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Soil and biomass carbon re-accumulation after landslide disturbances

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Abstract

In high-standing islands of the Western Pacific, typhoon-triggered landslides occasionally strip parts of the landscape of its vegetative cover and soil layer and export large amounts of biomass and soil organic carbon (OC) from land to the ocean. After such disturbances, new vegetation colonizes the landslide scars and OC starts to reaccumulate. In the subtropical mountains of Taiwan and in other parts of the world, bamboo (*Bambusoideae*) species may invade at a certain point in the succession of recovering landslide scars. Bamboo has a high potential for carbon sequestration because of its fast growth and dense rooting system. However, it is still largely unknown how these properties translate into soil OC re-accumulation rates after landslide disturbance. In this study, a chronosequence was established on four former landslide scars in the Central Mountain Range of Taiwan, ranging in age from 6 to 41 years post disturbance as determined by landslide mapping from remote sensing. The younger landslide scars were colonized by *Miscanthus floridulus*, while after approx. 15 to 20 years of succession, bamboo species (*Phyllostachys*) were dominating. Biomass and soil OC stocks were measured on the recovering landslide scars and compared to an undisturbed *Cryptomeria japonica* forest stand in the area. After initially slow re-vegetation, biomass carbon accumulated in *Miscanthus* stands with mean annual accretion rates of $2 \pm 0.5 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$. Biomass carbon continued to increase after bamboo invasion and reached ~40% of that in the reference forest site after 41 years of landslide recovery. Soil OC accumulation rates were $\sim 2.0 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$, 6 to 41 years post disturbance reaching ~64% of the level in the reference forest. Our results from this in-situ study suggest that

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recovering landslide scars are strong carbon sinks once an initial lag period of vegetation re-establishment is overcome.

Keywords

Typhoon; Climate change; Remote sensing; Bamboo forest ecosystem

1 Introduction

Organic carbon (OC) contained in near-surface rocks and soils plays an important role in the global carbon cycle. The global OC stocks stored in (sedimentary) rocks are estimated at 15×10^6 Pg (Sundquist and Visser Ackerman, 2013) and those in soils are estimated at 1500 to 2000 Pg in the top meter and 2300 Pg in the top three meters (Jobbágy and Jackson, 2000). The soil OC is two to four times the amount of C contained in terrestrial vegetation (approx. 550 ± 100 Pg) or in the atmosphere (approx. 800 Pg; Houghton, 2007). Considering these magnitudes, small changes in the C fluxes from (and to) the rock and soil pools may affect atmospheric CO₂ concentrations with subsequent feedbacks to climate change.

Natural disturbances are a major pathway of fast and large-scale ecosystem C loss (Kurz et al., 2008). Tropical cyclones (typhoons) deliver heavy rainfall and cause mass wasting processes which mobilize large amounts of rock-derived (petrogenic) as well as soil and biomass (biospheric) OC along with debris materials (Goldsmith et al., 2008; Hilton et al., 2011; West et al., 2011). In large fluvial systems, the exported OC is thought to be oxidized in transit thus releasing CO₂ to the atmosphere (Ward et al., 2013); however, in short, steep rivers that drain mountain islands, it may escape oxidation and (re-)enter geological storage due to rapid fluvial transfer to the ocean (Hilton et al., 2011; Kao et al., 2014; Bao et al., 2015). Globally, large amounts of water and sediments are intercepted by water reservoirs (Chao et al., 2008), which constitute an important intermediate sink for exported OC (West et al., 2011).

The discussion on whether soil erosion induces C emission (source) or C sequestration (sink) is a continuing debate, with some studies concluding that soil erosion was a C source (e.g. Lal, 2003; Post et al., 2004), others that it resulted in a net sink (e.g. Smith et al., 2001; McCarty and Ritchie, 2002) of atmospheric C. In mountainous terrain, mass wasting can mobilize significant amounts of C ($0.1\text{--}0.9$ Mg C ha⁻¹ yr⁻¹) but also contribute to C recovery ($0.3\text{--}1.5$ Mg C ha⁻¹ yr⁻¹) on landslide scars during ecosystem development (Ramos Scharrón et al., 2012). To refine model predictions about landslide-triggered C fluxes, the dynamics of terrestrial C reaccumulation needs to be accounted for, particularly in subtropical and tropical areas. Unfortunately, data about this are very scarce. Most of what we know about C re-accumulation on recovering landslide scars is derived from studies in Luquillo Experimental Forest (LEF) in Puerto Rico. In LEF, hurricanes induced high rates of mortality among the aboveground plant community (Teh et al., 2009) and a persistent loss of the organic layer (Wilcke et al., 2003). Rapid re-vegetation was observed (Guariguata, 1990) but post-disturbance erosion was high and considerably retarded OC accretion in the system (only 60% of loss recovered after 150 years; Walker and Shiels, 2008).

Natural disturbances are thus integral drivers of forest dynamics (Franklin et al., 2002) and contribute to the diversity and adaptive capacity of ecosystems (Gutschick and BassiriRad, 2003). In (sub-) tropical areas, landslide scars are frequently invaded by bamboos. Bamboos (subfamily *Bambusoideae*) comprise of twelve subfamilies within the grass family (*Poaceae*) and represent the only major grass lineage to diversify in forests (Clark et al., 2015) with an estimated total area of 37 million ha globally, accounting for about 1% of the world's forest area (Lobovikov et al., 2007). Taiwan is home to 46 species and varieties of bamboo, of which 20 are native (Lin, 1967). Because of its vigorous growth, bamboo is a particularly attractive plant for C sequestration (Kuehl et al., 2013), and there is an ongoing debate on whether bamboo is a useful tool to tackle the negative effects of climate change (Lobovikov et al., 2007; Liese, 2009; Lou et al., 2010; Düking et al., 2011). Attempts to determine how much C bamboo forests contain have shown great variation, demonstrating the need to harmonize measurements of C density across different sites, species, climates and other conditions (Fredenburg, 2014).

Landslides are often caused by cyclones or other heavy rainfall events, which are predicted to intensify in the western North Pacific (Emanuel, 2005; Elsner et al., 2008; Montgomery et al., 2014; Lee et al., 2015). In Taiwan, as elsewhere in the west Pacific rim, intense precipitation, combined with high tectonic movement rates, drives rapid mass wasting and fluvial sediment transfer (Dadson et al., 2003). Taiwan received an average of four to five typhoon hits per year between 1958 and 2008 (Chiang and Chang, 2011), and from 1910 to 2010, 70 major earthquakes ($M_w > 5$ and depth < 35 km) were recorded on the island (Central Weather Bureau, 2016). Average erosion rates in Taiwan are estimated at 3–6 mm y⁻¹ (Dadson et al., 2003). In light of intensifying disturbances and the need to refine our understanding of landslide-triggered C-fluxes, this study aimed at quantifying the dynamics of OC re-accumulation along an age sequence of recovering landslide scars in the Central Mountain Range of Taiwan.

2 Materials and methods

2.1 Environmental setting

After the Chi-Chi earthquake ($M_w = 7.6$) in 1999, the landslide potential in central Taiwan strongly increased due to not only steep topography, fragile geological formation and heavy precipitation but also seismic destabilization of slope materials (Lin et al., 2010). The earthquake itself triggered many landslides (~22,000; Keefer, 2002) from the steep, heavily vegetated slopes in the Central Mountain Range. Today, about 93% of the mountain ranges are forested. Of the total forest cover 21% are conifers, 53% are hardwood stands, mixed conifer/hardwood species stands cover 19% of the area and bamboo is at 7%. The main reason for forest loss has been due to the collapse of forest lands, mainly by landslides (Environmental Protection Administration, 2009). Makino bamboo (*Phyllostachys makinoi*) was the most dominant among several species of bamboo, which were frequently incorporated with different construction methods as an auxiliary approach for slope stabilization (Lin et al., 2011). The C storage in Taiwan's forests is about 591.6 Tg in total (Environmental Protection Administration, 2009) of which bamboo forests store an estimated 9.5 Tg C (Wang and Chen, 2015).

Of the several typhoons which strike Taiwan each year, typhoon Morakot was the most severe to hit the region in latest years. It made landfall on Taiwan on 08-Aug-2009. The 3-day accumulated rainfall exceeded 3000 mm, causing serious floods in southern Taiwan including overflowed levees, landslides and debris flows from sloped land. Typhoon Morakot produced a large debris volume from landslide ($49.86 \pm 6.36 \text{ Mm}^3$) in the Tsengwen watershed (Fig. 1) accounting for 88% of the total landslide volume in the watershed (Chen et al., 2013). The total Morakot yield of coarse woody debris ($50\text{--}111 \text{ Mg C km}^{-2}$) was over an order of magnitude higher than the 2.6 Mg C km^{-2} transported over the entire year from the Amazon River, the largest point source of terrestrial C to the oceans (Schlünz and Schneider, 2000; West et al., 2011). Because typhoons are the major triggers of shallow landslides in Taiwan, landslide activity is expected to increase (Chiang and Chang, 2011).

2.2 Study area

The study was conducted in the Alishan area of the Tsengwen watershed (Fig. 1); the entire watershed area of this river basin is 1176 km^2 . It is a well-studied area, located in the upstream region of the Tsengwen river system in Chiayi County, central-south Taiwan. Tsengwen has an annual precipitation of 2800 mm, with 90% of its rainfall occurring in the wet season (Water Resource Agency, 2008). The average slope of the watershed is approximately 26.1° . Forest covers 83% of the area and the lithology is homogenous, consisting of sandstone and shale (Chen et al., 2013) with a rock strength of $6.2\text{--}33.8 \text{ MPa}$ (Western Foothills) and a basin-average erosion rate of 4.8 mm yr^{-1} (Dadson et al., 2003). The average landslide erosion rate in the Tsengwen watershed is $13.15 \pm 1.47 \text{ mm yr}^{-1}$ (Chen et al., 2013) which affected an area of $2.68 (\pm 0.7) \text{ km}^2 \text{ yr}^{-1}$ between 2005 and 2013 (unpublished data, Taiwan Forestry Bureau, Council of Agriculture, Executive Yuan).

2.3 Landslide chronosequence: site selection

There are two approaches to quantify C re-accumulation on recovering landslides. The perspective approach is to select a newly eroded site and to monitor C accumulation over time. This approach can yield a good estimate of C accumulation rates; however, the representativeness of one single site is problematic, and the approach is time consuming and prone to disturbances through possible future landslide events. The other retrospective approach is to collect multiple timeframes of past landslide scars via remote sensing and to identify their ages and development. The age of a landslide scar is defined as the time span between the occurrence of the landslide and the time of taking samples on the landslide scar. This requires that the landslide had not been reactivated before taking samples. Analyzing various landslide scars of different ages, i.e. a space-for-time-substitution or chronosequence, allows estimates of C re-accumulation dynamics within a specific watershed. This second approach is time saving, and hence more feasible within ongoing research projects compared to the first one described. In this study, the latter approach was therefore applied.

Time series of aerial photos (before 2000) and satellite images (after 2000) were analyzed. The available aerial photos were taken in 1959, 1962, 1963, 1967, 1968, 1973, 1974, 1989, 1990, 1991, 1993, and 1997 (<http://gis.rchss.sinica.edu.tw/GIArchive>). The resolution and

extent of each aerial photo varied from year to year; the horizontal resolution was 2 to 3 m. The parameters of spatial aerotriangulation were not recorded well; therefore, we rectified the aerial photos via georeferencing in ArcGIS 10.3 to create orthophotos, which were the basis for the mapping and digitizing of the landslides. We manually digitized the landslides on the aerial orthophotos using ArcGIS 10.3.

Satellite images of SPOT (20 m spatial resolution) and Formosa II (2 m panchromatic and 8 m multi-spectral) covering the years 2001, 2002, 2004, 2005, 2007, 2008, and 2009 were obtained from National Central University, Taiwan. Digitized landslide maps, based on these satellite images were provided by Y.C. Chen (cf. landslide inventories: Chen et al., 2013).

After creating landslide maps from the available aerial photos and satellite images, the landslide map of each year was overlaid by the ones of the following years. A landslide was selected to be representative for a particular year only if it did not intersect with any younger landslides. This procedure ruled out the selection of re-activated landslide scars, whose succession had been rejuvenated through following events.

The landslide scars pre-selected according to the above description were investigated in the field before making the final site selection. We tried to select landslide scars of different recovery ages that were comparable with respect to their parent material, slope and climate zones, and that showed minimal human management. Although these prerequisites were difficult to meet, we were able to establish a chronosequence with four landslide scars ranging in age from six to 41 years post disturbance (Table 1 and Fig. 2). Additionally, we selected a reference site in a coniferous stand adjacent to the oldest landslide scar, for which the aerial photos did not show any disturbance in the past 55 years and the oldest trees had an age of > 100 years based on growth parameters.

The sampling areas for biomass, litter and soil were located on the former scarp areas of the selected landslides and were free of colluvial material transported downslope by past landslides. We adopted simple random sampling (Ravindranath and Ostwald, 2008) due to the homogenous vegetation on the respective plots. The project areas, on the landslide scars and reference plot were converted into equal sized grids and the sample points were laid out randomly to avoid bias in locating the plots. The soil sampling points as well as the litter sampling points were located within the selected biomass plots.

2.4 Estimation of aboveground biomass and carbon stocks

The biomass inventory was carried out between January and March 2015. Five plots were chosen at each site. The plot sizes were small compared to tropical forest inventory standards but were chosen according to the small sizes of some of the landslide scars studied (Table 1). At the older landslide scars, where bamboo and trees occurred as the main vegetation, we did not measure herbaceous plants (or animals). Small plants add to the biomass but in general, they contribute a very small proportion and are routinely neglected (Condit, 2008). Thus, our estimates of biomass C stocks focused on aboveground living trees and live and dead bamboos.

The younger landslide sites, i.e. 6 and 11 years post disturbance, were colonized by the C₄ species *Miscanthus floridulus*. It is an herbaceous perennial originating from East Asia. It is a commonly found pioneer plant on infertile collapsed lands in Taiwan (Chou et al., 2009). At the 11 years old site, *Mallotus japonicus* var. *floccosus* occurred sporadically. At both *Miscanthus* sites all biomass within four square meter subplots was cut. The biomass was weighed in situ and sampled for laboratory analysis. Using the dried subsamples, the weight was converted from fresh to dry weight and C and nitrogen (N) were determined (Carlo Erba NA 1500). To estimate OC stocks, the aboveground biomass (*AGB*) was multiplied by the OC content.

On the older landslide sites, i.e. 26 and 41 years post disturbance, two *Phyllostachys* (*P*) bamboo species had invaded. The 26 years old site was colonized by *P. makinoi* otherwise known as makino bamboo and the 41 years old site was dominated by *P. pubescens* Mazel ex J.Houz, alias moso bamboo. Five circular plots ($r = 3$ m) were chosen at each bamboo site. We recorded the diameter at breast height (*DBH*) of each bamboo culm in each plot in situ. The genus *Phyllostachys* has monopodial rhizomes; they grow horizontally. The rhizome buds develop either upward, generating a culm, or horizontally, with a new tract of the rhizomal net. Monopodial bamboos generate an open clump with culms distant from each other and can be invasive (Lobovikov et al., 2007).

At the reference site, we chose five circular plots ($r = 6$ m) and recorded the values of *DBH* (diameter at breast height) of all woody stems of *DBH* > 5 cm. The reference site was predominantly covered by *Cryptomeria* (*C*) *japonica* D. Don, alias Japanese cedar. In addition, *Schefflera heptaphylla* (L.) Frodin and *Aidia canthioides* (Champ. ex Benth) Masam were the recorded deciduous species with a mean *DBH* value of 4.3 ± 0.1 cm. They were not included in the tree OC and biomass estimations due to their overall small diameters (Chave et al., 2005; Condit, 2008). *Cryptomeria japonica* is an important plantation species in Taiwan and represents 10% of total plantation area. It was first introduced in 1910 and widely planted in the northern and central mountain areas of Taiwan. The tree species shows robust growth performance (Cheng et al., 2013).

For aboveground biomass estimation of bamboo, an allometric function was developed by the harvest method describing biomass with *DBH* as an independent variable. To this end, five *P. pubescens* and five *P. makinoi* culms were harvested from the 1974 and 1989 landslide sites, respectively. To estimate the bamboo deadwood stock, the weight of the standing dead bamboo culms (dry matter) was assumed to be 80% of the weight of the living bamboo (IPCC, 2003). For biomass estimation of the coniferous stand, an allometric function based on Yen et al. (2009) was used.

At all study sites, litter was collected from five 1 m² subplots during dry season and the fresh weight was measured in-situ. Well-mixed subsamples were collected to determine oven dry-to-wet weight ratios to convert the total wet mass to oven-dry mass. Total C and N were measured with a CNS Analyzer (Carlo Erba NA 1500) for the subsamples. The sample plot value was then extrapolated to a carbon per hectare value.

2.5 Soil sampling, analyses and calculation of soil organic carbon stocks

At each site, soil profiles (Fig. 3) were excavated and bulk soil samples were taken from the depth layers 0–10, 10–30, 30–50 cm and 50 cm to bedrock. In addition, we collected a known volume of soil at the different depths using a metal ring pressed into the soil (intact core), and determined the weight after drying to assess the bulk density (ISO 11272, 1998). The approximate gravel content was determined after sieving the soil (~3 kg for each soil horizon). Soil sampling was conducted in December 2014, January 2015 and March 2015.

Air-dried soil samples were sieved through a 2-mm mesh screen, homogenously mixed and used for further analyses. Particle size analysis was done by the pipette method according to ISO 11277 (2009). Soil pH was measured with a glass electrode (1:2.5, soil: water ratio). Total C and N were measured using dry combustion with a CNS Analyzer (Carlo Erba NA 1500). Calcium carbonate was determined in accordance with the Scheibler volumetric method (ISO 10693, 1995): up to 9 g of soil were mixed with hydrochloric acid (10%) and the emitted CO₂ volume was measured. Organic C was calculated as the difference between total C and inorganic C.

Soil OC stock (to 50 cm depth) were calculated as follows:

$$\text{Soil OC stock} = \sum (BD_i \times C_i \times (1 - RC_i) \times t_i) \quad (1)$$

where BD_i is the bulk density of the soil in layer i , C_i is the carbon concentration of the soil in layer i , RC_i is the fraction of rock fragments > 2 mm in the sampled soil, and t_i is the thickness of the corresponding layer i .

In addition, rock samples of sandstone and shale, representing the dominant lithology of the Tsengwen watershed, were taken. Rock samples were crushed into powder, homogenously mixed and analyzed for OC concentration using the methods described above. To estimate the rock-derived OC stocks in the < 2 mm fraction on a (hypothetical) freshly exposed landslide scar, we used Eq. (1) with the rock OC concentrations and the remaining properties taken from the youngest landslide site.

2.6 Statistical evaluation

Using mean values per site ($n = 5$ for each landslide and reference), differences in OC stocks of aboveground biomass, litter and soil between sites were analyzed with one-way ANOVA followed by a Tukey's post-hoc test. Significance was determined at $p < 0.05$ and means are presented $\pm$ standard error (SE). All statistical analyses were performed using the SPSS 20 software package for Windows.

3 Results and discussion

3.1 Carbon accumulation in the aboveground biomass

The allometric equation developed to estimate bamboo aboveground biomass and carbon storage is depicted in Fig. 4 and was similar to the ones determined by Yen et al. (2010) who had shown that age did not significantly explain bamboo biomass; their correlation between

AGB and age was low ($R^2 = -0.11$). The younger *Miscanthus* site had only $1.3 \pm 0.22 \text{ Mg C ha}^{-1}$ stored in the aboveground plants 6 years after the 2009 landslide event (Fig. 5, photograph in Fig. 2a), indicating that the initial colonization of fresh landslide scars may proceed very slowly. The 2009 typhoon event eroded by far the largest area studied (Table 1); due to the great distance of up to 200 m to the nearest forest edge (Fig. 2), seed supply at the youngest landslide scar could have been restricted (Rammig et al., 2006) contributing to the low biomass OC pool. Also, the barren surface remained susceptible to further erosion (Walker and Shiels, 2008) which could additionally have retarded re-vegetation. In Taiwan, however, vegetation regrowth typically takes three months up to more than one year after landslide clearing, depending on the phenological state and severity of the natural hazard (Chiang and Chen, 2011); and after 6 years of natural succession following a landslide, the vegetation recovery rate in Jou-Jou Mountain (central Taiwan) has been found to reach up to 86% (Chou et al., 2009). Eleven years after the 2004 landslide, the aboveground biomass stored $11.3 \pm 2.5 \text{ Mg C ha}^{-1}$ (Fig. 5; see photograph in Fig. 2b for grown vegetation). Based on the biomass estimates for the landslide scars, the mean annual OC accretion rate for the *Miscanthus* biomass was $2 \pm 0.5 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$ 6 to 11 years after disturbance. This shows that once a former landslide site is stabilized, re-vegetation can proceed quickly and sequester atmospheric C into the newly aggrading terrestrial system (Walker and Shiels, 2008). In the bamboo stands with growth parameters in Table 2, biomass OC was much higher than in the younger stands, but still significantly lower compared to the coniferous reference site (Fig. 5). Aboveground C for *P. makinoi* at the 26 years old site was $44.5 \pm 8.0 \text{ Mg C ha}^{-1}$ (Fig. 5) which is similar to C data for *P. makinoi* in Taiwan's forests, cited by Yen and Wang (2013) to average $49.8 \text{ Mg C ha}^{-1}$ with a mean C sequestration rate (*CSR*) of $9.9 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$ but higher compared to Wang and Chen (2015) who studied makino bamboo in the Shihmen watershed, North Taiwan. Aboveground OC of *P. pubescens* at the 41 years old site ($70.7 \pm 4.5 \text{ Mg C ha}^{-1}$) was higher compared to Yen and Wang (2013) who predicted C storage for *P. pubescens*, also in central Taiwan, to be on average $40.6 \text{ Mg C ha}^{-1}$ with a mean *CSR* value of $8.13 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$. Our estimates for *P. pubescens* were also higher compared to field data from a large variety of Chinese studies (Lou et al., 2010), where aboveground C estimates ranged between 15.5 and $64.6 \text{ Mg C ha}^{-1}$. The vegetation OC stocks in our study (Fig. 5) showed a linear increase with time after disturbance ($R^2 = 0.9$). However, studies have shown that unmanaged bamboo plantations reach a level at which no additional biomass is formed (e.g. Lou et al., 2010) and that forest succession is arrested in stands dominated by bamboo (Griscom and Ashton, 2003); therefore, we assume that no significant additional gains in *AGB* would occur on our investigated bamboo plots unless vegetation cover changes to trees.

Our reference site had a stand density of $884 \pm 56 \text{ trees ha}^{-1}$ and *DBH* ranged from 16.1–79.9 cm with a mean *DBH* value of $35.1 \pm 2.0 \text{ cm}$. The aboveground mean OC stocks of $175.6 \pm 20.9 \text{ Mg C ha}^{-1}$ were comparable to other forests in Taiwan and close to the average C stocks of *C. japonica* (182 Mg C ha^{-1}), based on 12 even-aged stands along a stand age gradient from 37 to 93 years (Cheng et al., 2013). Yen et al. (2009), Cheng et al. (2013), and Yen and Wang (2013) assessed C storage of Japanese cedar forests and plantations between 101 and 320 Mg C ha^{-1} with a mean *CSR* value of 2.66 – $5.23 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$ (Yen and Wang, 2013). McEwan et al. (2011) studied broad-leaved evergreen forests in central Taiwan

(600–733 m a.s.l.) and hypothesized that persistent typhoon disturbances could place strong limitations on biomass accumulation. In their study, the typhoon-exposed forests showed *AGB* stocks $< 200 \text{ Mg ha}^{-1}$; but a topographically protected old-growth forest had much higher *AGB* stocks ($\sim 400 \text{ Mg ha}^{-1}$). From a meta-analysis comparing data from > 600 secondary tropical forest sites, Martin et al. (2013) concluded that *AGB* takes approximately 80 years to recover to the original value following a disturbance. The lower biomass of bamboo-dominated forests compared to tree-dominated forests may result from a combination of intermediate disturbances (e.g. crushing of juvenile plants by collapsing bamboo stems) and the capacity of some bamboo species to invade tree canopies with multiple gaps and competitively exclude juvenile trees (Griscom and Ashton, 2003). Our conclusions on OC re-accumulation are based on one reference site only, but other studies, as discussed above, confirm a high C storage potential of forests in Taiwan, which, as our results showed (Fig. 5), could take decades to recover after a landslide disturbance took place.

3.2 Litter carbon storage

Corresponding to the low biomass on the youngest *Miscanthus* site, its litter OC storage ($1.2 \pm 0.1 \text{ Mg C ha}^{-1}$) was lowest among all sites (Fig. 6). The remaining landslide scars had similar OC stocks around 2 Mg C ha^{-1} , with no significant trend along the landslide chronosequence, as also found by Kiyono et al. (2007). As with aboveground biomass, litter OC on the reference site covered by *C. japonica* ($3.91 \pm 0.47 \text{ Mg C ha}^{-1}$) was significantly higher than on the landslide sites (Fig. 6). Compared to our results, Qi et al. (2009) estimated a higher C stock for *P. pubescens* litter of $3.91 \text{ Mg C ha}^{-1}$ and Zhou et al. (2006) determined a leaf litter production for *P. pubescens* of approximately $3.05\text{--}6.11 \text{ Mg ha}^{-1} \text{ yr}^{-1}$. Moorhead and Sinsabaugh (2006) pointed out that soils may sequester more C if decay rates are slow or inputs of organic matter increase. Monopodial bamboos shed only their 2 years old leaves every year (Kleinhenz and Midmore, 2001) and it has been shown that bamboo leaves decompose slowly (Liu et al., 2000; Mehta et al., 2013), which is one cause for their high C sequestration potential. All our investigated sites showed litter C:N ratios between 30 and 38 which are just within a published threshold of 30–40 below which C and N are not mineralized from decomposing tropical litter (Takeda, 1996). Consequently neither species had a comparative advantage in sequestering C based on litter C/N ratios; hence, leaf litter production might be of more significance.

Besides leaf litter OC, the bamboo stands had considerable amounts of OC stored in standing dead culms. Thirty-two percent of the standing culms of the *P. makinoi* community (26 years old site) were dead, resulting in approx. $23.3 \pm 3.8 \text{ Mg C ha}^{-1}$. The *P. pubescens* stand (41 years old site), on the other hand, had less dead culms, resulting in $7.4 \pm 0.5 \text{ Mg C ha}^{-1}$. This transfer of C between live and dead pools can substantially increase the pool of coarse woody debris (CWD) leading to net C releases within forest ecosystems, as observed by Rice et al. (2004) in a disturbed Amazon forest.

3.3 Soil properties on the recovering landslide scars

Soil depth increased along the four chronosequence sites with recovery age (Table 3). For the 6 years old site, soil depth was only 15–20 cm, for the 11 years old site soil depth partly

reached 40–50 cm, while on the older sites, the mean soil depth was ~50 cm (Fig. 3). Therefore, in < 15 years, soil thickness increased up to 50 cm, which was likely facilitated by the weathering-favourable climate and vegetation colonization, as also observed by Phillips et al. (2008). Gravel content was high for all sampling sites (Table 3) which reflects that the soils have formed on freshly exposed rocks (Fig. 3). Bulk density decreased along the chronosequence; this could be due to loosening effects of root growth, increasing incorporation of organic matter, and developing soil structure. Soil pH showed a marked decrease with progressing soil development on the recovering landslide scars (Table 3). Soil pH generally declines in primary succession as organic acids are increasingly formed (Walker and Del Moral, 2003), and soil pH ranges from 5.0 to 6.5, as observed in our study (plots ~26 years), are most suitable for bamboo (Banik, 2015). The high pH of the *Miscanthus* sites (6 and 11 years old sites) could be an indication that these soils were still less weathered than the older profiles, and that traces of carbonate originating from the underlying shale (not shown) and base cations have not yet entirely dissolved/leached. Pandey and Singh (1985), studying ecosystem recovery in the Himalaya, found that the concentration of OC in soil increased with time and reached a similar concentration to that in nearby forests after 40 years. Our results also showed a clear increase in the soil OC concentration over time on the recovering landslides. The soil OC concentration on the 41 years old site was indeed very similar to that in the reference coniferous site.

The younger landslide scars of our study were colonized by the pioneer plant *Miscanthus*, and the older ones by bamboo. This suggests that some pedogenic site preparation was required before bamboo could establish, probably in the form of weathering and the accretion of soil organic matter, as described by Phillips et al. (2008) for the establishment of trees in humid subtropical climate.

3.4 Soil organic carbon re-accumulation along the landslide chronosequence

The younger *Miscanthus* site (6 years old) accumulated $2.9 \pm 0.6 \text{ Mg C ha}^{-1}$ in the soil, reflecting the limited C input by its sparse vegetative cover (s. photographs in Figs. 2a and 3). Soil OC accumulation strongly increased after a dense vegetative cover had established, reaching $17.3 \pm 1.7 \text{ Mg C ha}^{-1}$ at the 11 years old site (Fig. 7). After bamboo invasion, soil OC stocks continued to increase, reaching $45.8 \pm 8.5 \text{ Mg C ha}^{-1}$ after 26 years and $75.6 \pm 5.0 \text{ Mg C ha}^{-1}$ after 41 years post-landslide disturbance. However, soil OC stocks of the reference site ($117.9 \pm 18.1 \text{ Mg C ha}^{-1}$) were not yet reached.

Based on the calculated SOC stocks at the four landslide scars, we approximated the relationship between recovery time since the last landslide disturbance and recovered SOC stock by the following equation:

$$\text{SOC stock (Mg C ha}^{-1}\text{)} = -7.26 + 2.03 \times \text{recovery time} \quad (2)$$

This linear function is shown along with the calculated stocks in Fig. 7. The slope shows a CSR value of $\sim 2 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$ from six to 41 years of secondary succession, indicating that the investigated landslide sites may be strong CO₂ sinks.

The studied landslides occurred on sedimentary rocks, containing petrogenic OC; thus geological OC may have played a role for the initial OC stocks on the landslide scars. The OC concentration of the bedrock samples from the Tsengwen watershed was within the range reported by Hilton et al. (2010) with 0.05% for sandstone and 0.23% for shale. We estimated the rock-derived OC stocks to be about 1.9 Mg C ha^{-1} , i.e. only slightly lower than on the youngest (6 years old) landslide site (Fig. 7). Hence SOC accumulation rates in our study were low in the early years after landslide disturbance, indicating an initial lag period of retarded OC accumulation. This is supported by the low biomass C pool on the youngest landslide scar (Fig. 5). Our results show that SOC accumulation proceeded slowly at first but continued at high rates for several decades following landslide re-vegetation. Similar patterns have been documented in other studies, i.e. Kolbe et al. (2016) demonstrated that the onset of reforestation is a key factor controlling SOC concentrations and that SOC increased with time (> 80 years).

Del Castillo and Blanco-Macías (2007) showed that severe losses of soil and C can partly be reversed at the early stages of secondary tropical succession and that the processes slowed with time. Silver et al. (2000) documented a continuous accumulation of SOC for 100 years following reforestation in tropical regions, with soil *CSR* values of $1.30 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$ during the first 20 years and $0.20 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$ for the subsequent 80 years period. Parfitt et al. (2013) sampled erosion sites in New Zealand and estimated SOC stocks (0–20 cm) to increase from 10 to 80 Mg C ha^{-1} within 70 years after a landslide event, resulting in a net accumulation rate of $1.0 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$. Zehetner et al. (2009) studied OC accretion in riverine floodplain soils and reported rapid OC accumulation during the initial 100 years of soil formation, with rates exceeding $1 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$. For bamboo, Singh et al. (2006) documented high accretion rates of OC ($3.2 \pm 1.1 \text{ Mg ha}^{-1} \text{ yr}^{-1}$) in re-developing soil under 5-yr-old *Dendrocalamus strictus* plantations on coalmine spoils. While we found a linear increase of SOC accumulation at a high *CSR* value from 6 to 41 years, our chronosequence does not reflect the duration to document a decline in *CSR* observed over an extended study period (Gleixner et al., 2009).

Soil OC is greatly affected by the vegetation cover present. *Miscanthus* was reported to sequester high amounts of SOC (mean $1.68 \pm 0.70 \text{ Mg ha}^{-1} \text{ yr}^{-1}$) in a linear manner over time (Poeplau and Don, 2014). The vegetation change from *Miscanthus* to bamboo involves a C_4 to C_3 vegetation change but *CSR* on our sites remained equally high. Bamboo belongs to the class of C_3 plants, like trees, which indicates that the CO_2 fixation process of bamboo is not quicker than that of fast-growing trees (Düking et al., 2011). It has been inferred that unmanaged bamboo stands do not sequester additional C once they reach maturity (Kuehl, 2015). Bamboo culms of most species reach maturity after approximately 7 to 10 years, after which the biomass deteriorates rapidly, releasing CO_2 into the atmosphere (Liese, 2009). Similarly, Kleinhenz and Midmore (2001) demonstrated that with increasing age of culms and/or leaves, bamboo's photosynthetic rates decrease significantly. The SOC stocks of our study, which ranged from 45.8 to $75.7 \text{ Mg C ha}^{-1}$ for bamboo and $117.9 \text{ Mg C ha}^{-1}$ for *C. japonica*, fall within the mid to upper range of values reported in aforementioned studies from Taiwan and China (Cheng et al., 2013; Wang and Chen, 2015; Zhuang et al., 2015), suggesting that our landslide scars may be approaching their capacities as SOC sinks.

3.5 Ecosystem carbon recovery

Table 4 summarizes C accretion in the different ecosystem compartments at our study sites. It is known that only tropical forests hold as much C in vegetation as in soils whereas in all other biomes, soils claim a much larger share (Ravindranath and Ostwald, 2008). In our study, the distribution of OC between the vegetation and soil pools was a function of ecosystem recovery. Soil OC stocks were clearly dominant at the young (*Miscanthus floridulus*) sites. Similarly, Walker and Shiels (2008) documented substantial post-landslide litter inputs on 8–13 months old landslides in Puerto Rico but found much more OC in landslide soils than in plant biomass. On our older landslide scars (bamboo forests), soil and biomass OC stocks were almost equal; and in the coniferous reference site, biomass OC stocks were considerably higher than soil OC stocks (Table 4). Our results show that after 41 years of recovery, the bamboo forest had accumulated approximately half of the ecosystem OC as the *C. japonica* reference stand (Table 4).

A restriction of our study was that belowground biomass was not measured due to limited resources. In general, belowground biomass in forest ecosystems accounts for about 20–26% of the total biomass (Santantonio et al., 1977; Cairns et al., 1997). However, estimates for bamboo belowground C stocks have shown great variation. Some research suggests that aboveground C is higher than belowground C (Kiyono et al., 2007; Qi et al., 2009), other studies have shown that the stocks are about equal (Huang, 1987) and yet others estimated belowground C stocks to be higher than aboveground C stocks (Wang and Chen, 2015). Kuehl (2015) observed that the distribution of total bamboo biomass between above- and belowground components varied with age, which could explain some of the discrepancies between the aforementioned studies. Generally, *AGB* is given the highest importance in C inventories and in most mitigation projects, and represents the most important pool for the afforestation and reforestation Clean Development Mechanism (CDM) projects under the Kyoto protocol as well as any inventory or mitigation project related to forest lands (Ravindranath and Ostwald, 2008). Therefore, we inferred that our study can contribute to the discussion about recognizing bamboo for climate change mitigation projects.

3.6 Erosional carbon fluxes

Recent studies have shown that export yields of biospheric and petrogenic particulate OC (POC) are positively related to the yield of suspended material, and therefore POC export is mostly controlled by physical erosion (Galy et al., 2015; Hilton, 2016). Soil erosion in steep mountain topography with high sediment yields promotes efficient burial of POC eroded from the biosphere, which has been confirmed by results from Taiwan (Kao et al., 2014; Bao et al., 2015; Hilton, 2016). Petrogenic OC is very efficiently preserved in fluvial systems characterised by rapid sediment transfer to the ocean with minimal oxidation (Hilton et al., 2011; Kao et al., 2014), such as those found in Taiwan. Based on the aforementioned studies, we can assume that a large portion of the eroded biospheric OC does indeed escape oxidation in the environmental setting of Taiwan, which renders the aggrading terrestrial ecosystem investigated in our study to be strong net CO₂ sinks for several decades since a landslide disturbance. Hilton et al. (2008) estimated average non-petrogenic POC fluxes in the LiWu watershed in Taiwan at 0.16–2.02 Mg C ha⁻¹ yr⁻¹. This erosional flux of biospheric POC, which is among the highest worldwide (Galy et al., 2015) could be easily

sustained by the *CSR* values of aggrading terrestrial ecosystems found in the Tsengwen watershed.

3.7 Bamboo and climate change mitigation

The influence of atmospheric C on climate is an ever more pressing global issue and C sequestration is an increasingly important concern for forest management (McEwan et al., 2011). Forests store an estimated 296 Pg of C in their above- and belowground biomass alone (FAO, 2015), and bamboo is included in most international definitions of forests (though not all of them) but currently not recognized under REDD+, Reducing Emissions from Deforestation and Forest Degradation (Fredenburg, 2014). Article five of the Paris Agreement under the United Nations Framework Convention on Climate Change called for the enhancement of forest C stocks in developing countries (UNFCCC, 2015). Bamboo, which is widespread in many developing countries, has been shown to sequester similar amounts of C than fast growing tree plantations (Lou et al., 2010); indeed, substantial quantities of C can be sequestered by bamboo species that have high phytolith OC yields (Parr et al., 2010; Li et al., 2014). Our results confirm a high C sink potential for bamboo forests in recovering subtropical slope land ecosystems. In light of more frequent landslide events and accelerated landscape turnover, bamboo might even have a comparative advantage to many trees: *P. pubescens*, for example, accumulates 75% of its biomass within the first 40 days of growth (Yen, 2016). It could hence accumulate high amounts of carbon in a comparatively short time before a landslide scar may potentially be re-activated.

4 Summary and conclusions

Our study shows that in the initial years after a landslide event, the re-vegetation of landslide scars can be slow and only little OC is re-accumulated. However, once a dense vegetation cover (in our case *Miscanthus floridulus*) has formed after several years, OC re-accumulation proceeds at high rates; in our study area in central-south Taiwan, soil *CSR* reached $\sim 2 \text{ Mg ha}^{-1} \text{ yr}^{-1}$. We saw that when bamboo invaded after 15 to 20 years of succession, C sequestration into biomass and soil remained high. Bamboo forests stored up to 156 Mg C ha^{-1} (in biomass, litter and soil) after only few decades of succession, which shows that bamboo communities in mountainous regions can act as important C sinks. Nevertheless, within four decades, ecosystem C on the recovering landslides did not come to rival a reference, undisturbed coniferous site.

Disturbances are predicted to increase in response to global warming. If typhoon and landslide frequency increase such that landslide scars cannot recover beyond an initial lag period of retarded revegetation, the re-accumulated C may quickly get lost again. However, if the lag period is overcome and vegetation gets sufficiently established, high *CSR* values render recovering landslide scars strong C sinks. It is therefore crucial to better understand the temporal patterns of OC re-accumulation especially in the initial phases after complete vegetation removal. Our results underline that we still need to learn more about the factors that control vegetation re-establishment after disturbances in order to adequately incorporate the thresholds related to lag periods, as observed in our study, into future ecosystem models.

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Fig. 1.

Topography of the Tsengwen watershed. (a) Illustrated watershed area includes the Tsengwen River (blue); the black features represent the landslides triggered by typhoons from 2004 to 2014. The black rectangular outline indicates the location of our study area. Inset map shows the geographical position of Taiwan and the Tsengwen watershed (purple). (b) Close-up satellite image (taken in 2014) of the study area in the north of the Tsengwen watershed. Black circles illustrate the position of the selected landslide scars with the year of landslide occurrence. The triangle illustrates the position of the reference site. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

Fig. 2.

Aerial photos and pictures of the study plots. a–d) On the left, aerial photos of the landslide scars at the respective time of the landslide event (2009, 2004, 1991 (substitute for 1989), and 1974). Arrows indicate the landslide direction. In the middle, aerial photos showing the same sites in 2014. On the right, pictures taken on-site in 2014. Black/white outlines illustrate the landslide borders. The 1989 plot in c) shows a picture of 1991 since no aerial picture was available immediately after the 1989 landslide event. e) Photograph of the reference stand, adjacent to 2d.

Fig. 3. Soil profiles excavated on the respective landslide scar and the reference (Ref) site. Pictures taken in 2014–2015.

Fig. 4. Relationship between *DBH* (diameter at breast height) and aboveground biomass (dry weight, black line), and that between *DBH* and aboveground carbon (grey line) of *P. makinoi* and *P. pubescens*.

Fig. 5.

Aboveground C along the chronosequence (grey bars) and on the reference plot (Ref plot). Dominant vegetation is given above the respective bars. The white bars represent *AGB* carbon data taken from literature (Yen et al., 2009; Lou et al., 2010; Cheng et al., 2013; Yen and Wang, 2013; Wang and Chen, 2015) for three vegetation types studied (*P. makinoi*, *P. pubescens* and *C. japonica*). Means and standard errors ($\pm SE$). Different letters indicate significant differences between sites ($p < 0.05$; ANOVA; Tukey's post-hoc test).

Fig. 6. Carbon storage in plant litter along the chronosequence and in the reference plot (Ref plot). Dominant vegetation is given above the bars. Means and standard errors ($\pm SE$). Different letters indicate significant differences between sites ($p < 0.05$; ANOVA; Tukey's post-hoc test).

Fig. 7.

Soil OC stocks (0 to 50 cm or 0 to bedrock). Function fitted to landslide chronosequence only, not including the reference site. Dominant vegetation is given above the marks. Means and standard errors ($\pm SE$). Different letters indicate significant differences between sites ($p < 0.05$; ANOVA; Tukey's post-hoc test). For clarity: at $t = 0$: estimated rock-derived OC stocks in the < 2 mm fraction to a depth of 20 cm (i.e. the estimated concentration of bedrock to the OC stocks of the youngest landslide site).

Table 1

Characteristics of the four landslide sites and the reference site.

Year of landslide	Coordinates (WGS 84)		Elevation [m a.s.l.]	Slope [°]	Aspect [° from North]	Landslide area [m ²]	Main vegetation
	Longitude	Latitude					
2009	120.7620	23.4724	1069	32.2	Northwest (304)	50,443	<i>Miscanthus floridulus</i>
2004	120.7585	23.4732	994	29.9	Southeast (157)	3121	<i>Miscanthus floridulus</i>
1989	120.6581	23.4212	1299	18.6	Southeast (128)	1726	<i>Phyllostachys makinoi</i>
1974	120.7474	23.4807	1520	32.0	Southeast (133)	15,419	<i>Phyllostachys pubescens</i>
Reference site	120.7475	23.4819	1562	32.0	East (135)	n/a	<i>Cryptomeria japonica</i>

WGS 84: World Geodetic System 1984; a.s.l.: above sea level.

Table 2

Characteristics of the investigated bamboo stands. Means and standard errors in parentheses. *DBH*: Diameter at breast height; *AGB*: Aboveground biomass.

Year of landslide	bamboo species	<i>DBH</i> [cm]	Culms [ha ⁻¹]	<i>AGB</i> [Mg ha ⁻¹]	Stem C [%]
1989	<i>P. makinoi</i>	4.5 (0.3)	25,181 (2085)	88.9 (15.9)	51.5
1974	<i>P. pubescens</i>	8.5 (0.1)	12,661 (703)	139.0 (8.8)	49.5

Table 3

Soil characteristics of the five sampling sites. Means of five soil profiles per site with standard errors in parentheses. OC: organic carbon.

Years since landslide disturbance	Soil depth [cm]	Sand	Silt [%]	Clay	Gravel content [%]	Bulk density [g cm ⁻³]	pH in H ₂ O	CaCO ₃ [%]	OC [%]	N [%]
6	0–10	57 (1.4)	6 (1.1)	37 (1.5)	71 (3.9)	1.19 (0.04)	7.7 (0.1)	–	0.4 (0.1)	0.06 (0.00)
	10–20	59 (0.9)	6 (0.7)	35 (0.5)	72 (2.5)	1.17 (0.12)	7.6 (0.1)	–	0.6 (0.2)	0.07 (0.00)
11	0–10	60 (1.4)	7 (0.4)	33 (1.1)	72 (2.5)	1.20 (0.08)	7.3 (0.0)	–	1.4 (0.1)	0.10 (0.01)
	10–30	59 (0.7)	8 (0.7)	32 (0.8)	74 (2.6)	1.16 (0.11)	7.3 (0.1)	0.23 (0.1)	1.1 (0.2)	0.08 (0.01)
	30–50	58 (0.5)	8 (0.4)	34 (0.7)	70 (5.2)	1.30 (0.12)	7.3 (0.1)	0.43 (0.2)	1.2 (0.5)	0.09 (0.26)
26	0–10	50 (3.1)	10 (0.9)	40 (3.2)	40 (7.3)	0.82 (0.08)	5.8 (0.2)	–	3.3 (0.4)	0.22 (0.01)
	10–30	56 (2.0)	8 (1.2)	35 (2.2)	45 (9.8)	1.05 (0.08)	6.0 (0.2)	–	1.5 (0.1)	0.11 (0.01)
	30–50	60 (4.5)	8 (2.0)	32 (2.6)	42 (8.0)	1.17 (0.04)	6.1 (0.1)	–	0.9 (0.1)	0.08 (0.01)
41	0–10	54 (3.4)	6 (1.3)	40 (3.5)	49 (3.9)	0.39 (0.05)	5.1 (0.1)	–	10.0 (1.0)	0.60 (0.05)
	10–30	48 (1.8)	10 (1.3)	42 (1.3)	61 (3.8)	0.73 (0.11)	5.1 (0.1)	–	5.4 (0.4)	0.33 (0.02)
	30–50	49 (1.1)	8 (1.0)	43 (0.5)	56 (5.3)	0.84 (0.02)	5.0 (0.1)	–	4.0 (0.3)	0.25 (0.02)
Reference	0–10	44 (4.1)	10 (1.3)	46 (3.0)	45 (5.4)	0.56 (0.09)	5.3 (0.1)	–	10.1 (1.3)	0.56 (0.07)
	10–30	48 (1.7)	9 (0.4)	43 (1.3)	45 (5.4)	0.70 (0.09)	5.1 (0.1)	–	5.8 (0.7)	0.34 (0.03)
	30–50	46 (3.6)	9 (0.5)	46 (3.5)	43 (5.9)	0.81 (0.05)	5.0 (0.1)	–	5.0 (0.5)	0.29 (0.02)

Table 4

Summary of the carbon storage in the different ecosystem compartments at the investigated sites. Mean values with standard errors in parentheses. *AGB*: Aboveground biomass.

Years since landslide	Soil [Mg C ha ⁻¹]	<i>AGB</i> alive	<i>AGB</i> dead (standing)	Litter	Total
6	2.9 (0.6)	1.3 (0.2)	n/a	1.2 (0.1)	5.4 (0.6)
11	17.3 (1.7)	11.3 (2.5)	n/a	2.4 (0.3)	31.0 (3.0)
26	45.8 (8.5)	44.5 (8.0)	23.3 (3.8)	2.2 (0.2)	115.8 (12.3)
41	75.7 (5.0)	70.7 (4.5)	7.4 (0.5)	2.3 (0.3)	156.1 (6.7)
Reference	117.9 (18.1)	175.6 (20.9)	n/a	3.9 (0.5)	297.4 (27.7)