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MASTER’S DISSERTATION

论文题目：中国宁波地区亚热带森林生物多样性与生态系统碳储量的关系

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Department: College of Ecological and Environmental Sciences
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Research direction: Forest Ecology
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Abstract

Evergreen broadleaved forests play a central role in mitigating global warming in subtropical region. However, most productivity-species richness studies have been conducted in grasslands ecosystems, and only a few in forest ecosystems, perhaps because of the difficulty of using biomass as a surrogate for productivity in long-lived trees. The relationships between biodiversity components (stand structural and functional diversity) and ecosystem carbon (C) stocks in forest ecosystems particularly subtropical forests are poorly studied. To address this knowledge gap, firstly I investigated how tree species, size and height diversity alone or jointly affect C stocks in subtropical evergreen broad-leaved forests (EBLFs). Secondly, I investigated how functional diversity components i.e., functional divergence (niche complementarity hypothesis) and dominance (mass ratio hypothesis) mutually affected ecosystem C stocks in EBLFs. However, stand age (A), density (N) and site productivity (P) may also influence the relationship between stand structural diversity (tree species, size (diameter) and height diversities) and C stocks. Currently, (1) how does structural diversity affect C stocks in forest ecosystems with differing A, N and P, and (2) how does functional divergence of wood syndrome and dominance of leaf nutrient-use syndrome jointly affect ecosystem C stocks, are poorly understood. Understanding the effect of stand structural diversity and functional diversity components on forested ecosystem C stocks is important for maintaining ecosystem productivity and mitigating global climate change.

The study regarding to stand structural diversity-C stocks relationship was conducted across 80 subtropical forest plots (each size of 20 × 20 m) in five forest sites located on Tiantai Mountain near to Ningbo City, Zhejiang province in eastern China. The aboveground C stocks (ACS), and tree species, size and height diversity indices, as well as A, N and P, were determined across all 80 plots while soil organic C (SOC) stocks were measured across 65 plots by considering all five forest sites. Although there are well-developed allometric equations for trees (diameter at breast height-DBH > 5 cm), those equations are not suitable for calculating the aboveground biomass (AGB) of shrubs and small trees (DBH < 5 cm) because of the restriction of DBH range. Thus, the accurate estimation of AGB for shrubs and small trees are important for the analysis of diversity-C stocks relationships in forest ecosystems. Thus, I developed species-specific biomass allometric equations for common shrubs and small trees, as well as a multispecies shrub equations on basis of different stand variables (height-H, DBH and crown area-CA), for subtropical evergreen broadleaved forests (EBLFs) in eastern China.

For testing the relationship between stand structural diversity and C stocks, I used the null hypothesis approach through the application of generalized log-likelihood ratio (*LR*) test to investigate whether there were singular or joint effects of tree species, size and height diversities on ecosystem C stocks, when A, P and N were controlled. I found that stand structural diversity was positively correlated with aboveground C stocks (ACS). Tree species, size or height diversities, either alone or jointly, affected ACS, but not SOC (SOC). Stand structural diversities together had greater effect on ACS than A, N and P. Forest community or plot with high stand structural diversity had greater levels of ACS. Species, tree size and height diversities were singularly and jointly linked with ACS in subtropical forest ecosystems, supporting the niche complementarity hypothesis.

For exploring the functional diversity-ecosystem C stocks relationships, the C stocks in both above- and belowground partitions as well as in total ecosystem C stocks, and 14 plant functional traits by considering main parts of the plant (stem, branch and leaf), were determined across 31 subtropical forest plots in eastern China. I used two different components of functional diversity on the basis of single-trait indices: (a) community-weighted mean (CWM, locally dominant) and (b) functional divergence (FDvar) of trait values, to test the competing hypotheses on functional diversity-C stocks association's both in above and below-ground partitions as well as in total ecosystem.

I found that mostly CWM of leaf traits and FDvar of woody traits had significantly pair wise associations with C stocks in different partitions. Also, mostly CWM of leaf traits and FDvar of woody traits were retained in the final model of multiple stepwise regressions to reveal their explanatory powers with respect to C stocks. Furthermore, *LR* test confirmed that each functional diversity components had an additional contribution to each partition C stocks. Plots with high functional diversity had greater levels of total ecosystem C stocks. My study strongly suggested that a set of coexisting plant species in an ecosystem with the high dominance of nutrient-use syndromes and the divergent woody syndromes, results in greatest C stocks in both above-and below-ground partitions, even at total ecosystem level. Therefore, my results supported both mass ratio and niche complementarity hypotheses enhancing total ecosystem C stocks.

Thus my study demonstrated that increasing stand structural diversity and maintaining functional divergence of woody traits and functional dominance of leaf nutrient-use syndromes could be an effective approach to increasing ecosystem C stocks in subtropical forest ecosystems.

Key-words: ecosystem carbon stocks, functional dominance, functional divergence, functional traits, mass ratio hypothesis, stand structure, niche complementarity hypothesis.

摘要

亚热带常绿阔叶林的碳固持对减缓全球变暖有十分重要的作用。以往存在较多利用生物量估算草地生态系统碳储量的案例，但在森林系统中，由于准确估算立木生物量的困难，致使有关森林碳固持量估算、生物多样性和森林碳固持量关系的研究，并不多见。基于此，为探索生物多样性和森林碳储量的关系，本文选择常绿阔叶林为对象，在控制森林年龄（A）、森林立木密度（N）和生产力（P）不变的前提下，拟解决 2 个科学问题。1）选择森林的 3 个不同结构多样性指标（物种多样性、个体大小多样性和个体高度多样性），分析其对森林碳固持量的独自或综合影响；2）在上述结论成立的条件下，选择当前 2 种最为重要的、有关生物多样性对森林系统碳固持量变化维持机理的假说（生态位补偿补偿假说和质量比例假说），基于 14 个功能性状，利用这些功能性状的分离指数（functional divergence，FDvar）和功能优势度指数（community-weighted mean，CWM），分析上述 2 种假说对常绿阔叶林碳固持量变化的解释能力。

研究点位于浙江宁波地区，对于论文的第一个科学问题（森林生物多样性和森林地上碳固持量间的关系），涉及研究区 5 个森林类型 20×20 m^2 的 80 个样地，测定和计算了地上生物量和生物多样性，对于其中的 65 个样地，也测定了土壤有机碳储量。各样地内，胸径大于 5 cm 个体的地上生物量采用生物量—胸径的异速方程计算。对于胸径小于 5 cm 个体，先利用个体的树高、胸径和树冠面积对个体分级，随后计算每一级个体的生物量—胸径异速方程。最后结合每一物种的异速方程和多物种的综合异速方程，综合计算小胸径个体的地上生物量。森林地上总生物量基于每个物种的重要值加权获得。对于论文的第二个科学问题（生物多样性对森林系统碳固持量变化的维持机理），在 5 类型的 31 内样地内进行。

为证明第一个科学问题，本文在广义线性模型应用过程中，设定森林发育年龄、森林内植株密度和森林生产力为常量，运用一般对数概似值比（generalized log-likelihood ratio，LR）检验 3 种生物多样性对森林生态系统碳固持量的单独和综合影响。结果表明，1）森林的结构多样性（个体大小个体高度多样性）与森林地上碳固持量显著相关（$P < 0.05$）；2）森林物种多样性、个体大小多样性、个体高度多样性均独立或综合显著影响森林地面生物量（$P < 0.05$），但对森林土壤有机碳库的影响不大（P >

0.05）；3）相对于森林发育年龄、森林内植株密度和森林生产力，3 种多样性对森林地上生物量的影响较大；4）结构多样性较高的森林具有较高储量的地上生物量。

为证明第二个科学问题，本文基于 14 个功能性状，先通过计算各样地内的每一功能性状的 CWM（代表质量比例假说）和 FDvar（代表生态位补偿假说），后运用 GLM 逐步回归分析 2 种假说对森林碳固持量、地上生物量和土壤有机碳量变化的解释能力。结果表明，叶性状的 CWM 和干材性状的 FDvar 均和森林碳固持量、地上生物量和土壤有机碳量显著相关（$P < 0.05$），即说明生态位补偿假说和质量比率假说均对森林碳固持的变化具有解释能力；2）拥有高功能多样性的样地具有高的碳固持量；3）高碳固持量的森林表现出高树高和高枝干材密度的多样性，且植物叶氮浓度较低。

综合可知：物种多样性、个体大小多样性、个体高度多样性与森林地面生物量显著正相关，具有生物多样性高的森林具有高的碳储量。功能性状的分离和功能优势度的优势占据表明：植物相对高度和叶养分资源利用特征的空间分离符合生态位补偿假说和质量比率假说，可以解释森林碳储的变化。森林内，结构多样性的增加、干材性状的分离、资源利用性状的主导优势性是增加亚热带森林碳储量的主要生态学机制之一。

关键词：生态系统碳储量，功能主导优势，功能分离，功能性状，质量比率假说，森林结构，生态位补偿假说

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Abbreviations

A	Stand age
ACS	Aboveground carbon stocks
AGB	Aboveground biomass
BEF	Biodiversity-ecosystem functioning
C	Carbon
CA	Crown area
CD	Crown diameter
CFC	Chlorofluorocarbons
CH_4	Methane
cm	Centimeter
CO_2	Carbon dioxide
CWM	Community weighted mean
D_{45}	Diameter at 45 cm of plant height
DBH	Diameter at breast height
EBLFs	Evergreen broad-leaf forests
FDvar	Functional divergence
g	Gram
GHGs	Greenhouse gases
Gt	Gigatonne
H	Plant height
H	Plant height
H_0	Null hypothesis
H_1	Alternative hypothesis
H_d	Tree size diversity
H_h	Tree height diversity
H_s	Tree species diversity
LDMC-F	Current year leaf dry matter content
LDMC-M	Mature leaf dry matter content
Ln	Natural log
LNC-F	Current year leaf nitrogen concentration
LNC-M	Mature leaf nitrogen concentration
LPC-F	Current year leaf phosphorus concentration

LPC-M	Mature leaf phosphorus concentration
LR	Log-likelihood ratio test
m	Meter
Mg ha^{-1}	Megagram per hectare
N	Stand density
N_2O	Nitrous oxide
NPP	Net primary productivity
P	Site productivity
P_0	Null prediction
P_1	Alternative prediction
PCA	Principal Component Analysis
SLA-F	Current year leaf specific leaf area
SLA-M	Mature leaf specific leaf area
SNC	Stem nitrogen concentration
SOC	Soil organic carbon stocks
SOM	Soil organic matter
SPC	Stem nitrogen concentration
SWD	Stem wood density
TDMC	Twig dry matter content
TEC	Total ecosystem carbon
TWD	Twig wood density

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1. General introduction

1.1. Research background

Carbon (C) is continuously removed from the atmosphere by forest ecosystem processes and stored both in vegetation and soils (e.g. McGuire *et al.* 2001). If C stocks of ecosystems increase, the C content of the atmosphere is reduced. On the other hand, if the C stocks in ecosystems were to reduce, the increase rate of carbon dioxide (CO_2) in the atmosphere would be much faster than currently observed (Reich 2011). Therefore, global forests are very diverse and provide a range of ecosystem services such as C sequestration, flood and landslide protection, biodiversity preservations, aesthetic and health benefits (Pan *et al.* 2013). Forests are usually defined by the presence of trees and absence of non-forest land use, even though trees are also numerous outside forests in savannas, pasture lands, and in suburban areas and green city centers (Nowak & Greenfield 2012). Hence, forest trees are recognized as very important in the global C cycling, because of the amount of C stored in plant biomass globally exceeds that of atmospheric CO_2, and nearly 90 percent of the plant biomass C is stored in tree biomass (Mooney *et al.* 2001). Forests are thus the worldwide fundamental C pool and sink sequestrating approximate 47% of terrestrial C (Dixon *et al.* 1994). The sequestration of CO_2 among forests depends on forest type, dominant tree species and forest stand age (Huy & Anh 2008). Furthermore, the role of biodiversity played in shaping ecosystem functioning becomes a basic concern in ecology (Loreau *et al.* 2001; Diaz *et al.* 2009). For developing the best ecological plans to protect biodiversity, maintain ecosystem productivity and mitigate the effects of global warming, the relationships between species diversity and ecosystem productivity and thus C stocks have been received particular emphasis in recent decades (Mittelbach *et al.* 2001; Vilà *et al.* 2003; Firn *et al.* 2007; Kirby & Potvin 2007; Bosworth *et al.* 2008; Liang *et al.* 2007; Lei *et al.* 2009). However, in the forested ecosystems, discrepancies among studies remain unresolved. This highlights the need to precisely determine the amount of C stored in each of specific forest ecosystems.

This chapter will introduce some important topics of the thesis, namely (1) The role of forest ecosystems in global C stocks (Section 1.2), (2) Biodiversity-ecosystem functioning (BEF) associations (Section 1.3), (3) Methods of forest C stocks measurements (Section 1.4), and (4) Concept of biodiversity in ecology (Section 1.5). Based on reviews of relevant literature, this introduction will clarify some scientific theory, research knowledge gap, methods and major C pools regarding biodiversity-C stocks relationships in typical subtropical evergreen broad leaf forests (EBLFs).

1.2. The role of forest ecosystems in global carbon stocks

The increase of CO_2 in the atmosphere is becoming a global issue. Global warming is the increase in the average temperature of the earth's surface resulting from the rise in the concentration of greenhouse gases (GHGs) like carbon dioxide CO_2, methane (CH_4), nitrous oxide (N_2O), and chlorofluorocarbons (CFC). The increased concentration of GHGs in the atmosphere attributes to the change in the world's climate. GHGs destroy the ozone layer allowing the ultra violet rays to pass towards the earth surface. The extreme heat emitted in the earth surface through radiation has dangerous effect on plants, animals, human race, and its surrounding environment. Forest vegetation are considered as an key factor in mitigating global warming because of their role in C sequestration - the process of removing carbon dioxide (CO_2) from the atmosphere and 'storing' it in plants that use sunlight to turn CO_2 into biomass and oxygen. Forest trees are recognized as very important in the global C cycling, because of the amount of C stored in plant biomass globally exceeds that of atmospheric CO_2, and nearly 90 percent of the plant biomass C is stored in tree biomass (Mooney *et al.* 2001). Forests are thus the worldwide fundamental C pool and sink sequestrating approximate 47% of terrestrial C (Figure 1.1; Dixon *et al.* 1994). The sequestration of CO_2 among forests depends on forest type, forest class, dominant tree species and forest stand age (Huy & Anh 2008). This highlights the need to precisely determine the amount of C stored and CO_2 sequestrated in each of specific forest ecosystems.

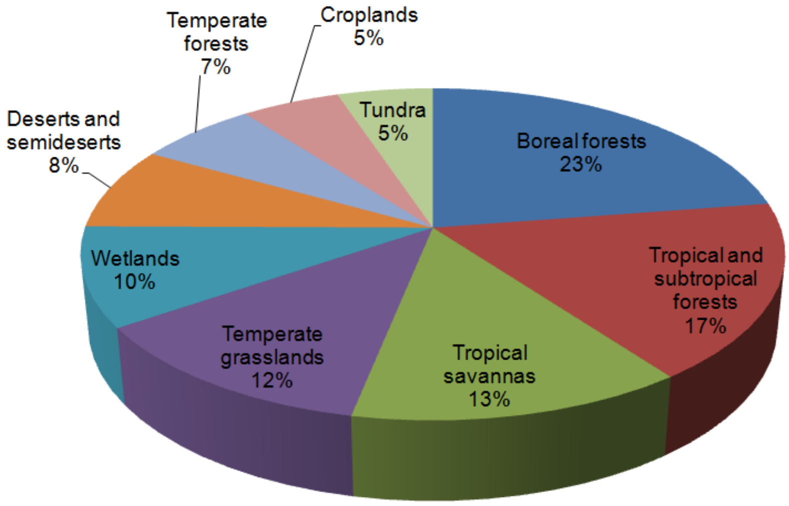

Figure 1.1 Contributions of different ecosystems in global C stocks. Boreal forests contribute more total C stocks than any other terrestrial ecosystem (23% of total terrestrial C stocks), tropical and subtropical forests account for 17% while temperate forests account for 7% (Dixon *et al.* 1994). In comparison with other forest in other terrestrial ecosystems, tropical and subtropical vegetation has a very high ACS stock (see Table 1.1).
Note: Total C stocks = 2477 Gt; 1 gigatonne (Gt) = 1 billion tonnes.
Source: (Dixon *et al.* 1994; IPCC 2001; Schlesinger 1997).

Release of certain GHGs into the atmosphere is being affected by human activities. Out of these gases, CO_2 in the atmosphere is GHGs which is drastically bringing climate change (IPCC 1995). Forests are the potential C sink as they store C as reservoir after sequestering from the atmosphere (Prentice *et al.* 2000). Deforestation is the main cause of global warming and because of which forest trees have been recognized as a potential ecosystem for measurement to mitigate global warming (FAO 2001).

According to the IPCC (Eggleston *et al.* 2006), there are five C pools of ecosystem involving biomass, namely the vegetation, belowground biomass, litter dead mass, dead woody debris and SOM. The CO_2 fixed by vegetation during photosynthesis is transferred across the different C pools. The vegetation constitutes the major portion of the C pool. It is the most important and visible C pool of the forest ecosystem (Ravindranath & Ostwald 2008). Any changes in the land use system like forest degradation and deforestation has a direct impact on this component of the C pool. The belowground biomass which constitutes all the live roots (Eggleston *et al.* 2006) plays an important role in the C cycle by transferring and storing C in the soil. The dead mass of litter and woody debris are not a major C pool as they contribute merely a small fraction to the C stocks of forests (Ravindranath & Ostwald 2008). Soil C pool is also a chief contributor to the C stocks of forests (Kumar *et al.* 2006) next only to the vegetation C pool (Ravindranath & Ostwald 2008). Forests vegetation and soils share almost 60 percent of the world's terrestrial C (Winjum *et al.* 1992), which are important sinks for atmospheric C and could significantly contribute to mitigation of global climate change with correct management (Bajracharya *et al.* 1998; Lal 2004; Smith 2004). These C pools continue to increase over the life cycle of a forest towards a state of equilibrium when respirational CO_2 losses by plants and soils and decomposition of biomass equals rate of growth (Smithwick *et al.,* 2002 and Acker *et al.,* 2002). The proportion of the total forest C stock stored in understory vegetation is usually only 1-2 percent (Pussinen *et al.* 1997), but disturbances or other exceptional situations may introduce wide variation.

However, the biomass of understory vegetation may plays role in ecosystems and its processes (Yarie 1980).

Table 1.1 Global C stocks in vegetation and soil C pools down to a depth of 1 m. Among the all biomes, tropical and subtropical forests has higher C stocks in vegetation C pool while boreal forests has higher C stocks in soil C pool.

Biome	Area (10^9 ha)	Global carbon stocks (Gt C)		
		Vegetation	Soil	Total
Boreal forests	1.37	88	471	559
Tropical and subtropical forests	1.76	212	216	428
Tropical savannas	2.25	66	264	330
Temperate grasslands	1.25	9	295	304
Wetlands	0.35	15	225	240
Deserts and semi-deserts	4.55	8	191	199
Temperate forests	1.04	59	100	159
Croplands	1.60	3	128	131
Tundra	0.95	6	121	127
Total	15.12	466	2011	2477

In absolute terms, soil C stocks are much larger than C sequestered in tree biomass (Lal 2004). A recent calculation of C storage values, including soil C stock estimates revealed significantly higher estimates in almost all biomes, including an approximately threefold increase in SOC estimates for tropical forests (Eglin *et al.* 2011). The tropical or sub-tropical forests contained more than 50% C stocks in soils and almost less than 50% in the vegetation (Table 1.1) because the dead biomass rapidly decomposes in the humid environment and the minerals rapidly leach out of tropical forest soils (Ross 2009).Therefore, C concentration in soils is highly significant; it is assumed that soils are having approximately three times more C than vegetation and twice as to that present in atmosphere (Batjes 1996).

1.3. Biodiversity-ecosystem functioning (BEF) association

Anthropogenic change threatens biodiversity worldwide and increased the accelerating rate of species extinction up to one thousand times the background rate by modifying the environment (Sala *et al.* 2000; Andelman & Willig 2003; Carney *et al.* 2004; Millennium Assessment 2005). According to the red list of International Union for Nature Conservation (IUCN 2009), threatened species contains 36% of 47,677 evaluated species are endangered with extinction counting 70% of the plant species evaluated. Humans are dependent on ecosystems for the services they offer; however, the capability for ecosystems to offer these services depends on how humans treat them. And, while it is plausible that ecosystem services are reliant on biota, there is incomplete knowledge in how exactly ecosystem services are reacting to the current trend of biodiversity loss. In an attempt to fill this gap in knowledge, ecologists have commenced a motivated research effort to quantify that how biodiversity loss affects ecosystems. An important goal in ecology is to understand ecosystem structure through quantification of biodiversity and ecosystem processes and determining the relationships between them (Hooper *et al.* 2005; Keddy 2005). In the past 30 year, interest has increased dramatically in determining the relationship of biodiversity-ecosystem functioning (BEF). One of the most commonly studied relationship is that between species richness and productivity, or its surrogates (see Waide *et al.* 1999; Mittelbach *et al.* 2001 for additional references). However, most productivity-species richness studies have been conducted in grasslands (Tilman *et al.* 1996; Hector *et al.* 1999; Chalcraft *et al.* 2009), and only a few in forests (Caspersen & Pacala 2001; Creed *et al.* 2009), perhaps because of the difficulty of using biomass as a surrogate for productivity in long-lived trees (Vance-Chalcraft *et al.* 2010). The associations between tree species diversity and ecosystem productivity have been reported to either positive (Tilman *et al.* 1996; Hector *et al.* 1999; Whittaker & Heegaard 2003; Balvanera *et al.* 2006; Erskine *et al.* 2006), negative (Huston 1980; Firn et *al.* 2007), or non-significant (Vilà et *al.* 2003). Early experiments conducted in grasslands showed mostly a decrease in plant productivity with a decline in plant species richness (*e.g.*, Naeem *et al.* 1994; Tilman et *al.* 1996; Hector et *al.* 1999; Whittaker & Heegaard 2003; Balvanera et *al.* 2006; Erskine et *al.* 2006).

Much of the discussion over the mechanisms behind observed BEF relationships has focused on whether species diversity effects are driven by niche partitioning and facilitation (i.e. the complementarity effect; Tilman *et al.* 1997) or by the selection of one or more highly productive species (i.e. the selection effect; Loreau & Hector 2001; Cardinale *et al.* 2012). The niche complementarity hypothesis postulates that a diverse array of species has a greater divergence of functional traits and can thus better utilize limiting resources, thus increasing total ecosystem functioning, than a less diverse community. The complementarity effect, is the increase in relative productivity among species in a mixture compared with the productivity of the species grown in monocultures due to positive interactions among species (*i.e.*, facilitation and partitioning of resources, *e.g.*, Naeem *et al.* 1994; Tilman et *al.* 1996). The mass ratio (selection effect) hypothesis states that most dominant species or traits drive ecosystem functioning and that positive BEF relationships arise simply because diverse communities are more likely to include high functioning species and traits. A better understanding of how functional divergence and dominance affect ecosystem function would help direct preservation and restoration plans for exploited ecosystems.

Positive relationships among species, rather than the existence of a highly dominant species, mostly determine the effect of diversity on plant productivity (Cardinale *et al.* 2007). In a meta-analysis of 44 separate BEF experiments (39 in terrestrial grassland, two in tundra, two in bryophyte, and one in estuarine), Cardinale *et al.* (2007) observed that plant productivity increased with diversity because of the complementarity effect in most experiments and that the effect increased over time. Complementarity occurs when species either facilitate the growth and insistence of other species (*e.g.*, legumes fixing nitrogen; Hooper & Dukes 2004) or partition the utilization of resources either temporally, spatially or chemically (*e.g.*, temporal differences in nitrogen use; Hooper 1998). Specific experiments testing for facilitation and resource partitioning are necessary to determine the mechanism resulting in a complementarity effect (*e.g.*, von Felten *et al.* 2009) and so far quite little is known about specific mechanisms motivating the effects of diversity. The current consensus among ecologists is that the loss of biodiversity alters ecosystem functions (Hooper *et al.* 2005). Due to logistic ease, most BEF research has been on productivity and nutrient retention in experimental grassland ecosystems; however the applicability of these results to forest ecosystems and non-experimental settings is questionable (Loreau *et al.* 2001). In a meta-analysis of 449 BEF studies, biodiversity effects were shown to vary among ecosystem types (Balvanera *et al.* 2006). However, because certain ecosystems are very scarce in the literature (*e.g.*, subtropical forest ecosystems, Figure 1.2), the reason of these scarcity is unknown (Schmid *et al.* 2009).

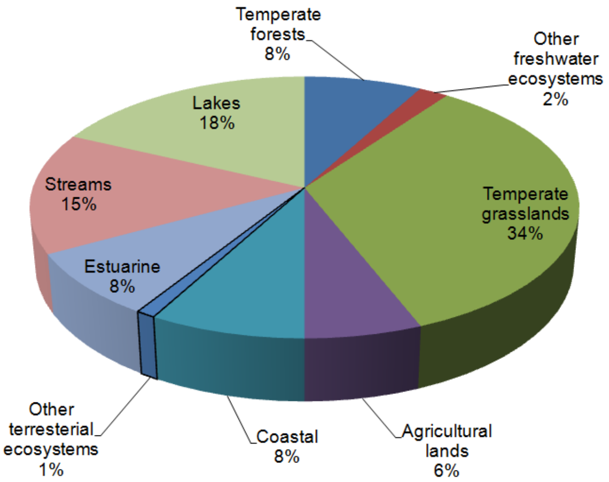

Figure 1.2 Representations of various ecosystem types in BEF literature "Other terrestrial ecosystems" refers to the category containing emergent subtropical forests studies. Modified from Cardinale *et al.* (2006), emphasis added.

Few studies have looked at the relationship between biodiversity and C stocks in forest ecosystems (Caspersen & Pacala 2001; Creed *et al.* 2009), perhaps because of the difficulty of using biomass as a surrogate for productivity in long-lived trees (Vance- Chalcraft *et al.* 2010). Furthermore, these relationships in the subtropical evergreen broad leaved forests (EBLFs) are relatively scarce in the literature. The existing research conducted in other forest ecosystems showed different relationships between tree species diversity and C stocks. For example, significant positive relationships between them was observed in dry tropical forests of India (Sagar & Singh 2006), deciduous forests of north America (Houle 2007) and agro-ecosystems of in south western Uganda (Nakakaawa et *al.* 2009). On the other hand, Szwagrzyk & Gazda (2007) reported that negative relationship exists between aboveground biomass (AGB) and tree species diversity in natural temperate forests of Central Europe and the same conclusion was found for Quercus communities in the transition zone of subtropical and warm temperate in northwest China (An-ning *et al.* 2008). Another case is a meta-analysis by Phillips *et al.* (1994) showing that there was a positive relationship between forest dynamic and tree species richness in tropical rain forests. Furthermore Bunker *et al.* 2005 reported that a simulation of 18 possible extinction scenarios using the data from a 50 ha tropical forest plot shows that C stocks in tropical forest depends on species composition. However, Chen (2006) reported a positive relationship between tree species diversity and SOC in the top 30 cm soil layer in an old-growth forest of northeast China, but Jonsson & Wardle (2009) reported that plant diversity significant weakly affects belowground C stocks in boreal forest ecosystems. On the other hand, Zhang *et al.* (2011) found no significant relationship between plant diversity and SOC in the top 30 cm soil in a subalpine coniferous forest in southwest China. Thus, this variability may probably be explained by the confounding effects resulting from other stand variables such as height, size, age, site productivity and stand density that significantly influence ecosystem productivity (Liang *et al.* 2007; Lie *et al.* 2009) and might also affect ecosystem C stocks differently.

Currently, only few studies have tested the assumption of above two hypotheses respectively, in terms of linkage between functional diversity and C stocks (Butterfield & Suding 2013; Conti & Díaz 2013). Recently, Conti & Díaz (2013) found that functional dominance of plant height (H) and to a much lesser degree stem wood density (SWD) predicts ecosystem C stocks, supporting the mass ratio hypothesis. Comparatively, Butterfield & Suding (2013) reported that functional divergence of H among rangeland herbs links with higher production, supporting niche-complementarity hypothesis. It is important to consider both the mass ratio hypothesis and the complementarity hypothesis when functional diversity-C stocks associations are examined. Based on the assumptions of the mass ratio hypothesis (Grime 1998), functional strategies of the dominant species have overwhelming influence on ecosystem C stocks. The increasing probability at high productivity levels that one or a few highly competitive species out-compete other species can counteract resource-use complementarity and finally diminish functional divergence (Schumacher & Roscher 2009). Unfortunately, no one has compared the explanatory power of both hypotheses simultaneously with respect to relationship between functional diversity components and each of aboveground C stocks (ACS) and soil organic C (SOC) stocks within forest ecosystems.

1.4. Methods of forest carbon stocks measurements

Assessment of the AGB has a major role in the quantification of forested ecosystem productivity, sustainability and functioning. It also gives us an idea of the potential C stocks that can be released in the form of CO_2 during deforestation (Vashum & Jayakumar 2012). Biomass estimation of the forest ecosystem enables us to estimate the C stocks, using an international standard conversion factor (0.5), by assuming that 50 % of the tree biomass has elemental C (Dixon *et al.* 1994; Nizami *et al.* 2009; Nizami 2012). Forest biomass can be estimated through direct field measurements, forest inventory data and remote sensing methods (Lu 2006; Ravindranath & Ostwald 2008).

Field measurement includes two methods of AGB estimation i.e. 1) destructive method and 2) non-destructive method. Among all the existing AGB estimation methods, the destructive method is the direct way for estimation of biomass and the C stocks stored in the forest ecosystems (Gibbs *et al.* 2007). This method includes harvesting of all the trees in the plot and measuring the wet weight of the different components of the cut tree i.e., tree stem, leaves and branches (Ravindranath & Ostwald 2008; Devi & Yadava 2009; Hashimotio *et al.* 2000; Lodhiyal & Lodhiyal 2003) and then measuring the oven dried weight of these components. Although this method estimates the biomass exactly for a specific area but it is time and resource consuming, destructive and costly, and it is not appropriate for a large scale area and degraded forests containing rare species (Montès *et al.* 2000). Commonly, this method designed for developing biomass allometric relationships with stand variables to be used for calculating biomass on a larger-scale (Segura & Kanninen 2005; Navár 2009). The second method, the non-destructive method, estimates the ABG without tree cutting, keeping biodiversity conservation. Non-destructive method contains different procedures for biomass estimation. For example, Montès *et al.* (2000) estimated AGB of the individual tree by considering the tree shape, physical samples of different components of the trees, dendrometric measurements, tree volume and bulk density of the different tree components. Although it is a non-destructive method but still need to validate the estimated biomass by taking physical samples for oven-dry weight. An additional way of estimating the AGB by non-destructive method is by climbing the tree to measure the different components (Aboal *et al.* 2005) or just measuring the DBH, height, volume of the tree and wood density (Montès *et al.* 2000) and calculate the biomass using existing allometric equations (Brown *et al.* 1989; Nowak 1993; Hughes *et al.* 1999). Although these methods do not require harvesting of tree species but is not easy to validate this method reliability. These methods are also best but human resources and time consuming.

Forest inventory-based approaches to estimate AGB and C stocks by using allometric equations have been used all over the world. Many researchers have developed allometric biomass prediction equations for different types of forest ecosystem and even for tree species (Brown *et al.* 1989; Montès *et al.* 2000; Basuki *et al.* 2009; Navár 2009; Chave *et al.* 2005). The allometric equations for biomass estimation are developed by establishing a relationship between the various stand indices such as the DBH, height of the tree stem, total tree height, crown area, tree species, etc. Specie or site-specific equations estimate the biomass for specific sites and for large-scale universal and local comparisons. Brown *et al.* (1989) developed different allometric regression equations to estimate the AGB of individual trees for tropical forests as a function of DBH, total height and wood density. The Brown's biomass equation estimation is only applicable for the live trees, and not the fallen litter and the dead wood debris. The forest C stocks are widely estimated from the allometric equations for forest biomass. Usually, the C concentration of the different components of a tree is assumed to be 50% of the biomass (Brown 1997) or 45% of the biomass (Whittaker & Linkens 1973). However, Losi *et al.* (2003) estimated in their study that the C concentration of dry bole sample to be approximately 48% of the dry bole biomass. Djomo *et al.* (2011) analyses the C content in wood with a CNS analyzer and found a mean value of 46.53%. The AGB estimation of the forest can be calculated out using any of the above methods or in combination of the methods mentioned above. But one should keep in mind the applicability of that method for the area or forest type or tree species when choosing a method for AGB estimation. The generalized allometric equations or regression models for AGB estimation also should not be used beyond their range of biometric variables (Brown *et al.* 1989; Chave *et al.* 2005). Although, the harvesting method gives a more accurate estimate of the AGB, it is destructive and time consuming. Therefore, allometric equations is often the preferred method for estimating stand AGB as this method provides a non-destructive and indirect measurement of large scale AGB, also it is less time consuming and less expensive. These allometric equations uses only the biometric variables obtained from the forest inventories to estimate the AGB. However, the allometric equations developed for AGB estimation need to be validated by cutting and weighting of tree components are required (Araújo *et al.* 1999; Montès *et al.* 2000; Djomo *et al.* 2011; Kim *et al.* 2011; Ryan *et al.* 2011).

Soil C consists of organic C as organic matter containing a range of organic materials and inorganic C as carbonates and bicarbonates. Organic C stocks are ~1500 Gt (1015 g) C and inorganic C stocks are ~720 Gt C in the top 1m of soil depth (Batjes 1996). Soil organic C is heterogeneous in nature and consists of several SOC pools, which can be broadly grouped based on their turnover rates in soil. For example, Parton *et al.* (1987) postulated three SOC pools: (i) the active or labile C pool; (ii) the slow C pool; and, (iii) the resistant or passive pool. These have turnover periods of, respectively, <10 years, 10–200 years, and >100 years.

Only recently, routine soil C assessments have been widely based on elemental measurements such as detection of CO_2 produced by dry combustion of a soil sample (Franks *et al.* 2001). Most soil analytical laboratories worldwide have produced large datasets for estimating soil C based on either dichromate oxidation or mass loss on ignition. The Walkley-Black procedure is widely used because it is rapid, easy, and has less tools needs (Nelson & Sommers 1996). However, this method has been shown to lead to the partial oxidation of organic C and is mainly poor for consuming elemental C forms. Different researches have shown that the organic C recovery using the Walkley-Black procedure ranges from 60 to 86% with an average recovery counted by 76% (Walkley & Black 1934). As a result of the partial oxidation and in the lacking of a site-specific correction factor, 1.33 as a correction factor is generally applied to the results to correct the organic C recovery. Nelson & Sommers (1996) listed 13 published variations in dichromate oxidation methodology for estimating soil C, including that described by Walkley & Black (1934), with each variation in methodology having its own conversion factor for translating the indirect measurement to estimates of soil C. Nelson & Sommers (1996) also listed 14 published variations for ignition conditions (temperature and time) used for mass loss on ignition protocols with each having its own regression equation to convert mass loss to estimates of soil organic matter.

1.5. Concept of biodiversity in ecology

According to Diaz *et al.* (2006), biodiversity as the number, abundance, composition, spatial distribution, and interactions of genotypes, populations, species, functional types and traits and landscape units in a given system. Furthermore, stand structure is an important factor of stand biodiversity (MacArthur & MacArthur 1961; Willson 1974; Franzreb 1978; Temple *et al.* 1979; Freemark & Merriam 1986). High biodiversity is linked with stands where there are numerous tree species and sizes (Buongiorno *et al.* 1994). For forested ecosystems, stand structural diversity can indicate overall species diversity (Kimmins 1997), as shown in research on avian and insect diversity (Whittaker 1972; Franzreb 1978; Temple *et al.* 1979; Moen & Gutierrez 1997). Managing forests for biodiversity may be accomplished by managing for structural diversity (Önal 1997).

In ecology, the literature on diversity is infinite (Dennis *et al.* 1979) but three measures of diversity are well-known. Species richness, a simple calculation of the species numbers, is easy but it ignores frequency of species. Shannon-Wiener's diversity index was a measure of entropy in the beginning but afterward applied to information theory (Shannon 1948; Shannon & Weaver 1949). The Gini-Simpson diversity index (Simpso 1949) had in fact been used formerly to measure economic discrimination. All above three ecological diversity indices are closely linked which can be derived from the same one-parameter family of diversity indices (Patil & Tailie 1979; Keylock 2005). I chose Shannon-Wiener's index for stand structural diversities (tree species, size and height diversity) in the present study because it reflects both species evenness (E) and richness (S) (Magurran 1988), without supporting either dominant or rare species while Simpson's index favoring dominant species.

Functional diversity is recognized to treat all species as ecologically non-independent and may therefore provide enough information about the relationships between biodiversity and ecosystem function such as productivity (Tilman *et al.* 1997; Mouchet *et al.* 2010). Two components of plant functional diversity were used to test the strength of relationship between biodiversity and ecosystem C Stocks (Fig. 1.3). Firstly, community weighted trait means (CWM), as an indicator for functional dominance, represents the expected functional trait value of a specific community (Díaz *et al.* 2007). Secondly, functional divergence (FDvar) of the single-trait, which basically represents the variance in trait values weighted by the abundance of each species in the community (Mason *et al.* 2003). I selected FDvar over other functional dissimilarity indices, because it incorporates species relative abundance and has demonstrated to have good performance (Pakeman, Lennon & Brooker 2011).

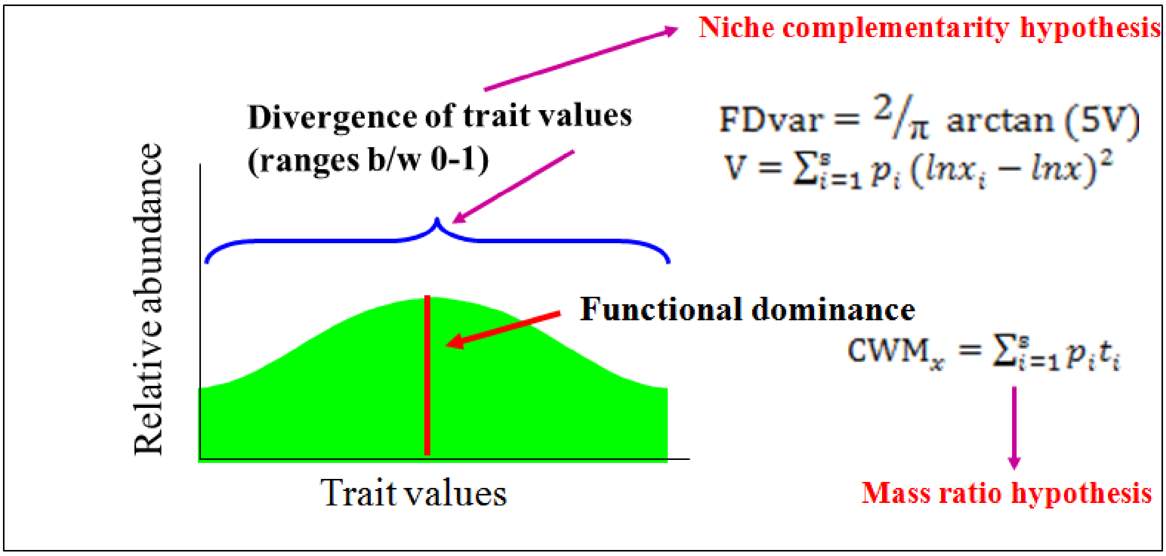

Figure 1.3 Conceptual diagram for functional diversity components i.e. functional dominance and divergence. The functional dominance support mass ratio hypothesis while functional divergence support niche complementarity hypothesis.

1.6. Thesis synopsis

My thesis is a biodiversity-C stocks relationship study in subtropical EBLFs of Zhejiang province in eastern China, focusing on three linked parts: (1) to enhance our understanding of quantify forest ecosystem C stocks both in aboveground (trees and shrubs) and soil, (2) to test stand structural diversity-C stocks relationship (niche complementarity effect) in subtropical EBLFs utilizing a natural experiment of involving forest developmental stages with differing diversity, and (3) to test the joint effect of functional dominance (defined here as CWM, based on the mass ratio hypothesis) and functional divergence (defined here as FDvar, based on the complementarity hypothesis) of plant syndromes on ecosystem C stocks in the subtropical forests.

For the development of shrub biomass allometric equations, data were collected from established three sampling plots, 10 × 10 m in size, in Tiantong National Forest Park, located in Zhejiang Province in eastern China. Then, 14 species and an average of seven individuals per species were selected on the basis of their dominance and abundance in each sampling plot. I measured AGB through a non-destructive method but the estimated AGB was validated by destructive sampling of plant components (stem wood, branches and leaves) for determining oven-dry weight (Montès *et al.* 2000). Furthermore, different biometric variables were measured for each individual plant. Fresh subsamples of stem wood, branch or twig and leaves were collected from an average of seven individuals per species, stored in sealed plastic bags, and then transported to the laboratory where fresh and oven-dried weights (oven-dried at 80 °C until constant weight) were recorded to estimate the water content (%). I also measured the wet and dry wood density for each plant. Therefore, the total fresh weight of each individual was calculated through the multiplication of wet wood density and total tree volume (Miles & Smith 2009). The water content was used to convert the fresh mass to aboveground dry biomass (AGB) (Whittaker &Woodwell 1968). Finally, simple linear regressions were used the development of allometric equations based on single variables while multiple regressions were used for the development of multivariable equations.

The stand structural diversity (species, size and height) and other stand variables (A, P and N) from young, pre-mature and mature forests were quantified in order to test (through log-likelihood ratio, *LR* test) whether the stand structural diversity mutually or individually in additions with other stand variables were plausible factors for enhancing forest ecosystem C stocks. I collated a forest data set of field measurements from 80 plots for estimation of C stocks in tree biomass while soil samples were collated from 65 plots for SOC determination in experimental laboratory, with accompanying stand structural diversity data (stand density, species richness, species diversity, size and height diversity) from three forest types of five mountains having contrasting diversity to quantify the stand diversity-C stocks relationship.

For functional diversity, 14 plant functional traits were measured on plant main organs (stem, branch and leaf) in 31 subtropical EBLFs plots. Firstly, the trait specific relationship between functional diversity components and ecosystem C stocks were determined. Then joint effect of functional dominance and divergence were explored on ACS, SOC and total ecosystem C (TEC) stocks through multiple regressions by including all functional diversity indices based on different functional traits. The relative contribution of functional diversity components were tested through *LR* test. Finally, the performance of each forest community for TEC stocks against functional divergence and dominance were also assessed.

1.7. Thesis aims, hypotheses and objectives

Thesis aims and hypotheses: to investigate the relationships of biodiversity components *i.e.,* stand structural diversity (tree species, size and height diversity) and functional diversity (divergence and dominance) with ecosystem C stocks in subtropical EBLFs in eastern China. My central hypotheses were that (1) when A, P and N are equal, stand structural diversity may have a positive effect on C stocks in evergreen broadleaved forests (EBLFs) widely distributed in subtropical China, support niche complementarity hypothesis (Chapter 3), and (2) functional dominance (defined here as CWM, based on the mass ratio hypothesis) and functional divergence (defined here as FDvar, based on the complementarity hypothesis) of plant syndromes do jointly enhance ecosystem C stocks in the subtropical forests (chapter 4). The specific objectives of the study were:

Objective 1: To quantify biomass estimation of shrubs in subtropical EBLFs in eastern China, a contribution to C stocks estimation (Chapter 2).

1.1. To develop species-specific allometric equations for biomass estimation of subtropical common shrubs or small trees (DBH < 5 cm).

1.2. To develop multispecies generalized allometric equations base on different stand variables for biomass estimation of subtropical shrubs and small trees, where species-specific equations are yet not available.

Objective 2: To quantify the effects of stand structural diversity on carbon stocks in the subtropical EBLFs (Chapter 3).

2.1. To determine the relationships between stand variables and ecosystem C stocks.

2.2. To determine the mutual and individual effects of stand structural diversity on ecosystem C stocks.

2.3. To investigate the performance of each forest community with respect to stand diversity-C stocks relationship.

Objective 3: To quantify the effects of functional diversity on ecosystem carbon stocks in the subtropical EBLFs (Chapter 4).

3.1. To determine the trait specific relationships between functional diversity components and ecosystem C (ACS, SOC, and TEC) stocks.

3.2. To determine the joint effects of functional dominance and divergence on each of the ecosystem C stocks.

3.3. To quantify the relative contribution of functional dominance and divergence to each of the ecosystem C stocks

3.3. To investigate the performance of functional dominance and divergence on ecosystem C stocks in each studied plot.

Objective 4: To provide implications for future biodiversity conservation and carbon stocks management (Chapter 5).

4.1. To discuss potential managerial implementations for C stocks management in subtropical EBLFs.

4.2. To recommend research questions for future work.

2. Biomass estimation for shrubs in evergreen broad-leaved forests in eastern China: a contribution to carbon stocks estimation

2.1. Introduction

The estimation of aboveground biomass (AGB) is a central issue in ecology (Flombaum & Sala 2007). Thus, AGB is a key ecosystem property resulting from the net balance between carbon (C) gains through photosynthesis and losses by respiration, tissue turnover, and natural or human disturbance (Chapin *et al.* 2011). AGB provides essential biological information and has been used for various purposes, such as estimations of nutrient cycling, net primary productivity (NPP), wood production, and recently, C stocks and their greenhouse effect (Ter-Mikaelian & Korzukhin 1997; Hierro *et al.* 2000; Keller *et al.* 2001). One of the major reasons of AGB estimation is to accurately quantify the relationship between forest ecosystem C stocks and biodiversity (Vilà *et al.* 2003; Erskine *et al.* 2006; Firn *et al.* 2007). The accurate estimation of AGB for shrubs and small trees (diameter at breast height-DBH < 5 cm) is important for the analysis of diversity-C stocks relationships in the forest ecosystems. Although there are well-developed allometric equations for trees (DBH > 5 cm, e.g., Brown *et al.*1989; Chave *et al.* 2005), those equations are not suitable for calculating the biomass of shrubs and small trees because of the restriction of DBH range.

Forest biomass can be estimated through direct field measurements, forest inventory data and remote sensing methods (Lu 2006; Ravindranath & Ostwald 2008). Field measurement includes two methods of AGB estimation i.e. 1) destructive method and 2) non-destructive method. Among all the existing biomass estimation methods, the destructive method, also called as the harvest method, is the direct way for estimation of AGB and the C stocks stored in the forest ecosystems (Gibbs *et al.* 2007). This method includes cutting of all the trees in the plot and measuring the green weight of the different components of the cut tree i.e., tree stem, leaves and branches (Ravindranath & Ostwald 2008; Devi & Yadava 2009; Hashimotio *et al.* 2000; Lodhiyal & Lodhiyal 2003) and then measuring the oven dried weight of these components. Even though this method determines the biomass more accurately for a specific area but it is time and resource consuming, destructive and expensive, and it is not applicable for a large scale area and degraded forests containing endangered species (Montès *et al.* 2000). The second method, the non-destructive method, estimates the AGB without tree cutting, keeping biodiversity conservation. Non-destructive method includes different approaches for biomass estimation. For example, Montès *et al.* (2000) estimated AGB of the individual tree by considering the tree shape, physical samples of different components of the trees, dendrometric measurements, volume and bulk density of the different components. Even though it is a non-destructive method, to validate the estimated biomass, still need to take physical samples for dry weight. I used this method because 1) it protects the forest from cutting and validated the biomass estimation through samples taken from each part of the plant for dry weight, 2) the destructive method include cutting the tree just for green weight measurement, which can also be calculated through direct drondrometric measurements of tree volume and wet wood density.

Shrub biomass is an important component of the total forest biomass, especially in the natural forest. However, because of the lack of methodology and difficulty in calculation, mostly researchers are avoiding the shrub vegetation in biomass estimation which results in underestimation of the total biomass (Karki 2002; Khanal 2001). In other cases, the basal area of shrub is estimated as cross-sectional area of a tree (by measuring diameter) and then biomass is calculated by applying the same equation of tree biomass estimation (Palm 2003), which may over or under estimate the biomass depending on species attributes. Unfortunately, shrub-specific allometric equations for shrubs and small trees are scarce for subtropical evergreen broad-leaved forests.

In this study, I aimed to develop species-specific allometric equations for common shrubs and small trees biomass estimation models in the subtropical EBLFs in eastern China. I tested the relative performance of height (H), diameter (D) and crown area (CA) variables over each other to predict individual shrub AGB. I also developed a multispecies shrub allometric equation for the EBLFs. Although this study includes only subtropical EBLFs shrub species, I provide a basis for the consistency of useful variables to predict more accurately the AGB shrub biomass component in other subtropical EBLFs.

2.2. Methods and materials

2.2.1. Study site and plot

The study was conducted in Tiantong National Forest Park (29°48′N, 121°47′E, 200 m a.s.l.), Zhejiang province, China, covering an area of 349 ha. The region has warm and humid sub-tropical climate with an average annual temperature of 16.2 °C, average annual precipitation of 1374.7 mm (mostly concentrated in the summer), annual average relative humidity is 82 % (shows little intra-annual variability) and mean annual evaporation is 1320.1 mm (less than annual precipitation). The soil is mostly mountain yellow-red soil, with parent material mostly including Mesozoic sedimentary rocks, some acidic igneous rocks, and granite residual weathered material (Song & Wang 1995).

The zonal vegetation in this region is subtropical EBLFs. In Tiantong National Forest Park, the majority of the EBLFs are *Schima superba* Gardn. et Champ dominated forests, which are considered as sub-climax monsoon EBLFs and have been severely disturbed in the history with only small tracks (approximate 10 ha) of semi-intact forests left around a Buddhist temple (Yan *et al.* 2009). Since forest age, community structure and plant species composition in this area are similar (Yan *et al.* 2013), three square plots with each size of 10 x 10 m was established to represent the ranges of both community and environmental properties in this study. The tree layer is 15~18m high and cover percentage is 80~90%, occupied by evergreen broadleaf species. The shrub layer is < 4m in height and coverage is 45~50%. The herb layer is < 0.5 m in height with coverage being 10~ 30 % and the dominant species are normally ferns.

2.2.2. Species selection and sampling

In this study, I considered shrubs and small trees (DBH < 5 cm). Thus, 14 species were selected on the basis of their locally dominance and abundance that were frequently appeared in each sampling plot. All selected species are common across the subtropical EBLFs ecosystem. Out of 14 species, *Schima superba*, *Castanopsis carlesii* and *Camellia fraternal* are closed related to young forest and widely distributed across all forest types in the area. But the first two species appeared as dominant in young forests.

An average of seven individuals per species was chosen in order to cover the possible range of plant sizes of each species. The selected species were sampled across the study plots having almost similar climatic and soil conditions. At each studied plot, several stand variables usually used in allometric shrub biomass estimations were measured in every individual plant prior to take sub-samplings (Table 2.1). Such as, total height (H), defined as the distance between the ground surface and the highest crown point; longest stem diameter (D) over the root collar (usually between 5 and 45 cm from the ground surface); maximum crown diameter (CD1), and its perpendicular diameter (CD2). Crown diameters (CD1 and CD2) were used to calculate crown area (CA) as follows (equation 2.1).

$$\mathrm{CA} = \pi \times (\mathrm{R}_1 \times \mathrm{R}_2) \qquad \text{eqn (2.1)}$$

where, CA is crown area (m^2), R_1 is the radius from the longest crown diameter (CD1, m) while R_2 is radius from the crown diameter, perpendicular to CD1 (CD2, m).

In addition, I tested whether the addition of crown shape improved the prediction level of models, based on different geometric volumes, with respect to stem variables to predict shrub AGB (Ludwig *et al.* 1975). I calculated an inverted cone-shaped crown (CoC), hemisphere-shaped crown (HsC) and parabolic-shaped crown (PrC) as additional AGB predictor variables using equations 2.2, 2.3 and 2.4 respectively.

$$\mathrm{CoC} = \pi \frac{\mathrm{CD}^2\mathrm{L}}{12} \qquad \text{eqn (2.2)}$$

$$\mathrm{HsC} = \pi \frac{\mathrm{CD}^3}{12} \qquad \text{eqn (2.3)}$$

$$\mathrm{PrC} = \pi \frac{\mathrm{CD}^2\mathrm{L}}{8} \qquad \text{eqn (2.4)}$$

where, CD is crown diameter (m) and L is crown depth (m), calculated as total tree height (H) minus height of crown lowest leaf.

Fresh subsamples of stem wood, branch or twig and leaves were collected from each individual per species, stored in sealed plastic bags, and then transported to the laboratory where fresh and oven-dried weights (oven at 80 °C until constant weight) were recorded to estimate the water content (%) per individual of each species. Also I measured the wet and dry stem wood density (SWD) for each individual plant. Therefore, the total fresh weight of each individual was calculated through the multiplication of wet wood density and total tree volume (Miles & Smith 2009). The water content was used to convert the fresh mass to dry AGB (Whittaker &Woodwell 1968) (Table 2.1).

I used tree increment cores for plant having diameter greater than 1 cm to determine SWD. Using a 5 mm-diameter increment corer, I collected one core at breast height or at 45 cm height from each plant which was also used for collecting leaf and twig samples. In the laboratory, the length and diameter of each tree core was measured using an electronic vernier caliper (accurate to 0.1 mm), then the volume of each tree core was calculated, supposing the tree increment core was estimated to a cylinder shape. For plant having diameter less than 1 cm, I cut the sample from the stem. In the laboratory, each fresh stem sample was totally immersed in a volumetric flask of known weight filled with distilled water, and the increase in weight was recorded. The weight increment equals the green volume of the wood sample. Wet SWD was measured and after that samples were then dried at 75ºC in an oven for 72 hrs to determine dry mass and to calculate dry SWD (Chave *et al.* 2006).

Table 2.1 Summary of the range values (min - max) of biometric variables and total aboveground dry biomass per species of shrub and small trees (DBH < 5 cm) of subtropical forests of eastern China.

Species name	*n*	H (m)	D (cm)	CA (m^2)	PrC (m^3)	DMC (%)	SWD (g cm^{-3})	AGB$_o$ (kg)
Adinandra millettii (Hook. Et Arn.) Benth. Et Hook. f. ex Hance	5	1.2 - 2.9	1.0 - 3.4	0.57 - 3.22	0.07 - 1.53	24.77 (2.74)	0.57 (0.01)	0.03 - 0.53
Camellia fraterna Hance	6	1.1 - 2.4	0.9 - 3.5	0.77 - 3.53	0.21 - 0.96	25.58 (2.52)	0.6 (0.03)	0.02 - 0.35
Castanopsis carlesii (Hemsl.) Hay.	13	0.5 - 4.0	0.7 - 4.5	0.14 - 3.22	0.01 - 2.17	28.41 (3.73)	0.51 (0.04)	0.01 - 2.07
Cyclobalanopsis glauca (Thunb) Oerst.	5	1.3 - 2.5	0.6 - 3.9	0.39 - 3.08	0.06 - 0.54	30.90 (1.37)	0.7 (0.07)	0.01 - 0.55
Cyclobalanopsis stewardiana (A. Camus) Y.C. Hsu et H.W. Jen	5	2.2 - 4.3	2.1 - 4.5	1.27 - 8.31	0.08 - 2.29	29.94 (3.19)	0.62 (0.01)	0.24 - 2.25
Diospyros kaki Thunb var. sylvestris Makino	5	2.2 - 3.8	2.3 - 4.5	1.9 - 4.54	0.23 - 1.77	22.80 (0.81)	0.53 (0.04)	0.27 - 0.93
Eurya nitida	8	1.2 - 2.7	0.8 - 3.6	0.57 - 9.82	0.12 - 1.96	24.72 (1.77)	0.57 (0.03)	0.02 - 0.70
Eurya rubiginosa var. attenuate H.T. Chang	7	0.7 - 3.2	0.4 - 2.0	0.10 - 2.26	0.01 - 1.30	25.77 (3.89)	0.59 (0.06)	0.003 - 0.27
Loropetalum chinense (R. Br.) Oliver	9	0.9 - 4.2	0.8 - 2.5	0.57 - 2.65	0.05 - 1.81	27.01 (1.87)	0.60 (0.09)	0.02 - 0.37
Machilus thunbergii (Siebold & Zuccarini)	5	1.7 - 4.1	1.1 - 4.9	1.27 - 9.05	0.22 - 4.52	25.57 (3.72)	0.53 (0.02)	0.04 - 1.95
Quercus fabri (Hance)	9	1.2 - 3.0	0.5 - 4.8	0.06 - 9.82	0.01 - 6.87	31.00 (2.62)	0.59 (0.05)	0.01 - 1.42
Schima superba Gardn. et Champ	6	1.9 – 4.6	1.5 - 4.2	1.27 – 5.09	0.17 - 2.31	27.70 (2.99)	0.54 (0.02)	0.09 – 1.71
Symplocos setchuensis Brand	8	2.4 - 4.4	1.8 - 4.6	0.57 - 3.08	0.10 - 2.46	25.75 (0.99)	0.51 (0.04)	0.17 - 2.06
Symplocos stellaris Brand	5	1.0 - 3.2	0.7 - 3.0	0.09 - 3.08	0.02 - 1.19	25.32 (4.14)	0.49 (0.04)	0.01 - 0.51
Total dataset	96	0.5 - 4.6	0.4 - 4.9	0.06 - 9.82	0.01 - 6.87	----------------	----------------	0.003 - 2.25

n, number of trees per species; H, total height (m); D, longest stem diameter (cm); CA, crown area (m^2); DMC, plant dry matter content per species (%); SWD, dry stem wood density (g cm^{-3}); PrC, parabolic crown variable (m^3); AGB$_o$, observed aboveground dry biomass (kg).

Values in brackets represent the standard deviation (SD).

2.2.3. Statistical analysis

Regressions analyses were conducted between observed dry AGB and tree variables (D, H and CA). If necessary, variables were log-transformed in order to apply linear models. I developed single-variable (D or H or CA) allometric equations for each species, best model was selected on the basis of high significant R^2. Then I also developed multi-variables equation for each species through multiple regression analysis ($P < 0.05$), the best fitted model including more than one variable was selected on the basis of high significant R^2. In addition, I constructed a set of multispecies AGB regression equations considering all individuals from the 14 species together (n =96).

Three different allometric equations were developed for multispecies: (1) single-variable model, (2) Two variables model and (3) multiple-variables model referred to include all three variables. Single-variable models were developed using simple linear regression analysis. Two variables and multiple-variables models were developed using multiple linear regression analysis. The best statistical model developed for multispecies was selected according to the R^2 and Akaike Information Criterion (*AIC*) (Johnson & Omland 2004), where the best model has the lowest value of *AIC*. I also mentioned the predictive mean squared error (*PMSE*) of the regression.

Dry wood density and crown shape were included after the best predicted model, if improve the R^2 value of the model. AGB estimations on the basis of log-transformed variables are expected to underestimate the real value. Thus, I calculated a correction factor (CF) (Baskerville 1972) to be applied to AGB in those cases where log-transformed variables included in the equation. The CF must be applied to AGB estimation as follows (equation 2.5).

$$\mathrm{AGB} = \mathrm{CF} \times \mathrm{e}^{\ln(\mathrm{AGB})} \qquad \text{eqn (2.5)}$$

where, $\mathrm{CF} = \mathrm{e}^{\frac{\mathrm{SEE}^2}{2}}$; CF is correction factor and SEE is standard error of estimation.

2.3. Results

2.3.1. Species-specific allometric models

The best-fitted variable (single and multiple) models developed per species, corresponding parameters, CF, and statistical criteria are presented in Table 2.2. Among the single-variable models, D was the best predictor variable of AGB estimation for all except three species, explaining variability in AGB at the range of 59-99 %. The best fit corresponded to

Symplocos stellaris, *Machilus thunbergii*, *Eurya nitida* and *Cyclobalanopsis glauca* ($R^2 = 0.99$; $P < 0.0001$). The exceptions were *Schima superba*, *Symplocos setchuensis* and *Quercus fabri*. In case of *Schima superba*, H instead of D was the best variable for AGB ($R^2 = 0.84$, $P < 0.05$), although D also performed well ($R^2 = 0.71$, $P < 0.05$). For *Symplocos setchuensis*, also H instead of D was the best variable for AGB ($R^2 = 0.79$, $P < 0.05$). Thus, after D, H was the second best predictor variable for species-specific AGB estimation in almost all cases. CA showed almost weak predictive capability, with the highest value and best predictor for *Quercus fabri* ($R^2 = 0.68$, $P < 0.05$).

Among the multiple-variables models, all three variables (D, H and CA) together or combination of D and H were the best predictor variables of AGB estimation for all except three species, explaining up to 99 % variability in AGB in all cases. The exceptions were *Cyclobalanopsis stewardiana*, *Schima superba* and *Quercus fabri*. The combination of H and CA performed better for *Cyclobalanopsis stewardiana* ($R^2 = 0.99$, $P < 0.01$) and *Schima superba* ($R^2 = 0.87$, $P < 0.05$). For *Quercus fabri*, combination of D and CA were best variables for AGB ($R^2 = 0.82$, $P < 0.05$). The inclusion of SWD also increased the prediction power of model for many species but if not included in model will never create so big difference. The inclusion of crown shape (parabolic) just increased the prediction power for *Castanopsis carlesii* both in single and multiple variable models.

Table 2.2 Best fitted species-specific regression models for the prediction of shrub and small trees (DBH < 5 cm) aboveground dry biomass of subtropical forests of eastern China. Parameters and statistical criteria are shown for the best fitted model ($n = 96$).

Species	Species-specific aboveground dry biomass model	R^2	CF
Adinandra millettii	**1)** $\mathrm{Ln}(AGB_p) = -10.7+2.50\times\mathrm{Ln}(D)+12.5\times SWD$	0.96*	1.06
	2) $\mathrm{Ln}(AGB_p) = -3.83+1.99\times\mathrm{Ln}(D)+0.860\times\mathrm{Ln}(H)$	0.99**	1.01
Camellia fraterna	**1)** $\mathrm{Ln}(AGB_p) = -3.54+2.33\times\mathrm{Ln}(D)$	0.87***	1.12
	2) $\mathrm{Ln}(AGB_p) = -4.03+1.16\times\mathrm{Ln}(H)+2.15\times\mathrm{Ln}(D)+0.0022\times\mathrm{Ln}(CA)$	0.99**	1.00
Castanopsis carlesii	**1)** $\mathrm{Ln}(AGB_p) = -1.70+1.18\times\mathrm{Ln}(D)+0.634\times\mathrm{Ln}(PrC)$	0.70**	1.94
	2) $\mathrm{Ln}(AGB_p) = -1.66+1.34\times\mathrm{Ln}(D)-0.21\times\mathrm{Ln}(H)+0.64\times\mathrm{Ln}(PrC)$	0.67**	2.05
Cyclobalanopsis glauca	**1)** $\mathrm{Ln}(AGB_p) = -17.4 +0.141\times\mathrm{Ln}(D)+21.6\times SWD$	0.99*	1.00
	2) $\mathrm{Ln}(AGB_p) = -3.55+2.01\times\mathrm{Ln}(D)+0.867\times\mathrm{Ln}(H)$	0.99*	1.00
Cyclobalanopsis stewardiana	**1)** $\mathrm{Ln}(AGB_p) = -3.67+3.07\times\mathrm{Ln}(D)$	0.98**	1.02
	2) $\mathrm{Ln}(AGB_p) = -3.86+3.36\times\mathrm{Ln}(H)-0.307\times\mathrm{Ln}(CA)$	0.99**	1.01
Diospyros kaki	**1)** $\mathrm{Ln}(AGB_p) = -2.80+1.64\times\mathrm{Ln}(D)$	0.70*	1.05
	2) $\mathrm{Ln}(AGB_p)= -5.57+1.89\times\mathrm{Ln}(D)+1.16\times\mathrm{Ln}(H)+2.72\times SWD$	0.99*	1.00
Eurya nitida	**1)** $\mathrm{Ln}(AGB_p) = -3.59+2.47\times\mathrm{Ln}(D)$	0.99***	1.01
	2) $\mathrm{Ln}(AGB_p) = -3.78+2.00\times\mathrm{Ln}(D)+0.75\times\mathrm{Ln}(H)+0.063\times\mathrm{Ln}(CA)$	0.99***	1.00

Eurya rubiginosa	**1)** $\mathrm{Ln}(AGB_p) = -3.42+2.22\times\mathrm{Ln}(D)$	0.59*	1.72
	2) $\mathrm{Ln}(AGB_p) = 8.92-0.71\times\mathrm{Ln}(D)-1.06\times\mathrm{Ln}(H)+1.10\times\mathrm{Ln}(PrC)-16.1\times SWD$	0.99*	1.03
Loropetalum chinense	**1)** $\mathrm{Ln}(AGB_p) = -9.50+2.70\times\mathrm{Ln}(D)+10.0\times SWD$	0.76*	1.20
	2) $\mathrm{Ln}(AGB_p) = -8.93+1.95\times\mathrm{Ln}(D)+0.769\times\mathrm{Ln}(H)+8.76\times SWD$	0.83*	1.16
Machilus thunbergii	**1)** $\mathrm{Ln}(AGB_p) = -3.51+2.59\times\mathrm{Ln}(D)$	0.99***	1.01
	2) $\mathrm{Ln}(AGB_p) = -3.14+2.49\times\mathrm{Ln}(D)-0.875\times\mathrm{Ln}(H)+0.503\times\mathrm{Ln}(CA)$	0.99*	1.01
Quercus fabri	**1)** $\mathrm{Ln}(AGBp) = -10.9+0.753\times\mathrm{Ln}(CA)+14.8\times SWD$	0.68*	2.11
	2) $\mathrm{Ln}(AGB_p) = -13.5+1.15\times\mathrm{Ln}(D)+0.346\times\mathrm{Ln}(CA)+18.2\times SWD$	0.82*	1.68
Schima superba	**1)** $\mathrm{Ln}(AGB_p) = -3.74+2.79\times\mathrm{Ln}(H)$	0.84*	1.11
	2) $\mathrm{Ln}(AGB_p) = -3.68+3.08\times\mathrm{Ln}(H)-0.361\times\mathrm{Ln}(CA)$	0.87*	1.12
Symplocos setchuensis	**1)** $\mathrm{Ln}(AGB_p) = -5.62+3.24\times\mathrm{Ln}(H)+2.06\times SWD$	0.79*	1.13
	2) $\mathrm{Ln}(AGB_p) = -4.77-0.384\times\mathrm{Ln}(D)+3.72\times\mathrm{Ln}(H)$	0.79*	1.13
Symplocos stellaris	**1)** $\mathrm{Ln}(AGB_p) = -3.63+2.66\times\mathrm{Ln}(D)$	0.99***	1.02
	2) $\mathrm{Ln}(AGB_p) = -3.24+3.83\times\mathrm{Ln}(D)-1.65\times\mathrm{Ln}(H)+0.0045\times\mathrm{Ln}(CA)$	0.99*	1.01

AGB_p, predicted aboveground dry biomass (kg); Ln, natural logarithm; H, total height (m); D, longest stem diameter (cm); CA, crown area (m^2); SWD, dry stem wood density ($g\ cm^{-3}$); PrC, parabolic crown variable (m^3).

R^2, regression coefficient indicated with asterisks if statistically significant. *: $P < 0.05$; **: $P < 0.01$; ***: $P < 0.001$)

PMSE, predictive mean squared error; CF, correction factor

1) Single-variable best fitted model (inclusion of SWD, if improve the model capacity)

2) Multiple-variables best fitted model (inclusion of SWD, if improve the model capacity)

2.3.2. Multispecies aboveground dry biomass regression models

Single-, two- and multiple-variables models for all species considered together ($n = 96$), their fitted parameters, and statistical criteria are shown in Table 2.3. Similar to the species specific allometric models, D was the best predicted single-variable for multispecies AGB estimation, followed by H (Figs 2.1a and 2.1b). The regression between CA and AGB clearly showed the very weak relationships (Fig. 2.1c) with almost all species.

Among the two-variable models, D and H were the only best predictable variables for multispecies AGB estimation. Furthermore, multiple-variable model explained the same variation (71%) as two-variable (D-H) model in AGB for multispecies. As expected, the inclusion of SWD improved while crown shaped variable didn’t improve the fitness of the models. Figure 2.2 shows the performance of the best fitted multispecies model (model 3b, Table 2.3) including D, H, CA, and WSG. In any case, it is important to note that regression models should not be used beyond their range of validity. Models proposed here are valid between the ranges of allometric variables shown in Table 2.1.

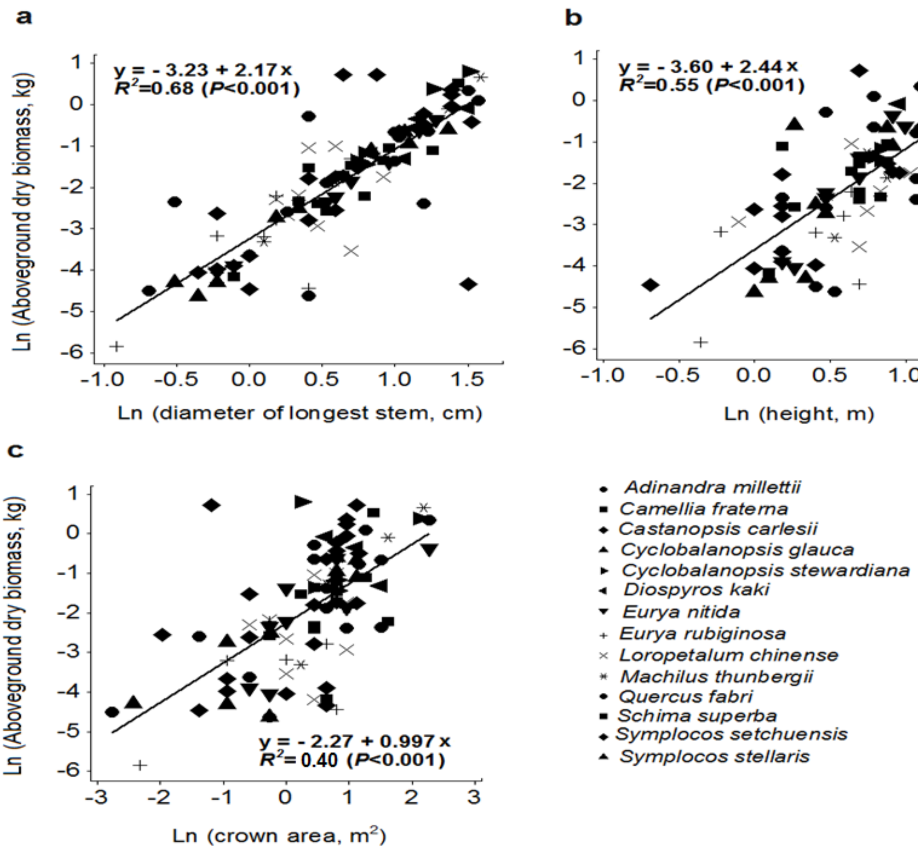

Figure 2.1 Relationship between the aboveground dry biomass (kg) and (a) diameter of longest stem (D, cm), (b) height (H, m) and (c) crown area (CA, m^2), for the complete dataset. Different symbols represent different species.

Table 2.3 Best fitted multispecies regression models for the prediction of shrub and small trees (DBH < 5 cm) aboveground dry biomass of subtropical forests of eastern China. Parameters and statistical criteria are shown for the best fitted model (n = 96).

Multispecies aboveground dry biomass model	R^2	*PMSE*	*AIC*	CF
1) Single- variable model				
a) $Ln(AGB_p) = -3.23+2.17×Ln(D)$	0.68***	0.82	250.70	1.46
b) $Ln(AGB_p) = -4.97+2.20×Ln(D)+3.06×SWD$	0.70***	0.79	247.68	1.45
2) Diameter-height model				
a) $Ln(AGB_p) = -3.50+21.65×Ln(D)+0.842×Ln(H)$	0.71***	0.75	244.48	1.42
b) $Ln(AGB_p) = -5.40+1.65×Ln(D)+0.885×Ln(H)+3.31×SWD$	0.73***	0.72	240.09	1.34
3) Multiple-variable model				
a) $Ln(AGB_p) = -3.43+1.50×Ln(D)+0.782×Ln(H)+0.16×Ln(CA)$	0.71***	0.79	244.72	1.42
b) $Ln(AGB_p) = -5.29+1.52×Ln(D)+0.83×Ln(H)+0.145×Ln(CA)+3.23×SWD$	0.73***	0.74	240.54	1.40

AGB_p, predicted aboveground dry biomass (kg); Ln, natural logarithm; H, total height (m); D, longest stem diameter (cm); CA, crown area (m^2); SWD, dry stem wood density (g cm^{-3}).

R^2, regression coefficient indicated with asterisks if statistically significant. *: $P < 0.05$; **: $P < 0.01$; ***: $P < 0.001$).

PMSE, predictive mean squared error; AIC, Akaike information criterion; CF, correction factor

a) Single-variable or multiple-variables models

b) Single-variable or multiple-variables models including stem wood density (SWD)

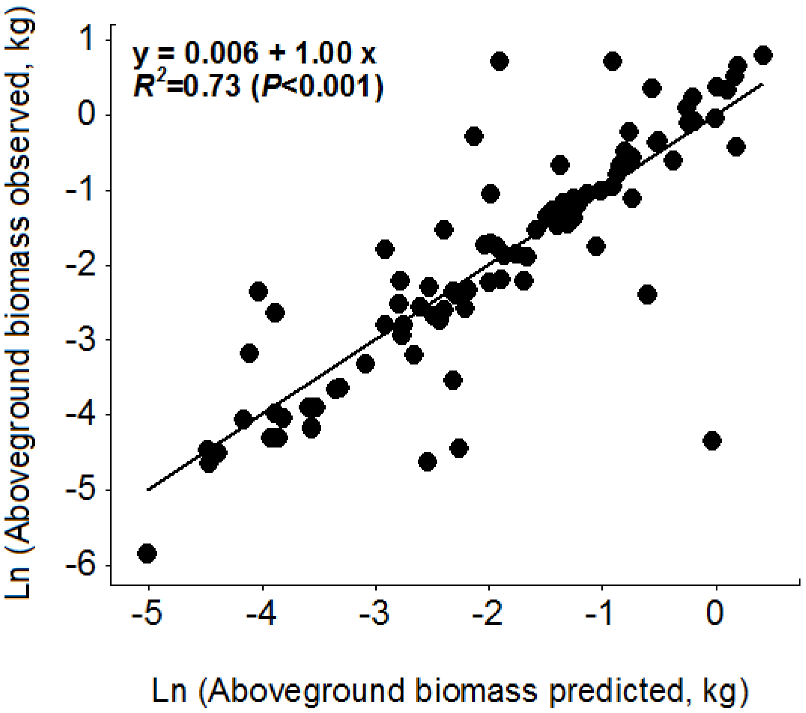

Figure 2.2 Capacity of model 2b (Table 2.3) for predicting shrub AGB, considering all species together. Each dot corresponds to an individual weighted shrub $\mathrm{Ln}(AGB_p) = -5.40+1.65\times\mathrm{Ln}(D)+0.885\times\mathrm{Ln}(H)+3.31\times SWD$

2.4. Discussion

Biomass allometric equations particularly developed for shrubs and small trees are relatively scarce in the literature. This study makes a contribution in that sense by testing the validity of several models including three main stand variables i.e., D, H and CA, for the estimation of individual shrub AGB in the subtropical EBLFs in eastern China. The results presented in this study showed that the models based on D-H variables performed better than CA and multiple variables based models for several species. For species-specific models, the multiple-variables performed better but just for three species out of 14 species. Additionally, D-H based model for multispecies model also fit significantly better (according to AIC and PMSE values) and explained the same variation to multiple-variables model. In practice, measurements of several variables could be very costly and concentration has to be paid to the range of validity of each variable separately, so models including fewer common variables are convenient and therefore preferred (Conti *et al.* 2013).

My study has made advances in two important aspects relevant to understanding of shrubs biomass estimation. At the individual level, the developed species-specific allometric equations allow a precise, quick and easy estimation of AGB of dominant and abundant shrub species in the studied forests. Moreover, the estimation is non-destructive and requires little efforts to validate the accuracy of AGB estimation. After developing 28 species-specific equations for 14 species and three multispecies allometric equations, I strongly recommend the use of D-H variables over other common allometric variables as best predictors of individual AGB of shrubs in subtropical EBLFs. Furthermore, my results also demonstrated that the use of natural log-log models is the most suitable form for providing a common statistical basis for quantifying size (D-H)-AGB relationships in almost all cases presented, as was found for others shrub species around the world (Hierro *et al.* 2000; Sah *et al.* 2004; Hofstad 2005). D and H variables were good predictor for species-specific models, as found by Yang et al. (2010) for several species, thus recommended for multispecies allometric equation because forest age, community structure and plant species composition of each forest ecosystem in this area are almost similar (Yan *et al.* 2013). My study results differ from Yang *et al.* (2010) where they developed species-specific equations for some species but their data set also included big size trees (DBH > 5 cm), thus not best applicable to shrub species.

D followed by H appeared as a more relevant variable in the case of species-specific single-variable based models. Additionally, several researchers found that D and H are the best predictors for AGB estimation and thus used in generalized equations developed for AGB estimation of big trees in different ecosystems (Brown 1997; Brown *et al.* 1989; Zianis & Mencuccini 2004; Chave et al. 2005; Pilli *et al.* 2006). The Brown *et al.* (1989) allometric equations require only DBH while the Chave *et al.* (2005) equations require species-specific information on SWD and provide a set of equations for each climatic zone that requires either DBH alone or both DBH and H to predict total AGB. Thus, equations developed in this study are recommended to estimate AGB in shrub and small trees in subtropical EBLFs. Unlike big trees, the measurement of shrub D and H is very simple, and can be made using a simple tape.

Different multispecies equations developed in this study for the estimation of AGB showed almost similar predictive capacity with no relevant difference in the number of variables included. Based on the AIC, the best predictive model in the present study was the one that included D, H, and SWD (model 2b in Table 2.3). Even when SWD improved the fit of the models, differences in their explanatory power were not sufficiently big to justify measuring SWD, since it involves the additional time-consuming activities of extracting the samples, processing, and measuring them in the laboratory. Accordingly, I recommend model 2a (Table 2.3) because of its good balance between number of input variables needed and predictive power (e.g., Chave *et al.* 2005). However, and for purpose of quantifying the relationship between biodiversity and C stocks, model 1a (Table 2.3) including just D as a predictive variable offer an accurate, practical, fast, and non-circulatory way to estimate shrub AGB on a large scale in subtropical EBLFs (e.g., Brown *et al.* 1989).

All models included in Table 2.3 should perform well in predicting shrub AGB in subtropical EBLFs because D and H variables are good predictors for AGB in big trees (Brown *et al.* 1989; Chave *et al.* 2005). The decision about the best predictive model to use for the biomass estimation is thus to be made on the basis of the available variables. Although species-specific equations often provide more accurate estimates of biomass than multispecies AGB allometric equations, as was found in present study, multispecies allometric equations could provide useful estimations in similar EBLFs ecosystems where species-specific equations are not yet available and the proposed variables could be correctly measured. Furthermore, it is preferable to use site-specific allometric equations whenever possible, since species size-biomass relationship could differ as plants vary allocation patterns in response to soils, environment and disturbance (Northup *et al.* 2005). The models provided here should contribute to the addition of shrubs and small trees species in subtropical EBLFs biomass and C stocks estimation.

3. Stand structural diversity increased aboveground carbon stocks in subtropical forests in eastern China

3.1. Introduction

The role biodiversity plays in shaping ecosystem function is a central topic in ecology (Loreau *et al.* 2001). The relationships between species diversity and ecosystem productivity and thus carbon (C) stocks have been received particular emphasis in recent decades (e.g., Mittelbach *et al.* 2001; Vilà *et al.* 2003; Firn, Erskine & Lamb 2007;Liang *et al.* 2007; Lei *et al.* 2009), because of their implications for protecting biodiversity, maintaining ecosystem productivity and mitigating the effects of global warming. However, the associations between species diversity and ecosystem productivity have been reported to be either positive (Erskine, Lamb & Bristow 2006), negative (Firn, Erskine & Lamb 2007), or non-significant (Vilà *et al.* 2003) in forested ecosystems. Also, different relationships have been observed between tree species diversity and C stocks in various terrestrial ecosystems. For example, Szwagrzyk & Gazda (2007) reported that negative relationship exists between aboveground biomass (AGB) and tree species diversity in natural forests of Central Europe. In contrast, there has been positive relationship between species diversity and AGB in both tropical forests of Vietnam (Con *et al.* 2013) and spruce-dominated forests in Canada (Wang *et al.* 2011). Also, relationships between tree species diversity and stocks of soil organic carbon (SOC) have been found to be positive in an old-growth forest in northeast China (Chen 2006), but non-significant in a subalpine coniferous forest in southwest China (Zhang *et al.* 2011).

In addition to species diversity, forest C stocks are also influenced by many other factors, such as tree size and stand characteristics (Buongiorno *et al.* 1994; Lindenmayer & Franklin 1997; Sullivan, Sullivan & Lindgren 2001; Franklin *et al.* 2002; Varga *et al.* 2005). It has been reported that stand structural diversity affects forest productivity (Edgar & Burk 2001; Liang *et al.* 2007; Lei *et al.* 2009) which may thus influence ecosystems C stocks (Gower, McMurttrie & Murty 1996). Also, site productivity (P), stand density (N) and age (A) affect stand growth and stand structural diversity (Liang *et al.* 2007; Lei *et al.* 2009). Hence, only considering species composition effects is not enough in explaining forest C stocks, because tree height and diameter are crucial for shaping stand structural biodiversity that has been recognized to substantially affect forest C stocks (Noss 1990; Lexerød & Eid 2006). Thus manipulating tree size diversity is a useful tool for maintaining greater biodiversity and ecosystem productivity (Varga *et al.* 2005).

Linking species diversity and stand structure with forest C stock is important for developing techniques to maintain forest biodiversity and ecosystem productivity sustainably with respect to both theoretical advancements and managerial interventions. Stand structural diversity is an indicator for overall biodiversity (Staudhammer & LeMay 2001) and is commonly described as species diversity, tree size diversity, height diversity or a combination of the above (Staudhammer & LeMay 2001). Therefore, stand structural diversity may not be adequately explained by species diversity alone. Even though the bulk of evidence suggests that forest C stocks such as AGB and SOC are ecologically connected to species diversity (Chen 2006; Szwagrzyk & Gazda 2007), their relationship with other stand structural diversity indices has been poorly studied. Currently, how species diversity, size diversity and height diversity combined and alone affect aboveground C stocks (ACS) and SOC, SOC stock in particular, in forest ecosystems is poorly understood.

It is well understood that stand structural diversity is shaped by recruitment of different sized trees in multistory canopies, because every species in different sizes and heights has its set of requirements for habitats such as nutrients and light (Liang *et al.* 2007; Lei *et al.* 2009). This feature, being approximate to tree size and height diversity, affects the quantity of light and precipitation received by secondary trees or understory vegetation (Anderson, Loucks & Swain 1969), thus affecting C stocks in forest ecosystems. It is commonly assumed that plant community containing species with a diverse array of ecological attribute are more efficient to access and utilize limiting resources (Tilman 1999). Complementarity effects, as niche complementarity theory stated, would occur when inter-specific niche differentiation leads to more efficient utilization of limiting resources, consequently high ecosystem productivity (Liang *et al.* 2007). It is also known that forests A, P and N influence stand growth (Fridley 2002; Pretzsch 2005; Firn, Erskine & Lamb 2007; Liang *et al.* 2007; Larson *et al.* 2008). This may hence affects relationships between stand structural diversity and C stocks. In this context, I hypothesize that, when all other factors (A, P and N) being equal, stand structural diversity has a positive effect on C stocks in evergreen broad-leaved forests (EBLFs) widely distributed in subtropical China (Hypothesis 1: H_1). If this is true, I predict that either species diversity (Prediction 1: P_1^1), size diversity (Prediction 2: P_1^2), or height diversity (Prediction 3: P_1^3) alone and jointly would affect ACS and SOC. If all of the above predictions are not true, my null hypothesis is that stand structural diversity has no effect on C stocks in the studied forest ecosystems (H_0).

I tested the above hypotheses in subtropical EBLFs of eastern China using a likelihood ratio (*LR*) test (Huelsenbeck & Crandall 1997). Specifically, I was interested in 1) exploring the stand diversity-C stocks relationship in subtropical EBLFs, expecting an positive effect of stand structural diversity on C stocks in different forest types, when other factors (particularly A, P and N) are controlled; 2) distinguishing the effect of stand structural diversities from other variables (A, P and N) on C stocks through the performance of each forest community; and 3) understanding how performance of stand structural diversity would affect C stocks in each forest community, when considering tree species, size and height diversities together.

3.2. Materials and methods

3.2.1. Study site, forest plots and measurements of forest structure

The study was conducted in the lower eastern extension of the Tiantai Mountain (29º41-50´N, 121º36-52´E) located in Ningbo city, Zhejiang province in eastern China (Fig. 3.1).

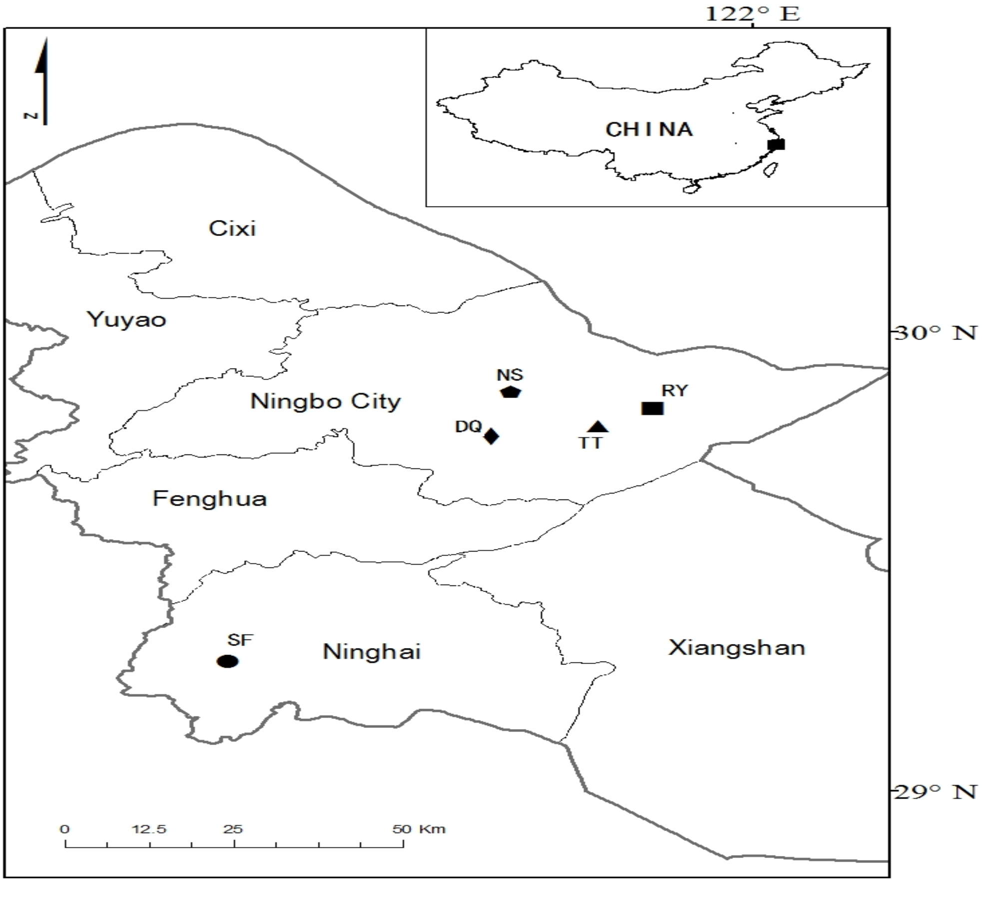

Figure 3.1 A map showing the distribution of the studied subtropical evergreen broad-leaved forests sites, located near Ningbo City, Zhejiang Province, in eastern China. Different symbols in the map represent different study sites.

Five sites were selected in this study including Tiantong Forest Park (TT), Ruiyan Forest Park (RY), Dongqian Lake Landscape Area (DQ), Shuangfeng Mountain (SF) and Nanshan Mountain (NS). The region has a typical subtropical monsoon climate with a hot humid summer and a dry cold winter. The highest peak in this area is at 800 m above sea level, while most other reliefs are in the 70-500 m range (Song & Wang 1995). These five sites had been subjected to different intensities of human disturbances (usually logging), but have been protected from logging and clear cutting from 25 yrs ago. Consequently, the integral plant community structure was different among sites. Thus, the vegetation in five sites represents different forest types, with all sites being mature (MF) except to DQ and RY, pre-mature (PF) except to DQ and young forests (YF) (Yan *et al.* 2009). In addition, among five sites, TT had the largest area of EBLFs with a relatively wider range of topographic features, including mesophytic, peak and ravine areas.

Forest plots within each of the five sites were systematically established 100-500 m apart, located on the same slope position and having similar vegetation history and soil type and soil texture (i.e., mainly loam textures). Overall 80 plots were established which covering all typical habitats in this region. Each plot (20 × 20 m) was located at least 100 m away from the stand edge. Because the 20 m × 20 m plot is the smallest reasonable area to be considered as a community for subtropical forests (Song & Wang 1995), I used the plot to represent a forest community with a set of species co-existing and interacting in a locality. A description of the vegetation and soil characteristics of each studied plot was provided in Appendix I.

In each studied plot, basal diameter and diameter at breast height (DBH) were measured for trees taller than 1.50 m, while basal diameter and 45cm diameter (D_{45}) were measured for trees with height less than 1.50 m. The total tree height was measured with a telescopic pole for heights up to 15 m, and with a clinometers for heights >15 m. Between 6 and 46 tree species inhibited each studied plot, among them *Schima superba* and *Lithocarpus glaber* were the dominant species in YF, *Choerospondias axillaris*, *Liquidambar formosana* and *S. superba* were dominant species in PF while *Castanopsis fargesii* and *Liquidambar formosana* were dominant species in MF sites.

3.2.3. Calculation of site productivity

Forest site productivity usually assessed through phytocentric (vegetation-based) method (e.g. Hägglund 1981). The phytocentric indicators usually relate to the forest stand, the trees comprising the stand or components of the individual trees, and can be classified as dendrometric or dentrocentric (Skovsgaard & Vanclay 2008). Thus, the dendrocentric or dendrometric method usually uses one or a combination of several easily measured tree or stand variables, depending on how closely the indicator is related to production of wood volume, to indicate site productivity in practice. Here, I assessed site productivity for each studied plot through direct volume measurement on the basis of dendrometric method, also suggested by Skovsgaard & Vanclay (2008) as best method for forest site productivity in forestry. Stand age data were from Yan *et al.* (2009). Thus, site productivity was calculated as the mean annual increment by stand volume (Lei *et al.* 2009), where a relationship $\mathrm{P} = \frac{\mathrm{V}}{\mathrm{A}}$ was determined. Where, P is site productivity (m^3 ha^{-1} $year^{-1}$), V is stand volume (m^3 ha^{-1}) and A is stand age (year).

3.2.4. Calculations of aboveground carbon stocks

AGB of individual trees (AGBt, DBH > 5cm) was calculated using an allometric equation (3.1) (Brown, Gillespie & Lugo 1989).

$$\mathrm{AGB}t = exp\{-2.134 + 2.530 \times Ln(\mathrm{D})\} \qquad \text{eqn (3.1)}$$

where, D is diameter at breast height (DBH > 5 cm).

AGB of small trees (DBH < 5 cm) was calculated using an allometric equation (3.2). More details on its development and its advantages are presented in Tables 2.1, 2.2 and 2.3, as well as Figures 2.1 and 2.2.

$$\mathrm{AGBs} = 1.423 \times \exp\{\text{-}3.50 + 21.65 \times \mathrm{Ln(D)} + 0.842 \times \mathrm{Ln(H)}\} \qquad \text{eqn (3.2)}$$

where D = (DBH < 5cm or D45 cm).

The sum of the AGB for trees and shrubs was considered as aboveground standing biomass (AGB). Then, I converted AGB to ACS (Mg ha^{-1}) by multiplying it with a factor of 0.5, assuming that 50 % of the total tree biomass is equal to elemental C (Dixon *et al.* 1994; Brown 1997).

3.2.5. Measurements of soil organic C stocks

Soil samples were collected from 0–20 cm depth from 65 sample plots by considering all five sites. To reduce variability, soil samples in each plot were collected from five randomly selected points, resulting in 325 samples. The samples were shipped to the laboratory and air-dried over 30 days. Then each soil sample was sieved through a 2 mm sieve and thoroughly mixed to determine SOC concentration using the oil bath-K_2CrO_7 titration method (Nelson & Sommers 1996). In each plot, soil bulk density was determined using a steel corer of known volume. Five soil cores were collected per plot. The soil cores were dried in an 105 °C oven for > 48 hours. Bulk density (g cm^{-3}) was calculated by dividing the oven dry weight (g) by the volume of the soil core. The amount of SOC (Mg ha^{-1}) was calculated by multiplying organic C content (g kg^{-1}) by soil depth (cm) and soil bulk density (g cm^{-3}) (Brown 2004).

3.2.6. Calculation of stand structural diversity

I chose Shannon-Wiener's index (Shannon 1948) to describe stand structural diversity, because it reflects both species evenness and richness (Magurran 1988). With Shannon–Wiener index, DBH and height had to be grouped into discrete classes. For DBH, 2, 4, 6, and 8 cm classes were tested; while for height, 2, 3, 4, and 5 m classes were tested to calculate the indices. It was discovered that tree size diversity by 8 cm increment for the DBH classes and height diversity by 3 m increment for the height classes showed the highest correlation coefficient ($r = 0.54$, $P < 0.001$) as compared to other class widths based on the same index. Therefore, 8 cm and 3 m were used for the DBH and height classes, respectively. Based on basal area, tree species diversity (H_s), size diversity (H_d) and height diversity (H_h) was calculated with each of equations 3.3, 3.4 and 3.5 (Magurran 2004; Buongiorno *et al.* 1994; Staudhammer & LeMay 2001) for each plot.

$$\mathrm{H_s} = -\sum_{i=1}^{s} \mathrm{p}_i \times Log\ \mathrm{p}_i \qquad \text{eqn (3.3)}$$

$$\mathrm{H_d} = -\sum_{j=1}^{d} \mathrm{p}_j \times Log\ \mathrm{p}_j \qquad \text{eqn (3.4)}$$

$$\mathrm{H_h} = -\sum_{k=1}^{h} \mathrm{p}_k \times Log\ \mathrm{p}_k \qquad \text{eqn (3.5)}$$

where p_i, p_j, and p_k was the proportion of basal area for each of *i*th species, *j*th diameter class and *k*th height class respectively, while *s*, *d* and *h* were the number of tree species, diameter classes and height classes.

The Shannon-Weiner indices analysis was performed using the diversity-vegan (Oksanen 2013) for the R statistical language (R Development Core Team 2013).

3.2.7. Statistical analysis

It is well known that A, P and N influence stand growth (Fridley 2002; Pretzsch 2005; Firn, Erskine & Lamb 2007; Liang *et al.* 2007; Larson *et al.* 2008), and thus affect relationships between stand structural diversity and C stocks. Therefore, these three parameters were chosen as the variables that reflect the effects of site factors and initial stand conditions; those variables were controlled so that their effects on the relationships between stand structural diversity and C stocks were constant in all models. Summary of variables and their categories are described in Table 3.1.

Table 3.1 Summary of plot variables used for the quantification of stand structural diversity and C stocks (aboveground and soil) in subtropical evergreen broadleaved forests in eastern China. *SD* = standard deviation. Number of sample plots used for ACS was 80 while for SOC were 65.

Variable	Unit	Aboveground carbon stock		Soil organic carbon (0-20 cm)	
		Mean	*SD*	Mean	*SD*
Dependent variable					
Carbon stocks	Mg ha^{-1}	80.53	53.67	84.44	25.69
Stand structural diversity variable					
Tree species diversity index (H_s)	unitless	1.80	0.54	1.72	0.52
Tree size diversity index (H_d)	unitless	1.31	0.38	1.23	0.37
Tree height diversity index (H_h)	unitless	1.49	0.36	1.48	0.38
Control variable					
Age (A)	year	72.60	40.38	70.82	40.68
Site productivity (P)	m^3 ha^{-1} year^{-1}	3.92	2.08	3.34	1.70
Stand density (N)	stems ha^{-1}	5144.03	2636.61	4791.80	2678.07

Firstly, I used a simple linear regression analysis to test for pair wise associations of C stocks (ACS and/or SOC) with stand structural diversity indices and control variables. Secondly, a generalized linear model using the Wald-Chi square test was applied to estimate the parameters for the quantification of effects of stand structural diversity on C stocks. My most sweeping null hypothesis was that all stand structural diversity had no effect on C stocks (ACS and/or SOC). In this case, the *LR* test (Huelsenbeck & Crandall 1997) was used to compare the log-likelihood (L) value in a full model (model 2), including all variables, with a restricted model (model 1) of null hypothesis, without any term of stand structural diversity indices, as listed in Table 3.2. If null hypothesis is rejected (if $P \leq 0.05$) and alternative hypothesis is accepted that all stand structural diversity affected C stocks, I alternatively predict that each diversity index alone has an effect on C stocks (ACS and/or SOC). In this case, the *LR* test was used again to compare the L-value in the full model (2) with restricted models (3, 5 and 7) of null predictions, models without any term of tree species diversity or size diversity or height diversity, respectively, as listed in Table 3.2.

Table 3.2 Statistical hypotheses, predictions and the number of model and its forms for the quantification of effects of stand structural diversity on ACS and SOC in subtropical forests of eastern China. The statistical models represent generalized linear model using Wald-Chi square test by keeping $P < 0.05$ for significance. In all models, age (A), site productivity (P) and stand density (N) were kept as control variables.

Hypothesis	Model number and statistical form	Prediction
H_0	**1:** $g_i=a_0+a_a\mathrm{A}+a_p\mathrm{P}+a_n\mathrm{N}$	All stand structural diversity had no effect on C stocks $H_{0:}\ \alpha_s = \alpha_d = \alpha_h = 0$
H_1	**2:** $g_i=a_0+a_s\mathrm{H_s}+a_d\mathrm{H_d}+a_h\mathrm{H_h}+a_a\mathrm{A}+a_p\mathrm{P}+a_n\mathrm{N}$	All stand structural diversity had effect on C stocks $H_{1:}\ \alpha_s \neq \alpha_d \neq \alpha_h \neq 0$
P_0^1	**3:** $g_i=a_0+a_d\mathrm{H_d}+a_h\mathrm{H_h}+a_a\mathrm{A}+a_p\mathrm{P}+a_n\mathrm{N}$	Tree species diversity ($\mathrm{H_s}$) had no effect $P_0^1: \alpha_d \neq \alpha_h \neq 0,\ \ \alpha_s = 0$
P_1^1	**4:** $g_i=a_0+a_s\mathrm{H_s}+a_a\mathrm{A}+a_p\mathrm{P}+a_n\mathrm{N}$	Tree species diversity ($\mathrm{H_s}$) alone had an effect $P_1^1: \alpha_d = \alpha_h = 0,\ \ \alpha_s \neq 0$
P_0^2	**5:** $g_i=a_0+a_s\mathrm{H_s}+a_h\mathrm{H_h}+a_a\mathrm{A}+a_p\mathrm{P}+a_n\mathrm{N}$	Tree size diversity ($\mathrm{H_d}$) had no effect $P_0^2: \alpha_s \neq \alpha_h \neq 0,\ \ \alpha_d = 0$
P_1^2	**6:** $g_i=a_0+a_d\mathrm{H_d}+a_a\mathrm{A}+a_p\mathrm{P}+a_n\mathrm{N}$	Tree size diversity ($\mathrm{H_d}$) alone had an effect $P_1^2: \alpha_s = \alpha_h = 0,\ \ \alpha_d \neq 0$
P_0^3	**7:** $g_i=a_0+a_s\mathrm{H_s}+a_d\mathrm{H_d}+a_a\mathrm{A}+a_p\mathrm{P}+a_n\mathrm{N}$	Tree height diversity ($\mathrm{H_h}$) had no effect $P_0^3: \alpha_s \neq \alpha_d \neq 0,\ \ \alpha_h = 0$
P_1^3	**8:** $g_i=a_0+a_h\mathrm{H_h}+a_a\mathrm{A}+a_p\mathrm{P}+a_n\mathrm{N}$	Tree height diversity ($\mathrm{H_h}$) alone had an effect $P_1^3: \alpha_s = \alpha_d = 0,\ \ \alpha_h \neq 0$

H_0 and H_1 are null and alternative hypothesis; P_0^1, P_0^2 and P_0^3 are null while P_1^1, P_1^2 and P_1^3 are alternative predictions 1, 2 and 3 respectively; *gi* is the carbon stocks (ACS or SOC); α_s is coefficient estimated from the plot data through generalized linear model; $\mathrm{H_s}$, $\mathrm{H_d}$, $\mathrm{H_h}$ are the tree species, size and height diversity respectively; A is stand age (years), N is stand density (stems $\mathrm{ha^{-1}}$) and P is site productivity ($\mathrm{m^3\ ha^{-1}\ year^{-1}}$).

In the case of that the alternative hypothesis could not be rejected ($P \leq 0.05$), a multiple backward elimination regression was used by including all variables together in order to prioritize and assess the relative importance of the stand structural diversities on C stocks. The minimal adequate significant model was selected according to significant parameters ($P < 0.05$) of stand diversities. In this process, the insignificant parameters with the highest *p*-value were removed and the model was re-established until the Akaike information criterion (*AIC*) (Akaike 1973) and Bayesian information criterion (*BIC*) (Schwarz 1978) could not be lowered, and the coefficients of all parameters were significant ($p < 0.05$) (MacNally 2000). The final minimal adequate models were expected to be consistent with the results of *LR* test.

Lastly, if all alternative predictions were accepted (if $P \leq 0.05$), I ran a multivariate principal component analysis (PCA) for the site conditions based on their diversity indices and control variables, in order to examine the percentage of variation in C stocks that can be explained by either diversity indices (individually or combined) or control variables. After PCA analysis, the association between PCA scores and C stocks in each of above- and below ground was tested by using a simple linear regression. The C stocking performance of each forest plot or community was assessed against the effects of stand structural diversities and/or control variables, through their scores along PCA axes. The complementarity performance of stand structural diversities (PC axis) on C stocks in each forest community was also assessed.

3.3. Results

3.3.1. Relationships between stand variables and carbon stocks

Tree size and height diversity (and thus stand structure diversity) increased significantly, but species diversity did not change statistically through forest development. Additionally, ACS and SOC increased through forest development but only ACS was significantly different among forest types (Table 3.3). Overall, higher ACS and SOC were found in mature stands compared with young stands (Table 3.3), mature stands were more likely to have a greater difference in tree size distribution because they had a greater number of big trees overall (Figs 3.2a and 3.2b). For other stand variable, maximum stand height, maximum stand DBH, species richness, forest age increased significantly, but stand productivity and stand density were invariant through forest development (Table 3.3). The pattern of expansive larger range of tree DBH in conjunction with forest development was accompanied by decreasing stand density and species evenness. Tree species diversity and site productivity did not follow this pattern. Species diversity decreased from young to pre-mature stage and then increased in the mature stage during forest development (Table 3.3). Although younger stands possessed smaller size trees overall, it is assumed that they undergo higher growth rates and possess greater stand density. Older stands, on the other hand, possessed higher structural complexity both horizontally and vertically (Figs 3.2a and 3.2b).

Table 3.3 Characterization of three forest developmental stages considered for the study on the linkage between stand structural diversity and carbon stocks (vegetation and soil) in subtropical evergreen broad leaved forests in eastern China. Values are mean ± *SD* for each developmental stage. Values with different letters in a given row are significantly different at $P < 0.05$ (*LSD* Fisher). The number of plots used (n) for young forests, pre-mature forests and mature forests was 21, 39, and 20, respectively, for the vegetation data, and 21, 25, and 19 for the soils data.

	Young forest (YF)	Pre-mature forest (PF)	Mature forest (MF)
Vegetation structure			
Maximum tree height (m)	14.23±7.45a	21.76±5.07b	24.12±4.98b
Maximum tree DBH (cm)	19.15±7.32a	38.64±11.51b	47.74±11.18c
Species richness	21.48±9.16a	26.13±9.59ab	29.05±8.19b
Species evenness	0.63±0.13a	0.56±0.17a	0.54±0.16a
Tree biomass (Mg ha^{-1})	48.24±31.30a	172.72±76.45b	256.81±105.75c
ACS (Mg ha^{-1})	24.12±15.65a	86.36±38.23b	128.40±52.87c
Tree species diversity index	1.88±0.56a	1.63±0.50a	1.77±0.46a
Tree size diversity index	0.93±0.31a	1.36±0.32b	1.54±0.26b
Tree height diversity index	1.20±0.48a	1.51±0.21b	1.70±0.22b
Age (years)	22.29±4.71a	79.23±5.99b	124.50±6.86c
Site productivity (m^3 ha^{-1} year^{-1})	3.28±2.10a	4.39±2.18a	3.68±1.72a
Stand density (stems ha^{-1})	6068.36±3370.96a	4970.38±2456.59a	4512.08±1856.54a
Soil property			
Bulk density (g cm^{-3})	1.18±0.22a	1.09±0.22ab	1.00±0.17b
SOC (Mg ha^{-1})	80.84±26.71a	85.31±28.81a	87.26±20.57a

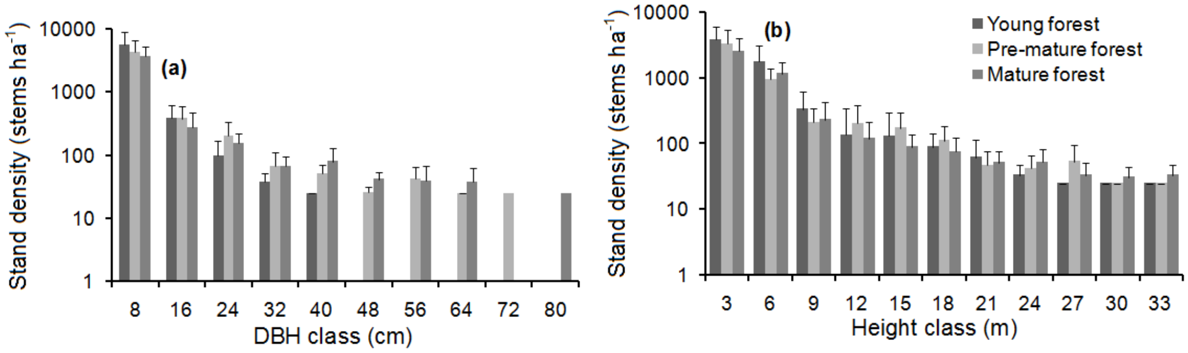

Figure 3.2 (a) Diameter and (b) height distributions in subtropical evergreen broad-leaved forests in different forest developmental stages. The diameter class interval is 8 cm while height class interval is 3 m. The vertical bars are mean + *SD*. Log$_{10}$ scale was used for Y-axis of each graph.

Stand variables showed different relationships with ACS and SOC (Fig. 3.3, Appendix II). ACS significant positively correlated with tree size diversity (Fig. 3.3b), tree height diversity (Fig. 3.3c), A (Fig. 3.3d) and P (Fig. 3.3e). There was no significant relationship between ACS and each of species diversity (Fig. 3.3a) and stand density (Fig. 3.3f). SOC did not significantly correlate to any stand variables (Figs 3.3g, 3.3h, 3.3i, 3.3j, 3k and 3.3l). It is necessary to note that tree species diversity had non-significant and negative pair wise association with ACS (Fig. 3.3a). The possible reason is that species richness increased while species evenness decreased through stand development in the studied forests (Table 3.3). Further, tree species diversity decreased from young to pre-mature stands while increased in the following stage (Table 3.3).

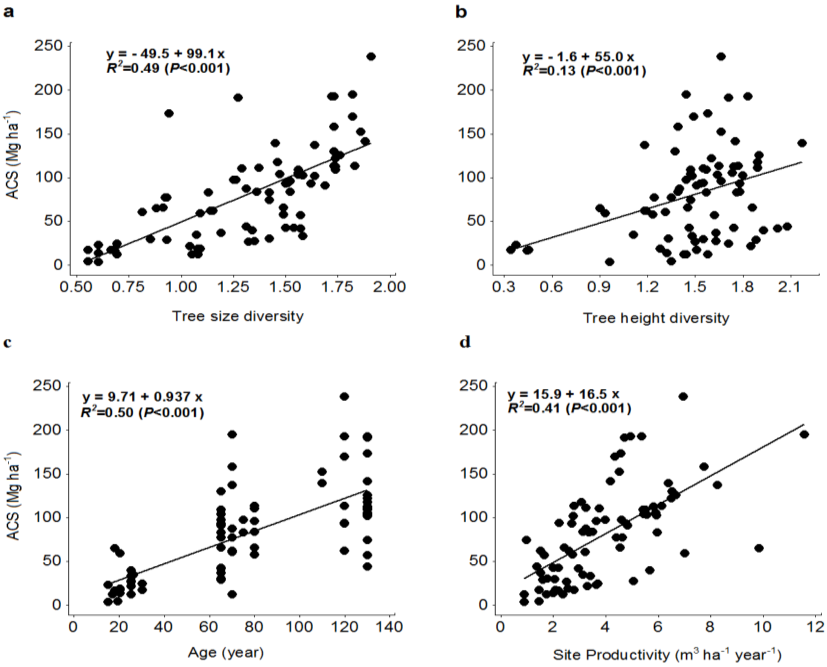

Figure 3.3 Simple linear regression analyses between stand structural diversity, control variables and the size of ACS in subtropical evergreen broadleaved forests in eastern China. Only significant associations (see Appendix II) are shown here.

3.3.2. Effects of stand structural diversity on carbon stocks

This study mainly focused that how these stand structural diversities jointly or individually affects C stocks, when controlling for A P and N. If all other factors (A, P and N) being equal, stand structural diversities showed different effect on ACS and SOC. Stand structural diversities affected ACS and SOC differently. The null hypothesis in this study was rejected with respect to ACS (*LR* test: $P < 0.001$) while accepted with respect to SOC (*LR* test: $P > 0.05$; Table 3.4). Therefore, I tested the alternative hypothesis that all stand structural diversities have combined effects on ACS, when A, P and N are controlled. However, it may be that only tree species, size, or height diversity affects C stocks alone. In case of ACS, three null predictions were also rejected (*LR* test: $P < 0.05$; Table 3.4), indicating that each of the three diversities was responsible for ACS. Moreover, *LR* test and *AIC* criteria demonstrated that the jointly effects of stand structure diversities have more influence on ACS as compared to alone effect of each of three diversities, because full model fits significantly better than restricted models, and the associated *P*-values of *LR* test for restricted models were less than 0.0001 (Table 3.4). For SOC, null predictions were accepted (*LR* test: $P > 0.05$), indicating that each of three stand structure diversities has no effect on soil C stocks (Table 3.4).

The multiple regression analysis showed that species, size and height diversities, as well as control variables of A and P mutually explained 83 % variation in ACS (Table 3.5). In terms of SOC, no predictable variables retained in the final model for three stand structure diversity indices and other control variables (Table 3.5).

Table 3.4 Significance tests of null hypothesis and predictions for the quantification of overall effects of stand structural diversity on C stocks (aboveground and soil) in the subtropical evergreen broad leaved forests of eastern China. The likelihood ratio (*LR)* chi-square test was used for comparing the log-likelihood (*L*) value in full model with each null hypothesis or predictions models. The parameters for each model were estimated through generalized linear model analysis (GLZ).

Hypothesis	*L*	*AIC*	*BIC*	*LR* test	*P*-value*
Aboveground carbon stock					
Full model	-360.83	737.66	756.72		
Null hypothesis	-368.39	746.79	758.70	15.12	**0.001**
Null prediction 1	-365.40	744.79	761.47	9.14	**0.011**
Null prediction 2	-364.27	742.55	759.22	6.88	**0.032**
Null prediction 3	-363.90	741.81	758.48	6.14	**0.046**
Soil organic carbon stock (0-20 cm)					
Full model	-301.78	619.56	636.96		
Null hypothesis	-301.89	613.79	624.66	0.22	0.974
Null prediction 1	-301.86	617.72	632.94	0.16	0.923
Null prediction 2	-301.79	617.57	632.79	0.02	0.990
Null prediction 3	-301.79	617.57	632.79	0.02	0.990

Null hypothesis, all structural diversity had no effect on C stocks: $H_0 = \alpha_s = \alpha_d = \alpha_h = 0$

Null prediction 1, tree species diversity (H_s) had no effect on C stocks: $P_0^1: \alpha_d \neq \alpha_h \neq 0,\ \alpha_s = 0$

Null prediction 2, tree size diversity (H_d) had no effect on C stocks: $P_0^2: \alpha_s \neq \alpha_h \neq 0,\ \alpha_d = 0$

Null prediction 3, tree height diversity (H_h) had no effect on C stocks: $P_0^3: \alpha_s \neq \alpha_d \neq 0,\ \alpha_h = 0$

L: log-likelihood for restricted models; *LR* test: log-likelihood ratio chi-square test

AIC: Akaike Information Criterion for restricted models (smaller is better).

BIC: Bayesian Information Criterion for restricted models (smaller is better).

*Reject H_0 if $P \leq 0.05$ while accept H_0 if $P > 0.05$

Table 3.5 Multiple regressions of ACS and SOC with tree species diversity (H_s), size diversity (H_d), height diversity (H_h), site productivity (P), stand age (A) and density (N) as predictable variables. Explanatory variables were selected using a backward elimination procedure for the selection purpose of significant model with significant parameters ($P < 0.05$). Values are regression coefficients (R^2) from multiple regression analysis, indicated with P-values for each model and each predictor variable within the model.

Model form	Predictor variables	P	R^2
Aboveground carbon stocks			
$ACS = -5.16 - 17.42 H_s + 33.67 H_d - 19.93 H_h + 0.77 A + 11.34 P$	*Model*	<0.001	0.83
	Constant	0.736	
	H_s	<0.001	
	H_d	0.003	
	H_h	0.037	
	P	<0.001	
	A	<0.001	
SOC (0-20 cm)			
None of the parameters were retained in the final model			

3.3.3. Performance of community structure in carbon stocking

The multivariate analysis (PCA) exhibited that a combination of stand structural diversity indices explained a greater proportion of the variation in ACS than the control variables (A, P and N) (Table 3.7). The first multivariate axis (PC 1) was mostly defined by stand structural diversity indices while the second multivariate axis (PC 2) was mostly defined by control variables. The ACS were significantly positively associated with scores of the sites along both PCs while, in contrast, no relationship was found between scores of both PCs and SOC (Table 3.6). This result suggesting that stand structural diversity had a greater effect on ACS than the control variables.

Table 3.6 Principal components analysis (PCA) of 80 forest plots for ACS and 65 forest plots for SOC, on the basis of their diversities and additional control variables. Values in parentheses indicate the variance (%) accounted for by each axis. Values in the top section of the table indicate the eigenvector scores of each of the variables on the two PCA axes. The bottom section of the table shows the regression coefficients (R^2) obtained from simple linear regression analyses between the C stocks of the forest sites and their scores on PC 1 and PC 2, indicated with asterisks if statistically significant. *: $P < 0.05$; **: $P < 0.01$; ***: $P < 0.001$; ns: not significant).

	Aboveground carbon stocks		Soil organic carbon stocks	
	PC 1 (39%)	PC 2 (23%)	PC 1 (41%)	PC 2 (24%)
Eigenvector scores of variables				
Tree species diversity index(H_s)	0.08	0.30	-0.08	-0.59
Tree size diversity index (H_d)	0.58	0.15	0.54	0.04
Tree height diversity index (H_h)	0.52	-0.27	0.53	-0.0039
Age (A)	0.53	0.002	0.51	-0.01
Site productivity (P)	0.26	0.55	0.12	0.70
Stand density (N)	-0.18	0.71	-0.38	0.40
Regression coefficient (R^2)	0.55***	0.08**	0.003ns	0.01ns

Finally, the C stocking performance of each forest community was examined to test that whether stand structural diversities jointly (*PC* 1 axis) or control variables (*PC* 2 axis) enhancing the ACS in the ecosystem. Thus, forest communities having higher values than -1 on the *PC* 1 axis also have higher levels of ACS than the *PC* 2 while communities with low values on both *PC* axes have a low average ACS values (Fig. 3.4a). Furthermore, Figure 3.4b shows the complementarity performance of stand structural diversities (*PC* 1 axis) on ACS in each forest community, with communities having higher values than -1 on the *PC* 1 axis also have higher levels of ACS and vice versa.

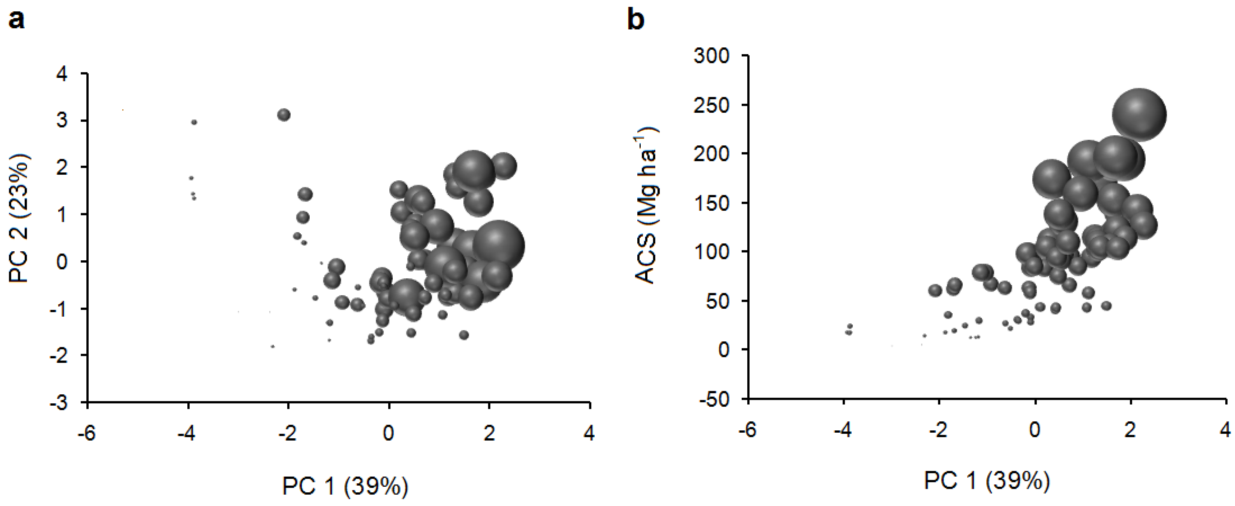

Figure 3.4 Relationships between community structure and ACS in subtropical evergreen broadleaved forests in eastern China. The performance analyses were done through the use of bubble graphs, and thus the circle size in the graph is proportional to the size of the C stock in each forest community. **(a)** C stock in each forest community vs. stand structural diversity

(PC 1) and control variables (PC 2); **(b)** Performance of stand structural diversity on ACS in each forest community. The circle size is proportional to the size of the ACS. See Table 3.6 for associated statistics.

3.4. Discussion

Estimating C stock is an important trend and should be more studied especially when tree species composition might change in mixed forest stand (Kellomäki, Kolström 1992). Increasing trend in size and height diversity during forest developmental stages were also observed in other studies (Spies 1998; Harper *et al.,* 2003; O'hara *et al.* 2007; Lei *et al.* 2009) while Lei *et al.* (2009) also found irregular trend in species diversity during forest development. Many other studies have found a similar increasing trend for both vegetation and soil C stocks that addresses age dependence of forest biomass or C stocks (Peichl & Arain 2006). Hence, forest developmental stages have a fundamental role in the C pools distribution for different forest ecosystems (Pregitzer & Euskirchen 2004).

I found that stand structural diversity indices statistically correlated with ACS, not yet with SOC in subtropical EBLFs in eastern China. This suggests that the relationship between stand structural diversity and C stocks is ecosystem compartment-specific. C stock variability among forests is more predictable by aboveground stand structural diversity than belowground. It is understandable that stand structural diversity derives mainly from characteristics of aboveground forest structure, such as vertical and horizontal changes of tree size, in steady of characteristics of belowground forest structure. Therefore, stand structural diversities jointly or individually has no effect on SOC, even controlling for A, P and N in this study. In this context, soil C stock may relate to stand structural diversity belowground. It can be evidenced by the observation that the effects of stand types on SOC stocks change significantly with soil depth (Laganière *et al.* 2013).

The significant effect of stand structural diversity on C stocks can be detected by testing whether those variables are statistically included in the specific generalized linear models. In this study, each of the three diversities was jointly and alone significant for explaining ACS, but insignificant for interpreting SOC. This result support my null hypothesis that stand structural diversity has no effect on C stocks belowground, but did not support my null hypothesis that stand structural diversity has no effect on C stocks aboveground (H_0). Rather, partly consistent with my alternative hypothesis that stand structural diversity affects ACS in EBLFs (Hypothesis 1: H_1). Therefore, predictions that either species diversity (Prediction 1:

P_1^1), size diversity (Prediction 2: P_1^2), or height diversity (Prediction 3: P_1^3) alone and jointly affect ACS are confirmed, on the conditional of A, P and N. More importantly, my results showed that effects of each stand structural diversity and control variables on ACS were lower than the jointly effects of stand structure diversities (Tables 3.3 and 3.4). Also, the effects of stand structural diversity on ACS were greater or equal than the effects of control variables (Fig. 3.3). This means that tree size-shaped stand structure is the key factor affecting tree C stocks (Vayreda *et al.* 2012). Recently study demonstrated that, at the global scale, AGB growth rate (thus C stocks accumulation rate) for the most of tropical and temperate tree species increases continuously with tree size (Stephenson *et al.* 2014). The large size trees do not act simply as senescent C stocks but actively accumulate large amounts of C compared to the smaller trees (Stephenson *et al.* 2014). Moreover, a single big tree can add the same amount of C to the forest within a year as compared to the entire mid-sized trees contained in the same forest (Stephenson *et al.* 2014). Because the big sizes trees in a given forest possess a higher structural diversity than those small or mid-sizes trees, biomechanically, tree basal area and height based structural diversity should thus associate with ACS positively (Figs 3.2a and 3.2b).

In this study, the most pre-mature and mature forests had been subjected to few logging, being protected from logging and clear cutting in the past several decades (Yan *et al.* 2009). In such condition, these unmanaged forests possess a more diverse tree structure that induces vegetation and soils to act as long-term C sinks. This is consistent with the observation in central Europe, where the unmanaged forests yield higher amount of ACS compared with the managed forests (Merino *et al.* 2007). In these relatively intact forests, tree size diversity may allometrically link with canopy depth and leaf area index. Thus a high efficiency of light utilization is expected in such type of stands. As a result of a complex tree size structure supporting intra-specific complementary effects, trees with high light use efficiency might cause a positive relationship between ACS and stand structural diversity (Lei *et al.* 2009).

The clear trends of ACS with changes of stand structural diversity might be discussed in the light of niche complementarity hypothesis. In forest ecosystems, complementarity effects occur when inter-specific niche differences lead to more efficient acquisition of limiting resources (Tilman 1999), consequently making the system more productive (Liang *et al.* 2007). It is well understood that stand structural diversity is shaped by different sized trees in multistory canopies (Liang *et al.* 2007; Lei *et al.* 2009). Maintaining high stand structural diversity supports tree species to meet their specific requirements; whereas sustaining low structural diversity or homogenous structural arrangements reduces complementarity effects

(Lei *et al.* 2009). In this study, it was examined that complementarity performance of stand structural diversities enhances the ACS in forest communities (Figs 3.4a and 3.4b), suggesting that ecosystem multifunctionality improves with increasing stand structural diversity values (Mouillot *et al.* 2011). This may attribute to the fact that tree size and height complexity (Figs 3.2a and 3.2b) may in itself enhance forest growth by promoting complementary resource utilization among tree species (Ishii *et al.* 2004). In this study, I found that greater variation in stem height and species richness in each height class creates several canopy strata (Table 3.7), which may allow light infiltrating more efficiently to optimize vertical growing space usages (Tilman 2001). Thus, my results strongly support the niche complementarity hypothesis, i.e., ACS increases with increasing stand diversities in the studied forests.

Table 3.7 Multi-strata subtropical evergreen broadleaved forests in eastern China. The number of species decreases with the increase in height class because the higher class included only dominant species that grow into the canopy.

Canopy stratum	Height class (m)	Number of species
A	3	138
B	6	119
C	9	76
D	12	52
E	15	48
F	18	30
G	21	25
H	24	14
I	27	12
J	30	9
K	33	7

It is necessary to note that tree species diversity had non-significant and negative pair wise association with ACS (Fig. 3.3a). The possible reason is that species richness increased while species evenness decreased through stand development in the studied forests (Table 3.3). Further, tree species diversity decreased from young to pre-mature stands while increased in the following stage (Table 3.3). Due to this complicated pattern of tree species diversity among forests, it is understandable that tree species diversity has a weak effect on C stocks (Szwagrzyk & Gazda 2007; Wang *et al.* 2011; Con *et al.* 2013).

Stand structural diversity varies across broad range of ecosystem processes (Loreau *et al.* 2001), but its application is not well developed. The link of stand structural diversities with belowground processes remains still open. In this study, I tested the effects of stand structural diversities on SOC with generalized *LR* test, regressions (simple and multiple) and multivariate analysis (PCA). Unfortunately, my potential predictors of SOC were not retained in the final regression model. The results of my study showed that stand structural diversities have no effect on SOC, even controlling for A, P and N. The possible reason may be that SOC is affected more by functional diversity than by stand structural diversities, because tree height and diameter are good functional predictors of total tree biomass (Chave *et al.* 2005). The functional changes of tree basal diameter and height are recognized to directly influence the amount of C stored in both the above and belowground portions of the standing vegetation, thus incorporating into the soil as litter at senescence (De Deyn, Cornelissen & Bardgett 2008; Lavorel & Grigulis 2012). Also, the large size trees are expected to shed more biomass in the form of leaf and woody litter per unit ground area and thus contribute directly to C stocks in the standing litter and in the organic soil (Lavorel & Grigulis 2012). Therefore, functional traits with respect to plant tissue quality, such as leaf nutrient concentration and leaf area displayed per unit C, influence the residence time of both living tissues and litters and eventually influence SOC (Aerts & Chapin 2000; De Deyn, Cornelissen & Bardgett 2008). Therefore, it is necessary for the further research to include functional diversity into the relationship between biodiversity and SOC, by considering different plant functional traits in terms of the quantity and quality of soil C source.

4. Functionally dominant leaf nutrient-use syndromes and divergent wood syndromes of trees jointly enhanced carbon stocks in subtropical forests

4.1. Introduction

Understanding the relationship between biodiversity and forest carbon (C) stocks is important for protecting biodiversity and mitigating the effects of global warming (Dixon *et al*. 1994; Díaz *et al*. 2009; Cavanaugh *et al*. 2014). However, when compared to species diversity, our understanding of how other axes of biodiversity such as functional diversity influence C stocks remains poorly studied. Functional diversity has been proposed as being beneficial for ecosystem functions such as productivity (Tilman *et al.* 1997; Mason et al. 2005), and may therefore important for shaping ecosystem C stocks. Functional diversity has three primary components i.e., functional evenness, functional richness and functional divergence (Mason *et al.* 2005). Together, these independent components of functional diversity reflect three complementary facets of the distribution of species or biomass and their abundances within the functional space (Mouchet *et al.* 2010). It is becoming increasingly accepted that functional diversity strongly affects ecosystem productivity (Díaz *et al.* 2009; De Deyn, Cornelissen & Bardgett 2008; Wardle *et al.* 2012). However, field tests of which component of functional diversity best predicts forest C stocks are relatively scarce.

The relationships between functional diversity and C stocks may depend on which functional characters are measured through niche space, because plant function in different tissues is not uniform but changeable (Baraloto *et al.* 2010). Therefore, it is essential that plant functional characters should be measured on the different tissues such as leaves, branches and stems, with which it can capture the leading dimensions of plant strategies (Westoby *et al.* 2002) and cover the full range of functional spaces (Mouchet *et al.* 2010). Recent evidence has shown that leaf and wood economics (Reich *et al.* 1997; Wright *et al.* 2004; Chave *et al.* 2009) are decoupled (Baraloto *et al.* 2010). Moreover, leaf chemical traits (e.g., leaf nitrogen (N) concentration-LNC and leaf phosphorus (P) concentration-LPC) between mature and current year leaves, wood chemical traits, as well as wood density in each of twigs and stems vary differently at the community level (Yan *et al.* 2013). In this case, it is unclear how the functional diversity-C stocks-relationship changes if decoupled trait axes are considered.

It is well known that leaf longevity significantly affects plant nutrient use strategies (Aerts & Chapin 2000; Westoby *et al.* 2002). Ecosystem C stocks thus may be influenced by plant species differing in the residence time of living tissues (Wardle *et al.* 2012; De Deyn, Cornelissen & Bardgett 2008). On the one hand, the 'exploitative' plants with short leaf longevity but high specific leaf area (SLA) and LNC are typically fast in nutrient acquisition and turnover. It is thus associated with high C sequestration through photosynthesis and high C loss through decomposition. On the other hand, the 'conservative' plants with long leaf longevity and nutrient-poor leaves (e.g., high leaf dry matter content-LDMC and low LNC) are generally slow in growth (Grime *et al.* 1996; Díaz *et al.* 2004; Wright *et al.* 2004; Cornwell *et al.* 2008; Poorter *et al.* 2009). Therefore, it is expected that, at the ecosystem level, prevailing acquisitive syndromes of plants should be conducive to higher C fluxes, whereas prevailing conservative syndromes should be conducive to higher C stocks (Wardle *et al.* 2012; De Deyn, Cornelissen & Bardgett 2008; Díaz, Hector & Wardle 2009). In addition to plant leaf syndromes, the ecosystem production and thus C stocks may also associate to variability of plant height (Butterfield & Suding 2013). In a given ecosystem, plants with high divergence of their sizes can cause a complex stand structure and high stand production, thus yielding large amount of C stocks (Ruiz-Benito *et al.* 2013).

There are two hypotheses for explaining the relationship between functional diversity and C stocks. The mass ratio hypothesis (Grime 1998) postulates that plant functional traits are related to species abundance, thereby the most dominant values of plant traits will be the main drivers of the ecosystem C stocks (Díaz, Hector, & Wardle 2009). More explicitly it implies that the community weighted means (CWM) of a trait can predict ecosystem C stocks (Garnier *et al.* 2004; Díaz *et al.* 2007; Conti & Díaz 2013). Additionally, niche complementarity hypothesis (Petchey & Gaston 2006) proposes that functional traits mediated niche partitioning leads to different resource utilization by the plants across space, such as stratified mixtures of sun- and shade-adapted species in a community (Pretzsch 2005; Lavorel & Grigulis 2012). To be applicable, niche complementarity hypothesis can be inferred from the functional divergence index (FDvar), which basically represents the variance of trait values, and the abundance-weighted variance of traits for each species in a community (Mason *et al.* 2003).

Currently, only few studies have tested the above two hypotheses separately, in terms of linkage between functional diversity and C stocks (Butterfield & Suding 2013; Conti & Díaz 2013). Unfortunately, no one has compared the explanatory power for both hypotheses with respect to the relationships between functional diversity and each of aboveground C stocks

(ACS) and soil organic C (SOC) stocks in forest ecosystems. In this study, I hypothesize that functional dominance (defined here as CWM, based on the mass ratio hypothesis) and functional divergence (defined here as FDvar, based on the complementarity hypothesis) of plant syndromes do jointly enhance ecosystem C stocks in the subtropical forests.

To test the above hypothesis, I measured 14 tissue-specific plant traits and C stocks in vegetation and soils across 31 plots in evergreen broadleaved forests (EBLFs) of eastern China. I focus my efforts on the relationships between functional diversity components and C stocks with emphasis on the explanatory power of CWM and FDvar of trait values on ecosystem C stocks. Specifically, I ask the following questions: 1) whether CWM and FDvar based on different trait axes associate with ecosystem C stocks? and 2) whether there are joint or additional effects of CWM and FDvar on ecosystem C stocks in terms of the mass ratio hypothesis and the complementarity hypothesis?

4.2. Materials and methods

4.2.1 Study site and sampling plots

The study was conducted in the lower eastern extension of the Siming Mountain (29º41-50´N, 121º36-52´E), Zhejiang province in eastern China. The study area has a typical subtropical monsoon climate with a hot humid summer and an arid cold winter. The maximum peak in this region is at 653 m above sea level, while almost other reliefs are in the range of 70-300 m (Song & Wang 1995). The zonal vegetation in this region is subtropical EBLFs, which have been severely disturbed in recent history with only small tracts of intact or semi-intact EBLFs left around a Buddhist temple in the Tiantong Forest Park (TT), as well as in Ruiyan Forest Park (RY) and in Dongqian Lake Landscape Area (DQ). These three sites, spaced approximately 15 km from each other, had been subjected to various intensities of human disturbances (usually clear cutting), but have been protected from clear cutting from 25 years ago. Consequently, the integral plant community structure was different among the three sites. Thus, the vegetation in those three sites represented different successional status or extents of maturity, with vegetation being mature in TT, pre-mature in both TT and RY, and young (i.e., shrubs) among three sites. In addition, soil N availability is highest in mature than in pre-mature and young vegetations (Yan *et al.* 2009), but soil P availability is lower in mature than in young vegetation (Yan et al. 2008).

Plots within each of those forest sites were systematically established 100-500 m apart, sited on the similar slope location and possessing similar vegetation history and soil type and soil texture (i.e., mainly loam textures). In these sites, 31 forest plots were established in total, which covering all typical habitats in this area, with nineteen in TT, four in RY, and eight in DQ. Each plot (20 × 20 m) was located at least 100 m away from the stand edge. Because the 20 × 20 m plot is the smallest reasonable area to be considered as a community for subtropical forests (Song & Wang 1995), I used the plot to represent a forest community with a set of species co-existing and interacting in a locality. A description of the vegetation and soil characteristics of each studied plot was provided in Appendix II.

4.2.2 Measurements and calculations of carbon stocks

In each plot, basal diameter and diameter at breast height (DBH) were measured for trees taller than 1.50 m, while basal diameter and diameter at 45 cm above the ground (D_{45}) were measured for trees shorter than 1.50 m. The total tree height was measured with a telescopic pole for heights up to 15 m, and with a clinometer for heights >15 m. Above-ground biomass of individual trees (AGB*t*, DBH > 5cm) was calculated using an allometric equation (1) (Brown, Gillespie, & Lugo 1989). Although more recent allometric equations exist (Chave *et al.* 2005), it includes plant height and stem wood density (SWD). Generally, plant height and SWD are important functional traits included in the functional diversity components analysis. To avoid any circularity in relating C stocks with plant functional diversity, equation (4.1) should be appropriate.

$$\mathrm{AGB}t = exp\{-2.134 + 2.530 \times Ln(\mathrm{D})\} \qquad \text{eqn (4.1)}$$

where D is diameter at breast height.

AGB of small trees (DBH < 5 cm) was calculated using an locally developed allometric equation (4.2). Also, equation (4.2) does not include any plant functional traits being used in estimating functional diversity components. More details on its development and its advantages are presented in Tables 2.1, 2.2 and 2.3, as well as Figures 2.1 and 2.2.

$$\mathrm{AGBs} = 1.460 \times \exp\{\text{-}3.23 + 2.17 \times \mathrm{Ln}(\mathrm{D})\} \qquad \text{eqn (4.2)}$$

Where, D is DBH < 5cm.

The sum of the aboveground biomass for trees and shrubs was considered as aboveground biomass (AGB). Then, Iconverted AGB to ACS (Mg ha^{-1}) by multiplying AGB with a factor of 0.5, as 50% of the total tree biomass is elemental C (Dixon *et al.* 1994).

Soil samples were collected from 0–20 cm depth from 31 sample plots distributed in all three sites studied. Soil samples in each plot were collected from five randomly selected points, resulting in 155 samples (5 samples per plot x 31 plots). The samples were shipped to the laboratory and air-dried over 30 days. Then each soil sample was sieved through a 2 mm sieve and thoroughly mixed to determine SOC concentrations using the oil bath-K_2CrO_7 titration method (Nelson & Sommers 1996). In each plot, soil bulk density was determined using a steel corer of known volume. Five soil cores were collected per plot. The soil cores were dried in at 105 °C in an oven for > 48 hours. Bulk density (g cm^{-3}) was calculated by dividing the oven dry weight of the soil (g) by the volume of the soil core. The amount of SOC (Mg ha^{-1}) was calculated by multiplying organic C content by soil depth and soil bulk density (Brown 2004).

Finally, I calculated total ecosystem C (TEC) stocks by adding ACS (as a representative of aboveground standing C stocks) and SOC stocks (as a representative of belowground C stocks).

4.2.3. Measurements of plant functional traits

For measuring functional traits, 3-6 healthy mature individuals per species were randomly selected and marked in each plot at the peak of the growing season, and recorded species identity and its coverage to estimate the relative abundance of different species. Then leaf, twig and stem samples were collected in the months of July and August. I measured 14 functional traits for each sampled individual according to methodology of Cornelissen *et al.* (2003) including LDMC, SLA, LNC and LPC in both mature (from one year old or older twigs) and current year leaves, twig dry matter content (TDMC), twig wood density (TWD), stem wood density (SWD), stem N concentration (SNC), stem P concentration (SPC) and plant height (H).

In the field, five branches were cut from five different positions in each sampled plant, i.e., the upper position and the four edges of the crown. After that according to the terminal set of internodes, the current year twigs were separated from the collected branches. Approximately 20 mature leaves without visible leaf damage were chosen and sampled, and then one current year twig without visible leaf loss was sampled from each branch. Each sample was covered in a moist paper towel and stored in a sealed plastic bag and kept cool until brought back to the laboratory for further analysis. In the laboratory, the leaves were removed from the twig, after that twig length and diameter at the midpoint were measured using an electronic vernier caliper (0.1 mm accuracy). Cross-sectional area of the twig was calculated from the diameter while twig volume was approximately assumed to a cylinder shape. At the same time, 20 mature leaves from each branch and all fresh leaves attached on the current year twig were collected to form separate samples. The leaves were then scanned using a leaf area meter (LI-3100C, Li-Cor, USA) to determine the mean leaf area for each mature and current year leaf. Then leaf and twig samples were dried at 75ºC for 48 hrs in an oven to determine leaf and twig dry mass, for calculating LDMC, SLA, TDMC and TWD. Finally, the leaf samples were milled to determine LNC and LPC using a flow-injection auto analyzer (Skalar, Netherland).

I used tree increment cores for plant having diameter greater than 1 cm to determine SWD. Using a 5 mm-diameter increment corer, I collected one core at breast height or at 45 cm height from each plant which was also used for collecting leaf and twig samples. In the laboratory, the length and diameter of each tree core was measured using an electronic vernier caliper (accurate to 0.1 mm), then the volume of each tree core was calculated, supposing the tree increment core was estimated to a cylinder shape. For plant having diameter less than 1 cm, I cut the sample from the stem. In the laboratory, each fresh stem sample was totally immersed in a volumetric flask of known weight filled with distilled water, and the increase in weight was recorded. The weight increment equals the green volume of the wood sample. Wet SWD was measured and after that samples were then dried at 75ºC in an oven for 72 hrs to determine dry mass and to calculate dry SWD (Chave et al. 2006). Finally, the stem samples were milled to determine SNC and SPC.

4.2.4. Calculations of functional diversity indices

I used plant coverage instead of basal area for defining species abundance (Pakeman & Quested 2007), because it avoids potential circularity derived from using DBH. in the calculation of both functional diversity and C storage. Species abundance in the study thus corresponds to its relative coverage in the plot. The coverage of each species in each forest ecosystem types, used in the study, was detailed in Appendix IV.

Two components of plant functional diversity were used to test the strength of relationship between biodiversity and ecosystem C Stocks. Firstly, community weighted trait means (CWM, equation (4.3) was calculated by averaging trait value in the community and then weighted by the species abundance (Garnier *et al.* 2004). This metric represents the expected functional trait value of a specific community (Díaz *et al.* 2007).

$$\mathrm{CWM_x} = \sum_{i=1}^{s} p_i t_i \qquad \text{eqn (4.3)}$$

where CWM_x is the CWM for a x trait, s is the number of species in the community, p_i is the relative coverage of ith species in the community and t_i is the trait value for the ith species.

Secondly, functional divergence (FDvar) of the single-trait was calculated based on equation (4.4), which basically represents the variance in trait values weighted by the abundance of each species in the community (Mason *et al.* 2003). I selected FDvar over other functional dissimilarity indices, because it incorporates species relative abundance and has demonstrated to have good performance (Pakeman, Lennon & Brooker 2011).

$$\mathrm{FDvar} = {}^{2}\!/_{\pi}\ \arctan(5\mathrm{V}) \text{ And } \mathrm{V} = \textstyle\sum_{i=1}^{s} \mathrm{p_i}\,(\ln \mathrm{x_i} - \ln \mathrm{x})^2 \qquad \text{(eqn 4.4)}$$

where x_i is the trait value for the ith species, $\mathrm{p_i} = \mathrm{a_i} / \sum_{n=1}^{s} \mathrm{a_i}$ and $\ln \mathrm{x} = \sum_{i=1}^{s} \mathrm{p_i} \ln \mathrm{x_i}$, where a_i is the relative coverage of the ith species in the community (Pla, Casanoves & Di Rienzo 2012). FDvar values range between 0 and 1.

The analysis on the both functional diversity indices was performed using the ''*vegan FD*'' package for the R statistical language (R Development Core Team 2013).

4.2.5. Statistical Analyses

I used a linear regression to test for the pair wise associations of two functional diversity components with C stocks in each of above- and belowground and ecosystems levels. In order to assess the joint effects of two functional diversity components on C stocks, a multiple step-wise regression was carried out by including all 28 functional diversity indices together. The minimal adequate significant model was selected according to significant parameters ($P < 0.05$; MacNally 2000). For each response variable of ACS, SOC or TEC, I performed multiple stepwise regression analysis for selecting the best fitted model. The final minimal adequate models were expected to be consistent to the results based on *LR* test.

Further, to distinguish the relative contribution of two functional diversity components to C stocks, I ran three linear models for each response variable (ACS, SOC or TEC). The first is a full model including both two functional components with 14 CWM indices and 14 FDvar indices based on 14 traits. The second and third models are the partial models, with the second one including 14 FDvar indices only (i.e., without any CWM indices), while with the third one keeping CWM indices only (i.e., without any FDvar indices). Among three models, the most parsimonious one for each response variable was selected according to *AIC* criteria (Akaike 1973). After this, to determine whether each functional diversity component has a significant additional contribution to the explanation of ecosystem C stocks, I used the generalized log-likelihood ratio (*LR*) test to compare the partial model (i.e., including either

CWM or FDvar indices) with full model (i.e., including indices from both diversity components) (Huelsenbeck & Crandall 1997). Then the parsimony of each model was assessed by considering *LR* test and associated *P* values. The summary of variables of each studied plot used in the analysis was shown in Appendix IV.

Finally, I used a bubble graph approach to assess the effects of the FDvar and CWM indices that were retained in the final minimum adequate model, on the TEC stocks across forest plots. Thereby, I selected the final significant predictors of TEC stocks (Table 4.1), i.e. CWM of LNC-F and multi-trait functional divergence (FDiv) index, calculated on the basis of H and TWD.

4.3. Results

4.3.1. Relationships between functional diversity indices and C stocks

Two functional diversity components (i.e., CWM and FDvar) based on 14 single plant functional traits showed different relationships with C stocks in each of above- and belowground partition and at the level of ecosystem (Fig. 4.1 and Appendix V). With respect to aboveground, CWMs of H (Fig. 4.1a) and LPC-F (Fig. 4.1c) and FDvars of H (Fig. 4.1d), TDMC (Fig. 4.1e) and TWD (Fig. 4.1f) were positively, but CWM of LNC-F (Fig. 4.1b) was negatively correlated to ACS. For belowground, there were significant negative relationships between SOC and FDvars of LNC-M (Fig. 4.1g) and SNC (Fig. 4.1h) only. In terms of ecosystem level, CWMs of H (Fig. 4.1i) and LPC-F (Fig. 4.1j) and FDvars of H (Fig. 4.1k) and TWD (Fig. 4.1l) were positively correlated to TEC stocks. The rest functional diversity components not significant correlated to C stocks (Appendix V).

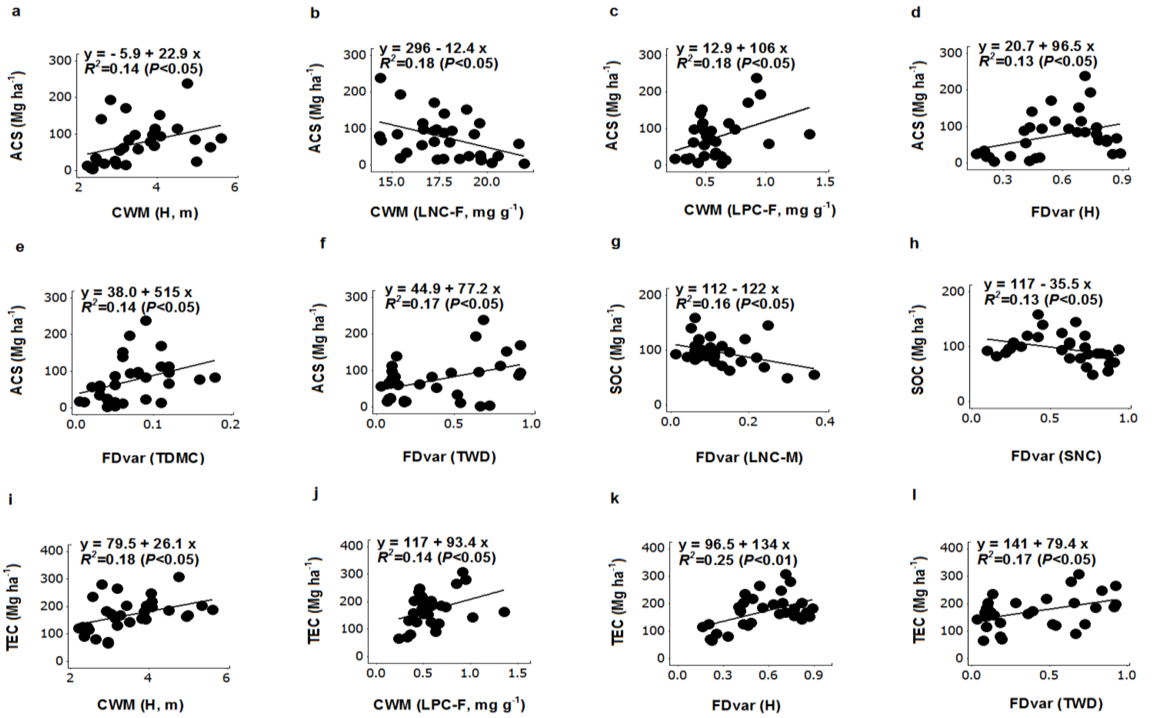

Figure 4.1 Simple linear regression analyses between the magnitude of C stocks and functional diversity components indices of the subtropical evergreen broad-leaved forests of eastern China. Only significant associations (see Appendix V) were shown here.

4.3.2. Joint effects of functional diversity on C stocks

For each of the ecosystem compartments, I applied stepwise regression to select the best fitted model by including 28 indices based on both functional diversity components (14 CWM and 14 FDvar indices) as predictive variables (Table 4.1). Thereby, the significant parameters from both functional diversity components (CWM and FDvar) were retained in the final model to reveal their explanatory powers with respect to C stocks. In case of ACS, CWM (LNC-F, LPC-F and SPC) and FDvar (H and TWD) jointly explained 74 % variation, with a positive prediction of the CWM of LPC-F and SPC, and the FDvar of H, TWD and LNC-F, but with a negative prediction of the CWM of LNC-F (Table 4.1). In terms of below-ground compartment, 55 % variation in SOC was explained positively by the CWM of LNC-F and LDMC-F and the FDvar of H, and negatively explained by the FDvar of LNC-M (Table 4.1). Considering the level of ecosystem, 62 % of the variation in TEC was explained positively by the FDvar of H and TWD, but negatively explained by the CWM of LNC-F (Table 4.1).

Table 4.1 Joint effects of functional dominance and divergence on ecosystem C stocks in subtropical evergreen broad leaved forests in eastern China.

Multiple regressions of aboveground C, soil organic C and total ecosystem C stocks with 28 trait indices of two functional diversity components (FDvar and CWM) as predictable variables. Explanatory variables were selected using a stepwise procedure for selecting significant parameters ($P < 0.05$).

Model form	Predictor variables	*P*	R^2
Aboveground carbon stocks			
ACS = 194.86 – 16.77 CWM of LNC-F + 75.95 CWM of LPC-F + 83.17 FDvar of TWD + 92.21 FDvar of H + 147.10 FDvar of LNC-F + 75.67 CWM of SPC	*Model*	<0.001	0.74
	Constant	0.003	
	CWM of LNC-F	<0.001	
	CWM of LPC-F	0.017	
	FDvar of TWD	0.001	
	FDvar of H	0.008	
	FDvar of LNC-F	0.013	
	CWM of SPC	0.030	
Soil organic carbon stocks (0-20 cm)			
SOC = -94.49 – 136.70 FDvar of LNC-M + 5.60 CWM of LNC-F + 0.21 CWM of LDMC-F + 41.16 FDvar of H	*Model*	<0.001	0.55
	Constant	0.045	
	FDvar LNC-M	0.004	
	CWM of LNC-F	0.003	
	CWM of LDMC-F	0.005	
	FDvar of H	0.013	
Total ecosystem carbon stocks			
TEC = 195.70 + 155.02 FDvar of H + 106.23 FDvar of TWD – 8.59 CWM of LNC-F	*Model*	<0.001	0.62
	Constant	0.008	
	FDvar of H	0.001	
	FDvar of TWD	0.001	
	CWM of LNC-F	0.022	

Values are regression coefficients (R^2) from multiple regression analysis, indicated with P-values for each model and each predictor variable within the model. Relationships of each predictor variables showing a negative slope within model are indicated with negative signs.

4.3.3 Contribution of each functional diversity component to C stocks

According to *AIC* criteria, the full model including both CWM and FDvar was the most parsimonious model, predicting a higher variability of C stocks for both the two ecosystem compartments alone and combined, relative to the two partial models, in which either CWM or FDvar was considered only (Table 4.2). The *LR* test showed that each functional diversity component had an additional contribution to C stocks, because the full model differed significantly to the partial models ($P < 0.001$) for each of the ecosystem compartments and the ecosystem as a whole (Table 4.2). With respect to the respective contribution of each functional diversity component to C stocks, CWM explained higher variability of ACS and SOC than that explained by the FDvar. In contrast, at the ecosystem level, FDvar explained more variation in TEC than that explained by CWM (Table 4.2).

Table 4.2 Contribution of each functional diversity component to C stocks both in above and below-ground partitions as well as total ecosystem.

The weight of support for the alternative models (FDvar: functional divergence, CWM: community weighted mean) and estimates of model parameters for each ecosystem C stocks (aboveground C, soil organic C and total ecosystem C stocks). Adjusted R^2 shows the contribution of each functional diversity components to each C stocks partition. Result of log-likelihood ratio (*LR*) test with associated *P*-value for comparing models is presented.

Model	*df*	*AIC*	R^2	*L*	*LR*	*P*
Aboveground carbon stocks						
FDvar + CWM	28	233.97	0.93	-86.98		
FDvar	14	341.38	0.30	-154.69	135.42	<0.001
CWM	14	332.44	0.47	-150.22	126.48	<0.001
Soil organic carbon stocks (0-20 cm)						
FDvar + CWM	28	245.67	0.44	-92.84		
FDvar	14	299.84	0.01	-133.92	82.16	<0.001
CWM	14	297.31	0.06	-132.65	79.62	<0.001
Total ecosystem carbon stocks						
FDvar + CWM	28	275.63	0.73	-107.81		
FDvar	14	327.48	0.56	-147.74	79.86	<0.001
CWM	14	348.61	0.12	-158.31	101.00	<0.001

df: degree of freedom

AIC: Akaike Information Criterion (smaller is better).

L: log-likelihood value for each model

LR and associated P-value: Log-likelihood ratio test and associated *P*-value exhibited that full model is better than nested model when $P < 0.05$

4.3.4. Performance of functional diversity components on C stocks

Figure 4.2 showed that forests with high functional divergence of woody traits (i.e., H and TWD) and high functional dominance of leaf conservative syndromes (i.e., low LNC-F) had high levels of TEC stocks, indicating that the TEC stocks closely link with the plant ecological strategies.

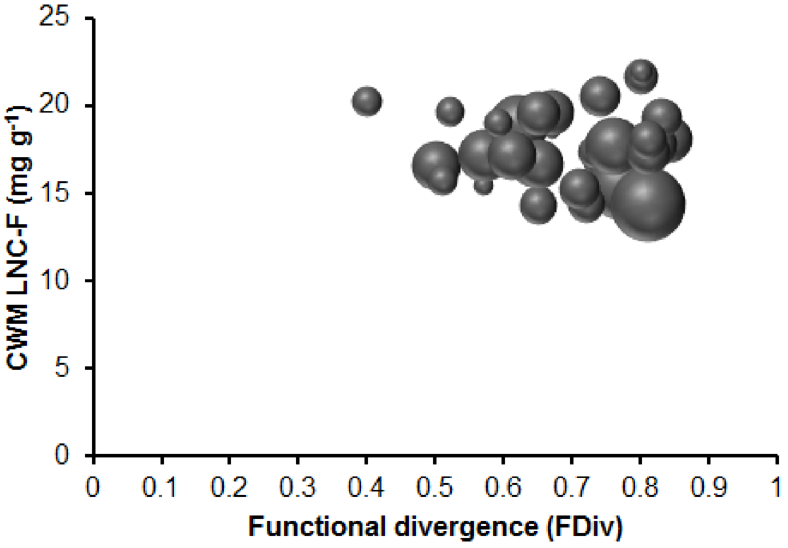

Figure 4.2 Performance of functional divergence and dominance on ecosystem C stocks in each studied forest plots in the subtropical EBLFs in eastern China. The performance analyses were done through the use of bubble graphs, and thus the circle size in the graph is proportional to the size of the total ecosystem C stock in each forest plot. Multi-traits functional divergence (FDiv) was calculated on the basis of H and TWD because FDvar of H and TWD were retained in the final model as best predictors. See Table 4.1 for associated statistics.

4.4. Discussion

This is the first study that reported the plant tissue-specific nature of the relationships between functional diversity and C stocks in forest ecosystems, with the two complementary components of functional diversity considered simultaneously. I found that coexisting plant species in an ecosystem with the high dominance of nutrient-use syndromes and the divergent woody syndromes result in the great ecosystem C stocks. The joint effects of functional dominance of nutrient-use syndromes and functional divergence of wood syndromes, and the additional contributions of the two independent functional diversity components on ecosystem carbon stocks suggest that plant species enhance ecosystem C stocks through utilizing the two complementary functional spaces. Therefore, this study provides a strong evidence for both the mass ratio hypothesis and the niche complementarity hypothesis in relation to the functional diversity-C stocks relationships in the subtropical forests.

4.5.1. Effects of plant nutrient-use and wood syndromes on ecosystem C stocks

I found that CWMs of H and LPC-F, and FDvars of H and TWD were significant positive associated with ACS and TEC, while FDvar of TDMC had significant positive pair wise association with ACS only. In contrast, CWM of LNC-F was negatively correlated with ACS, while FDvars of LNC-M and SNC had significantly negative pair wise associations with SOC (Fig. 4.1). These contrasting pair-wise associations of C stocks with two complementary functional components in relation to different functional characters suggest that plants with the dominant nutrient-use syndromes and the divergent wood syndromes result in great ecosystem C stocks in the studied forests.

The dominant effects of the nutrient-use syndromes on ecosystem C stocks may relate to the leaf economics spectrum (LES). The LES ranges from 'exploitative' plants characterized by N rich leaves and fast nutrient acquisition and turnover, thus being conducive to fast growth and high C accumulation, to 'conservative' plants with nutrient-poor leaves and slower growth associating with slow nutrient and C turnover (Wright *et al.* 2004). In which cases that ecosystem C stocks are influenced by the 'exploitative' or 'conservative' syndromes of plants may depend largely on the nutrient conditions for a specific site. On the one hand, in nutrient poor sites, plants with costly long-lived leaves are usually 'conservative' in their nutrient use strategy (Aerts & Chapin, 2000), thus providing slow litter decomposition and slow returns of carbon to soils (Reich *et al.* 1997; Wright et al. 2004). Plants in nutrient rich-sites, on the other hand, usually use nutrient in the way of 'exploitative' (Aerts & Chapin, 2000), i.e., with inexpensive short-lived leaves providing rapid returns of carbon and nutrient to soils.

In this study, the negative relationship between CWM of LNC-F and ACS and the positive relationships between CWM of LPC-F and each of ACS and TEC, suggest that LES is operating effectively for shaping ecosystem C stocks. The negative association of CWM of LNC-F with ACS indicates a 'conservative' syndrome in relation to the N use strategies, while the positive associations of CWM of LPC-F with both ACS and TEC imply an 'exploitative' syndrome with respect to the P use strategies across the studied plants. These patterns are partly consisted to the recent works that the dominance of either 'conservative' or 'exploitative' syndrome of plants is associated with the high C storages in temperate grassland ecosystems (Minden & Kleyer 2011; Laliberté & Tylianakis 2012; Lavorel & Grigulis 2012; Lienin & Kleyer 2012; Grigulis *et al.* 2013). The concurrence of N-'conservative' and P-'exploitative' syndromes in this study may be related to the successional status among the stands, where plant growth and/or vegetation productivity is N-limited for the pre-mature forests and young shrubs, but N-limited for the mature forests (Yan et al. 2008). It has been well documented that N limitation in the young stands and increasing P conservation in the naturally mature vegetation are the general patterns based on the long-term chronosequences worldwide (Wardle *et al*. 2004; Davidson *et al*. 2007). Since the most of stands in this study are pre-mature and young (i.e., 74 % in total), it is reasonable that in these types of vegetation, plant growth is N-limited. As a result, plant species in the N-limited condition may become excessively dominated by N-'conservative' and P-'exploitative' syndromes. In this case, because the most vegetation is not P-limited, plants with the prevailing P-'exploitative' syndromes are beneficial to leaf C assimilation, thus leading to a fast growth of the individual trees and a high C accumulation in the stands (Richter *et al.* 1999). Moreover, the prevailing N-'conservative' plant species increases residence time of living tissues and slows the organic matter decomposition (De Deyn, Cornelissen & Bardgett 2008). Eventually, the high accumulation of living tissues and the low C loss would result in high aboveground C stocks.

The dominance of the plant N-'conservative' effects on ecosystem C stocks could also be reflected by the negative correlations between SOC and each of functional divergence of LNC-M and SNC in this study. Functional divergence describes the tendency of species abundances to be on the outer margins of the functional space, when controlling for functional richness (Mason *et al.* 2005). The negative linear relationships between SOC and FDvars of LNC-M and SNC suggest that species with the low divergence of the N-based functional character are favored over those species being functionally distinct in terms of the N utilization in improving soil C stocks. In other words, this means that the high soil C stocks are contributed mainly by plant species that are clustered, instead of evenly distributed along the N-based functional axis. As discussed earlier, the most plant species in this study are N-'conservative'. Therefore, it suggests that plant species with the converging dominant proportion of the N-'conservative' strategies result in a high soil C stocks. The underlying mechanism may relate to the poor decomposability of leaf and wood litters that are contributed by the 'conservative' plant species (Cornwell *et al.* 2008; Mokany, Ash & Roxburgh 2008; Grigulis *et al.* 2013).

In addition to the low functional divergence of the N-'conservative' syndromes, this study showed that highly functional divergence of the wood syndromes (e.g., H, TDMC and TWD) also enhances ecosystem C stocks (Fig. 1). Consistently, this beneficial effect of FDvar of H on ecosystem C stocks has been found in the Californian rangeland (Butterfield & Suding 2013) and the semiarid ecosystems of Chaco (Conti & Díaz 2013). It is logical that an ecosystem containing plant species with the functionally diverse wood syndromes, e.g., a diverse array of height and other wood attributes, can cause a complex stand structure (Cavanaugh *et al.* 2014). Increasing stand structural diversity in an ecosystem could be more efficient for plants to accessing and utilizing resources that are limiting, thus increasing C stocks (Chave *et al.* 2005; Ruiz-Benito *et al.* 2013 Cavanaugh *et al.* 2014).

4.5.2. Joint effects of functional dominance and divergence on ecosystem C stocks

The multiple stepwise regressions analysis revealed that both CWM and FDvar were retained in the final model and jointly explained the great variation in each of ACS (74 %), SOC (55 %) and TEC (62 %) stocks (Table 4.1). Moreover, the generalized log-likelihood ratio (LR) test showed that the full model including both CWM and FDvar predicted a higher variability of ecosystem C stocks, than the partial model that includes either CWM or FDvar only (Table 4.2). These combined results suggest that C stocks in the studied forests are jointly enhanced by both CWM and FDvar of plant functional character, with each of these two functional diversity components having an additional contribution for enhancing C stocks. This is consistent to my hypothesis that both CWM and FDvar jointly explain more variation in C stocks than each component of functional diversity in the subtropical forests.

The joint effects of CWM and FDvar on C stocks might be explained by both the mass ratio hypothesis and the niche complementarity hypothesis. The mass ratio hypothesis proposes that ecosystem functioning is primarily driven by functional character of the most dominant species (Grime 1998), while the niche complementarity hypothesis suggests a complementarity effects among species with varying functional character (Petchey & Gaston 2006). Although several recent studies provided evidence for either the mass ratio hypothesis (Conti & Díaz 2013; Mokany, Ash & Roxburgh 2008; Laughlin 2011; Lavorel *et al.* 2011) or the niche complementarity hypothesis (Schumacher & Roscher 2009; Mouillot *et al.* 2011; Laliberté & Tylianakis 2012; Butterfield & Suding 2013; Grigulis *et al.* 2013), the joint effects of CWM and FDvar on C stocks in this study support both two hypotheses simultaneously, i.e., the great dominance of nutrient-use syndromes and the divergent plant height and wood syndromes together explaining great variation in ecosystem C stocks. The only study that I am aware is Cavanaugh *et al*. (2014), in which they partly supported the both hypotheses on the basis of functional dominance and taxonomic diversity (genus richness and Shannon diversity). However, they did not find significant relationship between functional diversity and ACS. In their study, the functional diversity components were calculated by using the maximum diameter of stem and SWD only, but did not include leaf syndromes. It has been recognized that leaf syndromes link plant resource use strategy and plant C economics functionally (Lavorel & Grigulis 2012). This thus might be one of reasons of why functional diversity did not correlate to ACS in their study.

The major advantage of my study is to test the associations of two biodiversity components with ecosystem C stocks on the basis of 14 functional traits representing both leaf and wood syndromes of plants. Therefore, the leading dimensions of plant ecological strategies and the large range of plant functional axes were captured to as the functional markers of ecosystem C stocks. The joint effects of dominant leaf nutrient-use syndromes and divergent wood syndromes on C stocks suggest that leaf and wood economics (Reich *et al.* 1997; Wright *et al.* 2004; Chave *et al.* 2009) cooperate closely to regulate C budget at the ecosystem level, even though they are decoupled at the plant species level (Baraloto *et al.* 2010). This means that, at the ecosystem level, a set of co-existing species improve C stocks though spreading the extreme of leaf economics and increasing variation of wood economics simultaneously. It is true that the orthogonal leaf and wood economics at species level decouple the leaf-mediated niche from the wood-mediated niche, thus increasing the dimensionality of niche space and thus efficiency of resource utility among co-existing species (Kang *et al.* 2013). As such, this would result in great ecosystem C stocks. Although this explanation is speculative, it can be evidenced by a recent meta-analysis of manipulative studies that has reported, on average, both niche complementarity and mass ratio effects contribute about 50% to the biodiversity-ecosystem function relationship (Cardinale *et al.* 2011).

5. Conclusions, implications and recommendations

5.1. Conclusions

The research presented in this thesis demonstrates the complexity of the relationships between biodiversity components (stand structural diversity and functional diversity) and ecosystem C stocks in subtropical EBLFs. The followings are the outlines of main findings and conclusions.

5.1.1. Consideration of shrub biomass of the forest ecosystem in biodiversity-C stocks relationships

By accounting for the shrub component of the vegetation, my developed allometric models provide a way to improve the quantification of biomass and C stocks in subtropical forests in eastern China. A better understanding of C stocks in the forest ecosystem by including the shrubs communities will provide useful information on the growth pattern of these species and on the resource dynamics of their communities. Furthermore, this information is important for biodiversity-C stocks relationships, and regional estimations of NPP in these areas. The specie-specific allometric equations developed in this work allowed the researchers to estimate the total AGB for 14 dominant shrub communities in subtropical forests. Multispecies allometric equations developed in this study could provide useful estimations in similar EBLFs ecosystems where species-specific equations are not yet available. The non-destructive method used proved accurate enough for large areas as well as for communities with specific compositions and complex structures. Furthermore, this non-destructive biomass estimation can conserve the forest structure and functioning from cutting and may be better understand the relationships between biodiversity and ecosystem C stocks. Thus, in this study, it was proved that the locally developed equations performed better for the quantifying the relationships between biodiversity and ecosystem C stocks because I found similar results of biodiversity-C stocks in all studied plots, relatively no over- or underestimation of ACS.

5.1.2. Stand structural diversity was crucial in increasing ACS but not for SOC in subtropical EBLFs in eastern China

High stand structural diversity was beneficial in increasing ACS but had no relationship with SOC in subtropical EBLFs in eastern China. Thus, I suggest that a combined effect of stand structural diversity in addition with stand A, P and N is the most parsimonious explanation, which consistently reached unprecedented levels of predictive accuracy (up to 83%) for ACS. Taken separately, each diversity component has weak explanatory power for ACS. The above conclusion is consistent with evidence that forest productivity and/or AGS are affected by either species (Con *et al.* 2013), tree size (Liang *et al.* 2007) or height diversity (Lei *et al.* 2009). My study demonstrated that the joint effect of stand structural diversities had a greater impact on ACS than that of each diversity indices alone. I provide evidence that high tree size and height diversities mutually improve ACS by promoting complementary resource utilization among tree species. Hence, the niche complementarity hypothesis with respect to the relationship between biodiversity and ecosystem C stock is expanded to the whole stand level, and to a new geographical area. The joint effect of stand structural diversity on ACS may have important implications for forest C management. To increase forest C stocks, I strongly suggest that modifying high stand structural diversity, tree size and height diversity in particular in the subtropical forests, could be an effective approach for mitigating the current global climate change.

5.1.3. A set of coexisting plant species in an ecosystem with the high dominance of nutrient-use syndromes and the divergent woody syndromes result in the great ecosystem C stocks

This is the first study that reported the plant tissue-specific nature of the relationships between functional diversity and C stocks in forest ecosystems, with the two complementary components of functional diversity considered simultaneously. We found that coexisting plant species in an ecosystem with the high dominance of nutrient-use syndromes and the divergent woody syndromes result in the great ecosystem C stocks. The joint effects of functional dominance of nutrient-use syndromes and functional divergence of wood syndromes, and the additional contributions of the two independent functional diversity components on ecosystem carbon stocks suggest that plant species enhance ecosystem C stocks through utilizing the two complementary functional spaces. Therefore, this study provides a strong evidence for both mass ratio hypothesis and niche complementarity hypothesis that might be the conjunctive mechanisms regulating C storage in the subtropical forests.

5.2. Implications for biodiversity conservation and carbon management

This study showed that the biodiversity components i.e., stand structural diversity and functional diversity (dominance and divergence) explain enough variation in ACS stocks in subtropical EBLFs while variation in SOC stocks were only assessed through functional diversity components. Thus, it was found that the jointly effects of stand structural diversity indices with addition of control variables (A, P and N) were stronger than the alone effects of each stand structural diversity or control variables on ACS. On the other hand, the mutual effect of functional dominance of leaf conservative resource use syndrome and divergence of woody traits increased C stocks both in above- and belowground as well as total ecosystem level. Additionally, the contribution of stand structural diversity and functional diversity to ACS were almost same. Therefore, the performance of each forest community showed that C stocks can be increased with the maintenance of high stand structural and functional diversities in EBLFs. Thus, my results suggest that management practices can be used to increase ecosystem C stocks by increasing tree species composition and stand structure. Thus, maintaining structural, functional and productive characteristic of forest stands are important for better ecosystem functions, particularly to store more C in trees and soils because SOC was more affected by functional diversity of the stands. Hence, for the purpose of increasing C stocks in both trees and soils, forest managers should focus more on tree size and height along with variety of productive species to maintain biodiversity for higher ecosystem productivity (Liang et al. 2007; Lei *et al.* 2009; Mouillot *et al.* 2011). For such pattern, mixed and uneven-aged stands should be encouraged because they create several canopy strata forests and variety of functional traits, will be more efficient in C stocking and productivity. For instance, green tree retention could be encouraged as an alternative management strategy to create structurally complex forest stands (Sullivan *et al.* 2001, Zenner 2000). Thus, I suggested that the initial forest structure (in terms of species, size and height class distribution) and conservative species composition having variety of functional traits, if not taken into account, may affect C stocks and productivity (Garcia-Gonzalo *et al.* 2007). Future forest management practices could be determined by the research presented above.

5.3. Recommendations for future research

I found that biodiversity components – stand structural diversity and functional diversity contribute to explain ecosystem C stocks at the forest ecosystem level but it is known that abiotic or environmental factors directly affect the ecosystem functioning (Diaz *et al.* 2007) and may also affect the ecosystem C stocks. Furthermore, it is also necessary to test the biodiversity-ecosystem C stocks relationships on different slope aspects of the mountain or at different altitude because may be different micro-environmental factors on different slopes or altitude may also affect the ecosystem C stocks. Therefore, it is important for the future research to test the combine effect of land use, abiotic, taxonomic diversity, and trait-mediated (functional diversity) effects on ecosystem C stocks by comparing statistical models including categorized combinations of effects. Thus the following research questions are important to be addressed in future researches:

1. What are the most important environmental or abiotic controls of taxonomic diversity, functional divergence, dominance and ecosystem C storage in subtropical EBLFs?
2. Is there a positive relationship between taxonomic diversity, functional divergence or functional dominance and ecosystem C stocks, after considering for environmental or abiotic factors?
3. Are there positive relationships between biodiversity components with ecosystem C stocks at different slope aspects or altitude of the mountain or landscape level?
4. How plant functional trait relationships scale-up to trade-offs and interactions with ecosystem C stocks.

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Appendix I Characterization of each forest plot considered for the study of the association between stand structural diversity and carbon stocks in the subtropical forests of eastern China, including general plants composition and soil (20 cm depth) properties.

Max. H: maximum stand height; Max. DBH: maximum stand diameter at breast height; BA: total stand basal area; VOB: total over-bark stand volume; AGB: total stand aboveground biomass; ACS: total stand aboveground C stocks; N: stand density; A: stand age; P: site productivity; S: species richness; E: species evenness; H_s: tree species diversity; H_d: tree size diversity; H_h: tree height diversity; SBD: soil bulk density; and SOC: total stand soil organic C stocks.

Plot	Max. H (m)	Max. DBH (cm)	BA (m^2 ha^{-1})	VOB (m^3 ha^{-1})	AGB (Mg ha^{-1})	ACS Mg ha^{-1})	N (stems ha^{-1})	A (year)	P (m^3 ha^{-1} year^{-1})	S	E	H_s	H_d	H_h	SBD (g cm^{-3})	SOC (Mg ha^{-1})
1	23.7	57	35.62	519.79	339.38	169.69	5700	120	4.33	45	0.39	1.47	1.82	1.49	1.01	94.94
2	24.1	50	40.82	641.39	386.84	193.42	6625	120	5.34	46	0.23	0.9	1.73	1.83	1.06	88.08
3	23.5	57	47.56	832.81	477.81	238.91	3550	120	6.94	39	0.4	1.45	1.91	1.66	0.91	68.81
4	21.9	30	18.55	294.94	132.04	66.02	4675	65	4.54	25	0.12	0.39	0.91	1.45	1.31	87.42
5	24.4	25	22.85	306.39	155.18	77.59	6250	70	4.38	26	0.27	0.88	0.93	1.35	1.25	78.6
6	25.7	31	25.61	387.47	166.66	83.33	10625	65	5.96	46	0.31	1.18	1.42	1.76	1.21	85.77
7	5.7	12	9.88	40.96	34.5	17.25	11500	20	2.05	20	0.62	1.85	0.67	0.45	1.3	62.73
8	12	23.5	7.79	50.88	37.71	18.86	7277.778	20	2.54	31	0.59	2.03	1.09	1.28	1.28	49.56
9	12	23.5	7.81	50.92	37.73	18.86	7277.778	20	2.55	31	0.59	2.04	1.08	1.28	1.32	55.03
10	8.9	22.3	23.77	139.95	118.94	59.47	12900	20	7.00	31	0.46	1.58	1.09	0.93	1.34	120.7
11	12	16	5.85	39.62	27.81	13.91	2875	20	1.98	8	0.44	0.9	0.6	1.32	1.45	117.6
12	17	18	10.14	131.33	58.41	29.21	5200	65	2.02	13	0.29	0.75	0.85	1.55	1.4	158.8
13	17.6	46.5	28.83	331.4	227.79	113.89	5450	120	2.76	32	0.46	1.61	1.73	1.65	1.19	89.31
14	16.3	35	28.45	262.92	187.82	93.91	6166.667	120	2.19	30	0.6	2.05	1.51	1.55	1.28	125.1
15	21	39	24.55	323.04	187.05	93.52	4250	120	2.69	30	0.5	1.7	1.5	1.78	1.19	105.2
16	18.5	41	24.8	224.81	175.22	87.61	2340	70	3.21	23	0.7	2.2	1.31	1.4	1.29	101
17	30.9	46	33.14	700.28	279.46	139.73	8300	110	6.37	36	0.6	2.14	1.45	2.17	1.17	97.17
18	26.8	58	32.64	496.21	304.57	152.28	3675	110	4.51	25	0.44	1.4	1.86	1.66	1.19	97.14

19	18.9	37	26.23	289.05	192.35	96.18	5350	80	3.61	36	0.55	1.98	1.52	1.66	1.14	107.8
20	16.4	40	21.85	250.08	166.29	83.15	2625	75	3.33	29	0.51	1.7	1.13	1.57	1.25	79.05
21	14.1	23	5.21	39.95	24.22	12.11	5475	17	2.35	39	0.67	2.45	1.08	1.43	1.31	107.7
22	25.7	50	25.22	489.49	227.42	113.71	3725	80	6.12	39	0.32	1.16	1.83	1.77	1.23	71.78
23	13.3	33	17.05	179.84	124.34	62.17	1625	70	2.57	26	0.67	2.2	1.15	1.19	1.27	140.7
24	16.6	31	27.33	297.63	195.33	97.67	4150	75	3.97	31	0.45	1.56	1.26	1.44	1.28	83.3
25	5.3	13	13.29	54.49	46.43	23.22	11900	15	3.63	34	0.85	2.99	0.6	0.37	1.38	93.86
26	15.1	20	21.42	223.83	121.52	60.76	10850	70	3.2	21	0.38	1.15	0.81	1.31	1.23	98.55
27	11	23	13.34	80.57	69.64	34.82	7080	26	3.10	28	0.57	1.91	1.07	1.11	1.34	92.32
28	16	31.5	8.64	113.07	60.43	30.22	1800	65	1.74	18	0.39	1.12	1.42	1.33	1.35	145.7
29	18	10	3.28	27.31	8.57	4.28	4200	19	1.44	14	0.73	1.92	0.55	1.35	1.32	120.4
30	10	20	22.02	177.41	130.69	65.35	5300	18	9.86	9	0.47	1.02	0.88	0.9	1.38	87.13
31	7	10	2.78	13.16	6.3	3.15	3900	15	0.88	12	0.7	1.75	0.6	0.96	1.38	88
32	20	10.4	11.48	110.35	48.37	24.18	5950	30	3.68	15	0.33	0.9	0.69	1.71	1	39.92
33	13.3	33	17.02	179.79	124.31	62.16	1325	120	1.50	23	0.7	2.19	1.14	1.18	0.75	126.2
34	24.1	50	40.66	641.13	386.71	193.35	4625	130	4.93	39	0.24	0.87	1.72	1.83	1.3	55.41
35	15	9.9	8.49	83.47	33.78	16.89	6200	30	2.78	13	0.5	1.27	0.66	1.51	1.3	80.35
36	5.3	13	10.25	42.61	34.38	17.19	8500	30	1.42	26	0.85	2.77	0.55	0.34	1.28	71.06
37	5.7	12	9.28	39.05	32.42	16.21	10500	18	2.17	16	0.6	1.67	0.68	0.44	1.13	14.84
38	16.7	12.41	5.6	60.46	23.99	12	4400	70	0.86	24	0.57	1.83	0.69	1.58	1.27	44.25
39	17.4	25	22.79	300.88	155.27	77.64	4700	65	4.63	21	0.29	0.88	0.92	1.24	0.96	37.01
40	16.6	31	27.25	297.56	195.27	97.64	3550	65	4.58	24	0.45	1.43	1.25	1.44	1.28	51.89
41	16.5	26.39	9.54	62.34	52.85	26.43	3325	25	2.49	28	0.72	2.41	1.32	1.5	0.83	74.36
42	21	45.45	23.21	356.06	203.66	101.83	2225	130	2.74	25	0.7	2.26	1.64	1.48	0.92	61.75
43	24	55.7	37.51	611.9	383.62	191.81	4350	130	4.71	23	0.59	1.87	1.27	1.71	0.98	58.17
44	25.1	38.9	28.84	419.07	223.04	111.52	5375	130	3.22	25	0.64	2.07	1.37	1.89	0.99	59.36
45	25.1	50.01	20.47	278.44	168.25	84.13	3600	80	3.48	23	0.68	2.12	1.52	1.78	0.98	65.82
46	30.7	37.88	15.64	217.79	115.42	57.71	3525	80	2.72	29	0.55	1.85	1.49	1.23	0.73	78.36
47	24.2	54.75	25.33	395.23	235.53	117.77	3000	130	3.04	27	0.68	2.24	1.46	1.89	0.81	81.95

48	18.5	49.34	22.88	298.59	221.45	110.73	2425	80	3.73	29	0.62	2.08	1.29	1.55	0.88	72.31
49	15	36.13	9.63	84.62	65.61	32.81	2375	25	3.38	17	0.79	2.24	1.58	1.48	0.95	71.77
50	21.2	36.7	10.22	96.66	73.53	36.77	2075	65	1.49	18	0.69	1.99	1.19	1.63	0.95	74.85
51	20	65.73	17.61	247.87	168.15	84.08	3000	80	3.10	20	0.53	1.59	1.36	1.39	0.76	72.9
52	26	61.27	30.59	593.3	346.85	173.42	3650	130	4.56	22	0.37	1.15	0.94	1.58	0.81	86.94
53	18.9	39.63	17.84	191.98	130.89	65.45	4525	80	2.4	20	0.6	1.79	1.49	1.86	0.84	83.74
54	28.5	66.53	26.83	539.45	283.16	141.58	3175	130	4.15	23	0.62	1.95	1.88	1.75	0.89	73.73
55	30	33.84	12.12	128.63	85.77	42.88	2725	65	1.98	17	0.7	1.98	1.54	1.46	0.86	72.01
56	24.5	26.52	13.22	141.76	79.94	39.97	3325	25	5.67	31	0.67	2.28	1.34	1.93	0.85	74.94
57	26.5	22.47	8.4	81.77	42.76	21.38	3450	25	3.27	17	0.69	1.95	1.04	1.85	0.85	95.49
58	27.7	21.77	10.27	103.22	57.51	28.75	3300	65	1.59	17	0.6	1.71	0.93	1.88	0.86	95.59
59	30.3	35.71	11.43	142.03	84.88	42.44	2250	65	2.19	16	0.54	1.49	1.5	1.74	0.85	74.42
60	31.2	38.39	12.08	190.25	84.33	42.17	2775	65	2.93	20	0.63	1.89	1.57	2.02	0.81	71.06
61	23	22.25	4.53	42.8	24.97	12.48	1525	25	1.71	17	0.62	1.75	1.05	1.44	0.87	91.06
62	28.9	61.75	16.36	124.35	149.37	74.69	4150	130	0.96	21	0.49	1.5	1.42	1.47	0.84	95.34
63	31.8	32.56	16.22	212.46	114.17	57.09	3950	130	1.63	22	0.68	2.1	1.57	1.62	0.87	96.95
64	31.4	26.83	8.48	125.72	55.35	27.67	2600	25	5.03	14	0.67	1.76	1.35	1.63	0.83	88.84
65	31.5	28.43	13.03	176.66	87.78	43.89	2425	130	1.36	17	0.8	2.25	1.31	2.08	0.84	96.36
66	18	33.7	27.14	354.1	208.1	104.05	6200	65	5.45	36	0.64	2.29	1.47	1.47		
67	23.5	45	23.66	313.84	182.2	91.1	6200	65	4.83	36	0.57	2.04	1.69	1.51		
68	17	49.5	31.75	424.71	260.95	130.48	6175	65	6.53	31	0.57	1.95	1.73	1.37		
69	29.5	41	24.37	309.07	187.12	93.56	6125	65	4.75	34	0.56	1.96	1.62	1.53		
70	17	52	27.39	351.44	218.57	109.29	6275	65	5.41	37	0.67	2.43	1.74	1.47		
71	25	49	31.46	467.36	251.74	125.87	8275	130	6.68	38	0.76	2.77	1.76	1.9		
72	20	47	30.87	455.44	244.1	122.05	8275	130	6.51	35	0.67	2.4	1.74	1.6		
73	20	42	29.49	406.57	226.3	113.15	8225	130	5.81	36	0.73	2.61	1.74	1.74		
74	19	40	28.81	384.79	217.74	108.87	8300	130	5.5	33	0.76	2.67	1.56	1.57		
75	20	38.5	28.27	412.18	211.44	105.72	8300	130	5.89	31	0.73	2.52	1.56	1.73		
76	20	38	27.85	389.13	207.2	103.6	8275	130	5.56	31	0.66	2.26	1.56	1.64		

77	28	37.5	28.02	414.75	205.72	102.86	8450	130	5.93	39	0.74	2.7	1.58	1.8
78	27	56.5	37.94	811.28	391.09	195.55	3850	70	11.6	9	0.82	1.79	1.82	1.44
79	25	56	31.75	541.37	316.35	158.17	3850	70	7.73	7	0.78	1.51	1.73	1.39
80	26	54.5	26.95	577.16	274.6	137.3	3280	70	8.25	6	0.51	0.91	1.64	1.18

Appendix II Simple linear regression analyses between stand structural diversity, control variables and C stock size in subtropical evergreen broadleaved forests in eastern China. Values indicate the regression coefficients (R^2) from simple linear regression analysis, indicated with asterisks if statistically significant (*: $P < 0.05$; **: $P < 0.01$; ***: $P < 0.001$; ns: not significant). Relationships showing a negative slope are indicated with negative signs.

Stand variables	ACS	SOC
Tree species diversity	-0.01ns	-0.000ns
Tree size diversity	**0.49*** **	0.000ns
Tree height diversity	**0.13*** **	0.000ns
Stand age	**0.50*** **	0.004ns
Site productivity	**0.41*** **	-0.02ns
Stand density	-0.000ns	-0.01ns

Appendix III Characterization of each forest plot representing different forest ecosystem types considered for the study of the association between functional diversity components and carbon (C) stocks in the subtropical forests of eastern China, including general plants composition and soil (20 cm depth) properties.

MF: mature forest; PF: pre-mature forest; YF: young forest; Max. H: maximum height; Max. DBH: maximum diameter at breast height; S: species richness; pH: soil pH; BD: soil bulk density; AGB: aboveground biomass; ACS: aboveground C stocks; SOC: soil organic C stocks and TEC: total ecosystem C stocks.

Plot	Forest type	Max. H (m)	Max. DBH (cm)	S	pH	BD (g cm^{-3})	AGB (Mg ha^{-1})	ACS (Mg ha^{-1})	SOC (Mg ha^{-1})	TEC (Mg ha^{-1})
1	MF	23.70	57.00	45	4.22	1.01	339.38	169.69	94.94	264.63
2	MF	24.10	50.00	46	4.19	1.06	386.84	193.42	88.08	281.50
3	MF	23.50	57.00	39	4.54	0.91	477.81	238.91	68.81	307.72
4	PF	21.90	30.00	25	4.04	1.31	132.04	66.02	87.42	153.44
5	PF	24.40	25.00	26	4.15	1.25	154.79	77.40	78.60	156.00
6	PF	24.10	31.00	46	4.32	1.21	166.66	83.33	85.77	169.10
7	YF	5.70	12.00	20	4.13	1.30	34.78	17.39	62.73	80.12
8	YF	12.00	23.50	31	4.60	1.28	32.99	16.50	49.56	66.06
9	YF	12.00	23.50	31	4.56	1.32	33.01	16.50	55.03	71.53
10	YF	8.90	22.30	31	5.78	1.34	105.76	52.88	120.73	173.61
11	YF	12.00	16.00	8	6.07	1.45	27.40	13.70	117.60	131.30
12	YF	17.00	18.00	13	5.80	1.40	50.96	25.48	158.81	184.29
13	MF	17.60	46.50	33	4.93	1.19	227.79	113.89	89.31	203.20
14	MF	16.30	35.00	30	5.01	1.28	187.82	93.91	125.11	219.02

15	MF	21.00	39.00	30	5.05	1.19	187.05	93.52	105.18	198.70
16	YF	18.50	41.00	22	5.14	1.29	175.22	87.61	101.00	188.61
17	MF	30.90	46.00	39	6.68	1.17	279.46	139.73	97.17	236.90
18	MF	26.80	58.00	26	5.86	1.19	304.57	152.28	97.14	249.42
19	YF	18.90	37.00	37	5.52	1.14	192.38	96.19	107.78	203.97
20	YF	16.40	40.00	30	6.12	1.25	166.30	83.15	79.05	162.20
21	YF	14.10	23.00	39	5.67	1.31	24.22	12.11	107.71	119.82
22	YF	25.70	50.00	38	6.09	1.23	227.42	113.71	71.78	185.49
23	PF	13.30	33.00	26	5.98	1.27	124.34	62.17	140.73	202.90
24	PF	16.60	31.00	31	5.89	1.28	195.33	97.67	83.30	180.97
25	YF	5.30	13.00	34	5.69	1.38	46.43	23.22	93.86	117.08
26	PF	15.10	20.00	21	6.74	1.23	121.54	60.77	98.55	159.32
27	YF	11.00	23.00	28	5.69	1.34	67.33	33.66	92.32	125.98
28	YF	16.00	31.50	18	6.75	1.35	45.43	22.72	145.68	168.40
29	YF	18.00	10.00	14	6.68	1.32	8.64	4.32	120.41	124.73
30	YF	10.00	20.00	9	6.53	1.38	113.95	56.98	87.13	144.11
31	YF	7.00	10.00	12	7.73	1.38	6.30	3.15	88.00	91.15

Appendix IV List of dominant species used to calculate for functional diversity components. Species are listed in descending order of abundance (% cover, see main text) across each type of forest ecosystem community.

Young forest (YF) species name	Mean % cover± SD	Pre-mature forest (PF) species name	Mean % cover± SD	Mature forest (MF) species name	Mean % cover± SD
Rhododendron ovatum (Lindl.) Planch. ex Maxim.	26.05±0.49	*Liquidambar formosana*	27.65	*Cornus japonicavar* chinensis	30.77
Symplocos paniculata (Thunb.) Miq.	25.46±17.87	*Castanopsis carlesii* (Hemsl.) Hay.	17.68±21.59	*Quercus fabri* Hance	23.61
Cinnamomum camphora	21.74	*Neolitsea aurata* var. chekiangensis (Nakai)	14.12	*Symplocos paniculata* (Thunb.) Miq.	17.31
Pinus massoniana Lamb.	18.98±13.41	*Rhododendron ovatum* (Lindl.) Planch. ex Maxim.	13.9	*Litsea elongata* (Wall. ex Nees) Benth. et Hook. f.	15.02±13.8
Diospyros glaucifolia	17.95	*Camellia fraterna* Hance	13.4±4.25	*Camellia fraterna* Hance	14.71±4.68
Liquidambar formosana	16.95±12.3	*Cyclobalanopsis glauca* (Thunb.) Oerst.	12.19±0.20	*Liquidambar formosana*	14.46±8.86
Loropetalum chinense (R. Br.) Oliver	16.64±7.47	*Eurya rubiginosa* var. attenuata H. T. Chang	10.88±2.01	*Ilex micrococca* Maxim	13.46
Castanopsis fargesii Franch.	16.38	*Cyclobalanopsis sessilifolia*	10.84	*Vernicia fordii* (Hemsl.) Airy Shaw	13.21
Choerospondias axillaris (Roxb.) Burtt et Hill.	15.38	*Schima superba* Gardn. et Champ.	10.27±5.29	*Illicium lanceolatum* A.C.Smith	13.13
Lithocarpus glaber	11.62±2.98	*Symplocos sumuntia* Buch.-Ham. ex D. Don	9.66±6.13	*Cyclobalanopsis gilva* (Blume) Oerst.	11.65±7.86
Cunninghamia lanceolata (Lamb.) Hook.	11.32	*Castanopsis fargesii* Franch.	9.55±4.09	*Pinus massoniana* Lamb.	9.67±0.08
Eurya rubiginosa var. attenuata H. T. Chang	11.03±4.09	*Cyclobalanopsis glauca*	8.96±0.89	*Schima superba* Gardn. et. Champ.	9.60±6.37
Pleioblastus amarus	10.43±0.02	*Lithocarpus glaber*	7.21±2.84	*Machilus leptophylla*	8.84
Mallotus japonicus var. floccosus S. M. Hwang	10.26	*Eurya loquaiana* Dunn	6.39±5.11	*Cyclobalanopsis sessilifolia*	8.55
Castanopsis carlesii (Hemsl.) Hay.	10.16±10.80	*Ficus erecta* var. beecheyana (Hook. et Arn.) King	6.02	*Neolitsea aurata* var. chekiangensis (Nakai)	8.31±8.09
Ilex chinensis Sims	9.69	*Vernicia fordii* (Hemsl.) Airy Shaw	5.97±1.89	*Castanopsis fargesii* Franch.	6.94±2.71
Cornus.japonica var. chinensis	9.52	*Ilex chinensis* Sims	5.88	*Machilus thunbergii* Siebold & Zuccarini	6.91±2.45
Schima superba Gardn. et Champ.	9.09±4.02	*Eurya muricata* Dunn	5.72±1.07	*Eurya loquaiana* Dunn	6.41±4.95
Rhus chinensis Mill.	8.81±5.67	*Clerodendrum cyrtophyllum* Turcz.	5.48	*Lithocarpus glaber*	6.15
Camellia fraterna Hance	8.8	*Damnacanthus indicus* Gaertn. f.	4.82±1.0	*Symplocos oblanceolata* Y. F. Wu	6.01±2.09
Cyclobalanopsis gilva (Blume) Oerst.	8.4	*Rhus chinensis* Mill.	4.81	*Photinia glabra* (Thunb.) Maxim.	5.96
Castanopsis sclerophylla (Lindl.) Schott.	7.41±2.05	*Symplocos stellaris* Brand	4.73±1.45	*Symplocos glauca* (Thunb.) Koidz.	5.7

Quercus acutissima Carruth.	7.28±3.17	*Castanopsis sclerophylla* (Lindl.) Schott.	4.49	*Sassafras tzumu* (Hemsl.) Hemsl.	5.66
Symplocos sumuntia Buch.-Ham. ex D. Don	7.02±1.92	*Cyclobalanopsis gilva* (Blume) Oerst.	4.46±0.50	*Symplocos stellaris* Brand	5.63
Rhododendron simsii	6.98	*Machilus thunbergii* Siebold & Zuccarini	4.22	*Lithocarpus henry*	5.39±0.66
Symplocos setchuensis Brand	6.98	*Syzygium buxifolium*	4.15	*Cyclobalanopsis glauca* (Thunb.) Oerst.	5.29±2.14
Albizzia kalkora	6.15±1.97	*Quercus myrsinifloia*	4.12	*Eurya rubiginosa* var. attenuata H. T. Chang	4.98±1.59
Syzygium buxifolium	5.89±2.82	*Ternstroemia gymnanthera* (Wightet Arn.)Sprague	4.12	*Carpinus turczaninowii* Hance	4.91±0.30
Celtis sinensis Pers.	5.23	*Symplocos lancifolia* Sieb. et Zucc.	3.95	*Cyclobalanopsis glauca*	4.75±3.24
Vaccinium bracteatum	4.27±1.28	*Symplocos oblanceolata* Y. F. Wu	3.39±0.45	*Rhododendron ovatum* (Lindl.) Planch. ex Maxim.	4.72±3.17
Gardenia jasminoides Ellis	4.2	*Loropetalum chinense* (R. Br.) Oliver	3.01	*Phoebe sheareri* (Hemsl.) Gamble	4.62
Cyclobalanopsis glauca	3.95	*Daphniphyllum macropodum* Miq.	2.94	*Symplocos sumuntia* Buch.-Ham. ex D. Don	4.51±1.63
Fraxinus insularis Hemsl.	3.36	*Neolitsea sericea* (Bl.) Koidz.	2.74	*Cyclobalanopsis stewardiana* (A. Camus)	4.47±0.17
Machilus thunbergii Siebold & Zuccarini	3.36	*Adinandra millettii* (Hook. et Arn.)	2.36±0.07	*Daphniphyllum macropodum* Miq.	4.38±1.22
Myrica rubra (Lour.) S. et Zucc.	3.36	*Fraxinus insularis* Hemsl.	2.28	*Alniphyllum fortunei*	4.36±3.18
Eurya muricata Dunn	3.1±0.82			*Castanopsis carlesii* (Hemsl.) Hay.	4.36±1.62
				Ilex chinensis Sims	4.18±3.38
				Albizzia kalkora	4.17
				Ternstroemia gymnanthera (Wightet Arn.)	4.13
				Clerodendrum cyrtophyllum Turcz.	4.09±2.27
				Toxicodendron vernicifluum	4.08
				Premna microphylla Turcz.	4.03
				Aralia chinensis Linn.	3.77
				Choerospondias axillaris (Roxb.) Burtt et Hill.	3.65±1.36
				Dalbergia hupeana Hance	3.43±1.05
				Diplospora dubia (Lindl.) Masam.	3.36±2.08
				Cinnamomum camphora	3.36±1.77
				Cinnamomum japonicum Sieb.	3.25±2.10
				Syzygium buxifolium	3.21

Fraxinus insularis Hemsl.	3.08
Ardisia crenata	3.02
Helicia cochinchinensis	3.02
Styrax japonicus Sieb. et Zucc.	2.82
Petiolus Trachycarpi Fortunei	2.68
Acer olivaceum Fang et P. L. Chiu	2.67±0.57
Elacocarpus japonicus	2.36±0.76
Lasianthus japonicus var. lancilimbus (Merr.) Lo	2.26
Indocalamus tessellatus (Munro) Keng f.	2.11
Rhus chinensis Mill.	2.11
Castanopsis sclerophylla (Lindl.) Schott.	1.75
Eurya nitida	1.75

* SD: standard deviations. Values without SD means specie found in only one studied plot.

Appendix V Summary of variables used in the analysis for the quantification of functional diversity (FD)-C stocks association in the subtropical forests of eastern China. Community weighted mean (CWM) and single-trait functional divergence (FDvar) are predictors while C (ACS, SOC and TEC) stocks are the dependent/response variables. See Table S1 for C stocks in each studied plot.

FDvar values are unitless while CWM values have the units as; H: plant height (m), LDMC-M: mature leaf dry matter content (mg g^{-1}), SLA-M: mature leaf specific leaf area (mm^2 mg^{-1}), LNC-M: mature leaf nitrogen concentration (mg g^{-1}), LPC-M: mature leaf phosphorus concentration (mg g^{-1}), LDMC-F: current year leaf dry matter content (mg g^{-1}), SLA-F: current year leaf specific leaf area (mm^2 mg^{-1}), LNC-F: current year leaf nitrogen concentration (mg g^{-1}), LPC-F: current year leaf phosphorus concentration (mg g^{-1}), TDMC (mg g^{-1}): twig dry matter content (mg g^{-1}), TWD: twig wood density (g cm^{-3}), SWD: stem wood density (g cm^{-3}), SNC: stem nitrogen concentration (mg g^{-1}) and SPC: stem nitrogen concentration (mg g^{-1}).

Plot	FD component	H	LDMC.M	SLA.M	LNC.M	LPC.M	LDMC.F	SLA.F	LNC.F	LPC.F	TDMC	TWD	SWD	SNC	SPC
1	CWM	3.2	409.2	11.88	17	0.58	387.4	13.3	17.2	0.85	364.4	0.17	0.49	2.31	0.42
1	FDvar	0.54	0.05	0.08	0.08	0.31	0.05	0.07	0.16	0.85	0.11	0.93	0.05	0.94	0.96
2	CWM	2.8	400.9	12.01	17.9	0.53	380.9	16.1	15.5	0.95	361.6	0.33	0.51	1.3	0.12
2	FDvar	0.74	0.05	0.15	0.07	0.24	0.06	0.4	0.24	0.89	0.07	0.64	0.35	0.83	0.93
3	CWM	4.79	395.1	11.81	15.7	0.49	376.4	13.4	14.4	0.92	408.9	0.54	0.53	2.68	0.15
3	FDvar	0.71	0.04	0.18	0.24	0.78	0.08	0.19	0.54	0.94	0.09	0.69	0.09	0.88	0.89
4	CWM	3.94	376.1	11.44	15.5	0.71	341.6	13.7	14.5	0.55	381.2	0.33	0.57	2.14	0.07
4	FDvar	0.87	0.03	0.09	0.04	0.15	0.03	0.04	0.08	0.8	0.12	0.08	0.06	0.82	0.82
5	CWM	3.84	357.9	10.79	15.6	0.88	375.4	13	14.4	0.5	468.1	0.39	0.59	3.23	0.19
5	FDvar	0.77	0.07	0.05	0.11	0.26	0.04	0.04	0.08	0.73	0.16	0.1	0.005	0.62	0.85
6	CWM	3.28	386.2	11.91	14.3	0.79	374.5	12.7	15.3	0.51	422.1	0.33	0.6	3.57	0.12

6	FDvar	0.71	0.06	0.16	0.06	0.84	0.12	0.15	0.13	0.15	0.09	0.12	0.09	0.87	0.85
7	CWM	2.64	410.9	11.74	16.2	0.37	406.8	11.8	15.5	0.36	422.2	0.23	0.63	2.7	0.33
7	FDvar	0.33	0.07	0.41	0.15	0.94	0.04	0.35	0.04	0.84	0.0046	0.18	0.08	0.73	0.93
8	CWM	2.96	428.1	11.21	18.4	0.1	456.4	11	18.6	0.24	427.5	0.29	0.62	3.15	0.16
8	FDvar	0.22	0.03	0.29	0.3	0.81	0.06	0.19	0.37	0.77	0.01	0.07	0.27	0.77	0.71
9	CWM	2.97	432	10.07	20.7	0.69	419.7	10.8	17.7	0.33	423.5	0.36	0.61	5.09	0.54
9	FDvar	0.21	0.03	0.28	0.37	0.52	0.01	0.3	0.38	0.84	0.05	0.19	0.13	0.87	0.88
10	CWM	3.03	431.9	11.08	17.7	0.63	431.1	10.3	16.6	0.48	429.6	0.38	0.58	4.22	0.15
10	FDvar	0.41	0.02	0.38	0.07	0.5	0.02	0.27	0.06	0.3	0.03	0.39	0.04	0.72	0.82
11	CWM	3.2	428.5	15.98	17.9	0.69	504	12.2	17.4	0.34	411.4	0.33	0.5	3.47	0.19
11	FDvar	0.48	0.06	0.4	0.07	0.33	0.2	0.05	0.16	0.11	0.11	0.18	0.05	0.42	0.92
12	CWM	2.93	389.6	13.85	19.1	0.56	444.6	10.8	19.6	0.57	412.3	0.37	0.57	2.89	0.13
12	FDvar	0.89	0.09	0.33	0.06	0.83	0.12	0.17	0.28	0.5	0.04	0.09	0.02	0.42	0.8
13	CWM	3.95	469.5	9.6	18.1	0.58	462.4	9.24	16.6	0.47	498.9	0.37	0.63	3.85	0.22
13	FDvar	0.69	0.04	0.09	0.09	0.25	0.03	0.12	0.1	0.47	0.11	0.1	0.06	0.21	0.88
14	CWM	4.09	445.5	10.68	17.5	0.49	446.1	11.1	17.2	0.48	428.7	0.36	0.55	3.02	0.43
14	FDvar	0.49	0.03	0.13	0.1	0.82	0.03	0.2	0.11	0.39	0.07	0.48	0.04	0.57	0.53
15	CWM	4.09	392	15.16	15.8	0.51	396	9.72	18.2	0.54	397.8	2.32	0.53	3.54	0.33
15	FDvar	0.63	0.08	0.42	0.1	0.55	0.09	0.63	0.04	0.13	0.08	0.93	0.03	0.62	0.22
16	CWM	5.65	420.8	12.9	17.2	0.44	413	9.38	17.8	0.52	417	2.42	0.52	3.29	0.21
16	FDvar	0.4	0.07	0.5	0.08	0.77	0.07	0.18	0.07	0.12	0.05	0.92	0.04	0.31	0.53

17	CWM	2.57	386.5	15.94	18.6	0.52	369	16.5	17.8	0.45	387.3	0.31	0.54	3.15	0.69
17	FDvar	0.44	0.07	0.44	0.11	0.09	0.1	0.32	0.13	0.17	0.06	0.13	0.11	0.24	0.85
18	CWM	4.07	400.4	14.53	18.1	0.79	376.8	16.3	18.9	0.46	377.3	1.83	0.6	3.6	0.45
18	FDvar	0.68	0.06	0.21	0.15	0.17	0.08	0.25	0.24	0.89	0.06	0.84	0.16	0.23	0.66
19	CWM	3.43	398.2	10.71	15.7	0.42	388.1	10.9	16.7	0.4	416.7	0.52	0.81	3.98	0.66
19	FDvar	0.43	0.05	0.09	0.13	0.48	0.06	0.06	0.16	0.38	0.12	0.66	0.08	0.26	0.52
20	CWM	4.98	392	12.03	17.4	0.42	405.6	13.2	19.3	1.36	415.1	0.34	0.64	3.66	0.26
20	FDvar	0.67	0.15	0.48	0.18	0.24	0.15	0.54	0.2	0.83	0.18	0.36	0.23	0.69	0.2
21	CWM	2.2	426.4	11.75	17.2	0.46	418.2	12	19.7	0.66	417	0.36	0.58	3.84	0.28
21	FDvar	0.46	0.05	0.18	0.06	0.28	0.89	0.19	0.08	0.4	0.06	0.54	0.07	0.62	0.87
22	CWM	4.53	428	15.15	19.4	0.45	227.5	14.5	19.6	0.69	405.6	0.33	0.56	2.06	0.17
22	FDvar	0.56	0.1	0.4	0.13	0.28	0.98	0.23	0.22	0.67	0.12	0.8	0.05	0.91	0.95
23	CWM	5.37	408.7	11.73	18.6	0.5	398.9	12.8	17.2	0.58	373.5	0.34	0.51	3.79	0.27
23	FDvar	0.82	0.05	0.05	0.05	0.07	0.05	0.13	0.12	0.68	0.05	0.28	0.11	0.45	0.96
24	CWM	3.89	415.1	11.07	18.7	0.53	416	11.4	17.4	0.74	415.1	0.39	0.57	5.04	0.51
24	FDvar	0.77	0.04	0.14	0.06	0.53	0.04	0.16	0.13	0.31	0.08	0.1	0.04	0.15	0.79
25	CWM	2.46	418.3	13.7	18.1	0.68	400.6	14.3	19.1	0.62	409.5	0.35	0.54	4.22	0.28
25	FDvar	0.16	0.06	0.27	0.11	0.45	0.06	0.21	0.18	0.51	0.04	0.09	0.06	0.57	0.66
26	CWM	3.13	424.6	12.03	18.7	0.66	398.5	12.7	18	0.39	404	0.31	0.54	3.65	0.49
26	FDvar	0.78	0.03	0.21	0.06	0.6	0.91	0.22	0.19	0.52	0.03	0.14	0.06	0.72	0.87
27	CWM	2.42	433	11.53	16.3	1.68	430.1	11.2	15.8	0.58	425.3	0.31	0.56	3.47	0.11

27	FDvar	0.2	0.01	0.12	0.01	0.81	0.01	0.06	0.12	0.8	0.03	0.52	0.03	0.09	0.94
28	CWM	5.02	448.6	11.29	20.7	1.05	447.9	10.4	20.6	0.48	452.3	0.41	0.49	3.61	0.27
28	FDvar	0.85	0.07	0.29	0.25	0.95	0.09	0.28	0.34	0.95	0.09	0.08	0.09	0.66	0.92
29	CWM	2.3	356.7	14.6	19.7	0.57	383.7	11.8	20.3	0.43	406.7	0.27	0.58	2.49	0.25
29	FDvar	0.43	0.07	0.07	0.19	0.27	0.05	0.05	0.17	0.61	0.05	0.73	0.03	0.35	0.92
30	CWM	3.51	357.8	17.71	24.1	0.92	334.6	17.7	21.7	1.02	377.6	0.29	0.51	4.01	0.37
30	FDvar	0.82	0.06	0.22	0.22	0.25	0.04	0.13	0.08	0.47	0.02	0.03	0.06	0.74	0.31
31	CWM	2.35	412.3	12.6	21.3	0.56	385	14.9	21.9	0.63	368.5	0.5	0.51	2.85	1.09
31	FDvar	0.25	0.05	0.12	0.11	0.41	0.03	0.24	0.06	0.38	0.04	0.67	0.31	0.8	0.94

Appendix VI Pair wise associations between the ecosystem C stocks and biodiversity components indices of the subtropical evergreen-broadleaved forests of eastern China. Values indicate the regression coefficients (R^2) from simple linear regression analysis, indicated with asterisks if statistically significant (*: P < 0.05; **: P < 0.01; ns: not significant). Relationships showing a negative slope are indicated with negative signs.

	Community weighted mean (CWM)			***Functional divergence (FDvar)***		
	ACS	SOC	TEC	ACS	SOC	TEC
Height (H)	**0.14***	0.01 ns	**0.18***	**0.13***	0.11 ns	**0.25****
Mature leaf dry matter content (LDMC-M)	-0.01ns	0.003 ns	-0.01 ns	0.0008 ns	0.01 ns	0.004 ns
Mature leaf specific leaf area (SLA-M)	-0.0002 ns	0.02 ns	0.0025 ns	-0.01 ns	-0.003 ns	0.02 ns
Mature leaf nitrogen concentration (LNC-M)	-0.1 ns	0.01 ns	-0.07 ns	-0.01 ns	**-0.16***	-0.07 ns
Mature leaf phosphorus concentration (LPC-M)	-0.02 ns	0.03 ns	-0.01 ns	-0.04 ns	0.0004 ns	-0.04 ns
Current year leaf dry matter content (LDMC-F)	-0.12 ns	0.10 ns	-0.04 ns	-0.003 ns	-0.00055 ns	-0.004 ns
Current year leaf specific leaf area (SLA-F)	0.10 ns	-0.05 ns	0.05 ns	0.02 ns	-0.02 ns	0.01 ns
Current year leaf nitrogen concentration (LNC-F)	**-0.18***	0.07 ns	-0.10 ns	0.07 ns	-0.03 ns	0.04 ns
Current year leaf phosphorus content (LPC-F)	**0.18***	-0.01 ns	**0.14***	0.03 ns	-0.07 ns	0.0035 ns
Twig dry matter content (TDMC)	-0.05 ns	0.001 ns	-0.06 ns	**0.14***	-0.003 ns	0.12 ns
Twig wood density (TWD)	0.04 ns	0.01 ns	0.05 ns	**0.17***	0.0008 ns	**0.17***
Stem wood density (SWD)	-0.01 ns	-0.06 ns	-0.03 ns	0.01 ns	-0.11 ns	-0.0007 ns
Stem nitrogen content (SNC)	-0.11 ns	0.0006 ns	-0.11 ns	0.01 ns	**-0.13***	-0.01 ns
Stem phosphorus content (SPC)	-0.0045 ns	-0.004 ns	-0.01 ns	-0.003 ns	0.0004 ns	-0.002 ns

Appendix VII Snapshots

Figure VII.1 An overview of subtropical evergreen broad leaved forests in eastern China

Figure VII.2 An overview of stand structure of subtropical evergreen broad leaved forests in eastern China

Figure VII.3 Author and other research fellows during wood samples collection

Figure VII.4 Author during leaf samples collection

Figure VII.5 Author and other research fellows during soil samples collection

Figure VII.6 Author conducting lab experiment at East China Normal University

Biography

Arshad Ali was born on 30th March, 1988 at Khyber Pakhtunkhwa province of Pakistan. During 2006-10, he successfully completed BS (Forestry) 4–years degree with the achievement of Gold Medal from Department of Forestry, Shaheed Benazir Bhutto University, KPK, Pakistan. After that, he worked for two years in different non-governmental organizations (NGOs) under various social and environmental projects. Then he got China Government Scholarship (2012-14) for conducting his Master's degree in Ecology at East China Normal University, Shanghai, PR China. Where his specialized research area was Forest Ecology mainly focused on forest biodiversity and ecosystem carbon stocks.