

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

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16

17 **Abstract**

18 Soil erosion is seriously threatening ecosystem functioning in many parts of the world. In this
19 context, it is assumed that tree species richness and functional diversity of tree communities
20 can play a critical role in improving ecosystem services such as erosion control. An
21 experiment with 170 micro-scale runoff plots was conducted to investigate the influence of
22 tree species richness and identity as well as tree functional traits on interrill erosion in a young
23 forest ecosystem. An interrill erosion rate of $47.5 \text{ Mg ha}^{-1} \text{ a}^{-1}$ was calculated. This study

24 provided evidence that different tree species affect interrill erosion differently, while tree
25 species richness did not affect interrill erosion in young forest stands. Thus, different tree
26 morphologies have to be considered, when assessing soil erosion under forest. High crown
27 cover and leaf area index reduced interrill erosion in initial forest ecosystems, whereas rising
28 tree height increased it. Even if a leaf litter cover was not present, remaining soil surface
29 cover by stones and biological soil crusts was the most important driver for soil erosion
30 control. Furthermore, soil organic matter had a decreasing influence on interrill erosion.
31 Long-term monitoring of soil erosion under closing tree canopies is necessary and a wide
32 range of functional tree traits should be considered in future research.

33

34 **1 Introduction**

35 Soil erosion is considered as one of the most severe environmental challenges globally
36 (Morgan, 2005). It is also a serious challenge in the PR China, especially in the southern
37 tropical and subtropical zone. Although important improvements in erosion control have been
38 achieved in this area in the last decades (Zhao et al., 2013), the annual soil loss rates range
39 between 0.28 Mg ha⁻¹ and 113 Mg ha⁻¹ (Guo et al., 2015). Thereby, soil erosion is negatively
40 affecting e.g. soil fertility or nutrient cycling (Pimentel et al., 1995; Richter, 1998).

41 Moreover, soil erosion can negatively influence biodiversity (Pimentel and Kounang, 1998),
42 but it is assumed that this relationship also acts vice versa (Körner and Spehn, 2002; Geißler
43 et al., 2012b; Brevik et al., 2015). It has been shown that a change in biodiversity can have
44 remarkable effects on ecosystem functions and stability (e.g. Hooper et al., 2005; Scherer-
45 Lorenzen, 2005). In many cases, increasing biodiversity enhanced ecosystem productivity and
46 stability (Loreau, 2001; Jacob et al., 2010). In particular, tree species richness as well as
47 functional diversity of tree communities can play a critical role in improving ecosystem

48 services such as water filtration or climate regulation (Quijas et al., 2012; Chisholm et al.,
49 2013; Scherer-Lorenzen, 2014). As forests are generally considered beneficial for erosion
50 control, afforestation is a common measure of soil protection (Romero-Diaz et al., 2010; Jiao
51 et al., 2012). This also applies to the south-eastern part of China, which is known as a hotspot
52 of biodiversity and woody plants (Barthlott et al., 2005; Bruelheide et al., 2011). Guo et al.
53 (2015) showed that forests in this area experienced the lowest soil loss rates of all land use
54 types in China. Considering that studies on soil erosion under forest have mostly focused on
55 deforestation (Blanco-Canqui and Lal, 2008) and counteracting measures such as afforestation
56 generally result in monoculture stands (Puettmann et al., 2009), it appears that the role of tree
57 species richness for soil erosion has been largely disregarded. Zhou et al. (2002) and
58 Tsujimura et al. (2006) demonstrated that tree monocultures have only limited mitigation
59 potential for soil losses, but further research is scarce. Nevertheless, there is growing evidence
60 that higher species richness can reduce soil erosion (Körner and Spehn, 2002). Bautista et al.
61 (2007) pointed out that an increase in functional diversity within a perennial vegetation cover
62 decreased soil losses in a semiarid Mediterranean landscape. Pohl et al. (2009) showed that an
63 increase in the diversity of root types led to higher soil stability on an alpine grassy hillslope
64 and most recently Berendse et al. (2015) found that a loss of grass species diversity reduced
65 erosion resistance on a dike slope.

66 Conceivable mechanisms underlying positive species richness effects on soil erosion are that
67 vegetation covers with a high number of species include a high number of plant functional
68 groups which complement one another. Thus, they are more effective in controlling erosion
69 processes than vegetative covers with few species (Pohl et al., 2012). For example, a high tree
70 species richness may result in an increased stratification of canopy layers (Lang et al., 2010).
71 As a consequence, crown overlap, biomass density and total canopy cover often are higher in
72 mixtures than in monocultures (Lang et al., 2012). In addition, a highly diverse structure

73 within the leaf litter layer on the forest floor seems to improve its protecting effect (Martin et
74 al., 2010). Recently, Seitz et al. (2015) pointed out that sediment discharge depends on the
75 species identities in the leaf litter cover, whereas there was no effect of leaf species richness
76 or functional diversity on soil erosion. Further research on the influence of tree species
77 richness on erosion control seems to be necessary, but the complex system of interacting
78 functional groups within the vegetation cover is also of great interest.

79 Vegetation covers are generally considered a key factor for the occurrence and dimension of
80 soil erosion (Thornes, 1990; Hupp et al., 1995; Morgan, 2005). A leaf litter layer on the forest
81 floor, for example, protects the soil from direct raindrop impact and modifies the water flow
82 and storage capacities on the soil surface (Kim et al., 2014). Moreover, forests can provide a
83 multi-storey canopy layer which largely influences rain throughfall patterns and leads to the
84 capture of raindrops as well as the storage of water within the tree crown (Puigdefábregas,
85 2005). Nevertheless, large drops can be formed at leaf apexes of tall trees (Geißler et al.,
86 2012a) and thus may increase the kinetic energy of throughfall in older forest stands up to a
87 factor of 2 to 3 compared to open fields (Nanko et al., 2008; Nanko et al., 2015). This leads to
88 considerable soil loss if the forest floor is unprotected, which may be the case if protecting
89 layers diminish e.g. under shady conditions (Onda et al., 2010) or fast decomposition
90 (Razafindrabe et al., 2010). Whereas the effects of soil surface covers on soil erosion is well
91 studied (Thornes, 1990; Blanco-Canqui and Lal, 2008), much less is known about the
92 influence of species-specific functional traits of the tree layer such as crown or stem
93 characteristics (Lavorel and Garnier, 2002; Guerrero-Campo et al., 2008). Moreover, most
94 research on the latter aspects was performed in old-grown forests (e.g. Zhou et al., 2002;
95 Nanko et al., 2008; Geißler et al., 2012a), whereas forests in an early-successional stage are
96 rarely mentioned. In those young forests, tree heights are lower than in later stages, but
97 structural and spatial complexity is high and species-specific growth rates differ considerably

98 (Swanson et al., 2011). It is assumed that these species-specific differences in structure and
99 growth will influence soil erosion rates.

100 This research focused on the influence of tree species richness, tree species identity and
101 species-specific functional traits on interrill erosion in young forests, when a leaf litter cover
102 is not present. Testing for those effects on soil erosion requires a common garden situation, in
103 which confounding factors such as different tree ages and sizes, inclination or soil conditions
104 are closely controlled. These requirements were met in the forest biodiversity-ecosystem
105 functioning experiment in subtropical China (BEF-China; cf. Bruelheide et al., 2014). Within
106 this experiment, 170 micro-scale runoff plots were established in a randomly dispersed and
107 replicated design. Thereby, the following hypotheses were postulated:

- 108 1. Increasing tree species richness decreases interrill erosion rates.
- 109 2. Tree species differ in their impact on interrill erosion rates.
- 110 3. The effects of different tree species on interrill erosion rates can be explained by
111 species-specific functional traits.

112

113 **2 Methodology**

114 **2.1 Study site and experimental design**

115 The study was conducted in Xingangshan, Jiangxi Province, PR China (29°06.450' N and
116 117°55.450' E) at the experimental sites A and B of the BEF China project (Bruelheide et al.,
117 2014). Together, both sites comprise an area of about 50 ha in a mountainous landscape with
118 an elevation range from 100 m to 265 m a.s.l.. Slopes range from 15 ° to 41 °. The bedrock of
119 the experimental site consists of non-calcareous slates with varying sand and silt contents and
120 is intermittent by siliceous-rich joints. Prevailing soil types are Cambisols with Anthrosols in

121 downslope positions and Gleysols in valleys (cf. IUSS, 2006) covering saprolites. Soil bulk
122 density is low (0.98 g cm^{-3}) and soil reaction acidic (mean pH in KCl 3.68). Soil texture
123 ranges from silt loam to silty clay loam. The climate in Xingangshan is humid and subtropical
124 and ranked as Cwa after the Köppen-Geiger classification. It is characterized by an annual
125 average temperature of $17.4 \text{ }^{\circ}\text{C}$ and a mean annual rainfall of 1635 mm (Goebes et al.,
126 2015b).

127 The experimental area has been used as a commercial forest plantation (*Cunninghamia*
128 *lanceolata* and *Pinus massoniana*) until 2007. It was clear-cut and replanted in 2009-2010
129 following an experimental plot-based design with different extinction scenarios (Bruehlheide et
130 al., 2014). The experimental site represented an early successional stage with tree ages from
131 four to five years at the time of measurements. In total, 566 experimental plots were
132 established using a pool of 40 native tree species, as well as bare ground and free succession
133 plots. Trees were planted randomly in seven species richness levels (div0, 1, 2, 4, 8, 16, 24)
134 with a planting distance of 1.29 m, following a broken stick design (Bruehlheide et al., 2014).
135 This study focused on the Very Intensively studied Plots (VIPs) of which 34 were used in this
136 study. The monocultures with tree heights lower than 1 m or crown covers less than 10 %
137 were excluded from the analysis. The selected set comprised a bare ground feature ($4 \times \text{div}0$)
138 and four levels of tree species richness ($20 \times \text{div}1$, $4 \times \text{div}8$, $4 \times \text{div}16$ and $2 \times \text{div}24$) with a
139 total of 22 tree species, two of which only appeared in mixtures (Appendix Table A1).

140 **2.2 Erosion measurements**

141 To determine initial sediment discharge and surface runoff, micro-scale runoff plots (ROP,
142 $0.4 \text{ m} \times 0.4 \text{ m}$) were used (cf. Seitz et al., 2015; without fauna treatment). Each ROP was
143 connected to a 20 L reservoir and a rainfall gauge was placed next to it (Fig. 1). All 34 VIPs
144 were equipped with five ROPs each, resulting in a total number of 170 ROPs. Within each
145 VIP, areas of 220 m^2 were sectioned for ROP measurements to avoid interferences with other

146 BEF China experiments. Those selected areas were representative for the range of surface
147 properties in the plot and the ROPs were placed randomly therein. All leaf litter was removed
148 from the ROPs prior to measurements. The ROPs were operated in May and June 2013 during
149 the rainy season. Runoff volume and rainfall amount were determined in situ and sediment
150 was assessed after sampling by drying at 40 °C and weighing. The capacity of the reservoirs
151 was not exceeded in any rainfall event.

152 [Figure 1]

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

162 **2.3 Rainfall patterns**

163 Weather conditions were recorded by an on-site climate station (ecoTech datalogger with
164 Vaisala weather transmitter and ecoTech tipping bucket balance) in 5-min intervals. The total
165 precipitation in the study area in 2013 was 1205 mm and lower than the mean of the
166 preceding three years (1635 mm). Of this amount, a fraction of 957 mm (33 events) were
167 strong enough to trigger soil erosion following Wischmeier and Smith (1978) who used an
168 event threshold of 12.7 mm. This threshold was confirmed by Yin et al. (2007) to be valid for
169 southeast China. In total, 10 rainfall events were captured at the study area in May and June.

170 Four of those events (E1 - E4) can be considered erosive. The total rainfall amount from May
171 to June was 185 mm, of which 135 mm fell during erosive rainfall events. The mean and peak
172 intensities as well as the total rainfall amount (except for E4) increased from May to June
173 (Table 1), reflecting a growing monsoon influence from beginning to mid-summer.

174 [Table 1]

175 **2.4 Statistical analysis**

176 Linear mixed effects models with restricted maximum likelihood were performed with R
177 3.0.2 (R Core Team, 2013) and “lmerTest” (Kuznetsova et al., 2014) to investigate the
178 influences on sediment discharge. Models were fitted with crown cover, leaf area index, tree
179 height, stem diameter, crown width, slope, surface cover, SOM, amount of precipitation and
180 tree species richness as fixed effects. As random effects, precipitation event nested in plot,
181 tree composition, site and ROP nested in plot were used. Tree and crown characteristics were
182 fitted one after the other, because they were highly correlated. Contrasts of diversity levels
183 (div0 to div1-24, div1 to div8-24) were introduced to quantify the effects of bare plots vs. tree
184 plots and tree monocultures vs. mixtures, respectively. The effect of individual tree species
185 (div1) was tested separately against the mean sediment discharge using crown cover, slope,
186 surface cover, SOM and amount of precipitation as fixed factors and site and ROP nested in
187 plot as random factor (n=200). The maximum likelihood approach was used to obtain model
188 simplification by step-wise backward selection, eliminating the least significant variable
189 except for tree species richness. If multicollinearity was detected (spearman $\rho > 0.7$), co-
190 variables were omitted. All variables were continuous and scaled, so model estimates could be
191 compared. The data was log-transformed and the residuals did not show any deviation from
192 normality. Hypotheses were tested with an ANOVA type 3 with Satterthwaite approximation
193 for degrees of freedom and p-values were obtained by likelihood ratio tests.

194

195 **3 Results**

196 The results were based on 334 ROP measurements out of a total of 378 measurements.
197 Sediment discharge over all VIPs and rainfall events ranged from 14 g m⁻² to 920 g m⁻² per
198 ROP. Event-based mean sediment discharge increased with peak intensity from precipitation
199 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
200 interrill soil erosion rate determined by micro-scale ROPs and extrapolated for all erosive
201 precipitation events (>12.7 mm rainfall amount) in 2013 was estimated to be 47.5 Mg ha⁻¹.

202 **3.1 Species richness effects on interrill erosion processes**

203 Tree species richness did not affect sediment discharge or runoff (Table 2 and Fig. 2).
204 Sediment discharge tended to decrease from diversity level 0 to 8 and to increase to diversity
205 level 24, while runoff volume tended to decrease from diversity level 0 to 16 and to increase
206 to diversity level 24, but shifts were non-significant. Sediment discharge and runoff volume
207 did not differ between bare plots (div0) and plots with trees (div1-div24), just as between
208 monocultures (div1) and species mixtures (div8, div16, div24). The standard deviations of
209 sediment discharge (g m⁻²) and runoff volume (l m⁻²) in relation to diversity levels were high
210 (Fig. 2 and Table 3). Mean crown cover in mixed stands was 44 % and mean tree height was
211 2.30 m compared to monocultures with 22 % and 1.63 m. In this experiment tree height in
212 mixed stands was not lower than 1.07 m and crown cover achieved at least 29 %.

213 [Table 2]

214 [Figure 2]

215 [Table 3]

216 **3.2 Species identity effects on interrill erosion processes**

217 Individual tree species in monocultures affected sediment discharge differently (Fig. 3) and
218 sediment discharge rates ranged from 90 g m⁻² (*L. formosana*) to 560 g m⁻² (*Ch. axillaris*) per
219 rainfall event.

220 [Figure 3]

221 The mean sediment discharge is 199 g m⁻² across all tree monocultures, among which *Ch.*
222 *axillaris*, *C. glauca*, *R. chinensis* and *K. bipinnata* showed above average and *M. yuyuanensis*,
223 *L. glaber*, *E. chinensis* and *L. formosana* below average sediment discharge. The growth
224 characteristics of these tree species differed considerably between the species (Table 4).

225 [Table 4]

226 **3.3 Effects of tree functional traits and site characteristics**

227 Crown cover was highly correlated with LAI, tree height, stem diameter and crown width
228 ($r=0.82, 0.80, 0.75, 0.77$, respectively). Crown cover ($p<0.01$) and LAI ($p<0.05$) negatively
229 affected sediment discharge. Tree height marginally positively affected sediment discharge
230 ($p<0.1$), whereas stem diameter and crown width had no influence (Fig. 4, Table 2). The soil
231 surface cover consisted of stones and biological soil crusts and covered on average one fifth
232 of the ROP surfaces in May and June 2013. It affected sediment discharge negatively
233 ($p<0.001$). Mean soil organic matter content in the top layer was high and reduced sediment
234 discharge ($p<0.05$). An indication of hydrophobic surface coatings and a significant role of
235 water repellency could not be found. The mean slope angle did not affect sediment discharge
236 (Fig. 4, Table 2).

237 [Figure 4]

238 Growth characteristics were highly variable between tree species, which was reflected by high
239 standard deviations of the respective variables. In contrast, site characteristics of these plots
240 showed a low variability (Table 5).

241 [Table 5]

242

243 **4 Discussion**

244 The soil loss rate determined by micro-scale ROPs ($47.5 \text{ Mg ha}^{-1} \text{ a}^{-1}$) for 2013 was
245 considerably higher than the average rate Guo et al. (2015) recently calculated for South
246 China (approx. $20 \text{ Mg ha}^{-1} \text{ a}^{-1}$) in a study based on small-scale and field ROPs. Pimentel
247 (1993) reported an average rate of $36 \text{ Mg ha}^{-1} \text{ a}^{-1}$ for the same area. Zheng et al. (2007) stated
248 an average soil loss rate of $31 \text{ Mg ha}^{-1} \text{ a}^{-1}$ determined with $^{137}\text{Cs}/^{210}\text{Pb}$ tracing techniques in
249 Sichuan Province, PR China. These different rates are due to different land use types and
250 measurement techniques, but also due to the scale-dependent nature of soil erosion and runoff
251 generation (cf. Boix-Fayos et al., 2006; Cantón et al., 2011). The micro-scale ROPs used in
252 this study quantified interrill wash and sediment detachment by raindrop impact (Agassi and
253 Bradford, 1999; cf. Cerdà, 1999; Parsons et al., 2003; García-Orenes et al., 2012). However,
254 an important part of erosion appears in the rilling system and the influence of interrill
255 processes on soil erosion varies greatly (Govers and Poesen, 1988). Sediment discharge and
256 runoff change with ROP length (cf. Abrahams et al., 1995) and boundary effects increasingly
257 influence the results with decreasing plot sizes (Mutchler et al., 1994). Nevertheless, Mutchler
258 et al. (1994) stated that micro-scale ROPs are suitable to study basic aspects of soil erosion
259 and furthermore, those measurements are particularly appropriate to define impacts of
260 vegetation by interplot comparison (Wainwright et al., 2000).

261 **4.1 Species richness effects on interrill erosion processes**

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

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

286 time of the experiment and increasing tree height increasing throughfall kinetic energy is
287 expected, which in turn increases the general soil erosion potential.

288 **4.2 Species identity effects on interrill erosion processes**

289 Trees in monocultures differed in their impact on interrill erosion and thus hypothesis 2 can
290 be confirmed. In a study on common European tree species, Augusto et al. (2002) showed that
291 the tree species composition of forests has an impact on chemical, physical and biological soil
292 properties. Several studies revealed that individual plants are important for erosion control in
293 arid and semi-arid Mediterranean landscapes (e.g. Bochet et al., 2006; cf. Durán Zuazo and
294 Rodríguez Pleguezuelo, 2008) and Xu et al. (2008) showed that different plant morphologies
295 may control soil loss and improved soil properties in a dry river valley in China.

296 In this study, four tree species (*Ch. axillaris*, *C. glauca*, *R. chinensis*, *K. bipinnata*) seemed to
297 foster interrill erosion rates, whereas another four species (*M. yuyuanensis*, *L. glaber*, *E.*
298 *chinensis*, *L. formosana*) showed a mitigating effect on interrill erosion at this initial stage of
299 the forest ecosystem. Thus, a species-specific effect on sediment discharge for this subtropical
300 experimental area can be confirmed. Species-specific effects can result from different
301 throughfall kinetic energy, which was recently shown by Goebes et al. (2015a) at the same
302 study site in China. The effect of throughfall kinetic energy was ascribed to different tree
303 architectural characteristics and leaf traits. The authors found three out of 11 tree species to
304 have distinct differences in mean throughfall kinetic energy. *Ch. axillaris* and *S. saponaria*
305 showed higher values, whereas *S. superba* was characterized by lower values of throughfall
306 kinetic energy. At the experimental site, varying tree species revealed heterogeneous growth
307 patterns, which were caused by species-specific growth variation and abiotic site conditions
308 (Li et al., 2014). *Ch. axillaris* was the tallest tree species with a nearly closed canopy and
309 caused the highest amount of sediment discharge in this study. Raindrops falling from leaves
310 of this species nearly reached terminal velocity and hence throughfall kinetic energy was high

311 (Morgan, 2005; Goebes et al., 2015a). This finding explained the high erosion rates below this
312 fast-growing species. Further stands with significantly higher erosion rates and the four tree
313 species with a mitigating effect on interrill erosion showed lower tree heights and thus lower
314 throughfall kinetic energy. Their effect on sediment discharge has to be explained by further
315 functional traits.

316 **4.3 Effects of tree functional traits and site characteristics**

317 Tree species differed widely in canopy characteristics and sediment discharge was
318 significantly related to crown cover, LAI and tree height. Therefore, the species-specific
319 effects of interrill erosion can be partially contributed to species-specific functional traits,
320 which confirms hypothesis 3. The falling velocities of throughfall drops are highly variable
321 under different tree species due to the species-specific growth pattern and crown
322 characteristics (Goebes et al., 2015a). Frasson and Krajewski (2011) showed that the
323 mechanisms of interception are manifold even within a single canopy and varying canopy
324 levels create different drop size distributions.

325 Increasing crown cover and LAI were mitigating interrill erosion in this early ecosystem
326 stage. The magnitude of canopy cover determines the proportion of raindrops intercepted
327 (Blanco-Canqui and Lal, 2008) and it has been shown that drop size distributions differ
328 between different canopy species (Nanko et al., 2006). High crown cover and leaf area
329 increase the interception of rain drops and the storage capacity of water in the canopy (Aston,
330 1979; Geißler et al., 2012a), which can lead to higher stemflow and thus decreasing
331 throughfall (Herwitz, 1987). Nevertheless, Herwitz (1987) equally showed that canopy
332 drainage can lead to larger throughfall drops and thus to increasing throughfall kinetic energy
333 depending on the leaf species (Hall and Calder, 1993; Geißler et al., 2012a; Goebes et al.,
334 2015a). Anyhow, LAI showed a weaker significance than crown cover, probably because
335 many trees had not yet developed a multi-layered canopy structure.

336 It has been shown that tree height is an import factor for sediment detachment under forest
337 (Geißler et al., 2013), mostly due to increasing drop falling heights (Gunn and Kinzer, 1949).
338 As trees did not yet reach adult height (mean height <2 m) in this study, the kinetic energy of
339 raindrops formed at leaf tips was lower than in grown up tree stands and drops did not reach
340 terminal velocities (Morgan, 2005; Geißler et al., 2013; Goebes et al., 2015a). Therefore, tree
341 height had a weak effect on sediment discharge ($p < 0.1$) in this study and sediment discharge
342 rates under trees were not exceeding those on bare ground. Nevertheless, high sediment
343 discharge under *Ch. axillaris*, by far the fastest growing tree in this experiment, showed the
344 potential of high trees to increase soil erosion on uncovered forest floors.

345 Stem diameter and crown width did not seem to influence erosion processes in early stage
346 forest ecosystems. Several other tree-related functional traits (Pérez-Harguindeguy et al.,
347 2013) could be used to explain sediment discharge such as branching architecture, specific
348 leaf area and root system morphology. Especially studies on leaf traits (Nanko et al., 2013) as
349 well as belowground stratification (Gyssels et al., 2005; Stokes et al., 2009) showed the
350 potential to influence soil loss and pointed out the complexity of factors mitigating soil
351 erosion in forest ecosystems.

352 Results showed that soil surface cover and soil organic matter affect interrill erosion. Even
353 though a leaf litter cover was not present in this experiment, the remaining soil surface cover
354 by stones and biological soil crusts was the most important driver to reduce sediment
355 discharge. This finding underlines the general importance of covered soil surfaces for erosion
356 control (cf. Thornes, 1990; Morgan, 2005) and shows that the protecting effect of leaf litter
357 could not only be replaced by soil skeleton but also by topsoil microbial communities in
358 young forest stands. The mitigating effect of leaf litter on soil losses has not been in the focus
359 of this experimental approach, but it is presumed that the fall of leaves even in young aged
360 forests reduces soil erosion considerably compared to bare land (Blanco-Canqui and Lal,

361 2008; Seitz et al., 2015). Furthermore, soil organic matter effectively prevented interrill
362 erosion by binding primary particles into aggregates (Blanco-Canqui and Lal, 2008). If soil
363 organic matter increases with increasing species richness, as it was recently demonstrated in a
364 grassland study by Cong et al. (2014), an indirect effect of biodiversity on soil erosion can be
365 presumed. At last, slope angle was not affecting interrill erosion due to the short plot length
366 that limits runoff velocities (cf. Seitz et al., 2015).

367

368 **5 Synthesis and conclusions**

369 An experiment with 170 micro-scale runoff plots was conducted to investigate the influence
370 of tree species richness and identity as well as tree functional traits on interrill soil erosion
371 processes in a young forest ecosystem. The results led to the following conclusions:

- 372 1. Tree species richness did not affect sediment discharge and runoff, although a
373 negative trend was visible from diversity level 1 to 8 and mixed stands showed a
374 more balanced and homogenous vegetation development than monocultures. This
375 finding was ascribed to the young successional stage of the forest experiment.
376 Future research should concentrate on how erosion rates change with increasing
377 stand age. Therefore, long-term monitoring of soil erosion under closing tree
378 canopies is necessary.
- 379 2. This study provided evidence that different tree species affect interrill erosion
380 processes. Different tree morphologies have to be considered, when regarding
381 erosion in young forest ecosystems. The appropriate choice of tree species for
382 afforestation against soil erosion becomes already important in an early
383 successional stage.

384 3. Species-specific functional traits and site characteristics affected interrill erosion
385 rates. High crown cover and leaf area index reduced soil erosion, whereas it was
386 slightly increased by increasing tree height. Thus, low tree stands with high
387 canopy cover were effectively counteracting soil loss in initial forest ecosystem. In
388 further studies, a wider range of functional tree traits such as leaf habitus or
389 belowground stratification should be taken into consideration. Moreover,
390 investigations on the influence of biological soil crusts, topsoil microbial
391 communities and their impact on organic matter accumulation will open the way to
392 new insights on soil erosion processes.

393

394 **Appendices**

395 [Table A1]

396

397 **Author contribution**

398 Thomas Scholten, Peter Kühn and Steffen Seitz designed the experiment and Steffen Seitz
399 carried it out. Steffen Seitz, Philipp Goebes and Helge Bruelheide developed the model code
400 and performed the statistics. Ying Li and Werner Härdtle provided data on tree growth and
401 species-specific functional traits. Steffen Seitz prepared the manuscript with contributions
402 from all co-authors.

403

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710 **Table 1: Characteristics of rainfall events considered erosive (threshold 12.7 mm) in Xingangshan, Jiangxi Province,**
711 **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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732 **Table 2: Results of the basic linear mixed effect model for sediment discharge (*** : p<0.001, ** : p<0.01, * : p<0.05, .**
 733 **: p<0.1, n.s. : not significant; n=334). Crown cover was highly correlated with the four other vegetation characteristics**
 734 **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		
<u>Vegetation characteristics fitted in exchange to crown cover</u>					
	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

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742 **Table 3: Mean sediment discharge in g m⁻² and surface runoff volume in L m⁻² (standard deviation in brackets,**
 743 **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)

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763 **Table 4: Discharge rates and growth characteristics (means) of tree species with significant differences in sediment**
 764 **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
<i>Ch. axillaris</i>	566	90	2.27	7.40	0.07	2.21
<i>C. glauca</i>	556	51	0.93	1.25	0.02	0.65
<i>R. chinensis</i>	502	47	0.85	1.82	0.03	1.62
<i>K. bipinnata</i>	378	19	0.30	1.97	0.03	1.15
<i>M. yuyuanensis</i>	64	11	0.14	1.62	0.04	0.95
<i>L. glaber</i>	114	20	0.28	2.32	0.03	1.09
<i>E. chinensis</i>	66	64	1.02	2.19	0.05	0.97
<i>L. formosana</i>	91	15	0.19	2.28	0.04	1.64

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777 **Table 5: Growth characteristics of the 20 tree species analysed and associated plot characteristics in Xingangshan,**
 778 **Jiangxi Province, PR China (mean, standard deviation (sd), maximum (max) and minimum (min)).**

	Mean	Sd	Max	Min
<u>Vegetation</u>				
Crown cover (%)	37	31	93	1
Leaf area index	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
<u>Site</u>				
Soil surface cover (%)	16	14	55	1
Soil organic matter (%)	6.4	1.4	9.4	4.3
Slope (°)	27	5	35	19

779 **Crown cover: proportion of soil surface area covered by crowns of live trees (%), leaf area index: one-sided green leaf area per unit**
 780 **soil surface area (dimensionless), tree height: distance from stem base to apical meristem (m), stem diameter: cross-section**
 781 **dimension of the tree stem at 5 cm above ground (m), crown width: length of longest spread from edge to edge across the crown (m),**
 782 **soil surface cover: proportion of soil surface area covered by stones, biocrusts and litter (%), soil organic matter: fraction of organic**
 783 **carbon containing substances in the soil (%), slope: inclination (°).**

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794 Table A1: 22 selected tree species used in the experiment according to the Flora of China (<http://www.efloras.org>).

795 Asterisks (*) mark species which only appear in mixtures.

Species name and author

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

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

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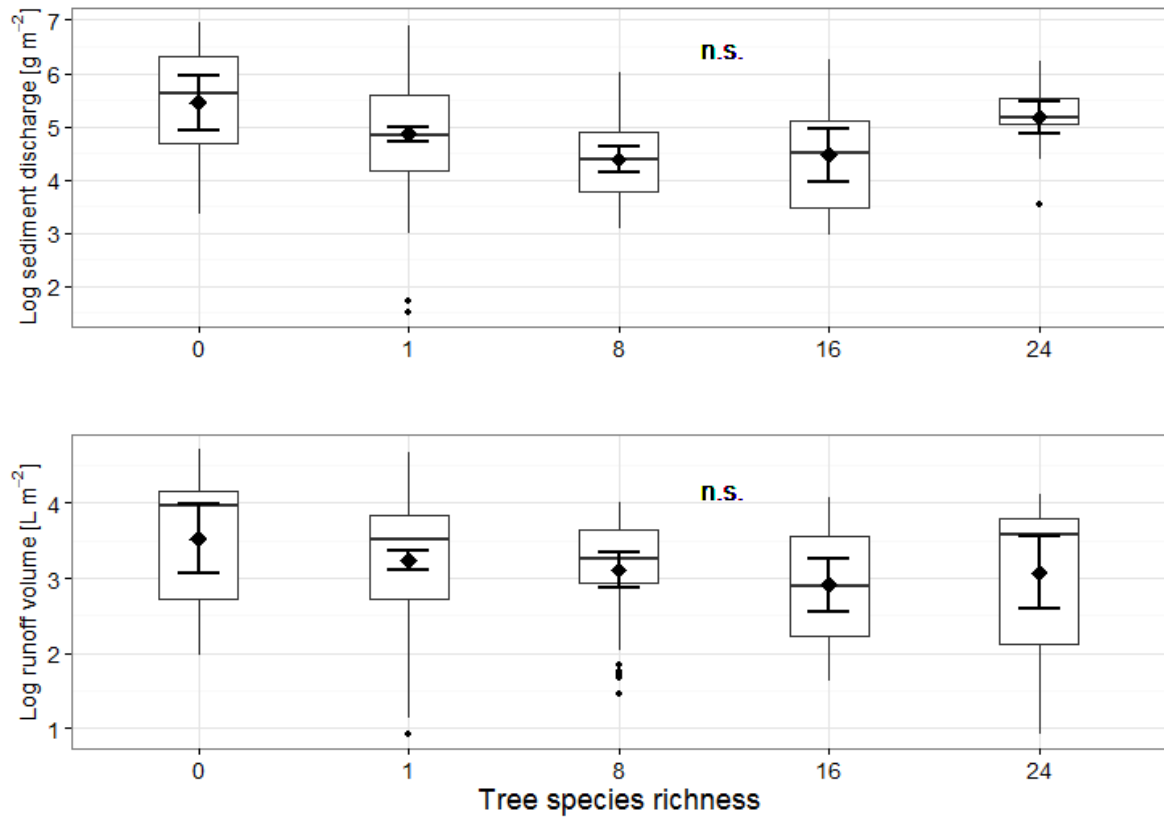
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821 **Figure 2: Sediment discharge and runoff volume at five diversity levels based on four rainfall events in May and June**
 822 **2013 in Xingangshan, Jiangxi Province, PR China (n.s.: not significant, n=334). Horizontal line within boxplot**
 823 **represents median and diamond represents mean.**

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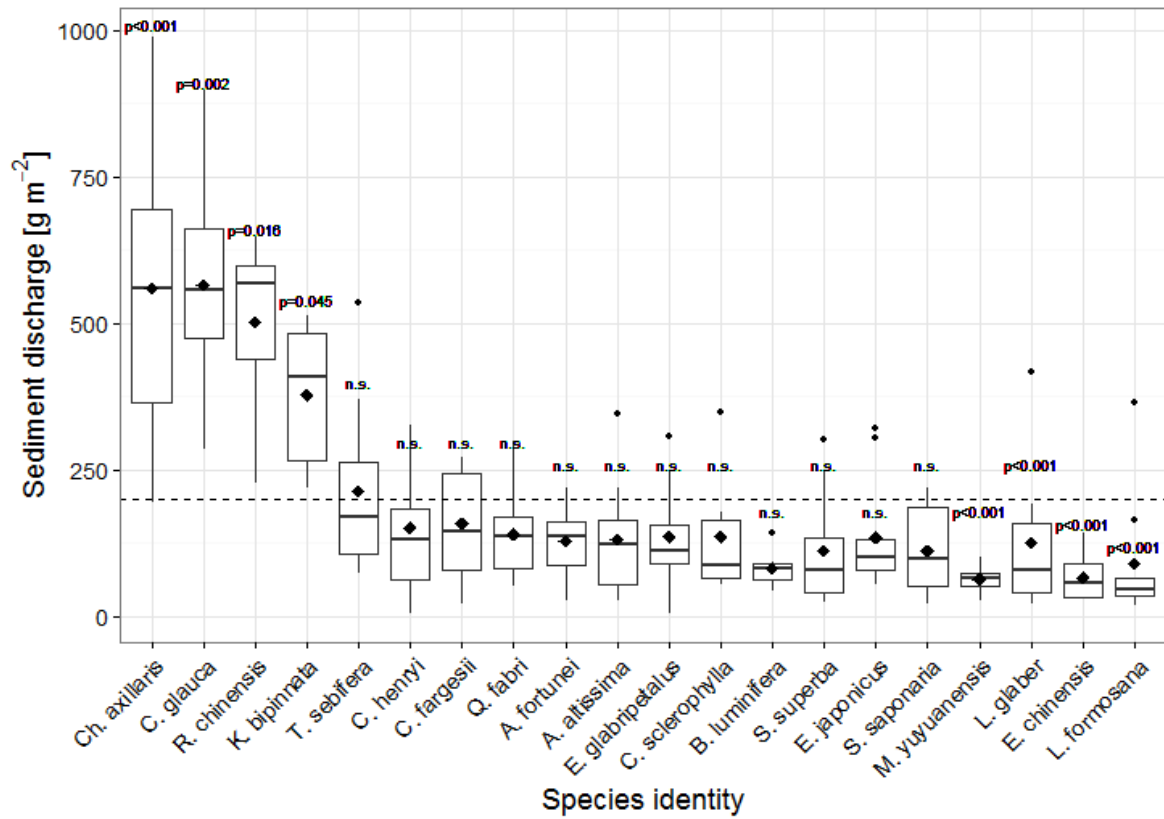
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836 Figure 3: Sediment discharge under 20 tree species in monocultures based on four rainfall events in May and June
 837 2013 in Xingangshan, Jiangxi Province, PR China. Dashed line indicates mean sediment discharge of all 20 species.
 838 Horizontal lines within boxplot represent median and diamonds represent mean values found for a respective species.

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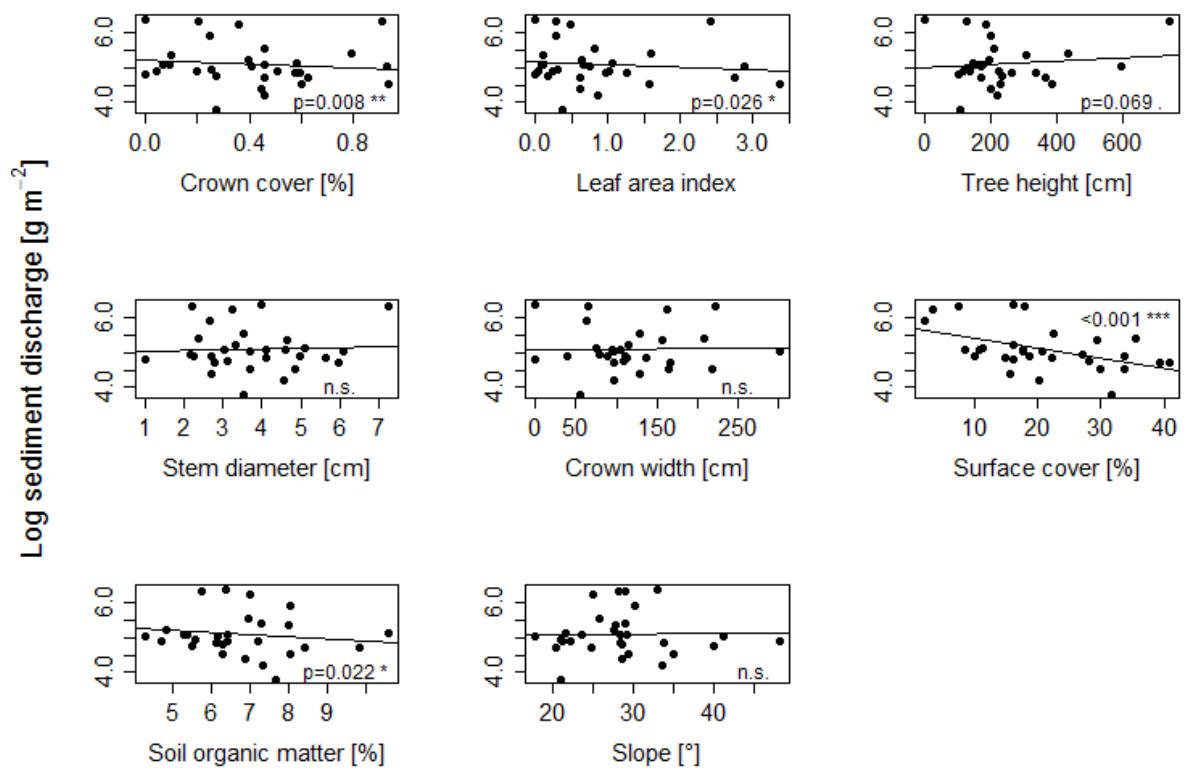
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851 **Figure 4: Effects of tree functional traits and site characteristics on sediment discharge. Analyses were based on four**
 852 **rainfall events in May and June 2013 in Xingangshan, Jiangxi Province, PR China. Black lines symbolize linear**
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