Davidson *et al.* 2006). It was calculated as shown in Equation 2:

$$Q_{10} = \exp(10b) \quad (2)$$

One-way ANOVA was used to compare SOC concentrations, total nitrogen (TN) and SOC stocks within the top 1 m of soil across land uses, with Duncan's test as the post hoc method. Repeated-measures ANOVA was applied (i) to detect seasonal changes in SOC and SMBC for each land use and (ii) to compare *in-situ* R_s, incubation-derived R_h or specific R_h, DOC, SMBC and SMBC/SOC ratio across land uses, with land use being the between-subject factor and season the within-subject factor. All statistical analyses were performed with SPSS 13.0 (SPSS Inc., USA).

RESULTS

Soil temperature and moisture

Soil temperature at 5-cm depth followed a common seasonal pattern under all land uses, with peak values in June (26.9°C) and lowest values in January (5.9°C; Fig. 1a). Despite the seasonal covariation of temperature and rainfall (Fig. 1b), there was no significant correlation between soil temperature and moisture at 5 cm (indicated by water-filled porosity, *WFP*). A seasonal pattern of *WFP* was not evident, except for CL, wherein the soil was submerged in summer and autumn and was drained in winter and spring (Fig. 1c).

SOC stocks and lability

Land use effect on SOC concentration was significant at each depth within 0–80 cm (Fig. 2a) but barely significant at 80–100 cm ($P = 0.06$). The order of SOC concentrations across land uses was depth dependent. At 0–5 cm depth, the two plantations showed slightly higher SOC concentrations than CL ($P < 0.01$), whereas CL had significantly higher SOC concentrations ($P < 0.01$) in soil

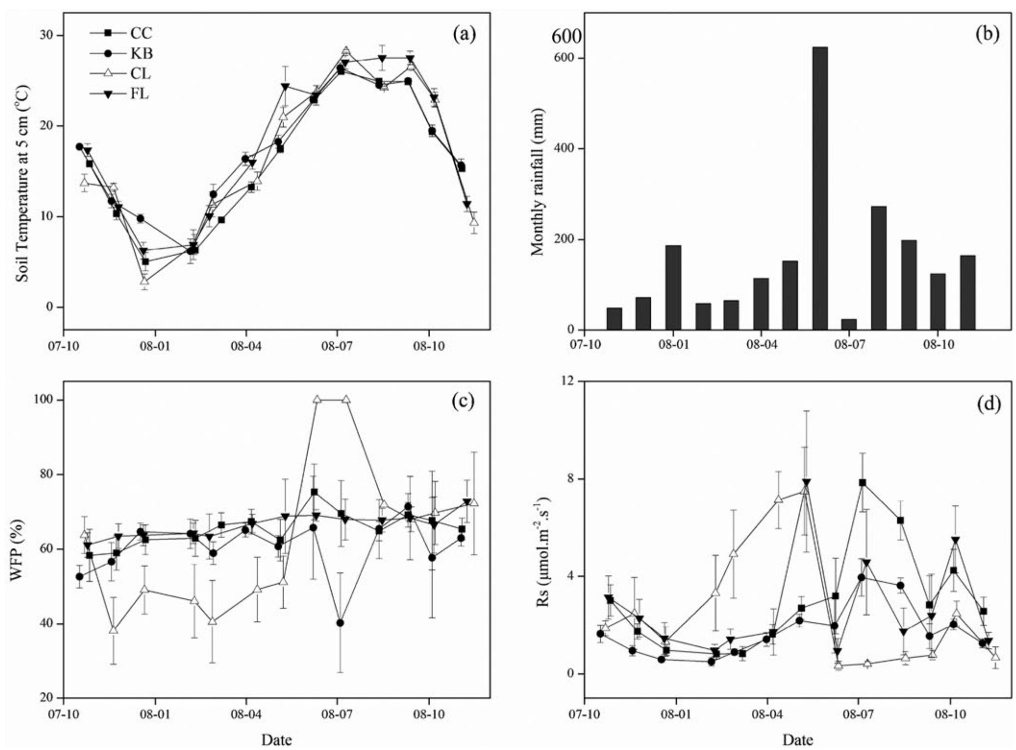

Figure 1: soil temperature at 5 cm depth (a), monthly rainfall (b), soil water-filled porosity (WFP) at 5 cm depth (c) and mean monthly soil respiration rates (R_s) under different land uses (d). Error bars represent standard errors. Symbols are the same in (a)–(d). CC and KB refer to plantations of *Cinnamomum camphora* and *Koelreuteria bipinnata*, respectively; CL: paddy rice croplands; FL: fallow land. The format of dates on X-axes is year-month.

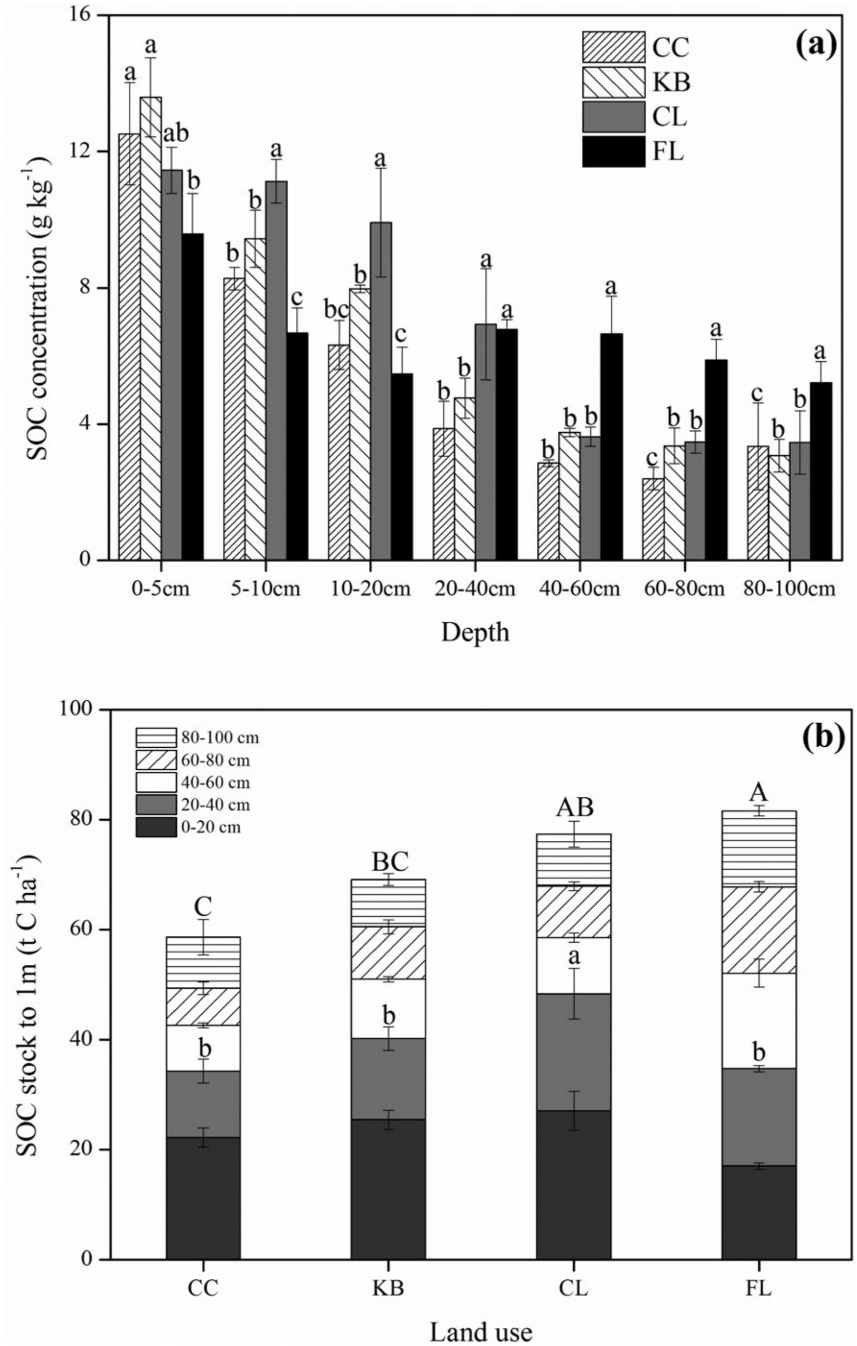

Figure 2: Profiles of SOC concentrations (a) and SOC stocks (b) up to the depth of 1 m. Error bars represent standard errors. Land uses with the same letters are not significantly different in SOC concentrations at each depth or in SOC stocks up to 1 m depth. See Fig. 1 for abbreviations.

layers within 5–40 cm. The two plantations did not differ significantly from each other in SOC concentration at different depths (Fig. 2a). Compared with other land uses, FL showed significantly lower SOC concentrations at 0–20 cm depths but higher concentrations between 40 and 100 cm.

SOC stocks in the top 20 cm of soil followed the order CL~KB>CC>FL ($P < 0.01$). For the top 40 cm, however, CL had an SOC stock of 47.92 t C ha^{-1}, compared with 37.24 t C ha^{-1} of the plantations and 38.58 t C ha^{-1} in FL. ANOVA showed that SOC stocks in the top 40 cm followed the order CL>KB~FL~CC ($P < 0.01$). Between 40 and 100 cm, SOC differed in the order FL>CL~KB~CC, i.e. CL had SOC stocks similar to that of the plantations but lower stocks than FL in horizons deeper than 40 cm. If calculated to the depth of 100 cm, CL had an SOC stock close to that in FL but higher than that in the plantations (Fig. 2b).

CL showed a significant increase of SOC storage in the top 20 cm after entering the rice-growing period, i.e. in June and

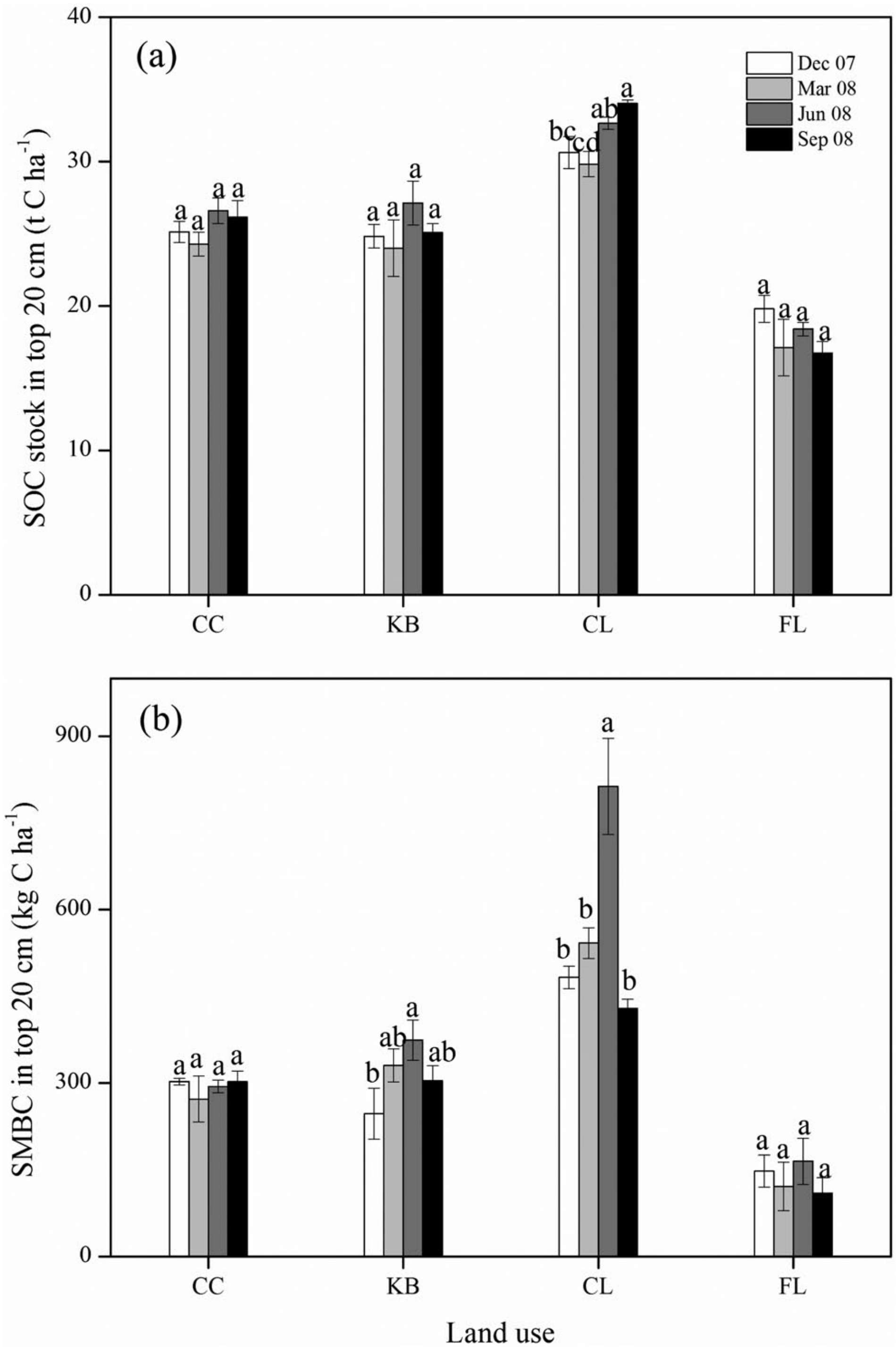

Figure 3: Seasonal changes of SOC (a) and soil microbial biomass carbon (SMBC) (b) within the top 20 cm of soil. Error bars represent standard errors. For each land use, SOC or SMBC levels with the same letters are not significantly different between seasons. See Fig. 1 for abbreviations.

September, 2008 (Fig. 3a). This temporal trend was not evident under any other land use.

The mean DOC concentration of CL was 24.38 mg kg^{-1}, significantly ($P < 0.01$) higher than those of other land uses (15.64–18.98 mg kg^{-1}, Table 1). CL also had the highest SMBC among all land uses, whereas FL had the lowest ($P < 0.0001$; Fig. 3b). SMBC and the SMBC/SOC ratio in CL were both significantly higher than those in any other site (Table 1). It was notable that SMBC in CL showed an abrupt increase in June shortly after rice was planted, which was not observed in any other land use.

Soil respiration rates in different land uses

R_s of the plantations was generally higher in summer (June–August) and lower in winter (December–February), with peak values in July and the lowest in February (Fig. 1d). The highest values of R_s in FL occurred in May, although the lowest was also observed in February as in plantations. The CL site showed a seasonal pattern of R_s quite distinct from others, with R_s gradually increasing from January to May and then decreasing sharply during the rice-growing period (June–November). Overall, soil CO_2 emission from CL mainly occurred in January–May and was minimal during June–September (Fig. 1d).

At the time when CL had been drained but soil had not been plowed to grow barley, mean R_s was as low as 1.59 ± 0.58 μmol m^{-2} s^{-1}. However, once the soil was plowed, R_s climbed up to 3.75 ± 1.94 μmol m^{-2} s^{-1}, i.e. R_s increased by 135.85% and became more variable. The highest value (8.28 μmol m^{-2} s^{-1}) after ploughing was even close to the R_s in May 2008, when the R_s of CL peaked (Fig. 1d). This ploughing-induced increase of R_s is clearly shown in Fig. 4.

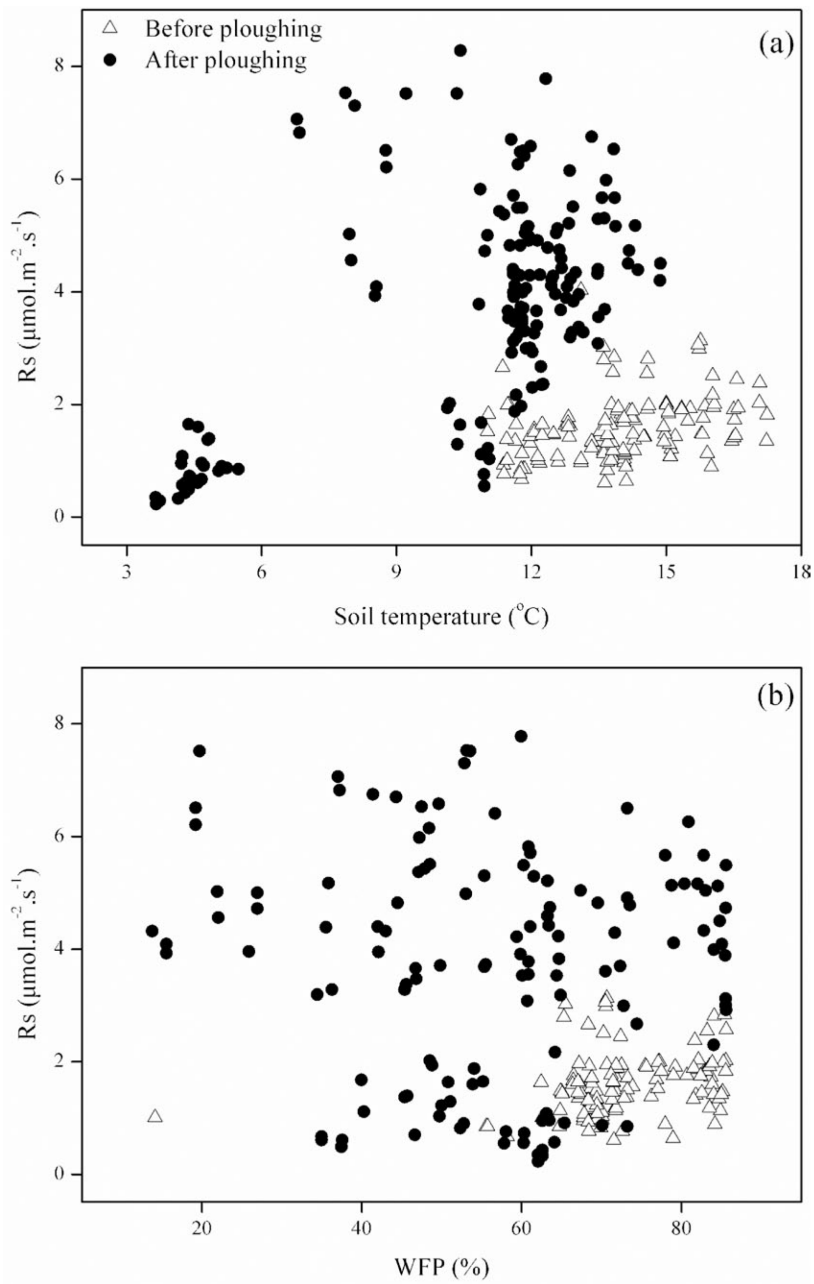

Figure 4: Relationship of soil respiration rates (R_s) to soil temperature (a) and to soil water-filled porosity (WFP) (b) before and after ploughing of paddy rice croplands after rice harvesting.

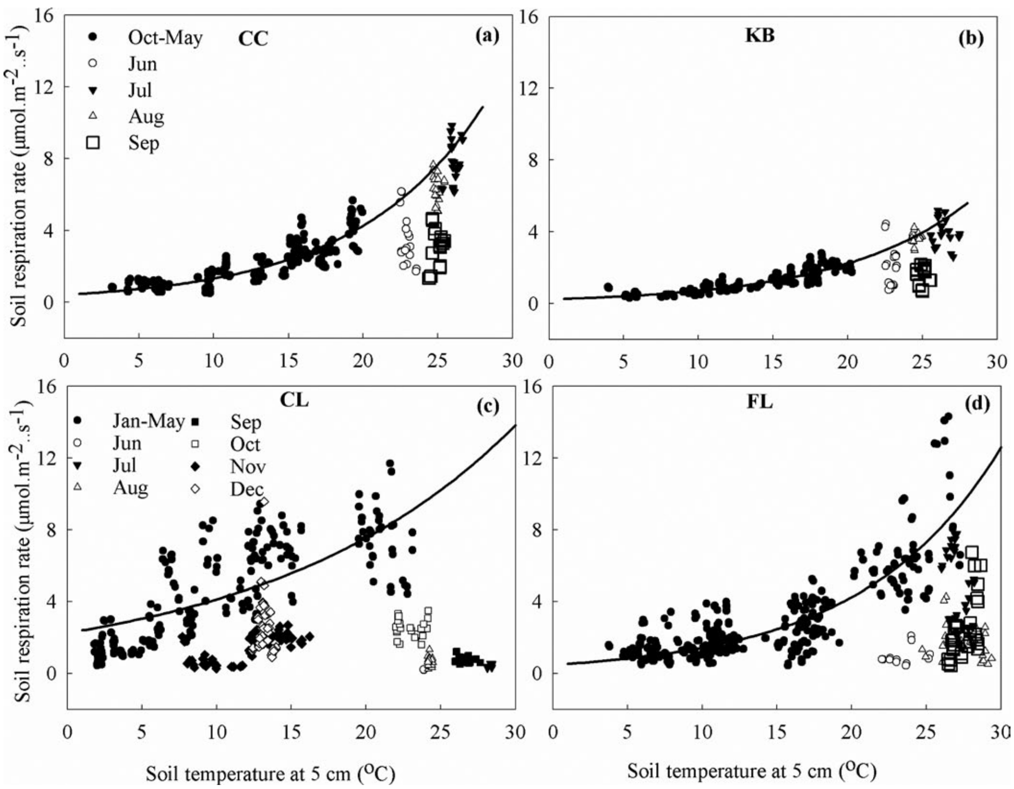

Figure 5: Soil respiration vs. temperature at 5 cm under CC (a), KB (b), CL (c) and FL (d). Curves are fitted to the van't Hoff equation with data during June–September where only data during January–May are used. See Fig. 1 for abbreviations.

Across land uses, the annual mean soil respiration rates followed the order CC~CL~FL> KB (Table 1). However, R_s during winter and spring (December–May) followed the order CL>FL>CC ~ KB, whereas that during summer and autumn (June–November) was in the order CC>KB~FL >CL. Afforested soils had an annual respiration rate of 2.36 μmol m^{-2} s^{-1} on average, slightly lower than that of paddy soils (2.67 μmol m^{-2} s^{-1}). Overall, afforestation on previous paddy fields decreased soil CO_2 emission.

Relationships between R_s and environmental factors

Except for CL, van't Hoff equation fitted the data fairly well at temperatures <20°C (Fig. 5), i.e. during October–May, when temperature explained >70% of variance in R_s. However, there was much scatter in the R_s–temperature relationship at temperatures >20°C, i.e. during June–September, when R^2 dropped to 0.19 and 0.44 for the plantations and the fitting was even insignificant in FL (Table 2). In CL, much larger

Table 2: Fitted parameters of the van't Hoff's equation (Equation 2) for soil respiration measured *in situ* and in laboratory incubation.

Land use	*In-situ* measurement[a]					Laboratory incubation				
	Period[b]	R_0	b	R^2	P	R_0	b	Q_{10}	R^2	P^c
CC	October–May	0.005	0.12	0.76	**	0.31	0.072	2.06	0.75	**
	June–September	0.41	0.28	0.44	**					
KB	October–May	0.23	0.11	0.76	**	0.36	0.058	1.78	0.93	**
	June–September	0.064	0.15	0.19	**					
CL	January–May	2.25	0.061	0.54	**	0.17	0.13	3.78	0.99	**
	June–September	ND	ND	ND	NS					
FL	October–May	0.48	0.11	0.72	**	0.42	0.053	1.71	0.89	**
	June–September	ND	ND	ND	NS					

[a]ND = not determined, because of insignificant fitting. [b]In CL, soils were drained during January–May 2008 and thus only data during this period was used for model fitting. **: $P < 0.01$; NS: $P > 0.05$ (not significant).

scatter in R_s vs. temperature was seen than forother land uses. Fitting to van't Hoff equation was only significant during the dry farming period of CL ($R^2 = 0.54$, $P < 0.01$).

Annually, there was no significant correlation between soil moisture and R_s at any site. However, R_s during June–September was negatively correlated to *WFP* at all sites, though <50% of the variance in R_s was explained (Table 3).

Changes in R_s could not be well explained by either temperature or *WFP* on the days before and after ploughing in CL. R_s did not significantly change with temperature before

Table 3: Fitted parameters in linear regression of soil respiration (R_s) vs. soil *WFP* ($R_s = a + b*WFP$) with field data for June–September 2008.

Land use	*a*	*b*	R^2	*P*
CC	13.02	−0.11	0.11	**
KB	4.56	−0.03	0.14	**
FL	15.41	−0.19	0.19	***
CL	1.58	−0.01	0.45	***

**: $P < 0.01$; ***: $P < 0.001$.

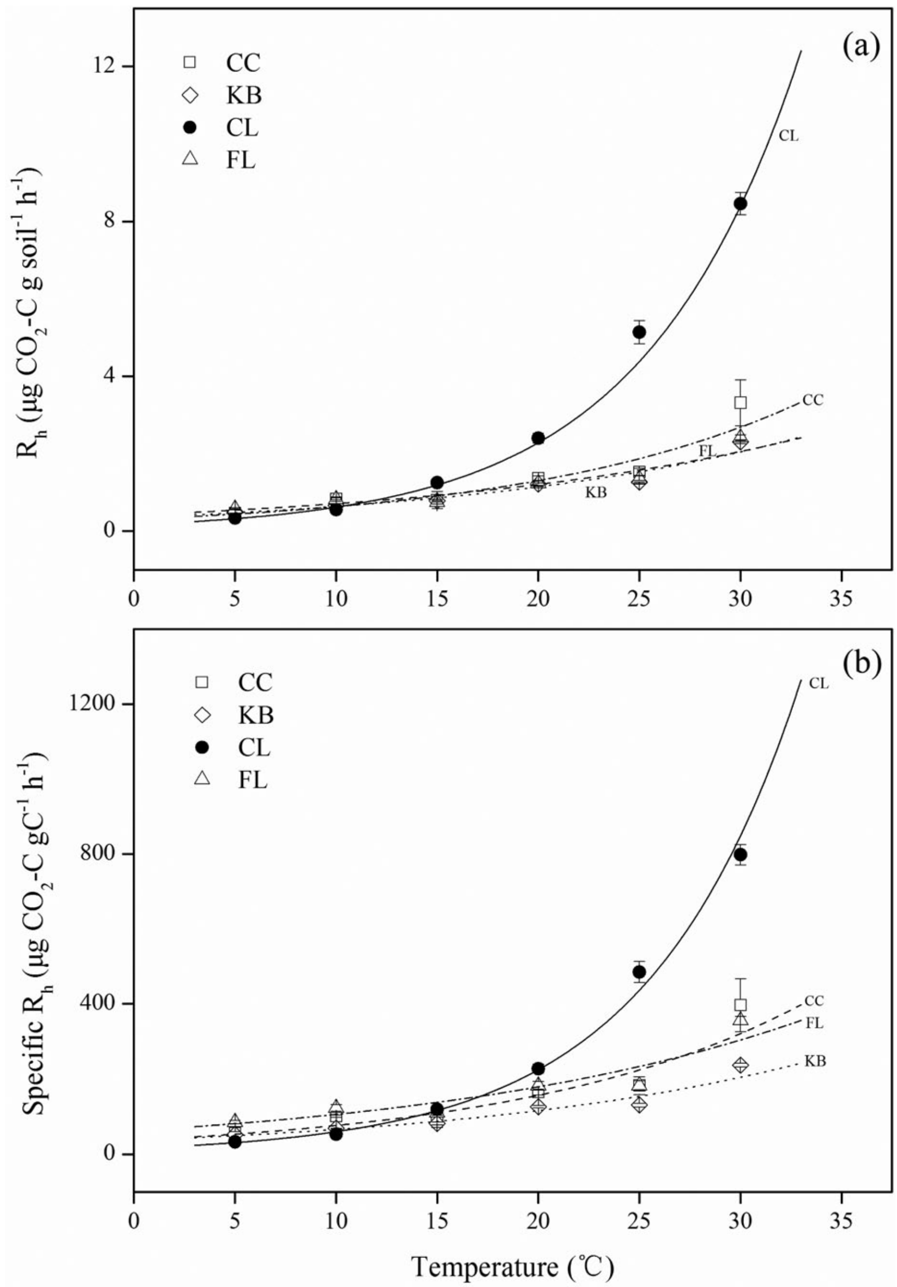

Figure 6: Relationship between heterotrophic soil respiration (R_h) rates and temperature (a); relationship between specific R_h rates and temperature (b) under laboratory incubation. Data were averaged at each temperature and error bars represent standard errors. Curves were fitted to the van't Hoff equation and land uses are labeled beside each curve. See Fig. 1 for abbreviations.

ploughing (P > 0.05, Fig. 4a). After ploughing, a weak relationship between R_s and temperature could be seen (R^2 = 0.24), with R_s increasing abruptly when temperatures rose above 6°C. *WFP* was not significantly correlated with R_s either before or after ploughing (P > 0.05, Fig. 4b).

Temperature sensitivity of SOC decomposition

Across all incubation temperatures, R_h differed in the order CL > CC ~ FL ~ KB (P < 0.01; Fig. 6a), whereas specific R_h followed the order CL > FL ~ CC > KB (P < 0.01; Fig. 6b). The responses of R_h or specific R_h to temperature showed similar patterns between land uses, and hence only the Q_{10} of R_h was estimated (Table 2). The results indicated a distinctively higher Q_{10} in CL than in other soils (3.78 vs. 1.71–2.06).

DISCUSSION

Changes in SOC sequestration following afforestation

After 15–20 years of afforestation on paddy rice croplands, we observed increases in SOC concentrations at the soil surface (0–5 cm). This was consistent with the study of Six *et al.* (2002), who pointed out that afforested soils were disturbed less and hence more SOC was physically protected in soil macroaggregates than in cultivated soils. However, there was significant SOC loss at 5–40 cm, leading to net decreases at 0–40 cm of total SOC stocks due to afforestation (Fig. 2b). Soil bulk density had not been reduced but slightly increased by afforestation (Table 1), suggesting that decreases in SOC storage would be larger if it was corrected for changes in soil bulk density. However, like most previous investigations on the land use effects on SOC, we could not know the background SOC stocks before land use changes at each site, which to some extent added uncertainties to our observation results. However, the uniform soil texture under different land uses (Table 1) and their common origin from estuarine sediments suggested that it was reasonable to assume similar SOC backgrounds at all sites before land use changes. Previous studies generally found decreases in SOC stocks when pastures were afforested (Guo and Gifford 2002; Post and Kwon 2000; Paul *et al*. 2002). In this sense, paddy rice croplands in China were somewhat comparable to USA pastures in terms of SOC sequestration, as suggested by Pan *et al*. (2003).

Our finding was consistent with the study of Cai (1996), who reported that SOC stocks up to a depth of 62.3 cm in paddy fields of eastern China were 23.6% higher than in agroforests. Iqbal *et al*. (2009) also noticed a 49.3% increase in SOC concentrations after conversion of woodlands to paddy fields, whereas declines in SOC were commonly observed by previous studies after afforestation on croplands (Guo and Gifford 2002; Post and Kwon 2000; Paul *et al*. 2002). Probably, afforestation was not as efficient as paddy rice cropping in sequestrating SOC in north subtropical China, at least in the first several decades of afforestation. One might suspect that a longer period (e.g. 10–40 years, as suggested by Degryze *et al*. (2004)) might be necessary to fully realize the SOC sequestration potential of afforestation. However, data from a national forest park of the Chongming Island revealed that even after 50 years of afforestation on paddy soils, SOC densities at 0–15 cm of afforested soils were only comparable to that of adjacent paddy soils (P > 0.05; our unpublished data). Therefore, at least within the first several decades following afforestation, afforested soils might be not as efficient as paddy soils in SOC sequestration.

Obviously, the larger SOC stocks in paddy fields were mainly due to their higher subsoil SOC concentrations. This probably resulted from incorporation of rice/barley straws into subsoil by tillage, as the plow layer of paddy fields was commonly down to 20 cm in China. Bashkin and Binkley (1998) reported that soil C inputs to 5–40 cm soil layers dropped substantially following afforestation on sugarcane fields, because soil homogenization by tillage stopped. In addition, paddy soils usually had high concentrations of DOC, leaching of which might contribute to SOC storage in the subsoil, as suggested by Kögel–Knabner *et al*. (2010) and Maie *et al*. (2004). This could also be the reason for the relatively high SOC concentrations below 20 cm in FL, which used to be frequently submerged under water before abandonment, leading to large SOC storage in subsoil. As a result, the paddy rice croplands and the FL showed the highest 1-m SOC stock among all land uses.

Changes in SOC lability following afforestation

Previous studies generally found larger microbial biomass in forest soils than in agricultural soils (Boyer and Groffman 1996). In contrast, we found that DOC and microbial biomass, as indicators of labile SOC pool (Lützow *et al.* 2007), were both higher in paddy rice croplands (Table 1). This reflected the higher quality of soil organic matter and substrate availability in agricultural soils than in forest soils (Hu *et al.* 1997). Iqbal *et al*. (2009) also found that paddy soils had significantly larger active and slow SOC pools but a smaller resistant SOC pool than woodland soils. Therefore, SOC sequestered in paddy soils seemed to be more labile than in afforested soils, despite the greater SOC sequestration of paddy fields.

The labile SOC in paddy fields should have mainly accumulated during the rice cultivation period. Topsoil SOC stock significantly increased shortly after paddy soils entered this period (Fig. 3a). The concurrent bursts of microbial biomass (Fig. 3b) suggested improved substrate availability, and the increased topsoil SOC was probably caused by the anaerobic decomposition products of straws from the last growing season. In agreement with this, Suetsugu *et al*. (2005) observed strong increases in dissolved and particulate organic matter during the rice-growing season of paddy fields. However, a slight decline in SOC was seen when soils were drained between December 2007 and March 2008, presumably due to the loss of labile SOC accumulated in the previous rice-growing season. This resembled the rapid decomposition of unstabilized SOC when natural wetlands were drained (Davidson and Janssens 2006). We thus hypothesized that SOC accumulation in paddy soils mainly occurred within the rice-growing period, whereas SOC loss mainly occurred in the dry-farming period, although more data are needed to confirm this. Such processes do not exist in afforested soils.

Changes in soil CO_2 emission following afforestation

Overall, our results suggested that soil CO_2 emission was lowered by afforestation on previous paddy fields. There was notably high potential of CO_2 emission in the paddy soils, where CO_2 was mainly released in bursts for only 5 months (January–May), while the annual mean R_s was still 53.4% higher than in the *Koelreuteria bipinnata* plantation and close to that in the *Cinnamomum camphora* plot (Table 1). This was obviously due to the distinctively high R_s after paddy fields entered the dry-farming period (Fig. 1d), which was somewhat like the rapid soil C emission after drainage of natural wetlands (Mitra *et al.* 2005). In mid-subtropical China, Iqbal *et al.* (2008) reported an annual soil CO_2 emission of 901 g CO_2-C m^{-2} $year^{-1}$ from a paddy rice cropland, compared with 533–727 g CO_2-C m^{-2} $year^{-1}$ from orchards and natural woodlands. Hence, afforestation could lower the CO_2 emission potential of paddy soils.

The high CO_2 emission potential of paddy soils should at least partly result from their high SOC lability, although root respiration was also involved. This was evidenced by the abrupt rises in R_s immediately after ploughing of paddy soils (Fig. 4). Soil temperatures were comparable on the two observatory days before and after ploughing and temperature had a poor relationship with R_s. Because crops had not been seeded, the distinctively high R_s after ploughing was presumably related to aeration of the labile SOC accumulated during the rice cultivation period. The sharp increase of R_s above 6°C also supported the conclusion that such SOC was not stable. One may expect even higher annual soil CO_2 emission from paddy soils than observed here if SOC decomposition had not been restricted by a long waterlogging rice-growing period.

Changes in temperature sensitivity of SOC decomposition following afforestation

Consistent with previous studies in subtropical China (Iqbal *et al.* 2008; Lou *et al.* 2004; Sheng *et al.* 2010), temperature exerted the primary control over soil CO_2 emission in this study because it explained >50% of variations in R_s during most of the year, especially in the afforested soils (Table 2). However, in the rainy summer season, soil respiration might have been inhibited by factors associated with rainfall, e.g. soil oxygen limitation, plant root activities and substrate availability (Luo and Zhou 2006). Summer R_s was significantly lower than that predicted by the van't Hoff equation (Fig. 5), and there was a negative correlation between soil moisture and R_s (Table 3). In the paddy rice croplands, flooding minimized soil respiration during the whole rice-growing period. Therefore, the temperature control over soil CO_2 emission under field conditions was confounded by other factors, particularly soil moisture in this region.

Considering the primary importance of temperature to soil C emission in subtropical China, we also evaluated how temperature sensitivity (i.e. Q_{10}) of SOC decomposition changed following afforestation, which might be relevant to future atmospheric CO_2 concentrations of China under global warming. Soils were incubated under fast-changing temperatures, which avoided the confounding effects of factors other than temperature under field conditions. The results clearly suggested that SOC sequestered by paddy rice cropping was more sensitive to temperature increases than by afforestation (Fig. 6). This should be related to the highest SOC lability and substrate availability of paddy fields among all land uses, because labile SOC is fast-cycling and can be easily accessed by microbes as temperature increases (Davidson and Jassens 2006; Hu *et al.* 1997).

Previous studies often adopted long-term incubation at constant temperatures to estimate Q_{10} values (e.g. Iqbal *et al.* 2009), which was problematic because declines in microbial biomass and substrate availability with incubation time would seriously bias Q_{10} values (Fang *et al.* 2005). Our incubation procedure effectively accounted for these problems (Fang and Moncrieff, 2001; Fang *et al.* 2005). Changing incubation temperatures for a certain soil sample within 1–2 hours ensured that soil microbial biomass and substrate availability did not change significantly. It might be suspected that labile substrates had been largely depleted at low temperatures because we started the incubation from high temperatures. However, previous studies revealed that changes in incubation temperatures, in decreasing or increasing order, would not affect Q_{10} values (Fang and Moncrieff 2001).

Implications for agricultural GHG mitigation in China

Afforestation has been considered an important measure to sequester C and mitigate GHG emission (IPCC 2001; Paustian *et al.* 1997). Smith (2004) estimated that conversion from cropland to woodland would have a realistic soil C–sequestration potential of 4.5 Mt C per year in Europe, the maximum among all management options for croplands. China has launched an ambitious nationwide "Grain-for-Green" program to promote forest area since 1999 (Zhang *et al.* 2010), which will contribute to national GHG mitigation.

However, this study revealed that large uncertainties still existed concerning the effects of afforestation on SOC sequestration in China, especially where paddy rice croplands were involved. At least at a time scale of several decades, paddy fields seemed to be superior to afforested systems in sequestrating SOC, mainly in the subsoil. The advantages of paddy soils in offsetting CO_2 emission over upland soils in China has long been realized (Lal 2004; Pan *et al.* 2003; Xie *et al.* 2007). However, the fact that paddy rice croplands might have an SOC sequestration potential equivalent to or even higher than afforested systems has not received enough attention (Cai 1996; Iqbal *et al.* 2008; Iqbal *et al.* 2009; Wang *et al.* 2006). If this widely exists in China, perhaps more available lands (e.g. wastelands) can be used for rice cultivation instead of afforestation, because rice cultivation has the 2-fold benefits of food production and efficient SOC sequestration. This would influence the calculation of regional carbon balance in China.

Interestingly, there were also indications that paddy soils had rather high potential of CO_2 emission. SOC sequestered in paddy fields was less stable and more sensitive to temperature changes than that in afforested soils. Considering future global warming concerns, it might be projected that the large amounts of SOC sequestered by paddy soils would be more easily lost into the atmosphere than that in afforested soils in the future, which deserves the caution of policy decision makers. Given the acceleration of CO_2 emission from paddy soils by tillage after rice harvesting (Fig. 4), management options such as nontillage or conservation tillage during the dry-farming period might be useful for mitigation of CO_2 release. To better understand the net benefit of afforestation relative to paddy rice cropping in SOC sequestration and GHG mitigation of China, a careful and comprehensive assessment has to be made based on more detailed data.

In summary, unlike the general observation of increased SOC stocks after croplands were afforested, this study found that afforestation on previous paddy rice croplands only slightly increased the SOC at 0–5 cm but led to losses of SOC stocks at other depths in the top 40 cm. In this sense, paddy rice croplands might be superior to afforested systems in terms of SOC sequestration in China, at least within the first several decades following afforestation. However, SOC sequestered by paddy soils was less stable and could be more easily released into the atmosphere with increasing temperatures than that in afforested soils, especially when paddy fields were drained. Overall, compared with paddy rice cropping, afforestation was less efficient in sequestering SOC but might lower the potential of soil CO_2 emission in a warming climate.

FUNDING

Ministry of Science and Technology of China (2010CB950604); National Major Scientific and Technological Project in China (2010BAK69B14); National Natural Science Foundation of China (30970556,31170386); Doctoral Program of Higher Education of China (20093227110004); and Key Project of the Shanghai Scientific and Technological Committee (10DZ1200700).

ACKNOWLEDGEMENTS

We appreciate the useful comments and suggestions from two anonymous reviewers. Thanks should also be given to workers at the Coastal Ecosystems Research Station of Yangtze River Estuary, who assisted in fieldwork.

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