

# CARBON DISTRIBUTION AND ITS CORRELATION WITH FLORISTIC DIVERSITY IN SUBTROPICAL BROAD-LEAVED FORESTS DURING NATURAL SUCCESSION

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Submitted November 2016; accepted May 2017

There are increasing demands for afforestation and reforestation aimed at sequestering carbon (C) and conserving biodiversity. This study provides quantitative results to evaluate C storage capacity and probes the relationship between C distribution of various forest components and their floristic diversity during natural recovery processes. We studied C allocation, total C storage and floristic diversity in a subtropical broad-leaved secondary forest in south-eastern China managed under three different regimes, namely, no logging, early selective logging and late selective logging. Diameter at breast height data from 33 plots were used to construct allometric equations and to evaluate total biomass and C storage. Carbon storage in trees, understorey, necromass and the top 100 cm of soil was responsible for 29.8–47.6%, 0.2–0.5, 2.2–2.7 and 49.9–67.1% of the total C stocks respectively under the three regimes. Tree C stock was positively correlated (Pearson coefficient = 0.35) with tree diversity and explained 12.4% of the overstorey Shannon and Simpson indices. Therefore, elimination of the coniferous species (*C. lanceolata*) and overstorey diversity would not harm C storage for subtropical, broad-leaved secondary forest management.

Keywords: Selective logging, carbon estimate, forest management, evergreen and coniferous broad-leaved forests

## INTRODUCTION

Forests play an irreplaceable role in maintaining regional ecological environments, regulating the global carbon (C) cycle and reducing atmospheric greenhouse gas emission (Zhao et al. 2015). Forest managers have expanded their objectives from traditional timber production to include broader goals of providing ecosystem goods and services (Burton et al. 2013). However, the primary aims of common international and national forest protection measures regarding afforestation and reforestation are to enhance productivity and mitigate CO<sub>2</sub> accumulation by sequestering C. Climate change and biodiversity loss are two interrelated global crises but are independent of forest protection programme objectives launched by most national governments. Forest protection, biodiversity conservation and climate change goals should be integrated to achieve mutual gains (Strassburg et al. 2010).

Subtropical broad-leaved forest is distributed throughout the monsoon climate zone of the eastern Asian coast of China, the southern parts of Japan and Korea, parts of the Indian subcontinent and coastal West Africa (Peel et al. 2007). Many ecological studies have reported community structure, composition, biodiversity and C storage assessment of subtropical broad-leaved forests (Legendre et al. 2009, Zhang et al. 2010, Lu et al. 2013, Feng et al. 2014). There are also studies focused on the relationship between C stocks and biodiversity for temperate (Hatanaka et al. 2011) and tropical forests (Williams et al. 2008, Cavanaugh et al. 2014, Magnago et al. 2015). However, not much research has been done towards examining the relationship between C stocks and biodiversity of different forest components under various management regimes during natural forest recovery processes in subtropical forests.

In China, subtropical broad-leaved forests are divided into three stages: (1) central timber production period (1950s–1980s), which was followed by a decrease in natural forest and increase in monoculture plantation establishment with cultivated species, e.g. *Cunninghamia lanceolata* (Chinese fir), *Pinus massoniana* (masson pine) and *Phyllostachys edulis* (moso bamboo) in south China, (2) transitional phase (1980s–1990s) and (3) sustainable forest management (1990s–present), which is characterised by the priorities given to ecological benefits while at the same time balancing economic and social objectives (Zhou 2002). During the third phase, China has experienced marked interest in conserving its biodiversity since the Biodiversity Conservation Action Plan was released in 1994. After 2010, improving present vegetation structure and floristic composition would be the next step for the local government to manage forests which have changed from timber forest to ecological forests. Despite a long-standing discussion about diversity and productivity, additional gains for biodiversity conservation are possible without compromising efficacy of climate change mitigation (Strassburg et al. 2010). However, trade-offs between dynamic C pools and biodiversity conservation in specific anthropogenically impacted ecosystems remain unknown. Therefore, understanding how to apply suitable methods for forest recovery succession in order to maintain the equilibrium of carbon stock and biodiversity conservation is needed for modern forest management projects and active response to climate change. Quantifying C stock patterns will facilitate C estimates for the voluntary C market, as well as landuse change planning, such as any shift from commercial to ecological forests as requested by local policy (Kalaba et al. 2013).

The primary objectives of this study were to use reliable empirical data to quantify C distribution and biodiversity changes and reveal the relationships, if any, between C stocks and biodiversity of different secondary broad-leaved forest components with different regeneration processes. Our specific research questions were as follows: (1) what were the C stocks and distribution in subtropical, broad-leaved secondary forests resulting from different forest management regimes during natural secondary succession and (2) what were the relationship between C stocks and floristic diversity in

broad-leaved secondary forests during natural secondary succession?

## MATERIALS AND METHODS

### Study site

The study site was located at Pushang State Forest Farm (26° 56' N, 117° 47' 5 E), Shunchang County, Fujian Province, China. The topography is characterised by low hills with an altitudes of 600–800 m above sea level and 20–30° slopes. The climate is subtropical oceanic monsoon, with mean annual temperature of 18.5 °C, average annual rainfall of 1880 mm, average annual sunshine of 1699 hours and 260 frost-free days. The red forest soil has heavy loam texture, medium fertility and large water holding capacity and depth greater than 100 cm.

### Field sampling

In August 2013, 33 plots (20 m × 20 m) located at 578–796 m above sea level were selected in a Chinese fir monospecific plantation under similar topographic features but under three different management regimes (no logging, early selective logging and late selective logging) over the past 30 years. The trees were 8 years old and had average density of 580 tree ha<sup>-1</sup>. The plots with no logging (9 plots) had undergone natural secondary succession and the dominant species after 30 years was Chinese fir in an evergreen broad-leaved mixed forest. In the other two regimes, Chinese fir was logged completely early in the 5<sup>th</sup> year (early selective logging, 12 plots) and later in the 15<sup>th</sup> year (late selective logging, 12 plots); the dominant species in these plots was the broad-leaved secondary forest. None of the plots caught fire during the past 30 years. In each plot, we identified the species and crown dimensions for all trees with diameter at breast height (DBH) ≥ 5 cm. Mean DBH was 13.8 ± 7.8 cm (± standard deviation) and mean tree height was 10.2 ± 4.1 m.

Trees from each species and DBH class were selected to estimate biomass. A total of nine dominant broad-leaved species (*Castanopsis fargesii*, *C. lamontii*, *C. tibetana*, *Lithocarpus glaber*, *Sloanea sinensis*, *Daphniphyllum oldhami*, *Alniphyllum fortunei*, *Manglietia yuyuanensis*, and *Engelhardtia fenzlii*), *C. lanceolata* and other minor broad-leaved species were destructively

harvested in each stand for a total of 178 trees and 28 species (Table 1). We harvested the entire shrub and herb layers in three random 1 m × 1 m subplots in each plot. The above- and belowground components were separated and all litter from each subplot was collected.

Three randomly located soil samples were obtained from each plot and divided into six layers, with a total depth of 100 cm, which constituted two 10-cm deep surface layers and the remaining four 20-cm deep layers. Bulk density was calculated using the cylindrical core method.

**Table 1** Allometric equations used to estimate different tree species and standing dead tree dry biomass based on standard sampling

Species	Component	a	b	r <sup>2</sup>	Species	Component	a	b	r <sup>2</sup>
<i>Castanopsis fargesii</i> (n = 21, DBH = 5.2–35.5, H = 6.3–19.5, B = 6.2–1204.2)	W <sub>S</sub>	0.1234	2.3429	0.98	<i>Alniphyllum fortunei</i> (n = 15, DBH = 5.1–21.0, H = 8.1–21.3, B = 10.5–301.4)	W <sub>S</sub>	0.1114	2.4465	0.92
	W <sub>B</sub>	0.0043	2.8734	0.90		W <sub>B</sub>	0.0042	2.5393	0.78
	W <sub>L</sub>	0.0020	3.1498	0.90		W <sub>L</sub>	0.0008	3.3098	0.84
	W <sub>A</sub>	0.1117	2.4898	0.98		W <sub>A</sub>	0.1121	2.4896	0.92
	W <sub>R</sub>	0.0126	2.7281	0.97		W <sub>R</sub>	0.0344	2.3037	0.84
<i>C. lamontii</i> (n = 15, DBH = 5.8–31.7, H = 6.7–17.7, B = 13.2–895.3)	W <sub>T</sub>	0.1230	2.5277	0.99	W <sub>T</sub>	0.1548	2.4354	0.93	
	W <sub>S</sub>	0.1178	2.3426	0.94	W <sub>S</sub>	0.0825	2.4562	0.94	
	W <sub>B</sub>	0.0025	3.0512	0.89	W <sub>B</sub>	0.0190	2.1851	0.93	
	W <sub>L</sub>	0.0015	3.2150	0.91	W <sub>L</sub>	0.0234	2.0938	0.92	
	W <sub>A</sub>	0.0921	2.5354	0.94	W <sub>A</sub>	0.1231	2.3836	0.95	
<i>C. tibetana</i> (n = 15, DBH = 5.2–30.6, H = 5.3–13.4, B = 10.4–586.2)	W <sub>R</sub>	0.0265	2.5053	0.88	W <sub>R</sub>	0.0213	2.4132	0.91	
	W <sub>T</sub>	0.1221	2.5211	0.94	W <sub>T</sub>	0.1463	2.3845	0.95	
	W <sub>S</sub>	0.1915	2.0980	0.96	W <sub>S</sub>	0.1958	2.0293	0.86	
	W <sub>B</sub>	0.0047	2.7655	0.93	W <sub>B</sub>	0.0021	3.1156	0.84	
	W <sub>L</sub>	0.0056	2.7888	0.90	W <sub>L</sub>	0.0021	3.1522	0.88	
<i>Lithocarpus glaber</i> (n = 15, DBH = 5.4–23.0, H = 5.9–14.8, B = 10.9–307.8)	W <sub>A</sub>	0.1753	2.2494	0.97	W <sub>A</sub>	0.1406	2.2912	0.87	
	W <sub>R</sub>	0.0366	2.3897	0.95	W <sub>R</sub>	0.0174	2.5009	0.85	
	W <sub>T</sub>	0.2149	2.2747	0.97	W <sub>T</sub>	0.1583	2.3232	0.88	
	W <sub>S</sub>	0.2927	2.0354	0.89	W <sub>S</sub>	0.0650	2.4192	0.98	
	W <sub>B</sub>	0.0281	2.1397	0.90	W <sub>B</sub>	0.0009	2.9360	0.98	
<i>Sloanea sinensis</i> (n = 15, DBH = 5.8–26.0, H = 2.9–13.8, B = 11.1–285.9)	W <sub>L</sub>	0.0113	2.4880	0.82	W <sub>L</sub>	0.0001	4.0296	0.90	
	W <sub>A</sub>	0.3525	2.0581	0.92	W <sub>A</sub>	0.0538	2.5604	0.98	
	W <sub>R</sub>	0.0585	2.2129	0.94	W <sub>R</sub>	0.0124	2.6275	0.94	
	W <sub>T</sub>	0.4086	2.0880	0.93	W <sub>T</sub>	0.0661	2.5764	0.96	
	W <sub>S</sub>	0.1635	2.2012	0.89	W <sub>S</sub>	0.1650	0.2715	0.91	
<i>Daphniphyllum oldhami</i> (n = 14, DBH = 6.7–22.9, H = 5.9–17.2, B = 18.8–299.1)	W <sub>B</sub>	0.0180	2.2849	0.71	W <sub>B</sub>	0.0172	2.2529	0.52	
	W <sub>L</sub>	0.0052	2.7326	0.76	W <sub>L</sub>	0.0070	2.6244	0.52	
	W <sub>A</sub>	0.1790	2.2704	0.92	W <sub>A</sub>	0.1899	2.3066	0.94	
	W <sub>R</sub>	0.0594	2.1316	0.88	W <sub>R</sub>	0.0760	2.1682	0.73	
	W <sub>T</sub>	0.2358	2.2483	0.94	W <sub>T</sub>	0.2830	2.2523	0.93	
<i>Daphniphyllum oldhami</i> (n = 14, DBH = 6.7–22.9, H = 5.9–17.2, B = 18.8–299.1)	W <sub>S</sub>	0.1369	2.3000	0.94	Standing dead tree	W <sub>SB</sub>	0.1380	2.3210	0.92
	W <sub>B</sub>	0.0061	2.6251	0.84					
	W <sub>L</sub>	0.0018	3.0804	0.84					
	W <sub>A</sub>	0.1370	2.3783	0.95					
	W <sub>R</sub>	0.0286	2.3983	0.92					
W <sub>T</sub>	0.1726	2.3686	0.96						

n = sample number, H = height (m), B = biomass (kg) of tree; all equations are in the form  $W_i = aD^b$  where D = diameter at breast height (DBH, cm) and a and b = statistical parameters; W<sub>S</sub> = stem biomass (kg), W<sub>B</sub> = branch biomass (kg), W<sub>L</sub> = foliage biomass (kg), W<sub>A</sub> = aboveground biomass, including stem, branch and foliage (kg), W<sub>R</sub> = root biomass (kg), W<sub>T</sub> = total biomass (kg), W<sub>SB</sub> = stem and branch biomass (kg) and r<sup>2</sup> = coefficient of determination of the equation

Soil containers were weighed first, and then the humid weights of the 100-cm<sup>3</sup> soil samples and the containers were measured at a flat site. Then, the samples were dried at 105 °C for 24 hours to constant weight. The water ratio of the soil was calculated and used to obtain bulk density, which was equal to dry soil weight divided by volume. Total soil C was determined using CNS elemental analyser based on air-dried samples.

### Biomass estimates

We estimated biomass using the Chinese National Forest Inventory methods (China SFA 2008). The trees were excavated from the ground and separated into three parts, namely, stem, branch and foliage, and root components. We cut stems into 1-m sections if tree total height was less than 10 m or 2-m sections otherwise. Each section was weighed to determine the stem fresh net weight. Branches were cut at the ramification point of the stem. Rachis and veins were considered parts of the leaves. Branches with foliage were divided into three groups, namely, upper (> 100 g), mid (100–50 g) and lower (< 50 g), and each group was individually weighed to estimate the average fresh weight of one branch with foliage. A total of three to five branches with foliage that were almost equal to the average weight were selected and weighed after clipping. Branch and foliage data were used to calculate the entire crown. In each plot, soil was removed from around tree roots, and the entire root system (diameter larger than 0.3 cm) was manually excavated and weighed after removing all soil particles before rinsing. Subsamples of 500 g of fresh stems, branches, foliage and roots were transported to the laboratory and oven dried at 105 °C for 2 hours and then at 85 °C to constant weight for determination of dry biomass. Fresh above- and belowground structures, i.e. stems, foliage, branches and roots from shrub and herb layers from three subplots (1 m × 1 m) located at boundaries of one plot were randomly collected and oven dried at 85 °C to constant weight. We fit the standard form of allometric equation (Table 1):

$$W_i = aD^b \quad (1)$$

where  $W_i$  = biomass of tree component  $i$ ,  $D$  = DBH and  $a$  and  $b$  = statistical parameters. From the best-fit results, the Richard growth curve, one

of the 11 growth curves commonly used in forest science, was used to obtain DBH of the Chinese fir trees which were logged in the 5<sup>th</sup> and 15<sup>th</sup> year (Liu et al. 2014). This curve, also known as the generalised logistic, is widely used and fits a wide range of S-shaped growth curves. The formula used to estimate DBH was as follows:

$$D = A(1 - e^{-C \times \text{AGE}})^B \quad (2)$$

where  $D$  = DBH,  $A$ ,  $B$  and  $C$  are coefficients and AGE is the tree age.

Necromass included litter and standing dead tree biomass. The fresh weight of litter from each subplot was recorded in-situ and subsequently oven dried to constant weight. For standing dead trees, we used total biomass ( $B$ ) of stems and branches of sampled live trees and DBH ( $D$ ) of dead trees to build an allometric equation ( $B = aD^b$ ) to estimate necrobiomass of standing dead tree (Table 1).

### Data analysis

Tree, shrub and herb diversity values were measured by calculating the Shannon-Wiener ( $H'$ ) and Simpson ( $D$ ) indices for the three forest management regimes:

$$H' = - \sum_{i=1}^s (P_i \ln P_i) \quad (3)$$

$$D = 1 / \sum_{i=1}^s P_i^2 \quad (4)$$

where  $P_i$  = total proportion of individuals belonging to the  $i^{\text{th}}$  species, calculated as  $n_i/N$ ,  $n_i$  = number of individuals of species  $i$ ,  $N$  = total number of individuals in the sample and  $S$  = total number of species.

We calculated tree, shrub, herb, forest floor and standing dead tree C stock as 50% of the biomass weight. This value represents a typical C proportion (Hernandez-Stefanoni & Ponce-Hernandez 2004). Allometric equations were used to estimate biomass and necromass from standard sampling data (DBH) measured from each of the 10 species and the mixed category. We used collective plot data of the understorey subplots and soil samples to determine C stocks per ha at the study site.

Soil C storage was calculated as follows:

$$C_{\text{soil}} = \sum_{p=1}^6 (C_s \rho_s H_s) \quad (5)$$

where  $C_{\text{soil}}$  = soil C stock ( $\text{Mg C ha}^{-1}$ ),  $p$  = soil layer;  $C_s$  = soil C concentration (%),  $\rho_s$  = measured soil layer bulk density ( $\text{g cm}^{-3}$ ) and  $H_s$  = soil layer thickness (cm). Total C storage in the ecosystem was calculated as follows:

$$C_f = C_{\text{tree}} + C_{\text{understorey}} + C_{\text{necro}} + C_{\text{soil}} \quad (6)$$

where  $C_f$  = forest ecosystem C storage ( $\text{Mg C ha}^{-1}$ ),  $C_{\text{tree}}$  = tree C stock,  $C_{\text{understorey}}$  = understorey C stock and  $C_{\text{necro}}$  = C stock of dead plant material.

We conducted regression analyses to develop exponential allometric equations using SPSS (version 16.0). Differences between different management regimes were tested using one-way analysis of variance. Least significant difference post-hoc test was carried out to evaluate differences between means when the F-test was significant ( $p \leq 0.05$ ). Two-tailed Pearson correlation and linear regression fit were calculated between biodiversity and total C stock. R-value from the regression analyses represented the explanatory power.

## RESULTS

### Carbon estimates

#### Biomass and necromass

Similar trends in biomass and necromass allocation for all three management regimes (no logging, early-selective logging and late selective logging) were observed (Table 2). Biomass was distributed between the different tree structural components in the following decreasing order: stem, root, foliage and branch. Shrubs contained most of the understorey biomass (Table 2). The biomass of the structural components of each tree was significantly lower in plots undergoing natural secondary succession (no logging) than in early and late selective logging plots. Biomass was generally higher in early selective logging compared with late selective logging plots. However, the difference was not significant. Stems contributed 60.9, 57.8 and 56.1% of the total tree biomass in the no logging, early selective logging

**Table 2** Forest biomass of the various components and necromass for three different succession processes

Component	Biomass ( $\text{Mg h a}^{-1}$ )		
	No logging	Early selective logging	Late selective logging
Total tree	175.13 ± 22.26 a	353.03 ± 117.19 b	302.05 ± 104.04 b
Aboveground	139.29 ± 17.39 a	282.16 ± 93.95 b	241.30 ± 81.16 b
Tree stem	106.70 ± 13.51 a	204.00 ± 61.68 b	169.34 ± 53.89 b
Tree branch	13.48 ± 2.53 a	33.89 ± 17.29 b	32.13 ± 15.59 b
Tree foliage	15.94 ± 2.64 a	40.87 ± 21.90 b	39.74 ± 21.31 b
Tree root	34.73 ± 4.92 a	70.53 ± 23.88 b	61.22 ± 21.43 b
Total understorey	3.02 ± 1.38 a	1.26 ± 0.97 b	2.14 ± 0.92 a b
Total shrub	2.76 ± 1.37 a	1.12 ± 1.00 b	1.92 ± 0.93 a b
Aboveground shrub	1.94 ± 1.24 a	0.64 ± 0.54 b	1.18 ± 0.63 b
Belowground shrub	0.82 ± 0.22 a	0.49 ± 0.49 a	0.74 ± 0.41 a
Total herb	0.26 ± 0.19 a	0.14 ± 0.07 b	0.22 ± 0.04 a b
Aboveground herb	0.05 ± 0.05 a	0.05 ± 0.03 a	0.06 ± 0.03 a
Belowground herb	0.21 ± 0.18 a	0.09 ± 0.03 b	0.15 ± 0.06 a b
Necromass	15.64 ± 7.10 a	17.81 ± 5.55 a	14.80 ± 5.90 a
Sum	193.79 ± 20.68 a	372.10 ± 117.77 b	318.99 ± 103.84 b

Total tree = aboveground + tree root, total understorey = total shrub + total herb, total shrub = aboveground shrub + belowground shrub, total herb = aboveground herb + belowground herb; data are mean values ± the standard deviations, the same letter in the same line indicates no significant difference at a  $p < 0.05$



and late selective logging plots respectively (Table 2). Aboveground biomass values were 79.5, 79.9, and 79.9% of the total tree biomass in no logging, early selective logging and late selective logging plots respectively (Table 2). Total understorey biomass was very low compared with tree biomass and necromass. Significant differences were detected between the three management regimes for understorey biomass but not necromass, which was highest in early selective logging plots but not significantly different from no logging and late selective logging plots.

*Carbon storage*

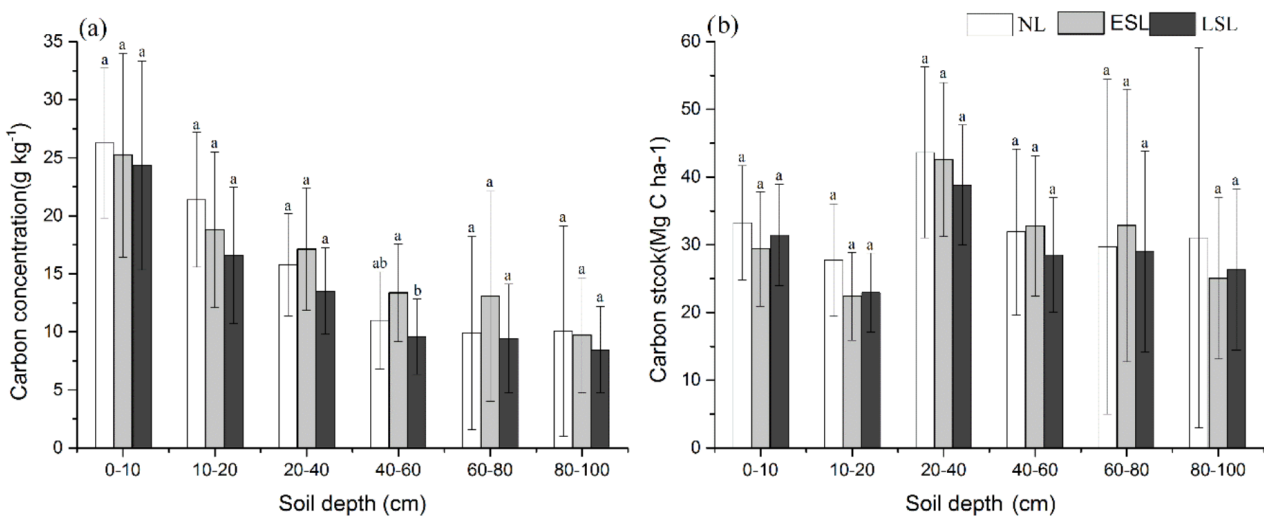
C stocks contained in trees (29.8–47.6%), understorey (0.2–0.5%), and dead plant materials (2.2–2.7%) followed the same pattern as biomass, suggesting trees contributed a large part of the total C stocks, with smaller contributions from understorey and necromass components. Total C stocks were the lowest in no logging plots, followed by late selective

logging and early selective logging plots (Table 3). However, the primary C sink was mineral soil, which contributed 49.9–67.1% to the total C stock. Soil C stocks were the highest in no logging plots but the values were not significantly different from other management strategies (Table 3). Soil C concentration decreased with increasing soil depth. In the top 20 cm of subsoil, C concentrations in no logging plots were higher than in early and late selective loggings (Figure 1). At intermediate depths (20–80 cm), early selective logging plots exhibited highest C concentration. No logging plots consistently showed highest soil C stocks compared with other management regimes, with the exception of the 40- to 80-cm soil depths (Figure 1). C stocks from the top 40 cm of soil represented 53.0, 51.0 and 52.6% of the entire soil profile (0–100 cm) in no logging, early selective logging, and late selective logging plots respectively (Figure 1). Analyses indicated no logging was significantly lower than the other two regimes in terms of forest C stocks.

**Table 3** C pools in different parts of the forest ecosystem (Mg C ha<sup>-1</sup>)

Management regime	Tree	Understorey	Necromass	Mineral soil (100 cm depth)	Total
No logging	87.56 ± 11.13 a	1.51 ± 0.69 a	7.82 ± 3.54 a	197.44 ± 37.52 a	294.33 ± 35.40 a
Ealy selective logging	176.51 ± 58.60 b	0.63 ± 0.49 b	8.90 ± 2.78 a	185.07 ± 48.40 a	371.12 ± 61.49 b
Late selective logging	151.03 ± 52.02 b	1.07 ± 0.46 a b	7.40 ± 2.95 a	174.61 ± 44.97 a	334.11 ± 70.34 ab

Data are mean values ± the standard deviations, the same letter in the same row indicates no significant difference at p < 0.05



**Figure 1** Soil (a) carbon concentrations and (b) stocks at different soil depths; NL = no logging, ESL = early selective logging, LSL = late selective logging; different letters indicate significant difference between three regimes at the same depth (p < 0.05), error bars represent standard deviations

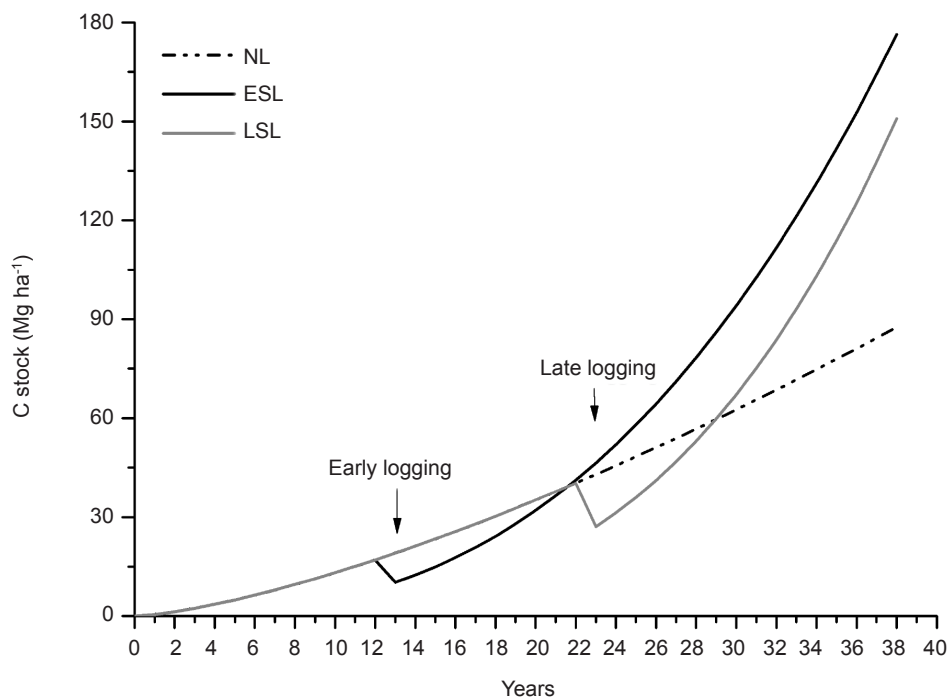
### Carbon loss

Selective logging plots maintained higher C stocks than no logging plots. We assumed the initial plot conditions for all three regimes were the same, and therefore used no logging as a base line (Figure 2). The nine no logging plots had 209 Chinese fir (average 23.2). Richard growth curve ( $D = 15.776 (1 - e^{-0.223 \times AGE})^{2.329}$ ) for suppressed Chinese fir at Jiangle Forest Farm (26° 26' N, 117° 40' E) of Fujian Province calculated the DBHs at different ages. Using the allometric equation of our samples ( $W_T = 0.0661D^{2.5764}$ ) (Table 1), average tree density and plot areas were applied to estimate Chinese fir C stocks in plots of early and late selective loggings in the 5<sup>th</sup> and 15<sup>th</sup> years respectively, when C stocks were exported during these management regimes. Early selective logging and late selective logging C losses were 8.89 and 13.21 Mg C ha<sup>-1</sup> respectively. Moreover, according to the final calculated biomass density (175.13 Mg ha<sup>-1</sup>) of no logging plots, the J-shaped growth curve equation was  $B = \text{age}^{1.5187}$ . In the fifth year, the average biomass density of the no logging plot was 38.17 Mg ha<sup>-1</sup>, from which 17.78 Mg ha<sup>-1</sup> were deducted to obtain the biomass density of early selective logging plot (7.21 Mg ha<sup>-1</sup>). The second point of the early

selective logging growth curve was 353.03 Mg ha<sup>-1</sup>. Then, we used nonlinear regression in SPSS to obtain constants for the early selective logging growth equation: biomass (B) = 0.022 × (AGE<sup>2.658</sup>). In the 15<sup>th</sup> year, the biomass density of no logging plot was 85.83 Mg ha<sup>-1</sup>, from which 26.42 Mg ha<sup>-1</sup> were deducted, and 302.02 Mg ha<sup>-1</sup> were fixed for late selective logging growth model:  $B = 0.011 \times (\text{AGE}^{3.423})$ . Biomass densities were converted to carbon densities using the 50% coefficient. Although significant differences in C stocks were not observed after 30 years, C accumulation rate was higher in late selective logging than in early selective logging plots.

### Floristic diversity

Both diversity indices (H' and D) showed that tree and herb diversity values in no logging stands were significantly lower than the values in selectively logged stands (early and late selective logging). Differences in shrub diversity between stand types were not significant (Table 4). The coefficients variation of diversity index (S) in no logging plots were higher than in early and late selective logging plots, particularly for tree and herbaceous layers, with values > 1. Floristic species richness in shrub and herb



**Figure 2** Changes in tree C stock under the three regimes during 30 years recovery with estimated carbon loss; NL = no logging, ESL = early selective logging, LSL = late selective logging

**Table 4** Floristic biodiversity index for three management regimes

Layer	Biodiversity index	No logging	Early selective logging	Late selective logging
Tree	H'	2.39 ± 0.26 a	2.50 ± 0.28 a	2.57 ± 0.27 a
	D	0.86 ± 0.04 a	0.89 ± 0.04 a b	0.91 ± 0.03 b
	S	21.33 ± 3.74 a	19.83 ± 3.79 a	19.67 ± 4.79 a
Shrub	H'	2.51 ± 0.36 a	2.55 ± 0.36 a	2.72 ± 0.24 a
	D	0.90 ± 0.08 a	0.90 ± 0.05 a	0.92 ± 0.04 a
	S	17.44 ± 3.43 a	19.17 ± 4.73 a b	21.50 ± 3.71 b
Herb	H'	0.44 ± 0.46 a	1.18 ± 0.41 b	0.75 ± 0.35 a
	D	0.27 ± 0.28 a	0.65 ± 0.10 b	0.44 ± 0.18 c
	S	4.92 ± 2.88 a	2.56 ± 1.01 b	3.17 ± 1.27 a

Data are mean values ± the standard deviation, different letters indicate significant difference at  $p < 0.05$ . H' and D are Shannon-Wiener and Simpson indices, S = total number of species.

layers was significantly different between the three management regimes, with the exception of the overstorey layer and total species richness ( $41.33 \pm 6.33$ ,  $43.92 \pm 6.20$  and  $44.33 \pm 4.56$  for no logging, early selective logging, and late selective logging plots respectively; results not shown).

#### Carbon stock and floristic diversity relationships

Both tree diversity indices (H' and D) were positively correlated (F test, both  $PC = 0.35$ ,  $p \leq 0.05$ ) with tree C stocks. Shannon-Wiener and Simpson diversity index of tree showed positive correlation 0.353 and 0.352 ( $p = 0.044$ , 0.045) at 0.05 level of significance with carbon stock of tree (Figure 3). Overstorey C stocks explained 12.4% of the H' and D indices. Shrub diversity was negatively correlated with necromass C storage (F test,  $p \leq 0.05$ ). There was no significant correlation between shrub and herb diversity and C stocks (Figure 3). Total floristic species richness exhibited weak correlation with tree stock ( $p = 0.047$ ) according to the one-tailed Pearson's correlation test. However, species richness of the different canopy layers did not correlate with C stock of each layer.

## DISCUSSION

### Distribution of biomass

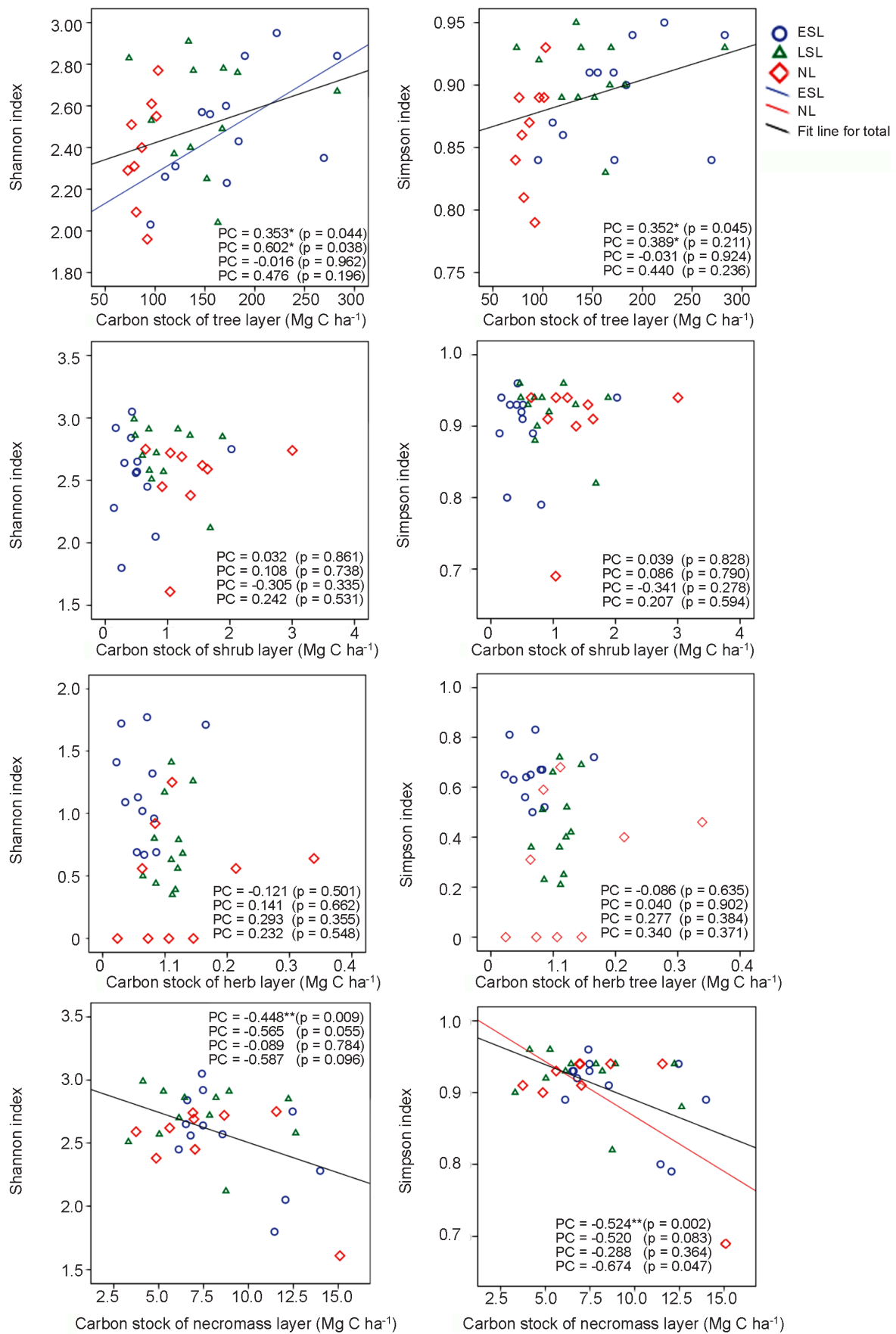
In the present study, forest management could possibly be responsible for differences in understorey biomass. The results for total biomass values for the three management regimes were similar to those reported by McEwan et al. (2011).

Branch biomass was lower than foliage biomass under all three regimes, likely due to better nutrient supplies and the low altitude (600–800 m) (Luo et al. 2013). The black humus depth of the plots (20 cm) was much deeper than the average 8.5 cm depth of soil in Shunchang county recorded by the Forest Management Planning Inventory in 2013 provided by Fujian Province Forestry Surrey Planning Institute. Based on functional equilibrium, biomass allocation patterns in plants maximise growth under given environmental conditions (Brouwer 1983). Under suboptimal conditions, light, CO<sub>2</sub> and water played only marginal roles in plant biomass allocation to leaves (Poorter & Nagel 2000). Instead, nutrient supply serve more important role than these factors in terms of changes in biomass allocation. Furthermore, species at low altitudes as in this study tend to have greater leaf mass than at middle (1000–1200 m) and high (1300–1500 m) altitudes (Xiang et al. 2009).

### Soil carbon stock distribution

C storage in each soil layer was higher than young stands of subtropical, evergreen broad-leaved forests in eastern China (12.70, 9.53 and 3.70 Mg C ha<sup>-1</sup> in 0–10, 10–30 and 30–50 cm depth respectively) (Zhang et al. 2010). Approximately 30% of soil C in the top 100 cm was stored in the 0–20 cm soil layer, which was lower than in a broad-leaved forest (40.9%) (Zhong & Zhao 2001). The difference is likely due to relatively higher soil C concentrations in the middle and deep soil layers (20–60 and 60–80 cm respectively) (Gong et al. 2011). Subsoil C has different





**Figure 3** Correlation between carbon stock of tree, shrub, herb and necromass layers and biodiversity index (Shannon and Simpson); NL = no logging, ESL = early selective logging, LSL = late selective logging; \*p-value < 0.05, \*\*p-value < 0.01, PC = Pearson coefficient

dynamics than the topsoil in tropical zones and could be disturbed by anthropogenic impacts (Schleuß et al. 2014). In policies for forest C management, we recommend promoting middle and subsoil C sequestration.

### Carbon stocks and floristic diversity relationships

Tree diversity was positively correlated with tree C stocks. The same trend was reported between aboveground live tree carbon stocks and tree species richness in miombo woodlands (Shirima et al. 2015). However, a negative correlation between diversity and total vegetation C stocks from 20 major forest types was detected in Garhwal Himalaya, India (Sharma et al. 2010). Results of studies varied in terms of the relationship between plant species composition or floristic richness and aboveground biomass, nutrient use efficiency, litter decomposition and other forest attributes (Day et al. 2014, Harguindeguy et al. 2008, Hiremath et al. 2002, Rivaie 2016). Positive, negative and not significant relationships have been reported between biodiversity and carbon stocks and included unimodal, bell-shaped, u-shaped and or kurtotic distribution (Loreau et al. 2001). The increased species richness and proportions of nitrogen-fixing species and species with low leaf mass per unit area increased C storage in mixed-species plantations, whereas increased proportions of large trees and species with high leaf mass per unit area increased tree C storage in natural forests (Ruiz-Jaen & Potvin 2011). The impacts of the characteristics, relative abundance, number and spatial pattern of species are, in principle, all likely to influence C sequestration (Lazzi et al. 2009). However, the interaction mechanisms, which attracted the interests of many ecologists, remained unclear. Facilitation from biodiversity was due to niche complementarity and sampling effect (Loreau et al. 2001). In this study, high overstorey layer diversity had high C stock of the subtropical broad-leaved forest.

### CONCLUSIONS

A positive relationship between the overstorey diversity indices ( $H'$  and  $D$ ) and tree C stocks was observed. However, significant correlation was not detected between shrub and herb diversity and their C stocks. This observation suggested

the elimination of the coniferous species (*C. lanceolata*) and the overstorey layer diversity could be favourable for C storage in subtropical, broad-leaved secondary forests during natural recovery processes.

### ACKNOWLEDGMENTS

This work was supported by the National Key R&D Program of China (2016YFC0502704), National Science Foundation of China (Nos. 31470578 and 31200363) and Fujian Provincial Department of Science and Technology Project (Nos. 2015Y0083, 2016Y0083, 2016T3037 and 2016T3032).

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