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Tree species identity and functional traits but not species richness affect interrill erosion processes in young subtropical forests

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Abstract

Soil erosion is seriously threatening ecosystem functioning in many parts of the world. In this context, it is assumed that tree species richness and functional diversity of tree communities can play a critical role in improving ecosystem services such as erosion control. An experiment with 170 micro-scale runoff plots was conducted to investigate the influence of tree species richness and identity as well as tree functional traits on interrill erosion in a young forest ecosystem. An interrill erosion rate of 47.5 $t\,ha^{-1}\,a^{-1}$ was calculated. This study provided evidence that different tree species affect interrill erosion, but higher tree species richness did not mitigate soil losses in young forest stands. Thus, different tree morphologies have to be considered, when assessing erosion under forest. High crown cover and leaf area index reduced soil losses in initial forest ecosystems, whereas rising tree height increased them. Even if a leaf litter cover was not present, remaining soil surface cover by stones and biological soil crusts was the most important driver for soil erosion control. Furthermore, soil organic matter had a decreasing influence on soil loss. Long-term monitoring of soil erosion under closing tree canopies is necessary and a wide range of functional tree traits should be taken into consideration in future research.

1 Introduction

Soil erosion is seriously threatening natural and agricultural ecosystems in many parts of the world. Therefore, it is considered as one of the most severe environmental challenges (Morgan, 2005). Pimentel and Kounang (1998) stated that about 75 billion tons of soil are eroded at global scale every year and soil is lost 13–40 times faster than it can regenerate. Soil erosion is also a serious challenge in the PR China, especially in the southern tropical and subtropical zone. Within this region, the annual soil loss rate ranges between 0.28 and 113 $t\,ha^{-1}$, depending on the annual precipitation, the landscape and the land use (Guo et al., 2015). Besides negative on-site effects like declining soil fertility, off-site effects triggered by the transport of sediment and included nutrients as well as pollutants cause high mitigation efforts and costs (Pimentel et al., 1995; Richter, 1998) and affect nutrient cycling and ecosystem functioning (Baumann et al., 2009; Zhao et al., 2009).

Moreover, soil erosion can negatively influence biodiversity (Pimentel and Kounang, 1998), but it is assumed that this relationship also acts vice versa (Körner and Spehn, 2002; Geißler et al., 2012b; Brevik et al., 2015). It has been shown that a change in diversity can have remarkable effects on ecosystem functions and stability (e.g. Hooper et al., 2005; Scherer-Lorenzen, 2005). In many cases, increasing biodiversity enhanced ecosystem productivity and stability (Loreau, 2001; Jacob et al., 2010). In particular, tree species richness as well as functional diversity of tree communities can play a critical role in improving ecosystem services such as water filtration, climate regulation or erosion control (Quijas et al., 2012; Chisholm et al., 2013; Scherer-Lorenzen, 2014). As forests are generally considered beneficial for erosion control, afforestation is a common measure of soil protection (Romero-Diaz et al., 2010; Jiao et al., 2012). This also applies to the south-eastern part of China, which is known as a hotspot of biodiversity and woody plants (Barthlott et al., 2005; Bruelheide et al., 2011). Guo et al. (2015) showed that forests in this area experienced the lowest soil loss rates of all land use types in China. Considering that studies on soil erosion under forest have mostly focused on deforestation (Blanco-Canqui and Lal, 2008) and counteracting measures like afforestation generally result in monoculture stands (Puettmann et al., 2009), it appears that the role of tree species richness for soil erosion has been largely disregarded. Although positive effects of mixed-species tree stands like increasing productivity or reduced pest risks were demonstrated (e.g. Vilà et al., 2007; Bauhus and Schmerbeck, 2010), the effects on erosion control are still unclear. Zhou et al. (2002) and Tsujimura et al. (2006) demonstrated that tree monocultures have only limited mitigation potential for soil losses, but further research is scarce. Nevertheless, there is growing evidence that higher species richness can reduce soil erosion (Körner and Spehn, 2002). Bautista et al. (2007) pointed out that an increase in functional diversity within a perennial vegetation cover decreased soil losses in a semiarid Mediterranean landscape. Pohl et al. (2009) showed that an increase in the diversity of root types led to higher soil stability on an alpine grassy hillslope, and recently Berendse et al. (2015) found that a loss of grass species diversity reduced erosion resistance on a dike slope.

Conceivable mechanisms underlying positive species richness effects on soil erosion are that vegetation covers with a high number of species include a high number of plant functional groups which complement one another. Thus, they are more effective in controlling erosion processes than vegetative covers with few species (Pohl et al., 2012). For example, a high tree species richness may result in an increased stratification of canopy layers (Lang et al., 2010). As a consequence, crown overlap, biomass density, and total canopy cover often are higher in mixtures than in monocultures (Lang et al., 2012). In addition, a highly diverse structure within the leaf litter layer on the forest floor seems to improve its protecting effect (Martin et al., 2010). Recently, Seitz et al. (2015) pointed out that sediment discharge depends on the species identities in the leaf litter cover, whereas there was no effect of leaf species richness or functional diversity on soil erosion. Further research appears to be necessary on the influence of tree species richness on erosion control, but also the complex system of interacting functional groups within the vegetation cover seems to be of interest.

Vegetation covers are generally considered a key factor for the occurrence and dimension of soil erosion (Thornes, 1990; Hupp et al., 1995; Morgan, 2005). A leaf litter layer on the forest floor, for example, protects the soil from direct raindrop impact and modifies the water flow and storage capacities on the soil surface (Kim et al., 2014). Moreover, forests can provide a multi-storey canopy layer which largely influences rain throughfall patterns and leads to the capture of raindrops as well as the storage of water within the tree crown (Puigdefábregas, 2005). Nevertheless, large drops can be formed at leaf apexes of tall trees (Geißler et al., 2012a) and thus may increase the kinetic energy of throughfall in older forest stands up to a factor of 2.6 compared to open fields (Nanko et al., 2008). This leads to considerable soil loss if the forest floor is unprotected, which may be the case if protecting layers diminish e.g. under shady con-

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ditions (Onda et al., 2010) or fast decomposition (Razafindrabe et al., 2010). Whereas the effects of soil surface covers on soil erosion is well studied (Thornes, 1990; Blanco-Canqui and Lal, 2008), much less is known about the influence of species-specific functional traits of the tree layer such as crown or stem characteristics (Lavorel and Garnier, 2002; Guerrero-Campo et al., 2008). Moreover, most research on the latter aspects was performed in old-grown forests (e.g. Zhou et al., 2002; Nanko et al., 2008; Geißler et al., 2012a), whereas forests in an early-successional stage are rarely mentioned. In those young forests, tree heights are lower than in later stages, but structural and spatial complexity is high and species-specific growth rates differ considerably (Swanson et al., 2011). We assume that these species-specific differences in structure and growth will influence soil erosion rates.

Our research focused on the influence of tree species richness, tree species identity, and species-specific functional traits on interrill erosion in young forests, when a leaf litter cover is not present. Testing for those effects on soil erosion requires a common garden situation, in which confounding factors such as different tree ages and sizes, inclination or soil conditions are closely controlled. These requirements were met in the forest biodiversity-ecosystem functioning experiment in subtropical China (BEF-China; cf. Bruelheide et al., 2014). Within this experiment, we established a high number of micro-scale runoff plots in a randomly dispersed and replicated design. Thereby, we addressed the following hypotheses:

1. Increasing tree species richness decreases soil erosion rates.

2. Tree species differ in their impact on soil erosion rates.

3. The effects of different tree species on soil erosion rates can be explained by species-specific functional traits.

First, we hypothesized that higher tree species richness leads to lower soil erosion rates. This is due to higher stratified and overlapping tree canopies, even when a leaf litter cover is not present. Second, we presumed that soil erosion rates change in rela-

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tion to different tree species due to species-specific functional traits. Third, we believed that tree height and canopy characteristics are good predictors for soil erosion rates.

2 Methodology

2.1 Study site and experimental design

The study was conducted in Xingangshan, Jiangxi Province, PR China (29°06.450′ N and 117°55.450′ E) at the experimental sites A and B of the BEF China project (Bruelheide et al., 2014). Together, both sites comprise an area of about 50 ha in a mountainous landscape with an elevation range from 100 m to 265 m a.s.l. Slopes range from 15 to 41°. The bedrock of the experimental site consists of non-calcareous slates with varying sand and silt contents and is intermittent by siliceous-rich joints. Prevailing soil types are Cambisols with Anthrosols in downslope positions and Gleysols in valleys (cf. IUSS, 2006) covering saprolites. Soil bulk density is low (0.98 $g\,cm^{-3}$), and soil reaction acidic (mean pH in KCl 3.68). Soil texture ranges from silt loam to silty clay loam. The climate in Xingangshan is humid and subtropical and ranked as Cwa after the Köppen-Geiger classification. It is characterized by an annual average temperature of 17.4 °C and a mean annual rainfall of 1635 mm (Goebes et al., 2015b).

The experimental area has been used as a commercial forest plantation (*Cunninghamia lancelota*) until 2007. It was clear-cut and replanted in 2009–2010 following an experimental plot-based design with different extinction scenarios (Bruelheide et al., 2014). The experimental site represented an early successional stage with tree ages from four to five years at the time of our study. In total, 566 experimental plots were established using a pool of 40 native tree species, as well as bare ground and free succession plots. Trees were planted randomly in seven species richness levels (div0, 1, 2, 4, 8, 16, 24) with a planting distance of 1.29 m, following a broken stick design (Bruelheide et al., 2014). We focused on the Very Intensively studied Plots (VIPs) of which 34 were used in this study. The monocultures with tree heights lower than 1 m or crown covers less than 10 % were excluded from the analysis. The selected set comprised a bare ground feature (4 × div0) and four levels of tree species richness (20 × div1, 4 × div8, 4 × div16 and 2 × div24) with a total of 22 tree species, two of which only appeared in mixtures (Appendix Table A1).

2.2 Erosion measurements

To determine initial sediment discharge and surface runoff, micro-scale runoff plots (ROP, 0.4 m × 0.4 m) were used (cf. Seitz et al., 2015; without fauna treatment). Each ROP was connected to a 20 L reservoir and a rainfall gauge was placed next to it (Fig. 1). All 34 VIPs were equipped with five ROPs each, resulting in a total number of 170 ROPs. Within each VIP, the ROPs were placed randomly in selected areas, which were representative for the range of surface properties of our study area. All leaf litter was removed from the ROPs prior to measurements. The ROPs were operated in May and June 2013 during the rainy season. Runoff volume and rainfall amount were determined in situ and sediment was assessed after sampling by drying at 40 °C and weighing. The capacity of the reservoirs was not exceeded in any rainfall event.

At each ROP ($n = 170$), tree crown cover and leaf area index (LAI) were measured using a fish-eye camera system (Nikon D100 with Nikon AF G DX 180°) and the HemiView V.8 software (Delta-T devices, Cambridge, UK). Total tree height, stem diameter at 5 cm above ground (hereafter, stem diameter) and crown width for each tree individual were measured to represent the tree characteristics (Li et al., 2014). Soil surface cover was measured photogrammetrically (grid quadrat method with GIMP 2.8) and slope with an inclinometer at each ROP ($n = 170$), respectively. Soil texture and soil organic matter (SOM) were identified for each VIP (5 cm depth, 9 replicates, $n = 34$) using a SediGraph III 5120 (Micromeritics, Aachen, Germany) and a Vario EL III elemental analyser (Elementar, Hanau, Germany). Furthermore, pH was measured in 1 M KCl using Sentix 81 electrodes.

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2.3 Rainfall patterns

Weather conditions were recorded by an on-site climate station (ecoTech datalogger with Vaisala weather transmitter and ecoTech tipping bucket balance) in 5-min intervals. The total precipitation in the study area in 2013 was 1205 mm and lower than the mean of the preceding three years (1635 mm). Of this amount, a fraction of 957 mm (33 events) were strong enough to trigger soil erosion following Wischmeier and Smith (1978) who used an event threshold of 12.7 mm. This threshold was confirmed by Yin et al. (2007) to be valid for southeast China. In total, 10 rainfall events were captured at the study area in May and June. Four of those events (E1–E4) can be considered erosive. The total rainfall amount from May to June was 185 mm, of which 135 mm fell during erosive rainfall events. The mean and peak intensities as well as the total rainfall amount (except for E4) increased from May to June (Table 1), reflecting a growing monsoon influence from beginning to mid-summer.

2.4 Statistical analysis

Linear mixed effects models with restricted maximum likelihood were performed with R 3.0.2 (R Core Team, 2013) and "lmerTest" (Kuznetsova et al., 2014) to investigate the influences on sediment discharge. Models were fitted with crown cover, leaf area index, tree height, stem diameter, crown width, slope, surface cover, SOM, amount of precipitation, and tree species richness as fixed effects. As random effects, precipitation event nested in plot, tree composition, site, and ROP nested in plot were used. Tree and crown characteristics were fitted one after the other, because they were highly correlated. Contrasts of diversity levels (div0 to div1–24, div1 to div8–24) were introduced to quantify the effects of bare plots vs. tree plantations and tree monocultures vs. mixtures, respectively. The effect of individual tree species (div1) was tested separately against the mean sediment discharge using crown cover, slope, surface cover, SOM, and amount of precipitation as fixed factors and site, and ROP nested in plot as random factor ($n = 200$). The maximum likelihood approach was used to obtain model

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simplification by step-wise backward selection, eliminating the least significant variable except for tree species richness. If multicolinearity was detected (spearman $\rho > 0.7$), co-variables were omitted. All variables were continuous and scaled, so model estimates could be compared. The data was log-transformed and the residuals did not show any deviation from normality. Hypotheses were tested with an ANOVA type 3 with Satterthwaite approximation for degrees of freedom and p-values were obtained by likelihood ratio tests.

3 Results

3.1 Species richness effects on interrill erosion processes

Tree species richness did not affect sediment discharge or runoff (Fig. 2 and Appendix Table A2). Sediment discharge tended to decrease from diversity level 0 to 8 and to increase to diversity level 24, while runoff volume tended to decrease from diversity level 0 to 16 and to increase to diversity level 24, but shifts were non-significant. Sediment discharge and runoff volume did not differ between bare plots (div0) and plots with trees (div1–div24), just as between monocultures (div1) and species mixtures (div8, div16, div24). The standard deviations of sediment discharge ($\mathrm{g\,m^{-2}}$) and runoff volume ($\mathrm{L\,m^{-2}}$) in relation to diversity levels were high (Fig. 2 and Appendix Table A3). Mean crown cover in mixed stands was 44 % and mean tree height was 2.30 m compared to monocultures with 22 % and 1.63 m. In our experiment tree height in mixed stands was not lower than 1.07 m and crown cover achieved at least 29 %.

3.2 Species identity effects on interrill erosion processes

Individual tree species in monocultures affected sediment discharge differently (Fig. 3).

The mean sediment discharge is 199 $\mathrm{g\,m^{-2}}$ across all tree monocultures, among which *Ch. axillaris*, *C. glauca*, *R. chinensis*, and *K. bipinnata* showed above average, and *M. yuyuanensis*, *L. glaber*, *E. chinensis*, and *L. formosana* below average sediment discharge. The growth characteristics of these tree species differed considerably between the species (Table 2).

3.3 Effects of tree functional traits and site characteristics

Crown cover was highly correlated with LAI, tree height, stem diameter, and crown width ($r = 0.82$, 0.80, 0.75, 0.77, respectively). Crown cover ($p < 0.01$) and LAI ($p < 0.05$) negatively affected sediment discharge. Tree height marginally positively affected sediment discharge ($p < 0.1$), whereas stem diameter and crown width had no influence (Fig. 4, Appendix Table A2). The soil surface cover consisted of stones and biological soil crusts and canopied on average one fifth of the ROP surfaces in May and June 2013. It affected sediment discharge negatively ($p < 0.001$). Mean soil organic matter content in the top layer was high and reduced sediment discharge ($p < 0.05$). An indication of hydrophobic surface coatings and a significant role of water repellency could not be found. The mean slope angle did not affect sediment discharge (Fig. 4, Appendix Table A2).

Growth characteristics were highly variable between tree species, which was reflected by high standard deviations of the respective variables. In contrast, site characteristics of these plots showed a low variability (Table 3).

3.4 Interrill erosion in young Chinese subtropical forests

Event-based mean sediment discharge increased with peak intensity from precipitation event 1 to event 4 with 42 g m^{-2} (E1), 85 g m^{-2} (E2), 120 g m^{-2} (E3) and 283 g m^{-2} (E4). The interrill soil erosion rate determined by micro-scale ROPs and extrapolated for erosive precipitation events (> 12.7 mm rainfall amount) was estimated to be 47.5 t ha^{-1} in 2013.

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4 Discussion

4.1 Species richness effects on interrill erosion processes

Tree species richness did not affect sediment discharge or runoff volume, and thus the first hypothesis has to be rejected. Nevertheless, a trend was visible from diversity level 0 to 8, where sediment discharge and runoff were decreasing. However, both parameters were nearly the same at diversity level 1 and 24 and standard deviations were high. Whereas tree growth patterns in monocultures were highly variable, mixed stands indicated a more balanced development (Kelty, 2006). All species mixtures in this experiment assured a minimum dimension of tree height and ground coverage after four to five years of tree growth, whereas in monocultures the canopy cover was highly tree species specific. Thus, we needed to exclude several monoculture plots before measurements, because some species could not provide any considerable ground coverage. At the same time, sediment discharge in mixture stands was lower than in monocultures. Nevertheless, contrasts in the model could not show any statistical difference between monocultures and mixtures or bare and covered plots.

The absence of a species richness effect on soil erosion is likely attributable to the early successional stage of the forest experiment with low tree ages. Full canopy covers with high stratification and overlap have not yet been developed at the study site, and the trees did by far not reach terminal height (Goebes et al., 2015b; Li et al., 2014). We assume that these vegetation characteristics will change with increasing tree age and tree species richness may become evident in adult stands. Young trees are functionally more equivalent than older trees (Barnes and Spurr, 1998) and specific crown traits may emerge more distinctly in later successional stages. Geißler et al. (2013) found that the erosion potential was higher in medium and old grown forests than in young forests. This effect is caused by raindrop transformation processes during the canopy passage, resulting in higher throughfall kinetic energy under forest than on fallow land (Geißler et al., 2010) and has only been proved for advanced successional forest stages (Nanko et al., 2008; Geißler et al., 2013). With ongoing time of the experiment and increasing tree height we expect increasing throughfall kinetic energy, which in turn increases the general soil erosion potential.

4.2 Species identity effects on interrill erosion processes

Trees in monocultures differed in their impact on soil erosion and thus hypothesis 2 can be confirmed. In a study on common European tree species, Augusto et al. (2002) showed that the tree species composition of forests has an impact on chemical, physical and biological soil properties. Several studies revealed that individual plants are important for erosion control in arid and semi-arid Mediterranean landscapes (e.g. Bochet et al., 2006; cf. Durán Zuazo and Rodríguez Pleguezuelo, 2008) and Xu et al. (2008) showed that different plant morphologies may control soil loss and improved soil properties in a dry river valley in China. In our study, four tree species (*Ch. axillaris*, *C. glauca*, *R. chinensis*, *K. bipinnata*) seemed to foster soil erosion rates, whereas another four species (*M. yuyuanensis*, *L. glaber*, *E. chinensis*, *L. formosana*) showed a mitigating effect on soil erosion at this initial stage of the forest ecosystem. Thus, we can confirm a species-specific effect on sediment discharge for our subtropical experimental area. Species-specific effects can result from different throughfall kinetic energy, which was recently shown by Goebes et al. (2015a) at the same study site in China. The effect of throughfall kinetic energy was ascribed to different tree architectural characteristics and leaf traits. The authors found three out of 11 tree species to have distinct differences in mean throughfall kinetic energy. *Ch. axillaris* and *S. saponaria* showed higher values, whereas *S. superba* was characterized by lower values of throughfall kinetic energy. On our experimental site, varying tree species revealed heterogeneous growth patterns, which were caused by species-specific growth variation and abiotic site conditions (Li et al., 2014). *Ch. axillaris* was the tallest tree species with a nearly closed canopy and caused the highest amount of sediment discharge in our study. Raindrops falling from leaves of this species nearly reached terminal velocity and hence throughfall kinetic energy was high (Morgan, 2005; Goebes et al., 2015a). This finding explained the high erosion rates below this fast-growing species. Further stands with significantly higher erosion rates and the four tree species with a mitigating effect on soil erosion showed lower tree heights and thus lower throughfall kinetic energy. Their effect on sediment discharge has to be explained by further functional traits.

4.3 Effects of tree functional traits and site characteristics

Tree species differed widely in canopy characteristics, and sediment discharge was significantly related to crown cover and LAI. Therefore, the species-specific effects of soil erosion can be partially contributed to species-specific functional traits, which confirms hypothesis 3. The falling velocities of throughfall drops are highly variable under different tree species due to the species-specific growth pattern and crown characteristics (Goebes et al., 2015a). Frasson and Krajewski (2011) showed that the mechanisms of interception are manifold even within a single canopy and varying canopy levels create different drop size distributions. Increasing crown cover and LAI were mitigating soil erosion in this early ecosystem stage. The magnitude of canopy cover determines the proportion of raindrops intercepted (Blanco-Canqui and Lal, 2008). Anyhow, LAI showed a weaker significance than crown cover, probably because many trees had not yet developed a multi-layered canopy structure. As trees did not yet reach adult height (mean height < 2 m) in this study, the kinetic energy of raindrops formed at leaf tips was lower than in grown up tree stands and drops did not reach terminal velocities (Morgan, 2005; Geißler et al., 2013; Goebes et al., 2015a). Therefore, tree height had a weak effect on sediment discharge ($p < 0.1$) in our study, and sediment discharge rates under trees were not exceeding those on bare ground. Nevertheless, high sediment discharge under *Ch. axillaris*, by far the fastest growing tree in our experiment, showed the potential of high trees to increase soil erosion on uncovered forest floors. Stem diameter and crown width did not seem to influence erosion processes in early stage forest ecosystems. Several other tree-related functional traits (Pérez-Harguindeguy et al., 2013) could be used to explain sediment discharge such as branching architecture, specific leaf area, and root system morphology. Especially studies on leaf traits (Nanko et al., 2013) as well as belowground stratification (Gyssels et al., 2005; Stokes

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et al., 2009) showed the potential to influence soil loss and pointed out the complexity of factors mitigating soil erosion in forest ecosystems.

Our results showed that soil surface cover and soil organic matter also affect interrill erosion. Even though a leaf litter cover was not present in this experiment, the remaining soil surface cover by stones and biological soil crusts was the most important driver to reduce sediment discharge. This finding underlines the general importance of covered soil surfaces for erosion control (cf. Thornes, 1990; Morgan, 2005) and shows that the protecting effect of leaf litter could not only be replaced by soil skeleton but also by topsoil microbial communities in young forest stands. The mitigating effect of leaf litter on soil losses has not been in the focus of this experimental approach, but it is presumed that the fall of leaves even in young aged forests reduces soil erosion considerably compared to bare land (Blanco-Canqui and Lal, 2008; Seitz et al., 2015). Furthermore, soil organic matter effectively prevented soil erosion by binding primary particles into aggregates (Blanco-Canqui and Lal, 2008). If soil organic matter increases with increasing species richness, as it was recently demonstrated in a grassland study by Cong et al. (2014), an indirect effect of biodiversity on soil erosion can be presumed. At last, slope angle was not affecting soil erosion due to the short plot length that limits runoff velocities (cf. Seitz et al., 2015).

4.4 Interrill erosion in young Chinese subtropical forests

The soil loss rate determined by our micro-scale ROPs ($47.5\,\mathrm{t\,ha^{-1}\,a^{-1}}$) for 2013 was considerably higher than the average rate Guo et al. (2015) recently calculated for South China (approx. $20\,\mathrm{t\,ha^{-1}\,a^{-1}}$) in a study based on small-scale and field ROPs. Pimentel (1993) reported an average rate of $36\,\mathrm{t\,ha^{-1}\,a^{-1}}$ for the same area. Zheng et al. (2007) stated an average soil loss rate of $31\,\mathrm{t\,ha^{-1}\,a^{-1}}$ determined with $^{137}Cs\,/\,^{210}Pb$ tracing techniques in Sichuan Province, PR China.These different rates are due to different land use types and measurement techniques, but also to the scale-dependent nature of soil erosion and runoff generation (cf. Boix-Fayos et al., 2006; Cantón et al., 2011). In an event-based approach, Zhu and Zhu (2014) pointed out that ROPs with short slope length yield higher sediment discharge than those with longer slope length and Bagarello and Ferro (2004) showed that increasing the size of ROPs from 0.04 to 0.16 m^2 decreased runoff and sediment discharge by a factor of 2.6. The micro-scale ROPs used in our study (0.16 m^2) quantified interrill wash and sediment detachment by raindrop impact (cf. Agassi and Bradford, 1999). However, an important part of erosion appears in the rilling system, and the influence of interrill processes on soil erosion varies greatly (Govers and Poesen, 1988). Sediment discharge and runoff change with ROP length (cf. Abrahams et al., 1995) and boundary effects increasingly influence the results with decreasing plot sizes (Mutchler et al., 1994). Nevertheless, Mutchler et al. (1994) stated that micro-scale ROPs are suitable to study basic aspects of soil erosion and further, those measurements are particularly appropriate to define impacts of vegetation by interplot comparison (Wainwright et al., 2000). A major advantage of micro-scale ROPs is the possibility to implement a high number of replications to tackle measurement variability (Wendt et al., 1986). A high number of ROPs in turn requires great efforts in maintenance and control, which are easier to ensure with plots of small scale and small-sized reservoirs (Boix-Fayos et al., 2006).

5 Synthesis and conclusions

An experiment with 170 micro-scale runoff plots was conducted to investigate the influence of tree species richness and identity as well as tree functional traits on soil erosion processes in a young forest ecosystem. Based on our findings we come to the following conclusions:

1. Tree species richness did not affect sediment discharge and runoff. Although a negative trend was visible from level 1 to 8 and mixed stands showed a more balanced and homogenous vegetation development than monocultures, higher tree species richness is not mitigating soil erosion. We ascribe this effect to the young successional stage of the forest experiment. Future research should concentrate

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on how erosion rates change with increasing stand age. Therefore, long-term monitoring of soil erosion under closing tree canopies is necessary.

2. This study provided evidence that different tree species affect interrill erosion processes. Different tree morphologies have to be considered, when regarding erosion in young forest ecosystems. The appropriate choice of tree species for afforestation against soil erosion becomes already important in an early successional stage. *Chorespondeas axillaris*,*Cyclobalanopsis glauca*, *Rhus chinensis*, and *Koelreuteria bipinnata* were related to increasing soil erosion rates, whereas *Magnolia yuyuanensis*, *Lithocarpus glaber*, *Elaeocarpus chinensis*, and *Liquidambar formosana* can help to mitigate soil erosion in young forest stands.

3. Species-specific functional traits and site characteristics affected soil erosion rates. High crown cover and leaf area index reduced soil erosion, whereas it was slightly increased by increasing tree height. Thus, low tree stands with high canopy cover are effectively counteracting soil loss in initial forest ecosystem. In further studies, a wider range of functional tree traits like leaf habitus or below-ground stratification should be taken into consideration. Even if a leaf litter cover was not present in this experiment, the remaining soil surface cover by stones and biological soil crusts was the most important driver for soil erosion control. Furthermore, soil organic matter had a decreasing influence on sediment discharge. Investigations on the influence of topsoil microbial communities and their impact on organic matter accumulation will open the way to new insights on soil erosion processes.

Author contributions. T. Scholten, P. Kühn and S. Seitz designed the experiment and S. Seitz carried it out. S. Seitz, P. Goebes and H. Bruelheide developed the model code and performed the statistics. Y. Li and W. Härdtle provided data on tree growth and species-specific functional traits. S. Seitz prepared the manuscript with contributions from all co-authors.

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Table 1. Characteristics of rainfall events considered erosive (threshold 12.7 mm) in Xingangshan, Jiangxi Province, PR China in May and June 2013.

Event	Mean intensity ($mm\,h^{-1}$)	Peak intensity ($mm\,h^{-1}$)	Total rainfall amount (mm)
E 1	1.38	11.4	20.29
E 2	2.34	23.04	25.74
E 3	3.19	45.24	54.42
E 4	14.60	83.04	34.01

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Table 2. Discharge rates and growth characteristics (means) of tree species with significant differences in sediment discharge at the experimental site in Xingangshan, Jiangxi Province, PR China.

	Sediment discharge ($g\,m^{-2}$)	Crown cover (%)	Leaf area index	Tree height (m)	Stem diameter (m)	Crown width (m)
Mean	199	32	0.75	1.84	0.03	0.94
Monocultures	202	22	0.63	1.63	0.02	0.78
Tree mixtures	135	44	1.18	2.30	0.04	1.26
Ch. axillaris	566	90	2.27	7.40	0.07	2.21
C. glauca	556	51	0.93	1.25	0.02	0.65
R. chinensis	502	47	0.85	1.82	0.03	1.62
K. bipinnata	378	19	0.30	1.97	0.03	1.15
M. yuyuanensis	64	11	0.14	1.62	0.04	0.95
L. glaber	114	20	0.28	2.32	0.03	1.09
E. chinensis	66	64	1.02	2.19	0.05	0.97
L. formosana	91	15	0.19	2.28	0.04	1.64

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Table 3. Growth characteristics of the 20 tree species analysed and associated plot characteristics in Xingangshan, Jiangxi Province, PR China (mean, standard deviation (SD), maximum (max) and minimum (min)).

	Mean	SD	Max	Min
Vegetation				
Crown cover (%)	37	31	93	1
Leaf area index (LAI)	0.88	1.08	4.20	0.03
Tree height (m)	2.55	1.64	7.40	1.16
Stem diameter (m)	0.04	0.02	0.07	0.02
Crown width (m)	1.25	0.61	3.00	0.40
Site				
Soil surface cover (%)	16	14	55	1
Bulk soil density ($g\,cm^{-3}$)	0.99	0.05	1.12	0.91
Soil organic matter (%)	6.4	1.4	9.4	4.3
pH	3.68	0.24	4.39	3.25
Slope (°)	27	5	35	19

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Table A1. 22 selected tree species used in the experiment according to the Flora of China (http://www.efloras.org). Asterisks (*) mark species which only appear in mixtures.

Species name and author	
Ailanthus altissima (Miller) Swingle	*Koelreuteria bipinnata* Franch.
Alniphyllum fortunei (Hemsl.) Makino	*Liquidambar formosana* Hance
Betula luminifera H. Winkl.	*Lithocarpus glaber* (Thunb.) Nakai
Castanea henryi (Skan) Rehd. et Wils.	*Magnolia yuyuanensis* Hu
Castanopsis fargesii Franch.	*Nyssa sinensis* Oliver*
Castanopsis sclerophylla (Lindl.) Schott.	*Rhus chinensis* Mill.
Choerospondias axillaris (Roxb.) Burtt et Hill.	*Sapindus saponaria* Gaertn.
Cyclobalanopsis glauca (Thunb.) Oerst.	*Schima superba* Gardn. et Champ.
Elaeocarpus chinensis Gardn. et Chanp.	*Triadica sebifera* (L.) Roxb.
Elaeocarpus glabripetalus Merr.	*Quercus fabri* Hance
Elaeocarpus japonicus Sieb. et Zucc.	*Quercus phillyreoides* A. Gray*

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Table A2. Results of the basic linear mixed effect model for sediment discharge. Crown cover was highly correlated with the four other vegetation characteristics and therefore, they have been exchanged and fitted in separate models.

		denDF	F	Pr		estimates
Fixed effects	Surface runoff	204	49.0	< 0.001	***	(–) 0.33
	Crown cover	120	7.25	0.008	**	(–) 0.18
	Slope	141	1.33	0.250	n.s.	(–) 0.05
	Surface cover	140	56.1	< 0.001	***	(–) 0.46
	Soil organic matter	42	5.61	0.022	*	(–) 0.07
	Precipitation	70	0.12	0.733	n.s.	(–) 0.01
	Tree species richness	25	0.30	0.589	n.s.	(–) 0.05
		SD	variance			
Random effects	Precipitation event: plot	0.204	0.042			
	Tree composition	0.332	0.110			
	Site	0.577	0.333			
	Plot: rop	0.503	0.253			
Vegetation characteristics fitted in exchange to crown cover						
	Leaf area index	95	5.16	0.026	*	(–) 0.17
	Tree height	31	3.58	0.069	•	(–) 0.10
	Tree stem diameter	30	0.20	0.661	n.s.	(–) 0.04
	Tree crown width	31	0.79	0.383	n.s.	(–) 0.08

*** $p < 0.001$, ** $p < 0.01$, * $p < 0.05$, • $p < 0.1$, n.s. not significant; $n = 334$.

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Table A3. Mean sediment discharge in $g\,m^{-2}$ and surface runoff volume in $L\,m^{-2}$ (standard deviation in brackets, $n = 334$) for tree species richness in May and June 2013.

	Diversity 0–24	Diversity 0	Diversity 1–24	Diversity 1	Diversity 8	Diversity 16	Diversity 24
Sediment discharge	199 (106)	361 (187)	188 (90)	202 (105)	103 (57)	135 (123)	204 (107)
Runoff volume	32.6 (21.4)	47.8 (32.1)	29.8 (18.5)	31.9 (20.9)	27.5 (14.5)	22.5 (15.7)	30.2 (19.7)

Figure 1. Measurement setup showing a runoff plot (ROP, 0.4 m × 0.4 m) with reservoir and rainfall gauge on the experimental site in Xingangshan, Jiangxi Province, PR China.

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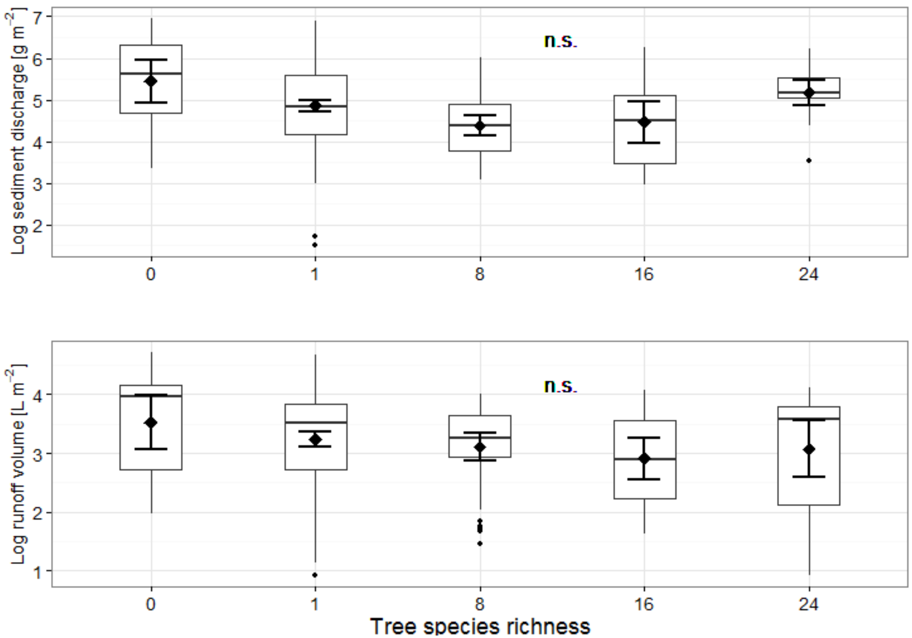

Figure 2. Sediment discharge and runoff volume at five diversity levels based on four rainfall events in May and June 2013 in Xingangshan, Jiangxi Province, PR China (n.s.: not significant, $n = 334$). Horizontal line within boxplot represents median and diamond represents mean.

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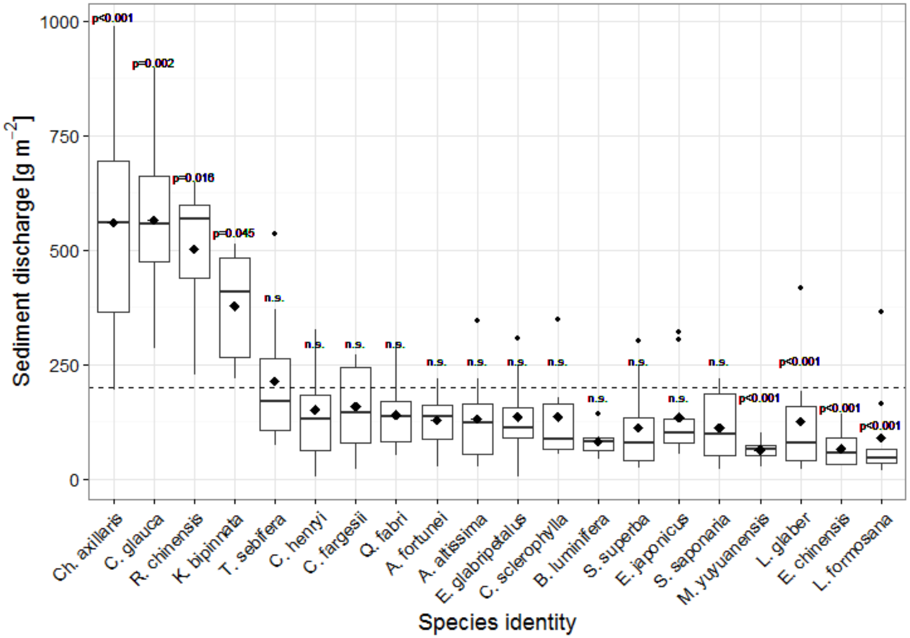

Figure 3. Sediment discharge under 20 tree species in monocultures based on four rainfall events in May and June 2013 in Xingangshan, Jiangxi Province, PR China. Dashed line indicates mean sediment discharge of all 20 species. Horizontal lines within boxplot represent median and diamonds represent mean values found for a respective species.

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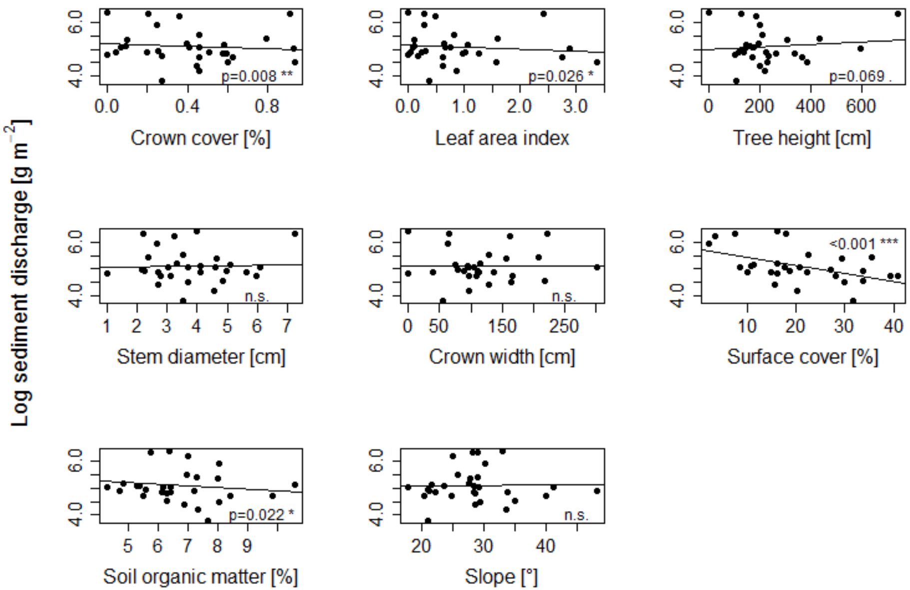

Figure 4. Effects of tree functional traits and site characteristics on sediment discharge. Analyses were based on four rainfall events in May and June 2013 in Xingangshan, Jiangxi Province, PR China. Black lines symbolize linear trends.

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