Pak. J. Bot., 47(5): 1855-1864, 2015.

IMPACTS OF CWD ON UNDERSTORY BIODIVERSITY IN FOREST ECOSYSTEMS IN THE QINLING MOUNTAINS, CHINA

JIE YUAN[1], XIN WEI[1], ZHENGCHUN SHANG[1], FEI CHENG[1], ZHAOYONG HU[1], XIAOFENG ZHENG[1] AND SHUOXIN ZHANG[1,2*]

[1]*College of Forestry, Northwest A&F University, Yangling, Shaanxi 712100, China*
[2]*Qinling National Forest Ecosystem Research Station, Huoditang, Ningshan, Shaanxi 711600, China*
**Corresponding author e-mail: sxzhang@nwsuaf.edu.cn; Tel: +86-29-87082993; Fax: +86-29-87082993*

Abstract

The stocks and characteristics of coarse woody debris (CWD) are expected to reflect forest stand features. However, despite their importance, there have been no reports of CWD stocks and characteristics in the Qinling Mountains. We measured the CWD stocks in different CWD types, decay classes and diameter classes of the five forest types in the Qinling Mountains. The highest biomass of CWD was the *Pinus tabulaeformis* forest (12.57 t·hm^{-2}), occupied 5.66% in the biomass of this forest, the lowest occupied 1.03% in *Betula albo-sinensis* forest (1.82 t·hm^{-2}). Our results revealed that there was a strong correlation between CWD and forest biomass. When the CWD biomass were 9.9 t·hm^{-2} and 11.6 t·hm^{-2}, the biomass of *Pinus armandi* forest and *P. tabulaeformis* forest reached maximum, respectively.

CWD is particularly important for biodiversity, but the importance of CWD in the control of diversity in forest systems has not been fully appreciated and certainly has not been evaluated intensively within China, especially in Qinling forests. In our research, we used species richness (S), Shannon-Wiener index (H), Simpson index (D) and Pielou's evenness index (J) to assess the diversity of plant community. According to our analysis, we found 1) the effect of CWD biomass onthese α diversity index was dependent on tree, shrub and herb in the five forest types, 2) the impacts of CWD biomass on understory biodiversity were more obvious, 3) With the increase of CWD biomass, the species richness (S), Shannon-Wiener index (H) and Simpson index (D) of understory increased significantly.

Our results suggested that there was a relatively lower CWD biomass in the Qinling Mountains, but it had significant effects on forest biomass and diversity of plant community. Reserving CWD was important for eco-forestry, but how many and how characteristic of CWD should be retained need further research. Development of CWD reasonable strategies was indispensable for future forest management.

Key words: Coarse woody debris; Biomass; Qinling Mountains; Biodiversity; Diversity index; Forest management.

Introduction

In forest ecosystems, CWD contributes significantly to key ecological processes. It plays an essential role in productivity (Stevens, 1997; Janisch & Harmon, 2002), nutrient cycling (Currie & Nadelhoffer, 2002; Ganjegunte *et al.*, 2004; Wilcke *et al*., 2005), carbon sequestration (Harmon *et al*., 1986; Gough *et al.*, 2007; Woodall & Liknes, 2008), community regeneration (Harmon & Franklin, 1989; Santiago, 2000), biodiversity (Harmon *et al*., 1986; Mac Nally *et al*., 2001; Stevenson *et al*., 2006), and geomorphological stability (Stevens, 1997). CWD is particularly important for fungal biodiversity (Bader *et al.*, 1995; Renvall, 1995; Bredesen *et al.*, 1997; Sippola & Renvall, 1999), and similar claims have been made with respect to bryophytes (Söderström, 1988; Andersson & Hytteborn, 1991) and saproxylic insects (Grove & Meggs, 2003; Similä *et al*., 2003).

Important insights into CWD dynamics have been gained by linking stand assessments to CWD characteristics (*i.e*., amount, type, diameter class and decay class), and which are a reflection of historical and present conditions (Hansen *et al*., 1991; Hardt & Swank, 1997; Sturtevant *et al*., 1997; Siitonen *et al*., 2000; Yan *et al*., 2007). In this respect, differences in CWD characteristics can be associated with stand age (Spies *et al*., 1988; Sturtevant *et al*., 1997; Yan *et al*., 2007), structure (Sturtevant *et al*., 1997; Motta *et al*., 2006; Schlegel & Donoso, 2008), productivity (Spetich *et al*., 1999; Brais *et al*., 2005), natural disturbances and management history (Hansen *et al*., 1991; Hardt & Swank, 1997; Eaton & Lawrence, 2006; Motta *et al.*, 2006; Shinwari, 2001; Passovoy & Fulé, 2006). Given that the absolute amounts of CWD vary along productivity gradients (Sippola *et al*., 1998), it is important to relate information on CWD to trees within the same forest type (Krankina & Harmon, 1995; Nilsson *et al*., 2002).

To develop forest management strategies that mimic natural processes and structures, baseline information on CWD from natural forests is needed (Kuuluvainen, 2002). The Qinling Mountains in central China provide an important climate boundary between the southern subtropics and the north temperate zone (Yu *et al*., 2013a). The region is distinguished by its high plant and animal diversity (Kang & Chen, 1996), including the last remaining natural habitat of the endangered Giant Panda (*Ailuropoda melanoleuca*) and Japanese Crested Ibis (*Nipponia nippon*). Dominant tree species include *Pinus tabulaeformis*, *Quercus aliena* var. *acuteserrata*, *Pinus armandi*, *Betula albo-sinensis* and *Larix principis-rupprechtii* (Lei *et al*., 1996; Yu *et al*., 2013b).

Little quantitative information has been reported on CWD characteristics in the Qinling Mountains, despite its ecological significance. The objectives of our study were to (1) quantify the types, decay stages and diameter classes of CWD in five forest types, (2) calculate their live biomass and CWD, and (3) assess diversity in these plant communities, and its relationship to CWD.

Materials and Methods

Study area: Taking an area of 2037 hm^2, the study area is located on the Huoditang Experimental Forest Farm of Northwest A&F University in the Qinling Mountains (33°18'–33°28'N, 108°21'–108°39'E), Shaanxi Province, China (Fig. 1), the altitudes ranging from 800 to 2500m. The climate is characteristic of warm temperate zone, with a mean annual temperature range of 8–10°C, annual precipitation ranging from 900–1200mm, and a frost-free period of 170 days.

The study area had intensively selective logging in 60s~70s of the 20th century, since then there were no massive anthropogenic interferences to happen, except for slightly tapping lacquer trees and illegal tree felling. Since the natural forest protection project was initiated 1998, the human activities or special managements have almost vanished in this region. The forests used for the current research was an average of 55-years old, mean stand height, diameter at breast height (DBH) and stand density are 16 m, 22 cm and 1585 trees·ha^{-1} respectively. Forest types are dominated by one or more tree species, including *P. tabulaeformis*, *Q. aliena*var. *acuteserrata*, *P. armandi*, *B. albo-sinensis* and *L. principis-rupprechtii*. An abrupt and broken topography consists of granite and gneiss parent material. The average slope is 35° and a mean soil depth of 45 cm. The soil is classified as mountain brown earth.

Fig. 1. Study location in Shaanxi Province, China.

Experimental design and field sampling: Five stands were selected with a minimum area of 60 m × 60 m (*P. tabulaeformis*, *P. armandi*, and *B. albo-sinensis*) or 40 m × 90 m (*Q. aliena* var. *acuteserrata* and *L. principis-rupprechtii*), site conditions were similar in each of the five forest types (Table 1). In July of 2012, each stand was divided into 6 sample plots with an area of 20 m × 30 m.

We used the USDA Forest Service and Long Term Ecological Research (LTER) definition of CWD (diameter ≥ 10 cm at the widest point). CWD was categorized in each plot by species, and assigned as logs, snags, or stumps, as follows (Harmon *et al*., 1996; Ringvall & Ståhl, 1999). Downed or leaning deadwood (> 45°from vertical) with a minimum diameter ≥ 10 cm at the widest point and length ≥ 1 m, was defined as a log. Deadwood ≤ 45°from vertical and the diameter at the widest point ≥ 10 cm was defined as a snag, and deadwood ≤ 45°from vertical with a height ≤ 1 m and diameter ≥ 10 cm at the widest point, defined as a stump. Each piece of CWD was assigned to one of five decay classes, as per Yan *et al*. (2007). In addition to CWD, living trees were recorded in each plot by species, DBH and abundance.

We set five 2 m × 2 m shrub, and 1 m × 1 m nested herbal subplots in the four corners and the middle of the each sample plot. Species, height, abundance and cover percentage were recorded in the subplots. This work was conducted based on Forestry Standards "Observation Methodology for Long-term Forest Ecosystem Research" of People's Republic of China (Wang *et al*., 2011).

Calculation of biomass and species diversity: A total of 225 CWD samples were collected. When sufficient sound wood was present, the CWD was cut by handsaw into disks, roughly 5 cm thickness. For the more advanced decay classes, the CWD samples were simply transplanted onto aluminum plates. Samples were immediately sealed in plastic bags, transported to the laboratory, and the sample volume (V_{sample}) determined gravimetrically by water displacement. The CWD samples were then dried to a constant weight at 70°C. The sample density (D_{sample}) was estimated as the ratio of dry mass to V_{sample}.

Prior to calculating CWD biomass, Smalian's formula was used to calculate the volume for each sample of logs and stumps, from the length and cross-sectional areas at the basal and distal ends of an assumed cylinder (Wenger, 1984). It should be noted that this formula tends to slightly overestimate volume due to the natural taper of the material (Baker *et al.*, 2007). For snags, we inserted the height and diameter of each relevant sample into a species-specific wood volume equation. Finally, the product of D_{sample} and the calculated sample volume was computed ($t \cdot hm^{-2}$).

The biomass of living trees was calculated using a series of regression models developed for each of the tree species (Table 2) (Chen & Peng, 1995). Aboveground biomass of shrub, herb and litter was quantified by a harvesting method, and underground biomass of shrub and herb was quantified by full-dig method.

Table 1. Plot characteristics of the investigated forest.

Forest types	Altitude (m)	Slope	Aspect	Age of stand (years)	Tree species composition	Canopy density	DBH (cm)	Height (m)	Stand density (trees·hm^{-2})
P. tabulaeformis	1585	35	231	55	1,2,3,5,9	0.70	25	20	1548
P. armandi	1651	27	214	60	2,4,7,12	0.65	28	18	1404
Q. aliena var. *acuteserrata*	1604	38	184	65	1,3,6,7,8,12	0.75	23	14	1831
L. principis-rupprechtii	2051	18	84	40	1,2,6,10	0.58	15	17	1681
B. albo-sinensis	2145	14	19	50	1,6,11	0.56	22	10	1058

1. *P. armandi*; 2. *Q. aliena* var. *acuteserrata*; 3. *Toxicodendron vernicifluum*; 4. *Tilia chinensis*; 5. *Carpinus cordata*; 6. *Acer davidii*; 7. *Cornus macrophylla*; 8.*Litsea pungens*; 9. *L. principis-rupprechtii*; 10. *B. albo-sinensis*; 11. *Salix hypoleuca*; 12. *P. tabulaeformis*

Table 2. The regression model of biomass and height in the five forest types.

Forest types	Contents	Regression equation	Correlation coefficient	Reliability of 95 % of the estimated accuracy
Q. aliena var. *acuteserrata*	Stem	$\ln W_S=0.99253\ln(D^2H)-3.78818$	0.99763	94.24
	Bark	$\ln W_{BA}=0.75632\ln(D^2H)-3.92450$	0.99708	95.37
	Branch	$\ln W_B=3.4993\ln D-6.50726$	0.96524	84.27
	Leaf	$\ln W_L=2.29344\ln D-4.88581$	0.97832	84.45
	Root	$\ln W_R=2.76345\ln D-4.20817$	0.99106	89.15
	Height	$1/H=8.01921/D^{2.59222}+0.05263$	0.78814	95.60
P. armandi	Stem	$\ln W_S=1.02363\ln(D^2H)-4.49970$	0.99802	97.09
	Bark	$\ln W_{BA}=0.88417\ln(D^2H)-5.38472$	0.99698	96.73
	Branch	$\ln W_B=2.57551\ln D-4.08452$	0.98656	90.60
	Leaf	$\ln W_L=2.75687\ln D-5.75891$	0.98004	81.56
	Root	$\ln W_R=0.97120\ln(D^2H)-5.26301$	0.97927	92.13
	Height	$1/H=1.34537/D^{1.70800}+0.07143$	0.88076	98.52
P. tabulaeformis	Stem	$\ln W_S=1.04086\ln(D^2H)-4.63143$	0.99558	93.70
	Bark	$\ln W_{BA}=0.77396\ln(D^2H)-4.69348$	0.99037	93.12
	Branch	$\ln W_B=2.57733\ln D-4.08026$	0.99159	86.37
	Leaf	$\ln W_L=2.57495\ln D-5.11712$	0.98652	73.77
	Root	$\ln W_R=2.28692\ln D-4.14198$	0.98792	82.60
	Height	$1/H=0.82960/D^{1.40330}+0.07692$	0.92899	97.80
B. albo-sinensis	Stem	$\ln W_S=0.91035\ln(D^2H)-3.79326$	0.99721	88.77
	Bark	$\ln W_{BA}=0.81021\ln(D^2H)-4.27750$	0.99674	91.22
	Branch	$\ln W_B=3.35934\ln D-5.93511$	0.98584	84.64
	Leaf	$\ln W_L=2.39007\ln D-5.56930$	0.98709	85.19
	Root	$\ln W_R=2.68879\ln D-4.33607$	0.99292	88.04
	Height	$1/H=4.98842/D^{2.43072}+0.06061$	0.87234	93.51
L. principis-rupprechtii	Stem	$\ln W_S=0.99794\ln(D^2H)-4.29251$	0.99312	86.62
	Bark	$\ln W_{BA}=0.80398\ln(D^2H)-4.53535$	0.98872	83.76
	Branch	$\ln W_B=2.04597\ln D-2.55078$	0.97720	82.50
	Leaf	$\ln W_L=1.90488\ln D-3.44704$	0.97436	76.46
	Root	$\ln W_R=2.18625\ln D-3.46236$	0.98725	90.67
	Height	$1/H=1.90568/D^{1.90809}+0.06897$	0.86281	95.39

Note: D= Diameter at breast height; H= Height of tree; W_S= Dry weight of stem; W_{BA}= Dry weight of bark; W_B= Dry weight of branch; W_L= Dry weight of leaf; W_R= Dry weight of roots

Species richness (S), the Shannon-Wiener index (H), Simpson index (D) and Pielou's evenness index (J)were used to estimate plant community diversity, as follows:

S = total number of species,

$$H=-\sum_{i=1}^{s} P_i \log_2 P_i$$ (Magurran, 1988)

where $P_i=n_i/N$ where n_i=number of individuals of the *i*th species and *N*=total number of individuals

$$D=1-\sum_{i=1}^{s} P_i^2$$ (Berger & Parker, 1970)

where S and P_i are as defined above.

$$J=H/H_{\max}$$ (Hurlbert, 1971)

$H_{\max}$ is the maximal Shannon-Wiener diversity index.

Statistical analyses: The effect of CWD type, decay class, and diameter class, on CWD biomass was tested using a One-Way ANOVA. Significant effects were followed by a Least Squares Difference (LSD) test to compare differences between CWD types, decay classes, diameter classes and forest types, separately. The dependence of the CWD biomass on forest biomass was analyzed with Pearson's correlation coefficient (r). All statistics were analyzed using the SAS 8.0 Statistical Package, with a *P*-value of 0.05 set as the limit for statistical significance.

Results

CWD biomass: Logs were the largest CWD biomass component in all forest types, followed by the snags, then stumps. Log biomass was significantly higher in *P. tabulaeformis* forest than the other four forest types (Table 3), followed by *P. armandi, Q. alienavar.acuteserrata* and *L. principis-rupprechtii*. Although *B. albo-sinensis* forest had the lowest log biomass, it was not significantly lower than *L. principis-rupprechtii*. *P. armandi* forest had the highest snag biomass, followed by *Q. aliena* var. *acuteserrata* and *P. tabulaeformis* (the latter of which were not significantly different), then *L. principis-rupprechtii* and *B. albo-sinensis*.

P. tabulaeformis stand had the highest CWD biomass for decay classes 1, 4 and 5 ($p<0.05$; Table 4). For decay class 2, the highest biomass was shared between *P. tabulaeformis* and *P. armandi*. Both *P. armandi* and *Q. aliena* var. *acuteserrata* had the greatest CWD biomass for decay class 3. Visual inspection indicated no consistent trends in CWD abundance across decay classes for the five forest types. In the case of *B. albo-sinensis* stand, CWD was present for only decay classes 1 and 2 (Table 4).

In terms of CWD size, only *P. tabulaeformis* stand had representation in the largest diameter class (>40 cm), while in *B. albo-sinensis* stand all CWD was ≤ 30 cm diameter (Table 5). *L. principis-rupprechtii* stand had the highest CWD biomass in the smallest size class ($p<0.05$), *P. tabulaeformis*, *P. armandi* and *Q. aliena* var. *acuteserrata* in the next size class (20-30 cm), both *P. tabulaeformis* and *P. armandi* in the 30-40 cm size class. *B. albo-sinensis* stand had markedly lower amount of CWD (Table 5).

Relationship between biomass of CWD and forest: Forest biomass, including biomass of the living trees, shrub, herb, litter and CWD, across the five forest types had a significant variability ranged from 176.68 t·hm^{-2} (*B. albo-sinensis* forests) ~255.25 t·hm^{-2} (*Q. aliena* var. *acuteserrata* forests) (Table 6). For all sample plots, Pearson's correlation coefficient (r) was 0.50 (P=0.005, n=30) (Fig. 2). This result revealed that there was a strong correlation between biomass of CWD and forest, but the correlation was different at various forest types. Analyzed by forest types (n=6, in each group), the correlation was highly significant in *Q. aliena* var. *acuteserrata* forest (r=0.97, P=0.0013), *B. albo-sinensis* forest (r=0.95, P=0.0034) and *L. principis-rupprechtii* forest (r=0.96, P=0.0024), but not in *P. tabulaeformis* forest (r=-0.13, P=0.7991) or in *P. armandi* forest (r=-0.16, P=0.7648).

Table 3. Biomass of the three CWD types in five forest types (t·hm^{-2}).

Forest types	CWD types		
	Logs	Snags	Stumps
P. tabulaeformis	10.92(3.75)a	1.46(0.31)b	0.19(0.17)a
P. armandi	7.15(2.54)b	3.63(0.53)a	0.18(0.04)a
Q. alienavar.acuteserrata	6.75(1.84)b	1.61(0.31)b	0.19(0.06)a
L. principis-rupprechtii	4.19(1.87)bc	0.92(0.18)c	0.12(0.04)ab
B. albo-sinensis	1.55(0.90)c	0.24(0.09)d	0.03(0.03)b

Note: Mean in each column with different letter is significantly different at p 0.05, standard error is provided in brackets

Table 4. Biomass in the CWD decay classes for five forest types (t·hm^{-2}).

Forest types	CWD decay classes				
	1	2	3	4	5
P. tabulaeformis	3.81(0.41)a	3.47(0.37)a	2.79(0.13)b	1.73(0.11)a	0.77(0.06)a
P. armandi	1.81(0.37)b	3.73(0.28)a	3.78(0.52)a	1.24(0.06)b	0.40(0.08)b
Q. alienavar.acuteserrata	0.25(0.06)d	1.87(0.26)b	4.57(0.94)a	1.32(0.13)b	0.54(0.07)b
L. principis-rupprechtii	1.75(0.33)b	1.87(0.34)b	1.03(0.21)c	0.37(0.04)c	0.21(0.03)c
B. albo-sinensis	0.87(0.09)c	0.95(0.04)c	0	0	0

Note: Mean in each column with different letter is significantly different at p 0.05, standard error is provided in brackets

Table 5. Biomass of the CWD diameter classes in five forest types (t·hm^{-2}).

Forest types	CWD diameter classes			
	10-20	20-30	30-40	>40
P. tabulaeformis	1.66(0.24)b	4.16(0.56)a	5.47(0.61)a	1.28
P. armandi	0.71(0.07)c	5.06(0.44)a	5.19(0.48)a	0
Q. alienavar.acuteserrata	0.74(0.09)c	4.66(0.38)a	3.15(0.35)b	0
L. principis-rupprechtii	2.51(0.34)a	2.29(0.13)b	0.43(0.08)c	0
B. albo-sinensis	0.5(0.04)d	1.32(0.14)c	0	0

Note: Mean in each column with different letter is significantly different at p 0.05, standard error is provided in brackets

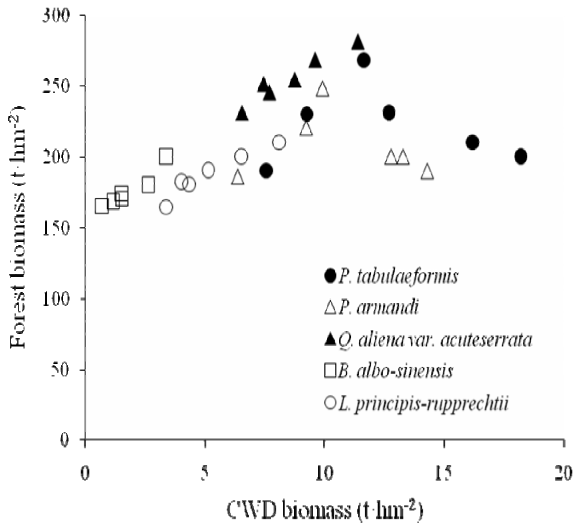

Fig. 2. Relationship between biomass of CWD and forest in each stand.

Biodiversity traits of various forest types: The effect of CWD biomass on four kinds of α diversity index was different among tree, shrub and herb in five forest types (Fig. 3). Especially for the impacts of CWD biomass on understory biodiversity were more obvious. With the increase of CWD biomass, the species richness (S), Simpson index (D) and Shannon-Wiener index (H) of understory increased at linear trend. Overall, the impacts of CWD biomass on understory four kinds of α diversity index were higher than that of trees. There was a positive correlation between CWD biomass and species richness (S), Shannon-Wiener index (H), Simpson index (D) in all five forest types (Table 7). As to Pielou's evenness index (J), the correlation was significant in *P. tabulaeformis* forest, *L. principis-rupprechtii* forest and *B. albo-sinensis* forest, but not in *Q. aliena* var. *acuteserrata* forest or in *P. armandi* forest. When trees, shrubs and herbs were analyzed separately, the correlation was more apparent in five forest types (Table 8). Compared to trees, the correlation between CWD biomass and species richness (S), Shannon-Wiener index (H), Simpson index (D) were more significant at understory, and the correlation coefficient (r) were also higher at understory.

Discussion

Character of CWD biomass: The global range in CWD biomass varies from 8 t·hm^{-2} to 200 t·hm^{-2} (Chen & Harmon, 1992). The accumulation in CWD is expected to be dynamically mediated by the various CWD production and decomposition-controlling factors (Stevens, 1997). Overall, the comparatively low CWD biomass could be explained by some aspect of carbon cycling in the five forests-either low rates of CWD input, and/or high rates of CWD decomposition (Baker *et al*., 2007). In present study, most importantly, the small quantity in CWD is possibly related to the near-mature forest developmental stage caused by the logging practice. This lower CWD biomass also seems to be caused by 1) the small pre-existing (before selective logging) or/and freshly created (natural disturbances, such as wind, rain, snow, fire, lightning, insect, invasion of fungi, etc.) CWD amount, 2) CWD amount declination over time due to decomposition (Harmon *et al*., 1986; Spies *et al*., 1988; Yan *et al*., 2007), 3) lower tree density and crown density in the study area. In forest ecosystems, different CWD types (*i.e*., logs, snags and stumps) can be an indicator of origin and legacy of CWD. In addition, it can be used to reflect forest management and stand development history. Yan *et al*. (2007) concluded that a higher proportion of CWD due to stumps in a given site may suggest extensive anthropogenic disturbances in the past, such as clear-cutting or selective logging. In 60~70s of the 20th century, intensively selective logging had occurred in this area. In present forests investigated, our data showed that logs were a principal CWD input source, followed by snags, which is in accordance with suggestions from other reports (Harmon *et al*., 1986; Keller *et al*., 2004). Generally, in natural forests most of CWD derive from gradual accumulation after ecosystems suffering from severe disturbances (e.g., wind throw) (Harmon *et al*., 1986). However, a surge in CWD biomass can be suddenly created by serious windstorms. For instance, in 1986, a catastrophic tornado produced up to 1000 ha of logs in the Changbai Mountain (Chen & Harmon, 1992). Thus, we believe that the substantial biomass in logs in this study area should primarily result from the latest local pulses in mortality, driven by a combination of strong winds and steep topography. The production in snags is expected to be relative to its shade intolerance. The trees grew well before canopy closing, thereafter, turned weak to contribute to the susceptibility of attack by insects and diseases following canopy closing (Stevens, 1997). The snags in the five forest types likely resulted from diseases and pests. The snags in *P. armandi* forest were mainly caused by a recorded infestation of *Dendroctonus armandi*. This explanation might reveal why snags in *P. armandi* forest had the highest biomass.

Our study showed that CWD in the five forest types had significantly different decomposition stages and diameter classes. The difference can partly be attributed to the vegetation composition, disturbance type and stand age. Also, the decomposition of CWD is complex and controlled by different factors such as the type of wood, humidity of wood, mean annual temperature and decomposers (Harmon *et al*., 1986; Mackensen *et al*., 2003; Garrett *et al*., 2007; Bond-Lamberty & Gower, 2008; Beets *et al*., 2008; Garrett *et al*., 2008). Such latest wind throw events could be propitious to shaded light on the origination for the high biomass in the early stage of decay class, especially in steep topography and younger stands. Carmona *et al*. (2002) and Motta *et al*. (2006) reported that the biomass of CWD in advanced decomposition classes increased with stand ages. It is generally acknowledged that CWD with similar ages is more abundant following severe disturbance events (Yan *et al*., 2006). Noticeably, the occurrence of large living trees is an essential prerequisite for creating large size CWD, but these are scarce in the five forest types. The removal of large logs after intensively selective loggings might partially explain why CWD with large size (> 40 cm) was scarce in our study area.

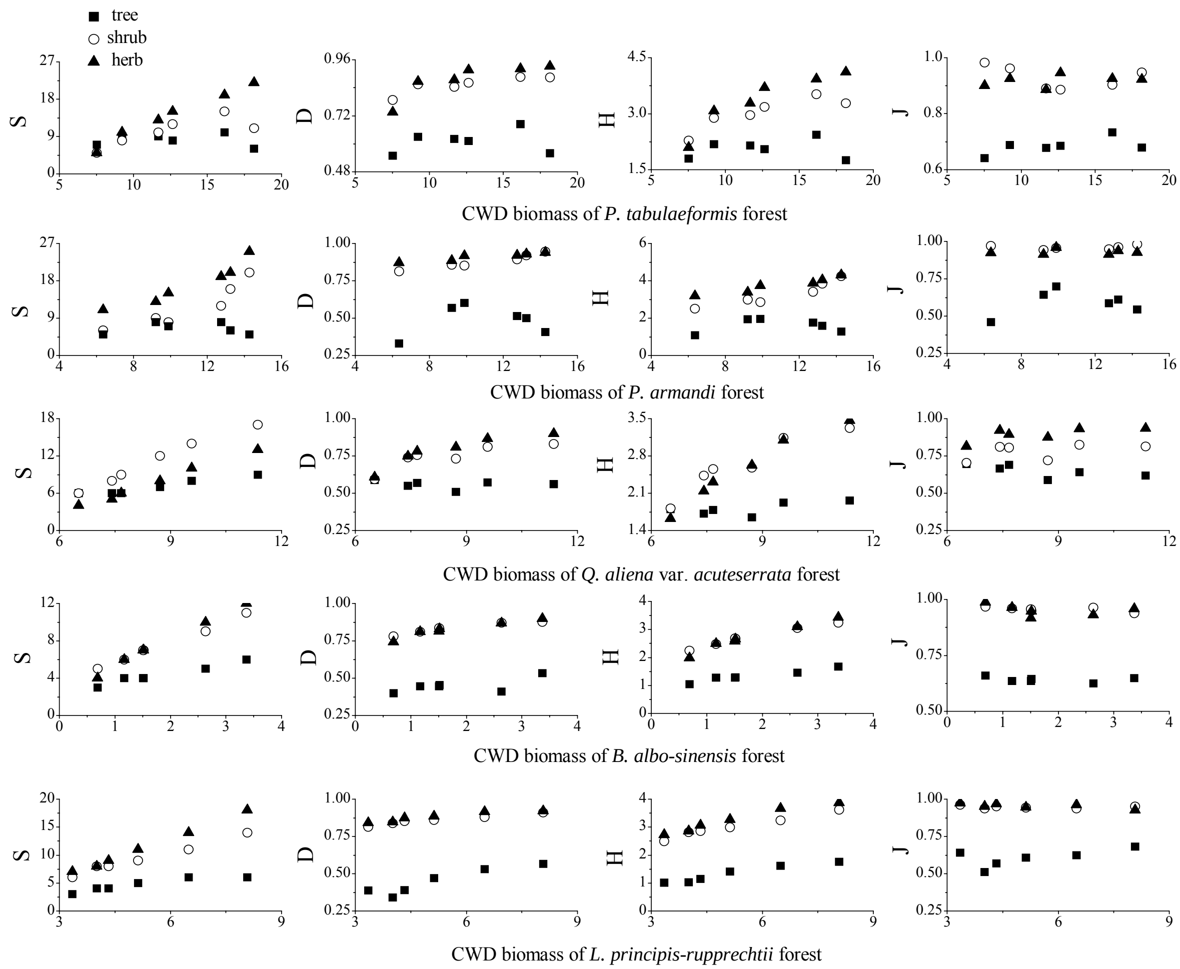

Fig. 3. The effect of the CWD biomass on α diversity index among tree, shrub and herb in the five forest types.

Table 6. Biomass in five forest types ($t \cdot hm^{-2}$).

Forest types	Tree	Shrub	Herb	Litter	CWD	Total
P. tabulaeformis	205.11b	3.39ab	0.45a	0.37c	12.57a	221.89b
	(26.57)	(0.64)	(0.08)	(0.05)	(4.03)	(27.99)
P. armandi	191.52b	4.21a	0.51a	0.65a	10.96ab	207.85bc
	(21.57)	(0.81)	(0.11)	(0.04)	(2.99)	(23.26)
Q. alienavar.acuteserrata	242.79a	3.16b	0.24c	0.51b	8.55b	255.25a
	(15.31)	(0.61)	(0.04)	(0.03)	(1.74)	(17.66)
L. principis-rupprechtii	180.19bc	2.14c	0.32b	0.39c	5.23c	188.27cd
	(14.35)	(0.34)	(0.07)	(0.06)	(1.77)	(16.07)
B. albo-sinensis	171.09c	2.91b	0.38ab	0.48b	1.82d	176.68d
	(11.55)	(0.12)	(0.06)	(0.05)	(0.99)	(12.78)

Note: Mean in each column with different letter is significantly different at $p \le 0.05$, standard error is provided in brackets

Table 7. Correlation analysis between CWD biomass and α diversity index.

Forest types	α diversity index			
	S	D	H	J
P. tabulaeformis	0.91103 *P*=0.0115	0.85298 *P*=0.0308	0.88901 *P*=0.0178	0.82054 *P*=0.0454
P. armandi	0.97333 *P*=0.0011	0.95615 *P*=0.0028	0.9849 *P*=0.0003	0.7815 *P*=0.0664
Q. aliena var. *acuteserrata*	0.99525 *P*<0.0001	0.94735 *P*=0.0041	0.97494 *P*=0.0009	0.5755 *P*=0.2321
L. principis-rupprechtii	0.99905 *P*<0.0001	0.95481 *P*=0.003	0.98767 *P*=0.0002	0.88406 *P*=0.0194
B. albo-sinensis	0.99718 *P*<0.0001	0.94843 *P*=0.0039	0.98404 *P*=0.0004	0.93792 *P*=0.0057

Table 8. Correlation analysis between CWD biomass and α diversity index on tree, shrub and herb.

Forest types	α diversity index											
	Tree				Shrub				Herb			
	S	D	H	J	S	D	H	J	S	D	H	J
P. tabulaeformis	-0.10854	0.2142	0.0922	0.6080	0.80263	0.8618	0.8689	-0.4090	0.9886	0.8303	0.9234	0.3438
	P=0.8378	*P*=0.6836	*P*=0.8621	*P*=0.2004	*P*=0.0546	*P*=0.0273	*P*=0.0246	*P*=0.4207	*P*=0.0002	*P*=0.0407	*P*=0.0086	*P*=0.5046
P. armandi	0.0163	0.1742	0.1128	0.2276	0.9160	0.9729	0.9477	0.1685	0.9550	0.9408	0.9611	0.0337
	P=0.9756	*P*=0.7413	*P*=0.8315	*P*=0.6645	*P*=0.0103	*P*=0.0011	*P*=0.004	*P*=0.7497	*P*=0.003	*P*=0.0052	*P*=0.0022	*P*=0.9494
Q. aliena var. *acuteserrata*	0.9705	-0.2289	0.6374	-0.6903	0.9925	0.8382	0.9397	0.4858	0.9962	0.9067	0.9816	0.7095
	P=0.0013	*P*=0.6627	*P*=0.1734	*P*=0.1290	*P*<0.0001	*P*=0.0372	*P*=0.0053	*P*=0.3286	*P*<0.0001	*P*=0.0127	*P*=0.0005	*P*=0.1144
L. principis-rupprechtii	0.9362	0.9362	0.9721	0.6113	0.9915	0.9781	0.9871	-0.2073	0.9989	0.9504	0.9841	-0.7704
	P=0.006	*P*=0.006	*P*=0.0012	*P*=0.1973	*P*=0.0001	*P*=0.0007	*P*=0.0002	*P*=0.6935	*P*<0.0001	*P*=0.0036	*P*=0.0004	*P*=0.0730
B. albo-sinensis	0.9835	0.6605	0.9751	-0.2913	0.9969	0.9576	0.9886	-0.6766	0.9962	0.9475	0.9788	-0.3424
	P=0.0004	*P*=0.1533	*P*=0.0009	*P*=0.5755	*P*<0.0001	*P*=0.0027	*P*=0.0002	*P*=0.1400	*P*<0.0001	*P*=0.0041	*P*=0.0007	*P*=0.5065

Relationship between biomass of CWD and forest: Our results revealed that there was a strong correlation between CWD and forest biomass. This conclusion has been reported by several studies (Sippola *et al*., 1998; Siitonen *et al*., 2000; Pedlar *et al*., 2002; Aakala *et al*., 2008). However, the dependence of the CWD biomass on forest biomass was different in forest types. *Q. aliena* var. *acuteserrata*, *B. albo-sinensis* and *L. principis-rupprechtii* all have significant positive correlation between the biomass of CWD and forest. Yet when the CWD biomass were 9.9 t·hm^{-2} and 11.6 t·hm^{-2}, the forest biomass of *P. armandi* and *P. tabulaeformis* reached the maximum, respectively. Moreover, the diameter classes, decomposition stages, species composition and quantities of CWD played a crucial role in forest biomass. Aakala *et al*. (2006) considered that the influence of the disturbance history of the stands was also an important factor for the dependence of the CWD biomass on forest biomass.

Impacts of CWD biomass on understory biodiversity: In our study, we found the impacts of CWD biomass on understory biodiversity were more obvious. With the increase of CWD biomass, the species richness (S), Shannon-Wiener index (H) and Simpson index (D) of understory increased significantly. Previous studies have reported the CWD could profoundly influence biodiversity (Brassard & Chen 2006), allow greater numbers of individuals and species to co-exist (Grove & Meggs, 2003), facilitate seed germination and seeding growth (Scheller & Mladenoff, 2002), initiate tree regeneration (Motta *et al*., 2006), and positively correlate with forest biodiversity (Tilman *et al*., 1996). These results have significantly enhanced our understanding of the impacts of CWD on biodiversity. However, little attention has been given to quantitative research, in particular understory four kinds of α diversity index were varied by CWD dynamics.

The reasons for the impacts of CWD biomass on understory biodiversity were diversity. With the increase of CWD biomass, the habitat fragmentation enhanced, the environmental heterogeneity enlarged, the habitat diversity increased, the species replacement rate accelerated. CWD contributes greatly to the structural complexity of the forest floor, this structural complexity increases the range of microclimates and microhabitats available for exploitation. Moreover, CWD may form gap, provide moisture and increase soil nutrients, these factors are necessary for understory to grow.

This study reported the CWD characteristics in the five forest types. A comparably low estimation in CWD biomass was a consequence of the present forest developmental stage. A small pre-existing or/and freshly created biomass before or/and after selective logging, removal of large logs, decomposition and human activities also contributed to the lower CWD biomass in the five forest types. Our results revealed that there was a strong correlation between CWD and forest biomass, moreover, CWD biomass affects the diversity of plant community. However, we currently do not fully understand how CWD characteristics, in particular the extent of decay and diameter classes, influence the plant communities. Some researchers suggested that plant species richness and abundance was highly correlated with CWD decay extent, size, and species type, and that unique plant community compositions were associated with specific CWD attributes (Lõhmus & Lõhmus, 2001; Humphrey *et al*., 2002; Lõhmus *et al*., 2007). Therefore, further research is required to reveal how CWD attributes (for example, types, decay stages, diameter classes and tree species) affect the plant communities in the Qinling Mountains. In addition, eco-forestry emphasized its importance to reserve CWD, but how many and how characteristic of CWD should be retained need further research. So, development of CWD reasonable strategies is indispensable for future forest management.

Acknowledgement

We are very grateful to Dr. Clive Welham and Professor Melvin Tyree for their comments on this manuscript. This work was funded by the project of "Technical management system for increasing the capacity of carbon sink and water regulation of mountain forests in the Qinling Mountains" (201004036) from the State Forestry Administration of China, and was also supported by CFERN & GENE Award Funds on Ecological Paper.

References

Aakala, T., T. Kuuluvainen, L.D. Grandpré and S. Gauthier. 2006. Trees dying standing in the northeastern boreal old-growth forests of Quebec: spatial patterns, rates, and temporal variation. *Can. J. Forest Res*., 37(1): 50-61.

Aakala, T., T. Kuuluvainen, S. Gauthier and L.D. Grandpré. 2008. Standing dead trees and their decay-class dynamics in the northeastern boreal old-growth forests of Quebec. *Forest Ecol. Manage*., 255(3-4): 410-420.

Andersson, L.I. and H. Hytteborn. 1991. Bryophytes and decaying wood–a comparison between managed and natural forest. *Ecography*, 14(2): 121-130.

Bader, P., S. Jansson and B. Jonsson. 1995. Wood-inhabiting fungi and substratum decline in selectively logged boreal spruce forests. *Biol. Conserv*., 72(3): 355-362.

Baker, T.R., E.N. Honorio Coronado, O.L. Phillips, J. Martin, G.M. van der Heijden, M. Garcia and J. Silva Espejo. 2007. Low stocks of coarse woody debris in a southwest Amazonian forest. *Oecologia*, 152(3): 495-504.

Beets, P.N., I.A. Hood, M.O. Kimberley, G.R. Oliver, S.H. Pearce and J.F. Gardner. 2008. Coarse woody debris decay rates for seven indigenous tree species in the central North Island of New Zealand. *Forest Ecol. Manage*., 256(4): 548-557.

Berger, W.H. and F.L. Parker. 1970. Diversity of planktonic foraminifera in deep-sea sediments. *Science*, 168(3937): 1345-1347.

Bond-Lamberty, B. and S. T.Gower. 2008. Decomposition and fragmentation of coarse woody debris: Re-visiting a boreal black spruce chronosequence. *Ecosystems*, 11(6): 831-840.

Brais, S., F. Sadi, Y. Bergeron and Y. Grenier. 2005. Coarse woody debris dynamics in a post-fire jack pine chronosequence and its relation with site productivity. *Forest Ecol. Manage*., 220(1-3): 216-226.

Brassard, B.W. and H.Y.H. Chen. 2006. Stand structural dynamics of North American boreal forests. *Crit. Rev. Plant Sci*., 25: 115-37.

Bredesen, B., R. Haugan, R. Aanderaa, I. Lindblad, B. Økland and Ø. Røsok. 1997. Vedlevende sopp som indikatorarter pa kontinuitet i østnorske granskoger [Wood-inhabiting fungi as indicators of continuity in spruce forests in eastern Norway]. *Blyttia*, 54: 131-140.

Carmona, M.R., J.J. Armesto, J.C. Aravena and C.A. Perez. 2002. Coarse woody debris biomass in successional and primary temperate forests in Chiloe Island, Chile. *Forest Ecol. Manage*., 164(1-3): 265-275.

Chen, C.G. and H. Peng. 1995. Standing crops and productivity of the major forest-types at the Huoditang forest region of the Qinling Mountains. *Journal of Northwest Forestry College*, 11(suppl.): 92-102 (in Chinese, with English abstract).

Chen, H. and M.E. Harmon. 1992. Dynamic study of coarse woody debris in temperate forest ecosystem. *Chinese Journal of Applied Ecology*, 3(2): 99-104 (in Chinese, with English abstract).

Currie, W.S. and K.J. Nadelhoffer. 2002. The imprint of land-use history: patterns of carbon and nitrogen in downed woody debris at the Harvard Forest. *Ecosystems*, 5(5): 446-460.

Eaton, J.M. and D. Lawrence. 2006. Woody debris stocks and fluxes during succession in a dry tropical forest. *Forest Ecol. Manage*., 232(1-3): 46-55.

Ganjegunte, G.K., L.M. Condron, P.W. Clinton, M.R. Davis and N. Mahieu. 2004. Decomposition and nutrient release from radiata pine (*Pinus radiata*) coarse woody debris. *Forest Ecol. Manage*., 187(2-3): 197-211.

Garrett, L.G., G.R. Oliver, S.H. Pearce and M.R. Davis. 2008. Decomposition of *Pinus radiata* coarse woody debris in New Zealand. *Forest Ecol. Manage*., 255(11): 3839-3845.

Garrett, L., M. Davis and G. Oliver. 2007. Decomposition of coarse woody debris, and methods for determining decay rates. *New Zealand Journal of Forestry Science*, 37(2): 227-240.

Gough, C.M., C.S. Vogel, C. Kazanski, L. Nagel, C.E. Flower and P.S. Curtis 2007. Coarse woody debris and the carbon balance of a north temperate forest. *Forest Ecol. Manage*., 244(1-3): 60-67.

Grove, S. and J. Meggs. 2003. Coarse woody debris, biodiversity and management: a review with particular reference to Tasmanian wet eucalypt forests. *Aust. Forestry*, 66(4): 258-272.

Hansen, A.J., T.A. Spies, F.J. Swanson and J.L. Ohmann. 1991. Conserving biodiversity in managed forests. *Bioscience*, 41(6): 382-392.

Hardt, R.A. and W.T. Swank. 1997. A comparison of structural and compositional characteristics of southern Appalachian young second-growth, maturing second-growth, and old-growth stands. *Nat. Area. J*., 17(1): 42-52.

Harmon, M.E. and J.F. Franklin. 1989. Tree seedlings on logs in Picea-Tsuga forests of Oregon and Washington. *Ecology*, 70(1): 48-59.

Harmon, M.E. and J. Sexton. 1996. *Guidelines for Measurements of Woody Detritus in Forest Ecosystems.* Publication No. 20. U.S. LTER Network Office: University of Washington, Seattle, WA, USA.

Harmon, M.E., J.F. Franklin, F.J. Swanson, P. Sollins, S.V. Gregory, J.D. Lattin, N.H. Anderson, S.P. Cline, N.G. Aumen, J.R. Sedell, G.W. Lienkaemper, K. Cromack Jr. and K.W. Cummins. 1986. Ecology of coarse woody debris in temperate ecosystems. *Advances in Ecological Research*, 15: 133-302.

Humphrey, J., S. Davey, A. Peace, R. Ferris and K. Harding. 2002. Lichens and bryophyte communities of planted and semi-natural forests in Britain: the influence of site type, stand structure and deadwood. *Biol. Conserv*., 107(2): 165-180.

Hurlbert, S.H. 1971. The nonconcept of species diversity: a critique and alternative parameters. *Ecology*, 52(4): 577-586.

Janisch, J. and M.E. Harmon. 2002. Successional changes in live and dead wood carbon stores: implications for net ecosystem productivity. *Tree Physiol*., 22(2-3): 77-89.

Kang, Y.X. and Y.P. Chen. 1996. Woody Plant Flora of Huoditang Forest Region. *Journal of Northwest Forestry College*, 11(1): 1-10 (in Chinese, with English abstract).

Keller, M., M. Palace, G.P. Asner, R. Pereira and J.N.M. Silva. 2004. Coarse woody debris in undisturbed and logged forests in the eastern Brazilian Amazon. *Global Change Biol*., 10(5): 784-795.

Krankina, O.N. and M.E. Harmon. 1995. Dynamics of the dead wood carbon pool in northwestern Russian boreal forests. *Boreal Forests and Global Change*: 227-238.

Kuuluvainen, T. 2002. Natural variability of forests as a reference for restoring and managing biological diversity in boreal Fennoscandia. *Silva Fenn*., 36(1): 97-125.

Lei, R.D., H. Peng and C.G. Chen. 1996. Types and Phytoenosis of Natural Secondary Forests at Huoditang Forest Region. *Journal of Northwest Forestry College*, 11(1): 43-52 (in Chinese, with English abstract).

Lõhmus, A., P. Lõhmus and K. Vellak. 2007. Substratum diversity explains landscape-scale co-variation in the species-richness of bryophytes and lichens. *Biol. Conserv*., 135(3): 405-414.

Lõhmus, P. and A. Lõhmus. 2001. Snags, and their lichen flora in old Estonian peatland forests. *Ann. Bot.Fenn*., 38: 265-280.

Mac Nally, R., A. Parkinson, G. Horrocks, L. Conole and C. Tzaros. 2001. Relationships between terrestrial vertebrate diversity, abundance and availability of coarse woody debris on south-eastern Australian floodplains. *Biol. Conserv*., 99(2): 191-205.

Mackensen, J., J. Bauhus and E. Webber. 2003. Decomposition rates of coarse woody debris - A review with particular emphasis on Australian tree species. *Aus. J. Bot*., 51(1): 27-37.

Magurran, A.E. 1988. ***Ecological Diversity and Its Measurement.*** Princeton University Press, New Jersey.

Motta, R., R. Berretti, E. Lingua and P. Piussi. 2006. Coarse woody debris, forest structure and regeneration in the Valbona Forest Reserve, Paneveggio, Italian Alps. *Forest Ecol. Manage*., 235(1-3): 155-163.

Nilsson, S.G., M. Niklasson, J. Hedin, G. Aronsson, J.M. Gutowski, P. Linder, H. Ljungberg, G. Mikusiński and T. Ranius. 2002. Densities of large living and dead trees in old-growth temperate and boreal forests. *Forest Ecol. Manage*., 161(1): 189-204.

Passovoy, M. D. and P.Z. Fulé. 2006. Snag and woody debris dynamics following severe wildfires in northern Arizona ponderosa pine forests. *Forest Ecol. Manage*., 223(1): 237-246.

Pedlar, J.H., J.L. Pearce, L.A. Venier and D.W. McKenney. 2002. Coarse woody debris in relation to disturbance and forest type in boreal Canada. *Forest Ecol. Manage*., 158(1-3): 189-194.

Renvall, P. 1995. Community structure and dynamics of wood-rotting Basidiomycetes on decomposing conifer trunks in northern Finland. *Karstenia*, 35(1): 1-51.

Ringvall, A. and G. Ståhl. 1999. Field aspects of line intersect sampling for assessing coarse woody debris. *Forest Ecol. Manage*., 119(1-3): 163-170.

Santiago, L.S. 2000. Use of coarse woody debris by the plant community of a Hawaiian montane cloud forest. *Biotropica*, 32(4): 633-641.

Scheller, R.M. and D.J. Mladenoff. 2002. Understory species patterns and diversity in old-growth and managed northern hardwood forests. *Ecol. Appl*., 12(5): 1329-1343.

Schlegel, B.C. and P.J. Donoso. 2008. Effects of forest type and stand structure on coarse woody debris in old-growth rainforests in the Valdivian Andes, south-central Chile. *Forest Ecol. Manage*., 255(5-6): 1906-1914.

Shinwari, Z.K. 2001. Participatory forest management: A review. In: *Proceedings Workshop on Ethnobotany and its application to Participatory Forest Management*. (Eds.): Shinwari, Z K. & Ashiq A. Khan. WWF-Pakistan 87-101.

Siitonen, J., P. Martikainen, P. Punttila and J. Rauh. 2000. Coarse woody debris and stand characteristics in mature managed and old-growth boreal mesic forests in southern Finland. *Forest Ecol. Manage*., 128(3): 211-225.

Similä, M., J. Kouki and P. Martikainen. 2003. Saproxylic beetles in managed and seminatural Scots pine forests: quality of dead wood matters. *Forest Ecol. Manage*., 174(1): 365-381.

Sippola, A.L., J. Siitonen and R. Kallio. 1998. Amount and quality of coarse woody debris in natural and managed coniferous forests near the timberline in Finnish Lapland. *Scand. J. Forest Res*., 13(2): 204-214.

Sippola, A.-L. and P. Renvall. 1999. Wood-decomposing fungi and seed-tree cutting: a 40-year perspective. *Forest Ecol. Manage*., 115(2): 183-201.

Söderström, L. 1988. Sequence of bryophytes and lichens in relation to substrate variables of decaying coniferous wood in northern Sweden. *Nord. J. Bot*., 8(1): 89-97.

Spetich, M.A., S.R. Shifley and G.R. Parker. 1999. Regional distribution and dynamics of coarse woody debris in midwestern old-growth forests. *Forest Sci*., 45(2): 302-313.

Spies, T., J. Franklin and T. Thomas. 1988. Coarse woody debris in Douglas-fir forests of western Oregon and Washington. *Ecology*, 69(6): 1689-1702.

Stevens, V. 1997. *The ecological role of coarse woody debris: an overview of the ecological importance of CWD in BC forests.* British Columbia, Ministry of Forests, Research Program.

Stevenson, S.K., M.J. Jull and B.J. Rogers, 2006. Abundance and attributes of wildlife trees and coarse woody debris at three silvicultural systems study areas in the Interior Cedar-Hemlock Zone, British Columbia. *Forest Ecol. Manage*., 233(1): 176-191.

Sturtevant, B.R., J.A. Bissonette, J.N. Long and D.W. Roberts. 1997. Coarse woody debris as a function of age, stand structure, and disturbance in boreal New found land. *Ecol. Appl*., 7(2): 702-712.

Tilman, D., D. Wedin and J. Knops. 1996. Productivity and sustainability influenced by biodiversity in grassland ecosystems. *Nature*, 379(6567): 718-720.

Wang, B., S.W. Lu and H.J. Li. 2011. Observation methodology for long-term forest ecosystem research (LY/T 1952-2011), Fenghuang Press. Beijing, China.

Wenger, K.F. 1984. *Forestry Handbook*. Edited for the Society of American Foresters. John Wiley & Sons, New York.

Wilcke, W., T. Hess, C. Bengel, E. Homeier, C. Valarezo and W. Zech. 2005. Coarse woody debris in a montane forest in Ecuador: mass, C and nutrient stock, and turnover. *Forest Ecol. Manage*., 205(1-3): 139-147.

Woodall, C.W. and G.C. Liknes. 2008. Relationships between forest fine and coarse woody debris carbon stocks across latitudinal gradients in the United States as an indicator of climate change effects. *Ecol. Indic*., 8(5): 686-690.

Yan, E.R., X.H. Wang and J.J. Huang. 2006. Concept and classification of coarse woody debris in forest ecosystems. *Frontiers of Biology in China*, 1(1): 76-84.

Yan, E.R., X.H. Wang, J.J. Huang, F.R. Zeng and L. Gong. 2007. Long-lasting legacy of forest succession and forest management: Characteristics of coarse woody debris in an evergreen broad-leaved forest of Eastern China. *Forest Ecol. Manage*., 252(1): 98-107.

Yu, F., D.X. Wang, X.X. Shi, X.F. Yi and G. Li. 2013a. Seed dispersal by small rodents favors oak over pine regeneration in the pine-oak forests of the Qinling mountains, China. ***Scand. J. Forest Res.***, 28(6): 540-549.

Yu, F., D.X. Wang, X.X. Shi, X.F. Yi, Q.P. Huang and Y.N. Hu. 2013b. Effects of environmental factors on tree seedling regeneration in a pine-oak mixed forest in the Qinling Mountains, China. ***J. MT. Sci-Engl.***, 10(5): 845-853.

(Received for publication 10 October 2014)