

1 **Tree species and functional traits but not species richness**
2 **affect interrill erosion processes in young subtropical**
3 **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 and tree species richness as well as 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 Soil erosion can negatively influence biodiversity (Pimentel and Kounang, 1998), but it is
42 assumed that this relationship also acts vice versa (Körner and Spehn, 2002; Geißler et al.,
43 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 (the diversity of
47 tree species) as well as functional diversity (the diversity of functional traits as morpho-

48 physiophenological attributes of a given species, cf. Violle et al., 2007) of tree communities
49 can play a critical role in improving ecosystem services such as water filtration or climate
50 regulation (Quijas et al., 2012; Chisholm et al., 2013; Scherer-Lorenzen, 2014). As forests are
51 generally considered beneficial for erosion control, afforestation is a common measure of soil
52 protection (Romero-Diaz et al., 2010; Jiao et al., 2012). This also applies to the south-eastern
53 part of China, which is known as a hotspot of biodiversity and especially tree species richness
54 (Barthlott et al., 2005; Bruelheide et al., 2011). Guo et al. (2015) showed that forests in this
55 area experienced the lowest soil loss rates of all land use types. Considering that studies on
56 soil erosion under forest have mostly focused on deforestation (Blanco-Canqui and Lal, 2008)
57 and counteracting measures such as afforestation often result in monoculture stands
58 (Puettmann et al., 2009), it appears that the role of tree species richness for soil erosion has
59 been largely disregarded. Zhou et al. (2002) and Tsujimura et al. (2006) demonstrated that
60 tree monocultures have only limited mitigation potential for soil losses, but further research is
61 scarce. Nevertheless, there is growing evidence that higher species richness can reduce soil
62 erosion (Körner and Spehn, 2002). Bautista et al. (2007) pointed out that an increase in
63 functional diversity within a perennial vegetation cover decreased soil losses in a semiarid
64 Mediterranean landscape. Pohl et al. (2009) showed that an increase in the diversity of root
65 types led to higher soil stability on an alpine grassy hillslope and most recently Berendse et al.
66 (2015) found that a loss of grass species diversity reduced erosion resistance on a dike slope.

67 Conceivable mechanisms underlying positive species richness effects on soil erosion are that
68 vegetation covers with a high number of species include a high number of plant functional
69 groups which complement one another. Thus, they are more effective in controlling erosion
70 processes than vegetative covers with few species (Pohl et al., 2012). For example, high tree
71 species richness can result in an increased stratification of canopy layers (Lang et al., 2010)
72 and a higher total canopy cover (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). Further research on the influence of tree species richness on erosion control
75 appears to be necessary, but the complex system of interacting functional groups within the
76 vegetation cover is also of great interest.

77 Vegetation covers are generally considered a key factor for the occurrence and dimension of
78 soil erosion (Thornes, 1990; Hupp et al., 1995; Morgan, 2005). A leaf litter layer on the forest
79 floor, for example, protects the soil from direct raindrop impact and modifies the water flow
80 and storage capacities on the soil surface (Kim et al., 2014). Moreover, forests can provide a
81 multi-storey canopy layer which largely influences rain throughfall patterns and leads to the
82 capture of raindrops as well as the storage of water within the tree crown (Puigdefábregas,
83 2005). Nevertheless, large drops can be formed at leaf apexes of tall trees (Geißler et al.,
84 2012a) and thus may increase the kinetic energy of throughfall in older forest stands up to a
85 factor of 2 to 3 compared to open fields (Nanko et al., 2008; Nanko et al., 2015). This leads to
86 considerable soil loss if the forest floor is unprotected, which may be the case if protecting
87 layers diminish e.g. under shady conditions (Onda et al., 2010) or fast decomposition
88 (Razafindrabe et al., 2010). Whereas the effects of soil surface covers on soil erosion is well
89 studied (Thornes, 1990; Blanco-Canqui and Lal, 2008), much less is known about the
90 influence of species-specific functional traits of the tree layer such as crown or stem
91 characteristics (Lavorel and Garnier, 2002; Guerrero-Campo et al., 2008). Moreover, most
92 research on the latter aspects was performed in old-grown forests (e.g. Zhou et al., 2002;
93 Nanko et al., 2008; Geißler et al., 2012a), whereas forests in an early-successional stage are
94 rarely mentioned. In those young forests, tree heights are lower than in later stages, but
95 structural and spatial complexity is high and species-specific growth rates differ considerably
96 (Swanson et al., 2011). It is assumed that these species-specific differences in structure and
97 growth will influence soil erosion rates.

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

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

110

111 **2 Methodology**

112 **2.1 Study site and experimental design**

113 The study was conducted in Xingangshan, Jiangxi Province, PR China (29°06.450' N and
114 117°55.450' E) at the experimental sites A and B of the BEF China project (Bruelheide et al.,
115 2014). Together, both sites comprise an area of about 50 ha in a mountainous landscape with
116 an elevation range from 100 m to 265 m a.s.l.. Slopes range from 15 ° to 41 °. The bedrock of
117 the experimental site consists of non-calcareous slates with varying sand and silt contents and
118 is intermittent by siliceous-rich joints. Prevailing soil types are Cambisols with Anthrosols in
119 downslope positions and Gleysols in valleys (cf. IUSS, 2006) covering saprolites. Soil bulk
120 density is low (0.98 g cm⁻³) and soil reaction acidic (mean pH in KCl 3.68). Soil texture
121 ranges from silt loam to silty clay loam. The climate in Xingangshan is humid and subtropical

122 and ranked as Cwa after the Köppen-Geiger classification. It is characterized by an annual
123 average temperature of 17.4 °C and a mean annual rainfall of 1635 mm (Goebes et al.,
124 2015b).

125 The experimental area has been used as a commercial forest plantation (*Cunninghamia*
126 *lanceolata* and *Pinus massoniana*) until 2007. It was clear-cut and replanted in 2009-2010
127 following an experimental plot-based design with different extinction scenarios (Bruelheide et
128 al., 2014). The experimental site represented an early successional stage with tree ages from
129 four to five years at the time of measurements. Trees were planted randomly in different
130 species richness levels with a planting distance of 1.29 m, following a broken stick design.
131 This study focused on the Very Intensively studied Plots (VIPs, cf. Bruelheide et al., 2014) of
132 which 34 were used (Table 1). The selected set comprised a bare ground feature (4 × div0)
133 and four levels of tree species richness (20 × div1, 4 × div8, 4 × div16 and 2 × div24) with a
134 total of 26 tree species, six of which only appeared in mixtures (Table 2). Monocultures with
135 tree heights lower than 1 m or crown covers less than 10 % were excluded from the analysis.

136 [Table 1]

137 [Table 2]

138 **2.2 Erosion measurements**

139 To determine sediment delivery (as initial interrill erosion) and surface runoff volume, micro-
140 scale runoff plots (ROP, 0.4 m × 0.4 m) were used (cf. Seitz et al., 2015; without fauna
141 treatment). Each ROP was connected to a 20 L reservoir and a rainfall gauge was placed next
142 to it (Fig. 1). All 34 VIPs were equipped with five ROPs each, resulting in a total number of
143 170 ROPs. Within each VIP, areas of 220 m² were sectioned for ROP measurements to avoid
144 interferences with other BEF China experiments. Those selected areas were representative for
145 the range of surface properties in the plot and the ROPs were placed randomly therein. All

146 leaf litter was removed from the ROPs prior to measurements. The ROPs were operated in
147 May and June 2013 during the rainy season. Runoff volume and rainfall amount were
148 determined in situ and sediment was assessed after sampling by drying at 40 °C and weighing.
149 The capacity of the reservoirs was not exceeded in any rainfall event.

150 [Figure 1]

151 At each ROP, tree crown cover, leaf area index (LAI), soil surface cover, slope and rainfall
152 amount were measured. Crown cover and LAI were determined using a fish-eye camera
153 system (Nikon D100 with Nikon AF G DX 180°) and the HemiView V.8 software (Delta-T
154 devices, Cambridge, UK) adjusted on the canopy area vertically above the ROP. Soil surface
155 cover was measured photogrammetrically (grid quadrat method with GIMP 2.8) and separated
156 into organic and inorganic covers by colour distinction. Slope was measured with an
157 inclinometer. Rainfall amount at each ROP was determined by rainfall gauges (see above). At
158 each VIP, total tree height, stem diameter at 5 cm above ground (hereafter, stem diameter)
159 and crown width were measured and calculated as the mean of 36 tree individuals per VIP (Li
160 et al., 2014). Additionally, soil organic matter (SOM) was identified for each VIP (5 cm
161 depth, 9 replicates) by measuring total organic carbon with a Vario EL III elemental analyser
162 (Elementar, Hanau, Germany) and multiplying it by the conversion factor 2 (Pribyl, 2010).
163 Tree species richness was known from the VIP-setup.

164 **2.3 Rainfall patterns**

165 Weather conditions were recorded by an on-site climate station (ecoTech datalogger with
166 Vaisala weather transmitter and ecoTech tipping bucket balance) in 5-min intervals. The total
167 precipitation at the study area in 2013 was 1205 mm and lower than the mean of the preceding
168 three years (1635 mm). In May and June, 10 rainfall events were captured with ROP
169 measurements at the study area. Events were determined by breaks in rainfall of at least 6

170 hours. Four of those events (E1 - E4) were strong enough to trigger soil erosion (out of 33
171 events over the entire year 2013) following Wischmeier and Smith (1978) who used an event
172 threshold of 12.7 mm. The total rainfall amount from May to June was 185 mm, of which 135
173 mm fell during erosive rainfall events. The mean and peak intensities as well as the total
174 rainfall amount (except for E4) increased from May to June (Table 3), reflecting a growing
175 monsoon influence from beginning to mid-summer.

176 [Table 3]

177 **2.4 Statistical analysis**

178 Linear mixed effects models with restricted maximum likelihood were performed with R
179 3.0.2 (R Core Team, 2013) and “lmerTest” (Kuznetsova et al., 2014) to investigate the
180 influences on sediment delivery. Models were fitted with crown cover, leaf area index, tree
181 height, stem diameter, crown width, slope, surface cover, SOM, amount of precipitation and
182 tree species richness as fixed effects. As random effects, precipitation event (E1 - E4) nested
183 in plot, tree composition (species pool), site (A or B) and ROP nested in plot were used.
184 Nesting was introduced to avoid pseudoreplication considering the degrees of freedom in our
185 hypotheses tests. Tree and crown characteristics were fitted one after the other, because they
186 were highly correlated. Contrasts of diversity levels (div0 to div1-24, div1 to div8-24) were
187 introduced to quantify the effects of bare plots vs. tree plots and tree monocultures vs.
188 mixtures, respectively. The effect of individual tree species (div1) was tested separately
189 against the mean sediment delivery using crown cover, slope, surface cover, SOM and
190 amount of precipitation as fixed factors and site and ROP nested in plot as random factor
191 (n=200). The maximum likelihood approach was used to obtain model simplification by step-
192 wise backward selection, eliminating the least significant variable except for tree species
193 richness. If multicollinearity was detected (spearman $\rho > 0.7$), co-variables were omitted. All
194 variables were continuous and scaled, so model estimates could be compared. The data was

195 log-transformed and the residuals did not show any deviation from normality. Hypotheses
196 were tested with an ANOVA type 3 with Satterthwaite approximation for degrees of freedom
197 and p-values were obtained by likelihood ratio tests.

198

199 **3 Results**

200 The results were based on 334 ROP measurements out of a total of 378 measurements. Invalid
201 measurements were caused by technical constraints such as plugged tubes or toppled rainfall
202 gauges. Sediment delivery over all VIPs and rainfall events ranged from 14 g m⁻² to 920 g m⁻²
203 per ROP. Event-based mean sediment delivery increased with peak intensity from
204 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
205 (E4). The interrill soil erosion rate determined by micro-scale ROPs and extrapolated for all
206 erosive precipitation events (>12.7 mm rainfall amount) in 2013 was estimated to be 47.5 Mg
207 ha⁻¹.

208 **3.1 Species richness effects on interrill erosion processes**

209 Tree species richness did not affect sediment delivery or runoff volume (Table 4 and Fig. 2).
210 Sediment delivery and runoff volume did not differ between bare plots (div0) and plots with
211 trees (div1-div24), just as between monocultures (div1) and species mixtures (div8, div16,
212 div24). The standard deviations of sediment delivery (g m⁻²) and runoff volume (l m⁻²) in
213 relation to diversity levels were high (Fig. 2 and Table 5). Mean crown cover in mixed stands
214 was 44 % and mean tree height was 2.30 m compared to monocultures with 22 % and 1.63 m.
215 In this experiment tree height in mixed stands was not lower than 1.07 m and crown cover
216 achieved at least 29 %.

217 [Table 4]

218 [Figure 2]

219 [Table 5]

220 **3.2 Species effects on interrill erosion processes**

221 Individual tree species in monocultures showed significant differences in sediment delivery
222 (Fig. 3) ranging from 90 g m⁻² (*L. formosana*) to 560 g m⁻² (*Ch. axillaris*) per rainfall event.

223 [Figure 3]

224 The mean sediment delivery is 199 g m⁻² across all tree monocultures, among which *Ch.*
225 *axillaris*, *C. glauca*, *R. chinensis* and *K. bipinnata* showed above average and *M. yuyuanensis*,
226 *L. glaber*, *E. chinensis* and *L. formosana* below average sediment delivery. The growth
227 characteristics of these tree species differed considerably between the species (Table 6).

228 [Table 6]

229 **3.3 Effects of species-specific functional traits and site characteristics**

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

239 [Figure 4]

240 Growth characteristics were highly variable between tree species, which was reflected by high
241 standard deviations of the respective variables. In contrast, site characteristics of these plots
242 showed a low variability (Table 7).

243 [Table 7]

244

245 **4 Discussion**

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

263 **4.1 Species richness effects on interrill erosion processes**

264 Tree species richness did not affect sediment delivery or runoff volume and thus the first
265 hypothesis has to be rejected. Nevertheless, a trend of decreasing sediment delivery and
266 runoff volume from diversity level 0 to 8 was visible. However, both parameters were nearly
267 the same at diversity level 1 and 24 and standard deviations were high. In contrast to tree
268 growth patterns in monocultures which were highly variable, mixed stands indicated a more
269 balanced development (cf. Kelty, 2006). All species mixtures in this experiment assured a
270 higher level of tree height and ground coverage after four to five years of tree growth,
271 whereas in monocultures the canopy cover was lower and highly tree species-specific. Thus,
272 several monoculture plots were excluded before measurements, because some species could
273 not provide any considerable ground coverage. At the same time, sediment delivery in 8- and
274 16-species mixtures was lower than in monocultures. Nevertheless, contrasts in the model
275 could not show any statistical difference between monocultures and mixtures or bare and
276 covered plots.

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

289 time of the experiment and increasing tree height increasing throughfall kinetic energy is
290 expected, which in turn increases the general soil erosion potential if an understory is missing.

291 **4.2 Species effects on interrill erosion processes**

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

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

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

319 **4.3 Effects of species-specific functional traits and site characteristics**

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

328 Increasing crown cover and LAI were mitigating interrill erosion in this early ecosystem
329 stage. The magnitude of canopy cover determines the proportion of raindrops intercepted
330 (Blanco-Canqui and Lal, 2008) and it has been shown that drop size distributions differ
331 between different canopy species (Nanko et al., 2006). High crown cover and leaf area
332 increase the interception of rain drops and the storage capacity of water in the canopy (Aston,
333 1979; Geißler et al., 2012a), which can lead to higher stemflow and thus decreasing
334 throughfall (Herwitz, 1987). Nevertheless, Herwitz (1987) equally showed that canopy
335 drainage can lead to larger throughfall drops and thus to increasing throughfall kinetic energy
336 depending on the leaf species (Hall and Calder, 1993; Geißler et al., 2012a; Goebes et al.,
337 2015a). Anyhow, LAI showed a weaker significance than crown cover, probably because
338 many trees had not yet developed a multi-layered canopy structure.

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

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

355 Results showed that soil surface cover and SOM affect interrill erosion. Even though a leaf
356 litter cover was not present in this experiment, the remaining soil surface cover by stones and
357 biological soil crusts was the most important driver to reduce sediment delivery. This finding
358 underlines the general importance of covered soil surfaces for erosion control (cf. Thornes,
359 1990; Morgan, 2005) and shows that the protecting effect of leaf litter could not only be
360 replaced by soil skeleton but also by topsoil microbial communities in young forest stands.
361 The mitigating effect of leaf litter on soil losses has not been in the focus of this experimental
362 approach, but it is presumed that the fall of leaves even in young aged forests reduces soil
363 erosion considerably compared to bare land (Blanco-Canqui and Lal, 2008; Seitz et al., 2015).

364 Furthermore, SOM reduced interrill erosion which could be explained by its ability to bind
365 primary particles into aggregates (Blanco-Canqui and Lal, 2008). If we assume that SOM
366 increases with increasing species richness, as it was recently demonstrated in a grassland
367 study by Cong et al. (2014), an indirect effect of biodiversity on soil erosion could be
368 supposed. At last, slope angle was not affecting interrill erosion due to the short plot length
369 that limits runoff velocities (cf. Seitz et al., 2015).

370

371 **5 Synthesis and conclusions**

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

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

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

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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712 **Table 1: Mean characteristics of the 34 selected Very Important study Plots (VIPs) in 2013 in the BEF China**
 713 **experiment, Xingangshan, Jiangxi Province, PR China.**

VIP no.	Species number	Crown cover (%)	Leaf area index	Tree height (m)	Stem diameter (m)	Crown width (m)	Slope (°)	Surface cover (%)	Soil organic matter (%)
F27	0	-	-	-	-	-	26	10	5.4
H28	0	-	-	-	-	-	34	15	5.9
L20	0	-	-	-	-	-	24	11	8.3
Q23	0	-	-	-	-	-	15	23	6.2
E31	1	16	0.19	1.25	0.02	0.80	22	39	5.5
E33	1	20	0.28	2.32	0.03	1.09	19	41	4.4
E34	1	87	2.07	5.96	0.06	3.00	21	11	6.1
I25	1	11	0.14	1.62	0.04	0.96	29	11	5.3
I28	1	15	0.19	2.28	0.04	1.64	26	32	8.9
K19	1	93	4.20	3.67	0.06	1.66	24	32	8.3
L11	1	10	0.11	1.36	0.02	0.90	28	19	7.1
M7	1	46	0.62	2.01	0.03	1.28	31	8	6.8
N05	1	9	0.10	1.16	0.03	0.40	32	0	6.3
N11	1	42	0.55	1.68	0.03	0.96	26	32	9.7
N13	1	13	0.13	3.05	0.05	1.56	31	30	7.9
N17	1	47	0.85	1.82	0.03	1.62	28	1	7.9
O27	1	90	2.27	7.40	0.07	2.21	21	9	5.7
Q13	1	19	0.30	1.97	0.03	1.15	30	1	6.9
Q27	1	24	0.47	3.37	0.04	1.37	35	3	6.0
R14	1	51	0.93	1.25	0.02	0.64	30	1	7.6
R29	1	21	0.24	1.44	0.03	0.95	33	18	6.3
U16	1	10	0.14	2.26	0.05	1.10	20	5	4.7
V24	1	64	1.02	2.19	0.05	0.96	32	11	4.3
W11	1	34	0.43	2.61	0.06	1.13	19	6	6.0
J29	8	29	0.34	1.47	0.05	0.76	31	13	9.4
Q17	8	30	0.37	1.74	0.05	1.05	22	6	5.2
S10	8	99	5.35	3.85	0.05	2.19	36	29	4.2
T15	8	31	0.38	1.96	0.03	1.15	30	20	4.8
M22	16	87	2.06	4.35	0.06	2.09	23	44	7.2
S22	16	34	0.42	1.07	0.04	0.56	33	24	6.6
U10	16	48	0.56	3.06	0.06	1.56	22	10	6.0
V27	16	42	0.54	2.09	0.05	0.99	34	9	6.4
N09	24	11	0.17	2.08	0.04	1.29	33	38	8.8
R30	24	37	0.46	1.67	0.04	0.97	27	19	4.2

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716 Table 2: 26 selected tree species used in the experiment according to the Flora of China (<http://www.efloras.org>).

717 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>Machilus grijsii</i> Hance *
<i>Castanopsis fargesii</i> Franch.	<i>Machilus leptophylla</i> Hand.-Mazz. *
<i>Castanopsis sclerophylla</i> (Lindl.) Schott.	<i>Magnolia yuyuanensis</i> Hu
<i>Celtis Biondi</i> Nakai *	<i>Nyssa sinensis</i> Oliver *
<i>Choerospondias axillaris</i> (Roxb.) Burtt et Hill.	<i>Rhus chinensis</i> Mill.
<i>Cyclobalanopsis glauca</i> (Thunb.) Oerst.	<i>Sapindus saponaria</i> Gaertn.
<i>Elaeocarpus chinensis</i> Gardn. et Champ.	<i>Schima superba</i> Gardn. et Champ.
<i>Elaeocarpus glabripetalus</i> Merr.	<i>Triadica sebifera</i> (L.) Roxb.
<i>Elaeocarpus japonicus</i> Sieb. et Zucc.	<i>Quercus fabri</i> Hance
<i>Idesia polycarpa</i> Maxim. *	<i>Quercus phillyreoides</i> A. Gray *

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731 **Table 3: Characteristics of rainfall events considered erosive (threshold 12.7 mm) in Xingangshan, Jiangxi Province,**
 732 **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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753 **Table 4: Results of the basic linear mixed effect model for sediment delivery (***: p<0.001, **: p<0.01, * : p<0.05, . :
754 p<0.1, n.s. : not significant; n=334). Crown cover was highly correlated with the four other vegetation characteristics
755 and therefore, they have been exchanged and fitted in separate models.**

		denDF	F	Pr	estimates
Fixed effects	Runoff volume	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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763 **Table 5: Mean sediment delivery in g m⁻² and surface runoff volume in L m⁻² (standard deviation in brackets, n=334)**
 764 **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 delivery	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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784 **Table 6: Sediment delivery and growth characteristics (means) of tree species with significant differences in delivery**
 785 **at the experimental site in Xingangshan, Jiangxi Province, PR China.**

	Sediment delivery (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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798 **Table 7: Growth characteristics of the 20 tree species in monocultures analysed and associated plot characteristics in**
 799 **Xingangshan, Jiangxi Province, PR China (mean, standard deviation (sd), maximum (max) and minimum (min)).**

	Mean	Sd	Max	Min
<i>Vegetation</i>				
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
<i>Site</i>				
Soil surface cover (%)	16	14	55	1
Soil organic matter (%)	6.4	1.4	9.4	4.3
Slope (°)	27	5	35	19

800 **Crown cover: proportion of soil surface area covered by crowns of live trees (%), leaf area index: one-sided green leaf area per unit**
 801 **soil surface area (dimensionless), tree height: distance from stem base to apical meristem (m), stem diameter: cross-section**
 802 **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),**
 803 **soil surface cover: proportion of soil surface area covered by stones, biocrusts and litter (%), soil organic matter: fraction of organic**
 804 **carbon containing substances in the soil (%), slope: inclination (°).**

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

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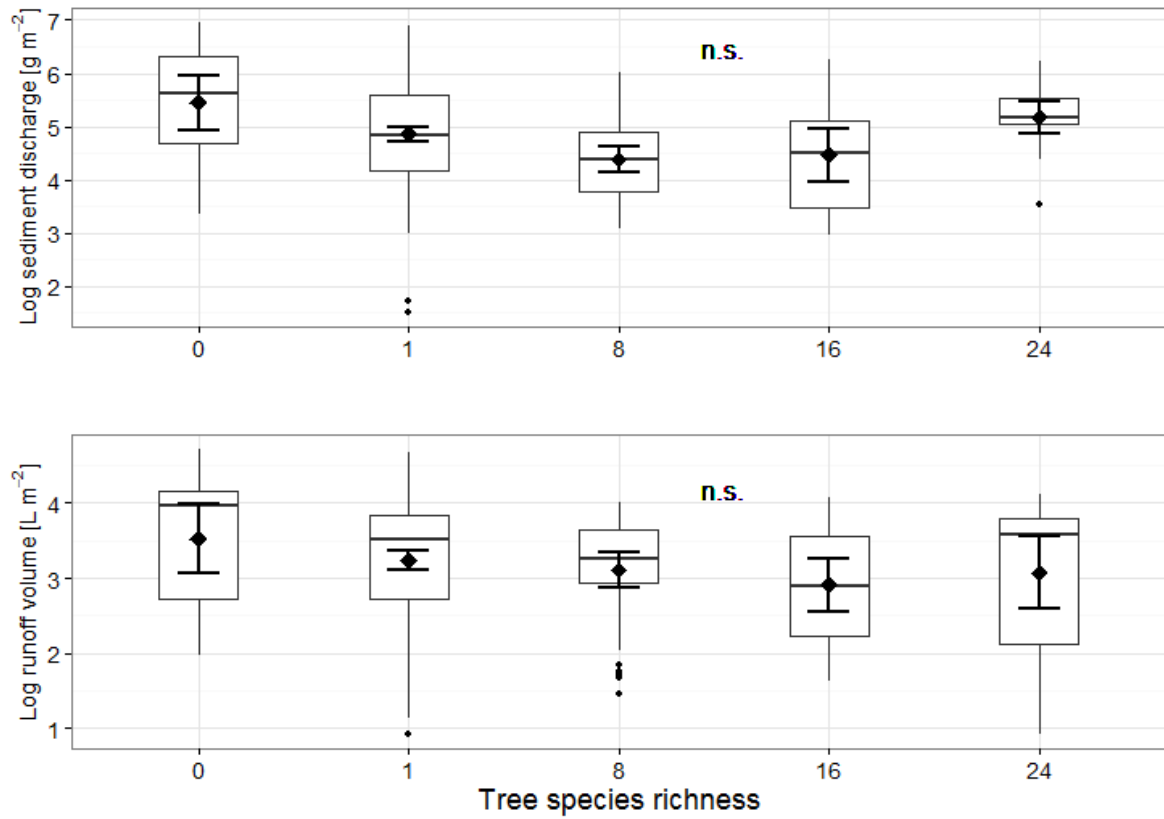
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827 **Figure 2: Sediment delivery and runoff volume at five diversity levels based on four rainfall events in May and June**
 828 **2013 in Xingangshan, Jiangxi Province, PR China (n.s.: not significant, n=334). Horizontal line within boxplot**
 829 **represents median and diamond represents mean.**

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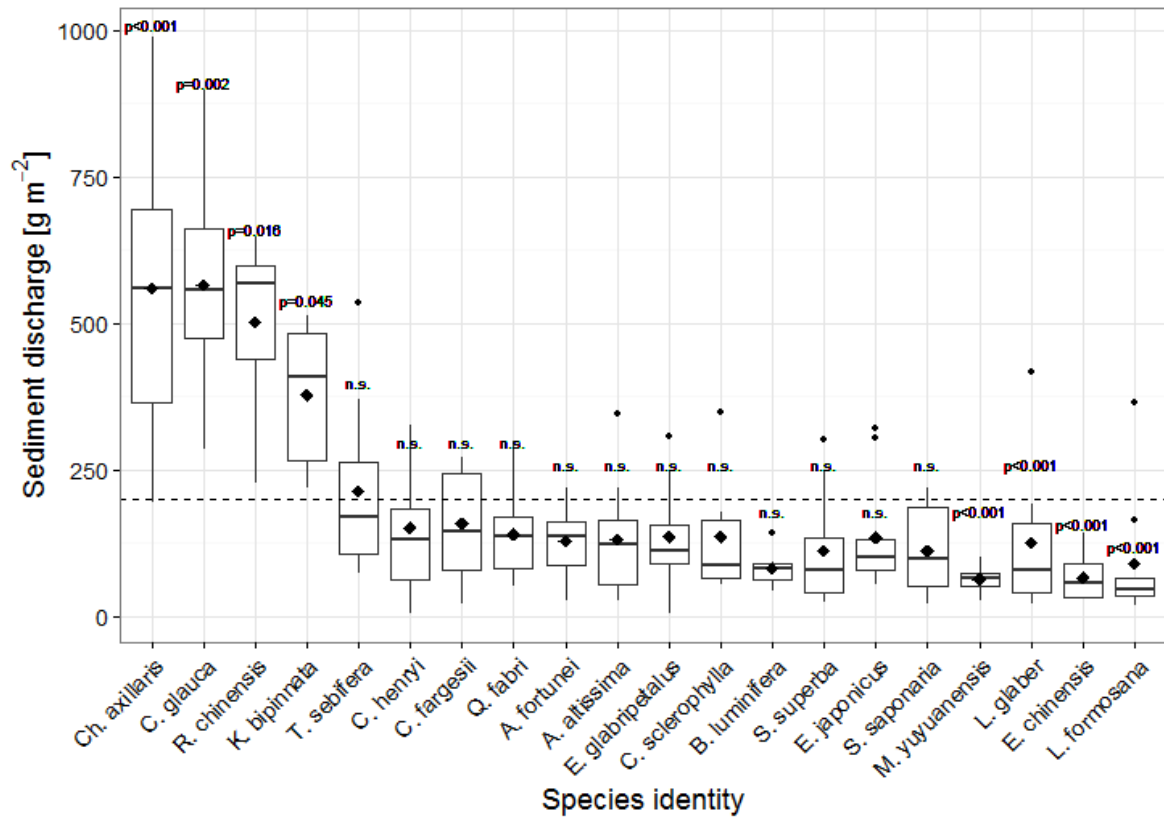
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842 **Figure 3: Sediment delivery under 20 tree species in monocultures based on four rainfall events in May and June 2013**
 843 **in Xingangshan, Jiangxi Province, PR China. Dashed line indicates mean sediment delivery of all 20 species.**
 844 **Horizontal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

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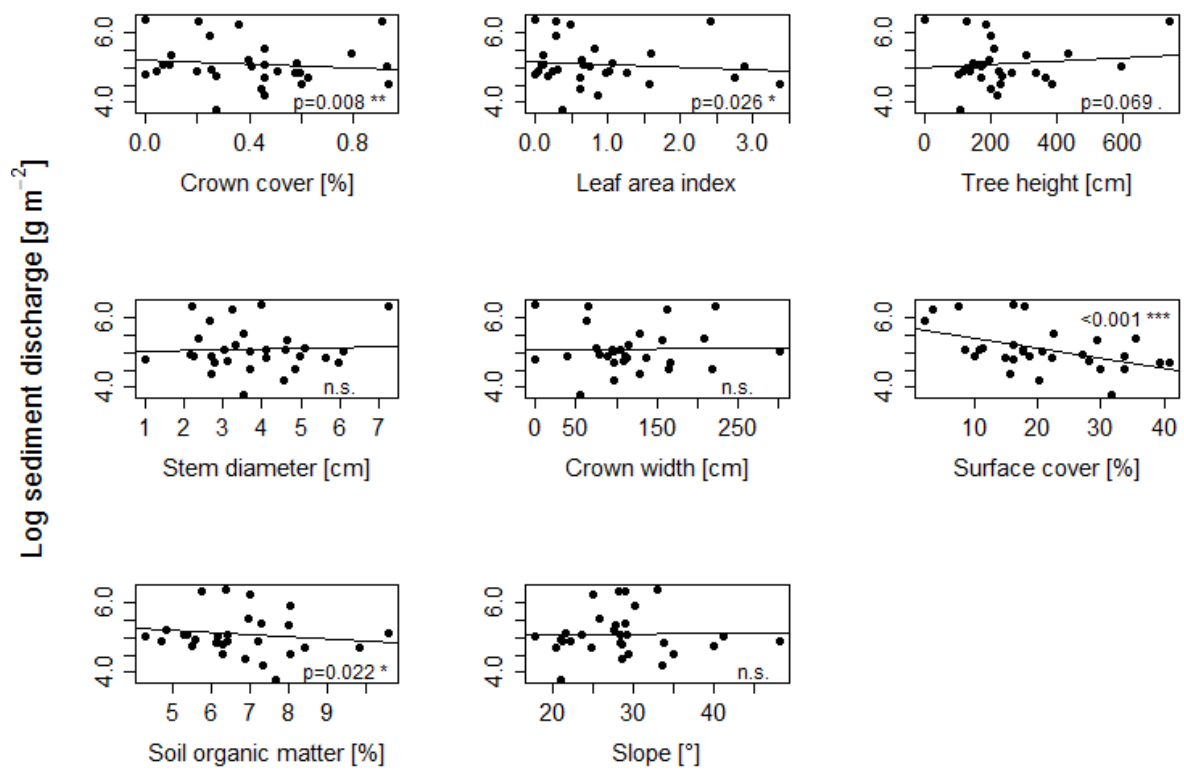
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857 **Figure 4: Effects of species-specific functional traits and site characteristics on sediment delivery. Analyses were based**
 858 **on four rainfall events in May and June 2013 in Xingangshan, Jiangxi Province, PR China. Black lines symbolize**
 859 **linear trends.**