Mechanisms of Soil Erosion in Subtropical Chinese Forests - Effects of Species Diversity, Species Identity, Functional Traits and Soil Fauna on Sediment Discharge

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Abbreviations

BEF	Biodiversity and Ecosystem Functioning
CBH	Crown Base Height
FAST	FArming System and Tillage experiment (Agroscope)
FKE	Freefall Kinetic Energy
LA	Leaf Area
LAI	Leaf Area Index
LME	Linear Mixed Effects (models)
NILEx	New Integrated Litter Experiment
ROP	RunOff Plot
SLA	Specific Leaf Area
SOM	Soil Organic Matter
TDR	Time Domain Reflectometry
TKE	Throughfall Kinetic Energy
VIP	Very Intensively studied Plot

Abstract

Soil erosion is a serious environmental problem in many parts of the world, especially in ecosystems with high anthropogenic influences. It is also a serious challenge in subtropical China, but forest stands mitigate soil loss rates in this area. Forests provide a multi-storey canopy layer which largely influences rain throughfall patterns as well as a leaf litter layer on the forest floor which protects the soil against direct raindrop impact and modifies the water flow and storage capacities. Nevertheless, only little research has been conducted on biodiversity and species effects on soil erosion control under forest stands. Furthermore, the processes within a protective leaf litter cover as well as the impact of soil mesofauna and macrofauna are not yet clear.

This thesis investigated the effects of species diversity, species identity, functional traits and soil fauna on soil erosion in subtropical forest ecosystems. It focused on interrill soil erosion rates determined by micro-scale ROPs under natural and simulated rainfall. Additionally, investigations with splash cups were carried out on changes in throughfall kinetic energy during the canopy passage of raindrops. Measurements took place in a forest biodiversity and ecosystem functioning experiment in the PR China (BEF China).

Results showed that tree species richness did not affect sediment discharge, runoff and TKE, although a negative trend was visible. This could be attributed to an absence of species richness effects on canopy characteristics in early successional forest stands. Nevertheless, stands with multiple species seemed to ensure a more balanced and homogenous soil cover. Furthermore, results showed that leaf litter species diversity did not influence the litter cover and thus soil erosion rates. Nonetheless, we could show that species identity influenced initial soil erosion processes under forest. That also applied to the leaf litter cover, where single leaf species showed significantly different influences on sediment discharge. Therefore, the appropriate choice of tree species during the establishment of plantations plays a major role in erosion control, even in young forest stages. Moreover, species-specific functional traits affected soil erosion rates. High crown cover and leaf area index reduced soil erosion, whereas it was slightly increased by increasing tree height. Investigations on the kinetic energy of raindrops showed that low LAI, low tree height, simple pinnate leaves, dentate leaf margins, a high number of branches and a low crown base height effectively minimized TKE. At last, the presence of soil mesofauna and macrofauna increased soil erosion and thus effects of this fauna group on sediment discharge have to be considered in soil erosion experiments.

Zusammenfassung

Die Bodenerosion stellt eines der weltweit bedeutendsten Umweltprobleme dar und tritt vor allem in Ökosystemen unter starker anthropogener Nutzung auf. Ein wesentlicher Einflussfaktor auf den Bodenabtrag ist hierbei die überdeckende Vegetation und insbesondere Waldökosysteme gelten als erosionsmindernd. Baumkronen beeinflussen die durch das Blätterdach fallenden Regentropfen und Laubstreu auf dem Boden schützt die Oberfläche gegen Abtrag. Biodiversitäts- und Arteffekte in der Baum- als auch in der bodenbedeckenden Laubschicht können hierbei eine Rolle spielen, wurden in der Forschung bislang aber wenig berücksichtigt.

Diese Dissertation liefert einen Beitrag zur Erforschung von Diversitäts-, Art- und Bodenfauna-Effekten auf initiale Bodenerosion in subtropischen Waldökosystemen. Hierzu wurden experimentelle Messungen mit kleinräumlichen Erosionsmessrinnen und Splash Cups durchgeführt. Neben der Nutzung des natürlichen Regenfalls erfolgte auch eine künstliche Beregnung mit einem mobilen Regensimulator. Die Messungen fanden in einem waldbaulichen Biodiversitätsexperiment in der Volksrepublik China (BEF China) statt.

In den Experimenten konnte kein Einfluss der Baum- und Blattdiversität, aber ein deutlicher Einfluss einzelner Baum- und Blattarten auf den Bodenabtrag festgestellt werden. Bei steigendem Diversitätsniveau war ein negativer Trend im Sedimentaustrag, im Oberflächenabfluss und der kinetischen Energie der Regentropfen zu verzeichnen. Es lag allerdings keine signifikante Einflussnahme vor, was mit dem frühen Sukzessionsstadium des untersuchten Waldökosystems begründet wird. Höher diverse Baumbestände zeigten eine ausgeglichenere und homogenere Bodenbedeckung als Monokulturen. Unterschiedliche Monokulturen und Blattarten unterschieden sich teilweise sehr deutlich in ihren Abtragsraten. Diese Unterschiede sind bei der Auswahl von Bäumen für Aufforstungen zu berücksichtigen und zeigen bereits in sehr jungen Sukzessionsstadien Wirkung. Weiterhin beeinflussten verschiedene funktionelle Gruppen der einzelnen Baumarten die Bodenerosion. Eine hohe Kronenüberdeckung und ein hoher Blattflächenindex führten zu geringeren Erosionsraten, während steigende Baumhöhen diese erhöhten. Die kinetische Energie der Regentropfen wurde von einem geringen Blattflächenindex, niedriger Baumhöhe, einfach gefiederten Blättern, gezähnten Blatträndern, einer hohen Anzahl an Ästen und geringer Kronenhöhe negativ beeinflusst. Außerdem führte das Auftreten von Bodenfauna in der Laubschicht zu einem erhöhten Sedimentabtrag.

List of publications and personal contribution

Accepted manuscripts

(1) SEITZ S, GOEBES P, SONG Z, BRUELHEIDE H, HÄRDTLE W, KÜHN P, LI Y, SCHOLTEN T. (2015): Tree species identity and functional traits but not species richness affect interrill erosion processes in young subtropical forests. *SOIL Discussions* **2**: 701-736. DOI: 10.5194/soild-2-701-2015

(2) SEITZ S, GOEBES P, ZUMSTEIN P, ASSMANN T, KÜHN P, NIKLAUS PA, SCHULDT A, SCHOLTEN T. (2015): The influence of leaf litter diversity and soil fauna on initial soil erosion in subtropical forests. *Earth Surface Processes and Landforms* **40**. 1439-1447. DOI: 10.1002/esp.3726

(3) GOEBES P, SEITZ S, KÜHN P, LI Y, NIKLAUS PA, VON OHEIMB G, SCHOLTEN T. (2015): Throughfall kinetic energy in young subtropical forests: Investigation on tree species richness effects and spatial variability. *Agricultural and Forest Meteorology* **213**: 148-159. DOI:10.1016/j.agrformet.2015.06.019

(4) GOEBES P, BRUELHEIDE H, HÄRDTLE W, KRÖBER W, KÜHN P, LI Y, SEITZ S, VON OHEIMB G, SCHOLTEN T. (2015): Species-specific effects on throughfall kinetic energy in subtropical forest plantations are related to leaf traits and tree architecture. *PLoS ONE* **10**(6): e0128084. DOI: 10.1371/journal.pone.0128084

Ready for submission manuscripts

(5) SEITZ S, GOEBES P, VAN DER HEIJDEN M, SONG Z, WITTWER R, SCHOLTEN T.: Soil erosion under organic and conventional farming and different tillage systems at a farming trial in Switzerland.

Share in publications

This thesis is a cumulative dissertation and thus based on publications carried out in teamwork. The personal contribution of the author to each publication is the following:

No.	Accepted for publication	Number of authors	Position of the candidate in list of authors	Scientific ideas of candidate	Data generation by candidate	Analysis and interpretation by candidate	Paper writing by candidate
(1)	yes	8	1	40	90	80	80
(2)	yes	8	1	40	50	60	70
(3)	yes	7	2	10	10	10	10
(4)	yes	9	2	10	10	5	15
(5)	no	6	1	50	50	60	90

1. Introduction and objectives

Soil erosion is a serious environmental problem in many parts of the world, especially in regions with high anthropogenic influences on the ecosystems (Morgan 2005). Although it is generally a natural process, soil erosion is often intensified by human impacts (Richter 1998). Soil erosion can reduce soil organic matter (SOM), diminish soil fertility or negatively affect the biodiversity of soil flora and fauna (Durán Zuazo and Rodríguez Pleguezuelo 2008; Montgomery 2007). Besides those on-site effects, off-site effects triggered by the transport of sediment and included nutrients as well as pollutants may cause high mitigation efforts and costs (Pimentel et al. 1995) and affect nutrient cycling and ecosystem functioning (Baumann et al. 2009; Zhao et al. 2009). Pimentel and Kounang (1998) stated that about 75 billion tons of soil are eroded at global scale every year and soil is lost 13 to 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 t ha^{-1} and 113 t ha^{-1}, depending on the precipitation, the topography and the land use (Guo et al. 2015). The degradation of soils will remain one of the principal environmental problems as an augmentation of extreme weather events and changes in mean precipitation are stated for the future (IPCC 2014; Xu et al. 2007; Lal 2003).

Processes of soil erosion

Factors that generally affect soil erosion rates are climate, topography, soil structure and composition as well as the vegetation cover. Losses of soil occur when effects of water, wind or gravity detach and translocate soil particles (Morgan 2005). Water is a principle agent of soil erosion and high intensity rain storms are responsible for a significant proportion of global soil losses (Blanco-Canqui and Lal 2008). Erosion by water can be divided into different stages from splash erosion to interrill and rill erosion (Richter 1998, Figure 1).

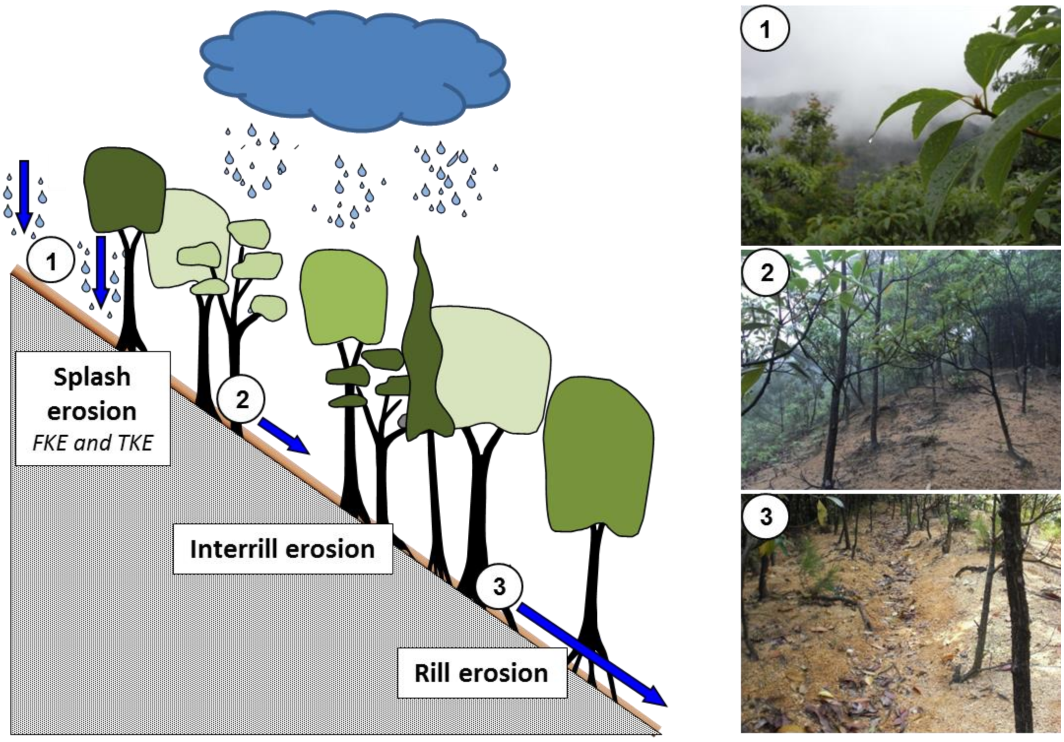


Figure 1: Different stages of water erosion from splash erosion (1) to interrill (2) and rill erosion (3). Splash erosion is mainly influenced by freefall kinetic energy (FKE) on fallow land and throughfall kinetic energy (TKE) under vegetation.

Splash erosion represents the first stage of the erosion process. It occurs when raindrops hit bare soil and the impact breaks up soil aggregates (Morgan 2005). Individual soil particles are than splashed from the point of impact and redistributed on the soil surface (Wang 2015). The splashed particles can rise more than 0.6 m above the ground and move up to 1.5 m from the source (Morgan 2005). Further, splashed particles fill the spaces between soil aggregates and thus a crust is formed that reduces infiltration and increases runoff (Richter 1998). The power of rain drops to detach soil particles is indicated as kinetic energy of rainfall (Salles et al. 2002). While freefall kinetic energy (FKE) characterises the drop spectra on fallow land, throughfall kinetic energy (TKE) characterises spectra under vegetation. The kinetic energy of raindrops can be measured directly by e.g. splash cups (Scholten et al. 2011), the paper stain method (Brandt 1988) or piezo-electric sensors (Hall and Calder 1993). In addition, it can be derived indirectly by converting rainfall intensity measurements (Lal 1994).

Interrill or sheet erosion is the second stage of water erosion. It is defined as the discharge of sediment in thin layers by shallow surface runoff after raindrop impact. Interrill erosion appears when the amount of rainfall exceeds the infiltration rate of the soil and the soil pores are saturated with water (Blanco-Canqui and Lal 2008). Mainly fine soil particles are lost in this stage and therewith a part of the available nutrients and organic matter in the soil. Interrill erosion is gradual and thus can go unnoticed by the naked eye, but the cumulative impact accounts for large soil losses (Morgan 2005; Toy et al. 2002). This process can be measured in situ by micro-scale runoff plots (ROPs) under natural or simulated rainfall (Lal 1994). Furthermore, sub-processes are determined by e.g. measurements of aggregate stability, infiltration capacity or interflow (Richter 1998). The surface water flow causing interrill erosion runs for only short distances (1 to 2 m) before rill processes occur (Toy et al. 2002).

Rill erosion is caused by the concentration of runoff in depressions or low points and represents the third stage of water erosion. Rills are shallow drainage lines of less than 0.3 m depth (Morgan 2005). They can further develop into gullies which are deeply incised channels of more than 0.3 m depth and can no longer be removed by normal cultivation (Toy et al. 2002). In order to determine those processes, photogrammetrical and modelling approaches as well as larger scaled ROPs and erosion sticks are commonly used (Blanco-Canqui and Lal 2008; Richter 1998; Lal 1994).

The influence of forest vegetation on soil erosion

Vegetation covers are generally considered as a key factor for the occurrence and dimension of soil erosion and forests in general reduce the risk of considerable soil loss (Morgan 2005; Hupp et al. 1995; Thornes 1990). For instance, Guo et al. (2015) showed that grown forests experience the lowest soil loss rates of all land use types in China. Forests 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). On fallow land, the kinetic energy of rainfall is affected by abiotic factors like rainfall intensity and amount (Levia and Frost 2006), but below forest canopies biotic factors come into play (Nanko et al. 2006). Species-specific effects on TKE are evoked by the individual tree structures and plant traits, such as leaf area index (Geißler et al. 2013) or tree height (Geißler et al. 2010). 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 may lead to considerable soil loss, if the forest floor is unprotected, which can be the case if protecting layers diminish e.g. under shady conditions (Onda et al. 2010) or fast decomposition (Razafindrabe et al. 2010). A leaf litter layer on the forest floor protects the soil from direct raindrop impact and modifies the water flow and storage on the soil surface (Kim et al. 2014). Leaves of different species vary in their size, shape, decomposition rate (Cornelissen 1996) and water storage capacity within a litter layer (Kim et al. 2014), having an important influence on ground coverage and surface runoff. Sidle et al. (2007) described a shallow preferential flow in soil-overlaying organic horizons (biomat flow), that is not initiating soil erosion. Moreover, it has been demonstrated that leaf litter quality is an important ecosystem factor in temperate climates with direct impact on soil characteristics (Kooijman and Cammeraat 2010)

Although the protective role of litter cover against soil erosion has been known for a long time (e.g. Smith 1914), only little research has been conducted on the processes involved. Even less is known about the influence of species-specific functional traits of the tree layer, such as crown or stem characteristics on sediment discharge and TKE (Guerrero-Campo et al. 2008; Lavorel and Garnier 2002). Moreover, most research on the latter aspects was performed in old-grown forests (e.g. Geißler et al. 2012a; Nanko et al. 2008; Zhou et al. 2002), 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.

The influence of soil mesofauna and soil macrofauna on soil erosion

In general, quality and quantity of litter determines decomposer communities ranging from microbes and fungi to animals of different size classes (Hättenschwiler et al. 2005). Leaf litter provides habitats, maintains a favourable microclimate for soil fauna and in addition is an important food source (Sayer 2006). Although the main part of litter decomposition is performed by microbes (Bardgett 2005), the mesofauna and macrofauna constitute the dominant physical litter transformers. It is consequently essential for promoting both, litter decomposition (Hättenschwiler and Gasser 2005) and physical–chemical soil parameters (Gabet et al. 2003). Therefore, these organisms have the potential to influence geomorphological processes like soil erosion (Butler and Sawyer 2012; Wheaton et al. 2011; Butler 1995). Allen et al. (2014) recently linked ecological principles to geomorphological aspects in general.

However, knowledge about how these principles affect soil erosion is still rudimentary. Cammeraat and Kooijman (2009) consider both, soil fauna and tree type, as important ecosystem engineers. At the same time, most of the relevant studies on faunal effects deal with bioturbation and illustrate the role of one or few functional groups of soil mesofauna and macrofauna as geomorphic agents (Viles 1988; Hupp et al. 1995). For instance, the crucial role of earthworms influencing soil structure and related soil physical properties is relatively well studied (Don et al. 2008; Blanchart et al. 2004). Earthworms can decrease surface water runoff by increasing soil porosity (Edwards and Bohlen 1996). Porosity is indirectly influenced by leaf litter and SOM, as they are a major food supply to earthworms, but also arthropods (Sayer 2006). Ants and termites can also have an effect on soil turnover (Butler 1995). For example, Cerdà and Jurgensen (2011) indicated that the presence of ant nests decreases surface runoff, but increases sediment loss due to unconsolidated soil mounds. Nevertheless, the impact of other soil mesofauna and macrofauna on soil erosion, especially in ecosystems without the important influence of earthworms, is not yet clear.

The influence of biological diversity on soil erosion

Biological diversity (hereinafter referred to as biodiversity) signifies the variability among living organisms from all sources and the ecological complexes they are part of (UNEP 1992). Soil erosion can negatively influence biodiversity (Pimentel and Kounang 1998; Harvey and Pimentel 1996), but it is assumed that this relationship also acts vice versa (Brevik et al. 2015 ; Geißler et al. 2012b; Körner and Spehn 2002). It has been shown that a change in diversity can have remarkable effects on different ecosystem functions and ecosystem stability (Cardinale et al. 2012; Hooper et al. 2005; Scherer-Lorenzen 2005). In many cases, increasing biodiversity enhanced ecosystem productivity and stability (Jacob et al. 2010; Loreau et al. 2001). In particular, the biological diversity of trees (hereinafter referred to as 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 (Scherer-Lorenzen 2014; Chisholm et al. 2013; Quijas et al. 2012). Although positive effects of mixed-species tree stands, like increasing productivity or reduced pest risks, were demonstrated (e.g. Bauhus and Schmerbeck 2010; Vilà et al. 2007), the effects of higher species mixtures on erosion control are still unclear. For instance, Tsujimura et al. (2006) and Zhou et al. (2002) demonstrated that tree monocultures have only limited mitigation potential for soil losses, but further research on the influence of more diverse tree stands is scarce.

Nevertheless, there is growing evidence that higher species richness can generally 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. It is suggested that plant species and functional diversity as well as a highly structured and diverse litter cover is an important factor for soil erosion control on mountain slopes (Martin et al. 2010). Furthermore, a positive tree diversity effect on TKE was observed in a subtropical secondary forest along a range of successional stages (Geißler et al. 2013) and most recently, Berendse et al. (2015) found that a loss of grass species diversity reduced erosion resistance on a dike slope. Nonetheless, most of these studies did not focus on soil erosion under forest vegetation and thus further investigations in woodland ecosystems seem to be necessary. In addition, studies on soil erosion under forest have mostly focused on deforestation (Blanco-Canqui and Lal 2008). As forests are considered beneficial for erosion control, afforestation is a common measure of soil protection, but generally results in monoculture stands (Jiao et al. 2012; Romero-Diaz et al. 2010; Puettmann et al. 2009). Thus, it appears that the role of tree species richness for soil erosion control has been largely disregarded.

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 (Kelty 2006; Wilson 1988). Thus, they are more effective in controlling erosion processes than vegetative covers with few species (Pohl et al. 2012). For example, high tree species richness may result in an increased stratification of canopy layers (Lang et al. 2010). Hence, crown overlap, biomass density and total canopy cover are often higher in mixtures than in monocultures (Lang et al. 2012). TKE reacts strongly to these tree characteristics and may affect sediment discharge as a consequence (Geißler et al. 2013; Nanko et al. 2008). 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). Further research on the influence of tree species richness on erosion control seems to be necessary, but the complex system of interacting functional groups within the vegetation cover is also of great interest.

Objectives of this thesis

In this thesis, mechanisms of soil erosion in subtropical forest ecosystems were investigated. In this context, the effects of species diversity, tree species identity, functional traits and soil fauna were observed. As those mechanisms are complex compared to open field conditions and single parts of the erosion process were of interest, the field work was separated into different parts. This thesis focused on interrill soil erosion rates determined by micro-scale ROPs under natural (Seitz et al. 2015a) and simulated rainfall (Seitz et al. 2015b). Additionally, investigations with splash cups were carried out on changes in TKE during the canopy passage of raindrops (Goebes et al. 2015a; Goebes et al. 2015b). Rilling processes have not yet been in the centre of interest of this thesis. In a final step, the technical setup with micro-scale ROPs and a portable rainfall simulator was further tested and improved in a Swiss agricultural trial (manuscript 5).

Testing for effects of species diversity and identity 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 a forest biodiversity and ecosystem functioning experiment in subtropical China (BEF China; cf. Bruelheide et al. 2014, www.bef-china.de). This part of China is known as a hotspot of biodiversity and woody plants (Bruelheide et al. 2011; Barthlott et al. 2005). Furthermore, it is the first time that soil erosion processes are included as a principal component in a BEF project under forest vegetation.

The objectives of this thesis were to:

- investigate the influence of tree species richness, tree species identity and species-specific functional traits on interrill soil erosion (manuscript 1)
- investigate the influence of leaf litter covers and soil mesofauna and macrofauna on interrill soil erosion (manuscript 2)
- investigate the influence of tree species richness and tree species identity as well as tree architectural and leaf traits on TKE (manuscripts 3 and 4)
- improve and adapt the technical setup for measurements of interrill erosion in different environments (manuscript 5)

First, it is hypothesised that higher tree species richness leads to lower soil erosion rates. This is due to higher stratified and overlapping tree canopies, even when there is not any leaf litter cover. It is presumed that soil erosion rates change in relation to different tree species due to species-specific functional traits and that tree height and canopy characteristics are good predictors for soil erosion rates.

Second, it is hypothesised that full leaf covered soil surfaces are better protected against soil erosion by greater overlap and gap-filling in highly diverse leaf litter mixtures. At the same time, highly diverse leaf litter mixtures may lead to faster leaf decomposition caused by more active decomposer communities and thus soil surfaces will faster lie fallow. As a consequence, this effect may lead to an inferior conservation of soil surfaces and therefore to higher erosion rates. Furthermore, it is hypothesized that soil mesofauna and macrofauna have an impact on leaf litter cover and sediment discharge.

Third, it is hypothesised that tree species richness affects TKE and TKE is spatially variable due to differences in tree characteristics. Moreover, it is presumed that TKE is higher than FKE. Furthermore, it is hypothesised that TKE below forest canopies is highly species-specific due to different tree characteristics. At last, it is assumed that leaf and tree architectural traits mediate species-specific effects on TKE.

Fourth, a well replicated and highly portable field setup with micro-scale ROPs and a lightweight rainfall simulator is a major improvement to study single vegetation and faunal effects on initial soil erosion processes.

2. Methodology

2.1. Study site

The studies were conducted at the Biodiversity and Ecosystem Functioning China (BEF China) experiment (Bruelheide et al. 2014) in Xingangshan, Jiangxi Province, PR China (29°06.450′ N and 117°55.450′ E). The project maintains two main experimental sites (Site A and B) of about 50 ha and a side experiment (New Integrated Litter Experiment - NILEx) of about 0.12 ha. The area comprises a mountainous landscape with an elevation range from 108 m to 250 m a.s.l.. Slopes range from 0 ° to 45 ° with a mean slope of 29 °. The bedrock of the sites consists of non-calcareous slates with varying sand and silt contents and is intermittent by siliceous-rich joints. Prevailing soil types are Endoleptic Cambisols with Anthrosols in downslope positions and Gleysols in valleys (cf. IUSS 2006) covering saprolite. 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 characteristic for subtropical summer monsoon regions with a mean annual temperature of 17.4 °C and a mean annual rainfall of 1635 mm (Goebes et al. 2015b) and is ranked as Cwa according to the Köppen-Geiger classification.

The experimental sites A and B have been used as a commercial forest plantation (*Cunninghamia lancelota* and *Pinus massoniana*) until 2007. Afterwards, they were clear-cut and replanted in 2009-2010 following a plot-based design with different extinction scenarios (Yang et al. 2013). In 2013, the main sites represented an early successional stage with tree ages from four to five years. In total, 566 experimental plots (25.8 m × 25.8 m) 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) following a broken stick design with 400 trees per plot and a planting distance of 1.29 m (Bruelheide et al. 2014). The main experiments of the project were focused on 70 Very Intensively studied Plots (VIPs), which represent a selection of all tree species and species richness levels. The NILEx has been placed in a *Castanea mollissima* plantation which was established in 1996. It is situated next to the main experimental site B and is surrounded by region-specific secondary forest. It shows an elevation difference of 11 m and is characterized by moderately steep to steep slope without an ectorganic layer.

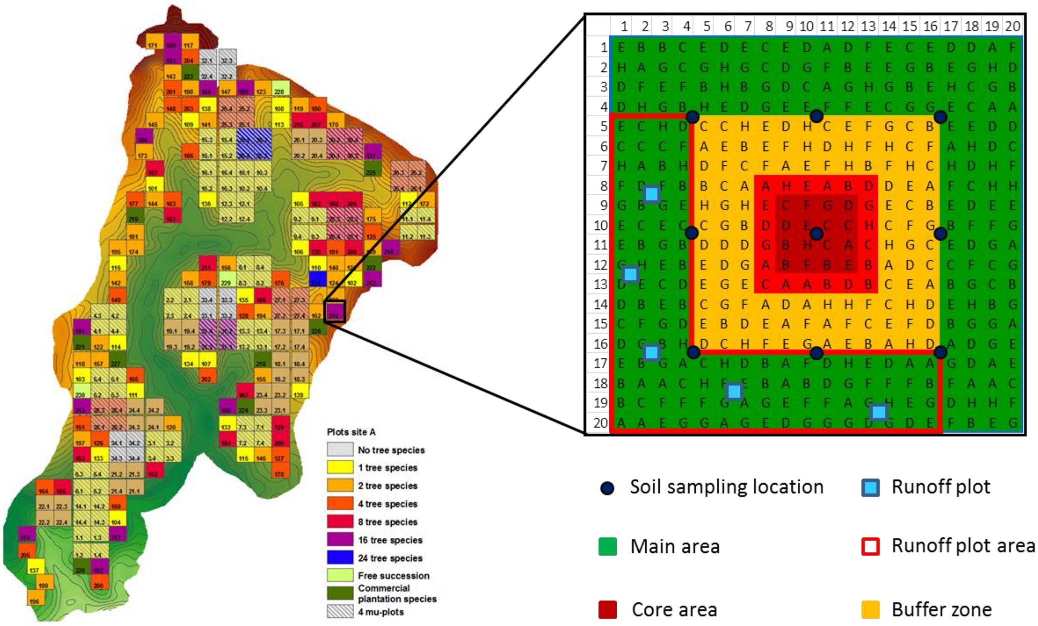
2.2. Experimental design

The measurement of sediment discharge in the main experiment (manuscript 1) was conducted with micro-scale ROPs under natural rainfall on a selection of 34 VIPs. 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 (Table 1).

Table 1: 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 *

All 34 VIPs in the main experiment 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 the study area (Figure 2). All leaf litter was removed from the ROPs prior to measurements. The ROPs were operated in May and June 2013 during the rainy season and the capacity of the reservoirs was not exceeded during any rainfall event.



	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20
1	E	B	B	C	E	D	E	C	E	D	A	D	F	E	C	E	D	D	A	F
2	H	A	G	C	G	H	G	C	D	G	F	B	E	E	G	B	E	G	H	D
3	D	F	E	F	B	H	B	G	D	C	A	G	H	G	B	E	H	C	G	B
4	D	H	G	B	H	E	D	G	E	E	F	F	E	C	G	G	E	C	A	A
5	E	C	H	D	C	C	H	E	D	H	C	E	F	G	C	B	E	E	D	D
6	C	C	C	F	A	E	B	E	F	H	D	H	F	H	C	F	A	H	D	C
7	H	A	B	H	D	F	C	F	A	E	F	H	B	F	H	C	H	D	H	F
8	F	D	F	B	B	C	A	A	H	E	A	B	D	D	E	A	F	C	H	H
9	G	B	G	E	H	G	H	E	C	F	G	D	G	E	C	B	E	D	E	E
10	E	C	E	C	C	G	B	D	D	E	C	C	H	C	F	G	B	F	F	G
11	E	B	G	B	D	D	D	G	B	H	C	A	C	H	G	C	E	D	G	A
12	G	H	E	B	E	D	G	A	B	F	B	E	B	A	D	C	C	F	C	G
13	D	E	C	D	E	G	E	C	A	A	B	D	B	C	E	A	B	G	C	B
14	D	B	E	B	C	G	F	A	D	A	H	H	F	C	H	D	E	H	B	G
15	C	F	G	D	E	B	D	E	A	F	A	F	C	E	F	D	B	G	G	A
16	D	G	B	H	D	C	H	F	E	G	A	E	B	A	H	D	A	D	G	E
17	E	B	G	A	C	H	D	B	A	F	D	H	E	D	A	A	G	D	A	E
18	B	A	A	C	H	F	E	B	A	B	D	G	F	F	F	B	F	A	A	C
19	B	C	F	F	F	G	A	G	E	F	F	A	G	H	E	G	D	H	H	F
20	A	A	E	G	G	A	G	E	D	G	G	G	D	G	D	E	F	B	E	G

Figure 2: General views of the experimental site A (left) and a Very Intensively studied Plot (VIP, right) in Xingangshan, Jiangxi Province, PR China. On the left side, colours indicate different diversity and treatment levels. On the right side, colours indicate different zones as well as exemplary positions of runoff plots and soil sampling locations are shown. Different letters represent different tree species (n=400) within the plot area (25.8 m × 25.8 m).

Additionally, measurements of sediment discharge were carried out on the NILEx site (manuscript 2) with micro-scale ROPs and a rainfall simulator. Leaf litter of seven domestic tree species [*Liquidambar formosana* (Hance, species A), *Machilus thunbergii* (Sieb. et Zucc., species B), *Quercus serrata* (Thunb., species C), *Schima superba* (Gardn. et Champ., species D), *Castanopsis eyrie* (Champ. et Benth., species E), *Castanea henryi* (Rehd. et Wils., species F), *Cyclobalanopsis glauca* (Thunb., species G), author and abbreviation for given species in parentheses] was used. The litter was collected in the Gutianshan National Nature Reserve close to Xingangshan (Geißler et al. 2013). Species were organized in two pools with one overlapping species (pool I: species A, B, C and D; pool II: species D, E, F and G). One, two and four leaf species mixtures as well as a bare ground feature (diversity 0) were used. In addition, a fauna exlusion feature was introduced. Both species pools were equipped with 24 ROPs. Moreover, all plots were replicated and hence, a total number of 96 ROPs was achieved. Leaf litter blends were mixed before deployment to avoid local patches of individual species (Burghouts et al. 1998) and distributed randomly (60 g for each ROP). Prior to this, alien leaves, twigs and loose stones (>2 mm) have been removed from the plot surfaces.

Rainfall was simulated for 20 minutes at each ROP. To consider the effect of different water saturations, we applied a first 20 minute run on actual field water saturated ground and a second run 15 minutes later, when soils were higher saturated. Measurements were carried out when ROPs were fully covered (May 2012, timestep 1) and after decomposition had led to reduced leaf litter coverage (September 2012, timestep 2). Leaf covered ROPs have not laid bare completely in the second timestep and no rills were present in the experimental plots.

The measurement of throughfall kinetic energy (manuscripts 3 and 4) was carried out on 40 VIPs on experimental site A with splash cups. Therefore, 24 different tree species and six levels of tree species richness (17 × div1, 10 × div2, 6 × div4, 4 × div8, 1 × div16, 2 × div24) were used. The measurements were located in the core area of each VIP, which consists of the central 6 × 6 tree individuals (Figure 1). The core area was divided into 9 sections and in each section a TKE measurement was located. The positions were (1) 0.15 m away from the stem, (2) in the middle of two tree individuals, (3) in the middle of four individuals, (4) 0.45 m away from the stem, (5) at the 0.45 m × 1.2 m intersection between two individuals, (6) below the first branch of an individual, (7) at the 0.75 m × 0.75 m intersection between two individuals, (8) 0.3 m away from the stem and (9) outside the plot as a no-vegetation reference. In total, 1800 splash cups were measured in 2013 during five rainfall events.

2.3. Field measuring techniques

Measurement of sediment discharge with runoff plots

To determine initial sediment discharge and surface runoff, micro-scale ROPs (0.4 m × 0.4 m) were used. They were made of stainless steel panels and equipped with a covered runoff-gutter at the lower end. Each ROP was connected to a 20 L reservoir and a rainfall gauge was placed next to it (Figure 3). Runoff volume and rainfall amount were determined in situ. An aliquot of 1 L was taken from the reservoir after agitating it for 1 minute. Sediment discharge was separated from runoff by vacuum filtration (fibreglass filters) and the solid subsample was oven dried (40 °C) before weighing.

Figure 3: 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.

Micro-scale ROPs were chosen to establish a maximum number of replicates within a factorial random design to tackle arbitrary natural and measurement variability (Wendt et al. 1986). A high number of ROPs requires great efforts in maintenance and control (cf. Hudson 1993), which in turn are easier to ensure with plots of small scale and small sized reservoirs (Boix-Fayos et al. 2006). The micro-scale ROPs used in this thesis (surface: 0.16 m^2) quantified interrill wash and sediment detachment by raindrop impact (cf. Agassi and Bradford 1999). The stability of soil aggregates influenced e.g. by SOM or mesofauna and macrofauna are of major importance at this scale (Morgan 2005). However, an essential 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. Bagarello and Ferro 2004; Abrahams et al. 1995) and boundary effects increasingly influence the results with decreasing plot sizes (Mutchler et al. 1994). Further, the average splash distance has to be considered and is approximately one quarter of the ROP length for the silt loam determined at the BEF study sites (0.12 m, cf. Legout et al. 2005).

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).

In NILEx (manuscript 2), a modified version of ROPs with a fauna exclusion feature was used in order to investigate the mesofaunal and macrofaunal influence. Four holes were incorporated to the side panels (circular, diameter 50 mm). Those holes were equipped with mesh (size 20 mm) to exclude bigger animals (e.g. mice and toads), but allow access to litter decomposing mesofauna and macrofauna. Half of the plots were additionally equipped with pitfall traps (diameter 0.09 m, depth 0.15 m, capacity 0.55 L) and fine mesh (size <0.1 mm) to exclude or at least strongly reduce soil mesofauna and macrofauna from ROPs (fauna treatment). The traps were filled with 0.15 L of preserving solution (40 % ethanol, 30 % water, 20 % glycerol, 10 % acetic acid, some drops of detergent to reduce surface tension), which has shown its reliability in subtropical climates (Schuldt et al. 2011). In between the measuring campaigns all ROPs have been provided with 20-mm-mesh to fix the experimental leaves against moving downslope and to exclude falling leaves from *Castanea mollissima* (Figure 4).

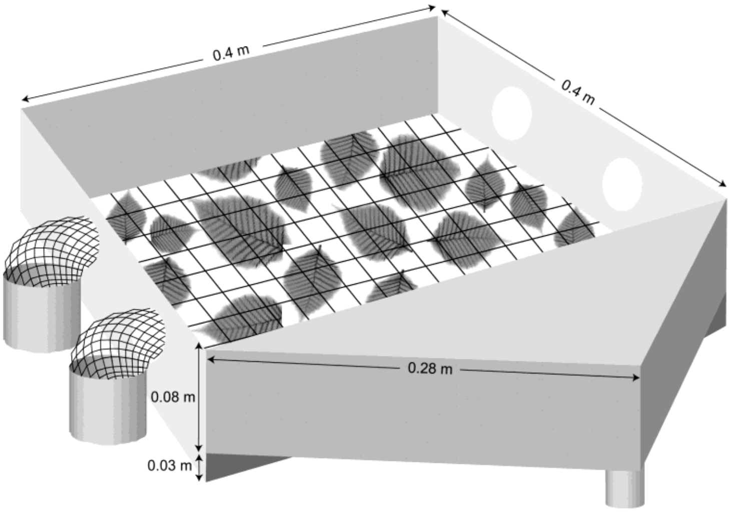


Figure 4: Runoff plot (ROP) with fauna treatment. ROPs consist of stainless-steel side panels and a triangular covered runoff gutter. Falling leaves are separated from experimental leaf mixtures by a 20-mm-mesh. Pitfall traps (two at each side) are installed to exclude soil mesofauna and macrofauna.

Rainfall simulation

Whereas erosion measurements in the main experiment relied on natural rainfall, a rainfall simulation was carried out at the NILEx site (manuscript 2). It was performed in two timesteps (May and September 2012) with the portable Tübingen rainfall simulator (Iserloh et al. 2013, Figure 5). The simulator was modified by a Lechler 460.788.30 nozzle and adjusted to a falling height of 3.5 m (pressure at nozzle: 150 hPa). The sprinkle area was 1 m^2 and protected from outer influences by a light frame tent (inside volume 16 m^3). Drop size spectrum and intensity of the simulator were calibrated using a 'Laser Precipitation Monitor' by Thies (Lanzinger et al. 2006) to obtain homogeneous characteristics.

Figure 5: Tübingen rainfall simulator with light frame tent during runoff plot measurements in 2012 on the NILEx site, Xingangshan, Jiangxi Province, PR China.

Measurement of throughfall kinetic energy with splash cups

The kinetic energy of raindrops (manuscripts 3 and 4) was measured with Tübingen splash cups (T-Cup, Scholten et al. 2011). The T-Cup consists of a water-filled polyethylene flask with a carrier system that is connected by cotton wick and an inner removable cup. Those removable cups have a diameter of 4.6 cm and a height of 4 cm to exclude rim and wash-off effects during rainfall. They are connected to the carrier system by a silk mesh that is in hydraulic contact to the carrier system and provides constant moisture over a long time. Splash cups were filled with fine-sized sand (0.125 µm). Detached sand by rain splash was calculated by subtracting the weight of dried sand after a rainfall event from initial sand weight. Sand loss and the kinetic energy of raindrops follow a linear function, which was used to calculate TKE (Goebes et al. 2015b; Scholten et al. 2011). The T-Cup was approved in field studies in China and showed its reliability in subtropical conditions (Geißler et al. 2012a, Geißler et al. 2010).

2.4. Soil, vegetation and rainfall characteristics

Soil

Soil surface cover was measured photogrammetrically (grid quadrat method with GIMP 2.8) and slope with an inclinometer at each ROP, respectively. Soil texture and SOM contents were identified for VIPs in the main experiment (5 cm depth, 9 replicates per plot, n=34) and at the NILEx site (n=48) using a SediGraph III 5120 (Micromeritics, Aachen, Germany) and a Vario EL III elemental analyser (Elementar, Hanau, Germany). Likewise, pH was measured in 1 M KCl using Sentix 81 electrodes and bulk soil density (5 cm depth) was determined with the mass-per-volume method (100 cm^3 intact core). During the NILEx, soil moisture was assessed at both timesteps and every ROP with a Wet2-Sensor (TDR, Delta-T devices, Cambridge, UK). Furthermore, soil meso- and macrofauna from pitfall traps were counted, identified and classified to higher taxonomic levels.

Vegetation

At each ROP, 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). To specify the influence of tree architectural traits, tree height, stem diameter at 5 cm above ground (hereinafter referred to as stem diameter), crown width, crown base height and the number of branches of neighbouring trees were measured (Li et al. 2014). Tree height was measured with a measuring pole from stem base to the apical meristem. Stem diameter was measured with a calliper to the nearest millimetre. Crown width was determined with a linear tape along two directions (north-south and east-west). Based on the crown diameters, crown area was calculated as an area of ellipse. Crown base height (CBH) was measured to the bifurcation point of the lowest living crown branch.

Nine leaf traits were analysed by Kröber et al. (2012) and Kröber and Bruelheide (2014). Those traits included leaf area (LA), specific leaf area (SLA), leaf pinnation (simple or pinnate), leaf margin (entire or dentate), trichome cover of upper leaf surface, leaf thickness, leaf toughness and leaf habit (deciduous or evergreen).

Rainfall

Weather conditions were recorded by an on-site climate station (datalogger with Vaisala weather transmitter and tipping bucket balance, ecoTech, Bonn, Germany) in 5-min intervals. The total precipitation in the study area was 1205 mm in 2013 and 1921 mm in 2012.

In summer 2013, work was carried out on the main experimental sites under natural rainfall. From May to June, a fraction of 957 mm from total rainfall (33 events) was 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 were used for ROP measurements (manuscript 1) and five were used for splash cup measurements (manuscripts 3 and 4). The total rainfall amount from May to June 2013 was 185 mm, of which 135 mm fell during erosive rainfall events.

In summer 2012, a rainfall simulation was performed (manuscript 2). Rainfall intensity was set to 60 mm h^{-1} for all simulation runs according to a typical regional rain storm event, as monitored in Xingangshan (2009–2012). Meteorological data showed typical monsoon patterns in natural rainfall with a mean monthly precipitation of 200 mm from May to August 2012. In September the end of the monsoon season led to a considerably lower monthly precipitation (116 mm). During timestep 1 in May 128 mm of natural rainfall occurred, whereas during timestep 2 in September no natural rainfall was recorded during measurements.

2.5. Data analysis

Linear mixed effects (LME) models with restricted maximum likelihood were performed using R 3.0.2 (R Core Team 2013), ASReml (Butler 2009) and “lmerTest” (Kuznetsova et al. 2014) in order to investigate the influences on sediment discharge and TKE. These models account for pseudo-replication and remove unnecessary variance in the hierarchical data by using random effects like plot or species composition. The maximum likelihood approach was used to obtain model simplification by step-wise backward selection, eliminating the least significant variable except for tree species richness. When trends were indicated, specific diversity levels were tested against each other using linear contrasts (e.g. whether monocultures were different from mixtures). If multicolinearity was detected (spearman ρ >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 ANOVA type 1 or ANOVA type 3 with Satterthwaite approximation for degrees of freedom and p-values were obtained by likelihood ratio tests (Wald test). The distinction between FKE and TKE was investigated using Student’s t-test and TKE was log10 transformed before modeling. To specify possible effects of species identity on TKE, mediation trait variables were fitted before the species identity term.

3. Results and discussion of the manuscripts

3.1. The influence of tree species richness, identity and functional traits on interrill erosion (manuscript 1)

This study was conducted in order to investigate the influence of tree species richness and identity as well as tree functional traits on soil erosion processes in young forest ecosystems. Therefore, 170 micro-scale ROPs were sampled in the main experiment in summer 2013. The soil loss rate determined by those ROPs (47.5 t ha^{-1} a^{-1}) was considerably higher than the average rate Guo et al. (2015) recently calculated for South China (approx. 20 t ha^{-1} a^{-1}) in a study based on small-scale and field ROPs.

We found that tree species richness did not affect sediment discharge and runoff (Figure 6). 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 significantly between bare plots (div0), monocultures (div1) and different species mixtures (div8, div16, div24).

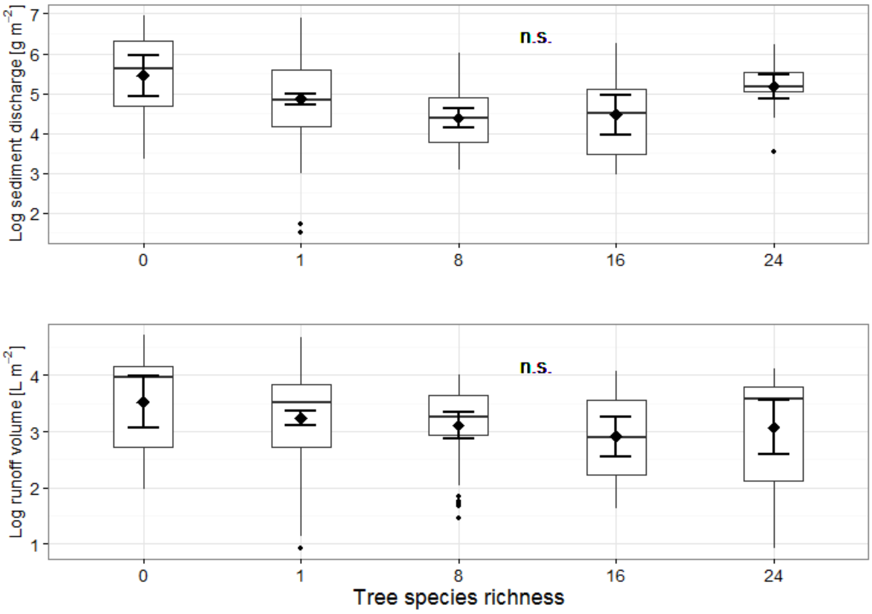


Figure 6: Sediment discharge and runoff volume at five diversity levels based on four rainfall events in May and June 2013 in Xingangshan, Jiangxi Province, PR China (n.s.: not significant, n=334). Horizontal line within boxplot represents median and diamond represents mean.

The standard deviations of sediment discharge (g m^{-2}) and runoff volume (L m^{-2}) in relation to diversity levels were high (Figure 6 and Table 2). 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 the experiment tree height in mixed stands was not lower than 1.07 m and crown cover achieved at least 29 %. Thus, all species mixtures in this experiment assured a higher level of tree height and ground coverage after four to five years of tree growth, whereas in monocultures the canopy cover was lower and highly tree species specific.

Table 2: 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)

Although a negative trend in sediment discharge was visible from level 1 to 8 and mixed stands showed a more balanced and homogenous vegetation development than monocultures (cf. Kelty 2006), higher tree species richness did not mitigate soil erosion. The absence of a species richness effect on soil loss 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. 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.

Moreover, this study provided evidence that different tree species affect interrill erosion processes as several species showed significant variation from mean sediment discharge (199 g m^{-2}, Figure 7). Thus, it could be shown that different tree morphologies have to be considered, regarding erosion in young forest ecosystems. 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. Durán Zuazo and Rodríguez Pleguezuelo 2008, Bochet et al. 2006) and Xu et al. (2008) showed that different plant morphologies may control soil loss and improve soil properties in a dry river valley in China. In this study, *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* mitigated soil erosion in young forest stands. Thus, we can confirm a species-specific effect on sediment discharge for this subtropical experimental area. Furthermore, it appears that the appropriate choice of tree species for afforestation against soil erosion becomes already important in an early successional stage.

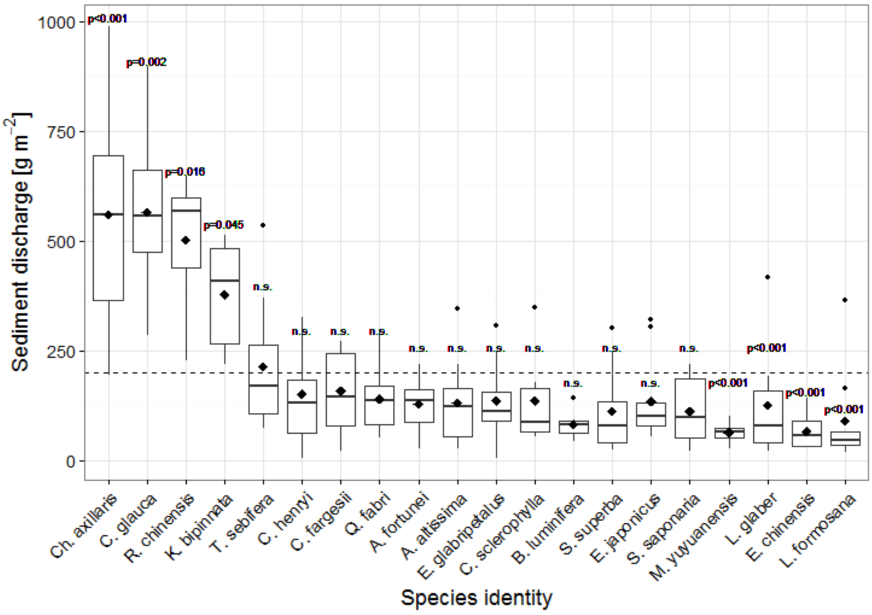


Figure 7: Sediment discharge under 20 tree species in monocultures based on four rainfall events in May and June 2013 in Xingangshan, Jiangxi Province, PR China. Dashed line indicates mean sediment discharge of all 20 species. Horizontal lines within boxplot represent median and diamonds represent mean values found for a respective species (n.s.: not significant).

Species-specific effects can result due to different throughfall kinetic energy and were ascribed to different tree architectural characteristics and leaf traits (cf. manuscript 4). For instance, *Chorespondeas axillaris* was the tallest tree species with a nearly closed canopy and caused the highest amount of sediment discharge in this study. Raindrops falling from leaves of this species nearly reached terminal velocity and hence throughfall kinetic energy was high (Goebes et al. 2015a; Morgan 2005). 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, which could be explained by other functional traits.

This study revealed that species-specific functional traits and site characteristics affected soil erosion rates (Figure 8). 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. In this study, high crown cover ($p<0.01$) and LAI ($p<0.05$) reduced soil erosion, whereas it was slightly increased by increasing tree height ($p<0.1$). Thus, low tree stands with high canopy cover are effectively counteracting soil loss in this initial forest ecosystem. Stem diameter and crown width did not influence erosion processes in this system. Crown cover was highly correlated with LAI, tree height, stem diameter and crown width ($r=0.82$, 0.80, 0.75, 0.77, respectively). 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. 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 ($p<0.001$). This finding underlines the general importance of covered soil surfaces for erosion control (cf. Morgan 2005; Thornes 1990) 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. Furthermore, soil organic matter had a decreasing influence on sediment discharge ($p<0.05$) 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.

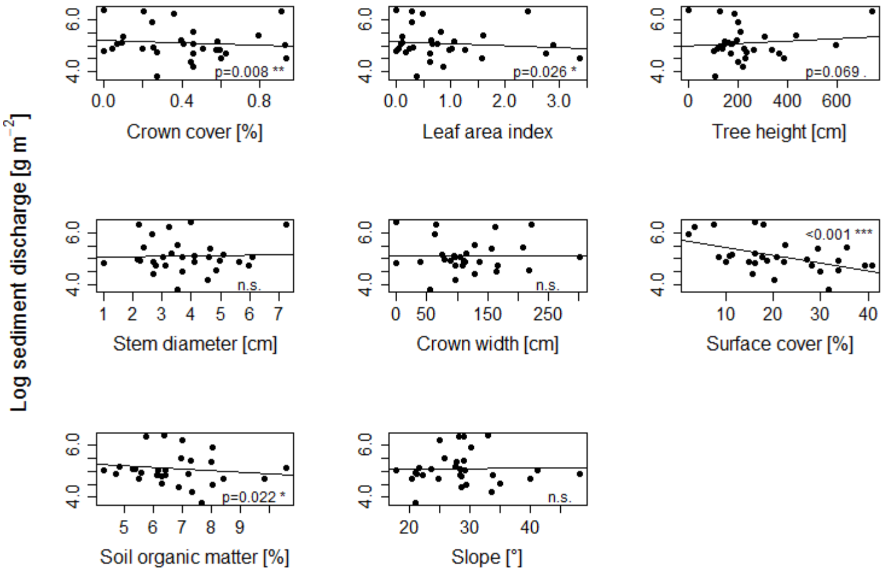


Figure 8: Effects of tree functional traits and site characteristics on sediment discharge. Analyses were based on four rainfall events in May and June 2013 in Xingangshan, Jiangxi Province, PR China. Black lines symbolize linear trends (n.s.: not significant).

3.2. The influence of leaf litter diversity and soil fauna on interrill erosion (manuscript 2)

In this manuscript, the influence of leaf litter cover and litter diversity as well as soil mesofauna and macrofauna on initial soil erosion processes in subtropical forest ecosystems was studied. Therefore, a full-factorial random design with seven domestic leaf species was established at the NILEx site. Sediment discharge and runoff volume were measured with a rainfall simulator on 96 micro-scale ROPs modified with pitfall traps.

This study showed that neither leaf litter species diversity nor functional diversity influence sediment discharge, leaf litter cover and thus soil erosion (Figure 9). Even if rising leaf litter diversity seemed to lead to a slightly smaller sediment discharge, these effects were not statistically significant. Moreover, no change in the influence of leaf litter diversity was detected from May (full leaf coverage) to September (reduced leaf coverage). Hence, better overlap and gap-filling or different decomposition rates in highly diverse litter mixtures seem not to be considerable parameters for soil erosion control. Nevertheless, mean sediment discharge was 60 % higher in September than in May.

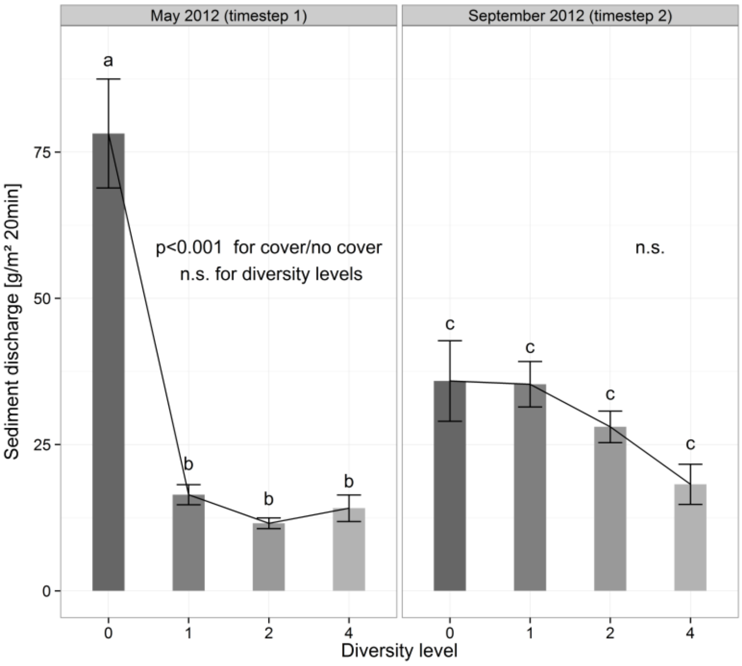


Figure 9: Effect of diversity levels and bare ground (diversity = 0) on sediment discharge at timestep 1 (May 2012) and timestep 2 (September 2012). Small letters indicate significant mean differences (n.s.: not significant).

Functional diversity of leaf traits did not influence erosion parameters but notwithstanding, single leaf species in monocultures showed rather different impacts on sediment discharge and thus erosion control (Figure 10). In this experiment, runoff plots with leaf litter from *Machilus thunbergii* showed the highest sediment discharge (68.0 g m^{-2}) whereas plots with *Cyclobalanopsis glauca* showed the smallest rates (7.9 g m^{-2}). This can be related to variable leaf habitus, different decomposition rates and food preferences of litter decomposing fauna. Nevertheless, it might not be detectable in the functional diversity index, as different leaf traits are merged. Further, different leaf species showed rather different decomposition rates and thus developments of surface litter cover from 10 % remaining cover (*Liquidambar formosana*) to 56 % remaining cover (*Schima superba*) in September. This variability leads to positive and negative feedbacks of leaf litter species on soil erosion, but within leaf mixtures those differences are levelled and no directional pattern can be detected (cf. Hättenschwiler et al. 2005). In addition to the latter, different decomposition rates of single leaf species resulted in a more heterogeneous influence on soil erosion in September.

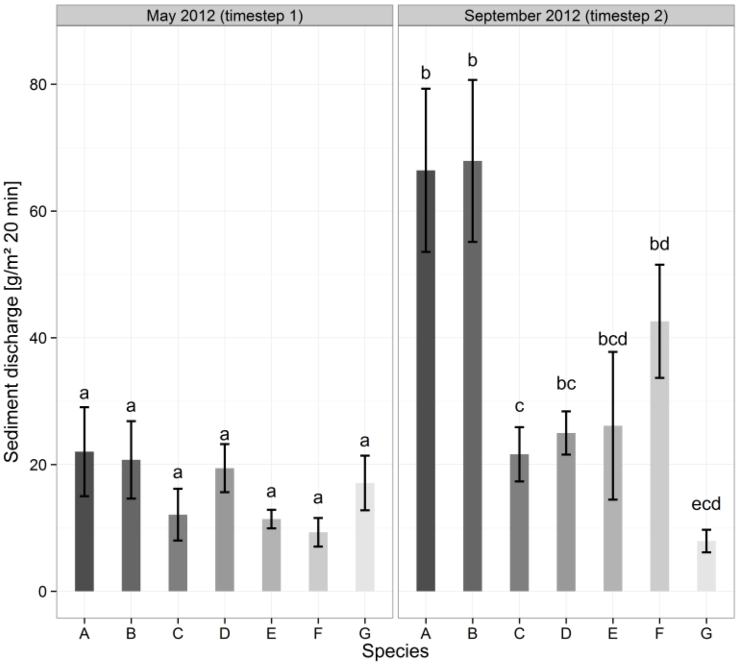


Figure 10: Relation between sediment discharge and monocultures of leaf litter species at timesteps 1 and 2. Small letters indicate significant mean differences.

The protective effect of a leaf litter cover was influenced by the presence or absence of soil mesofauna and macrofauna at both timesteps (May: $p < 0.001$, September: $p < 0.05$, Figure 11). Fauna presence increased soil erosion rates significantly by 58 % and this effect was slightly decreasing from spring to autumn.

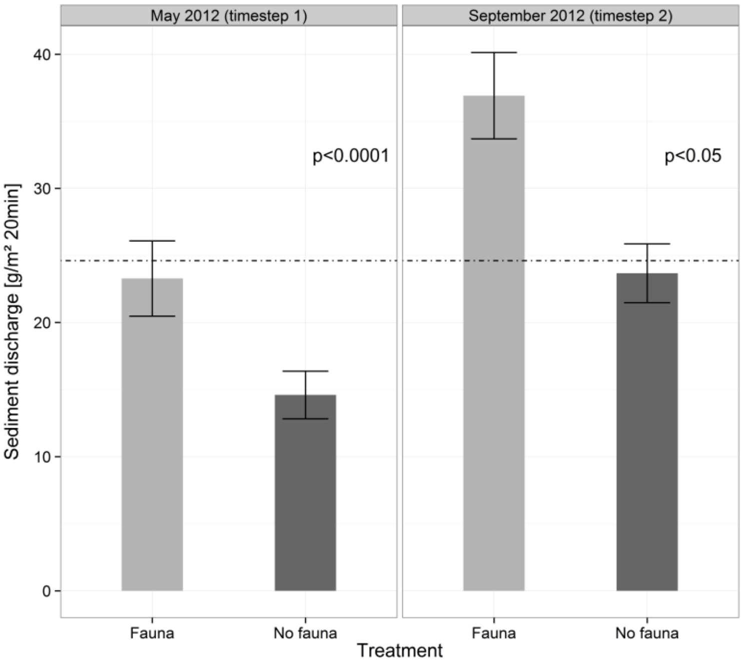


Figure 11: Effect of the fauna treatment on sediment discharge at timestep 1 (May 2012) and timestep 2 (September 2012).

It is assumed that the activity of soil-dwelling and surface-active mesofaunal and macrofaunal organisms led to the loosening and translocation of soil particles within the first centimetres of the soil surface. In the ROPs with fauna exclusion 2101 individuals belonging to the mesofaunal and macrofaunal size class were found. Springtails (Collembola) were the dominating taxon in the catches followed by mites (Acari), ants (Formicidae), worms (Oligochaeta) and beetles (Coleoptera). While springtails and mites were particularly abundant and might have contributed directly or indirectly to this process (by promoting the decomposition of the soil-protecting litter layer; Hättenschwiler and Gasser 2005), larger-sized and highly active ants and beetles can be assumed to have contributed significantly to the modification of the soil surface. Many species of these macrofauna groups influence soil processes in terrestrial ecosystems (Lavelle et al. 1997) and can show burrowing behaviour when searching for food or when trying to evade short- or long-term unsuitable microclimatic conditions on the soil surface (Dostál et al. 2005; Gabet et al. 2003; Swift et al. 1979). In September, litter cover on the plots was reduced and microclimatic conditions were less optimal than earlier in the season, so the soil faunal activity was on the decline. Exclusion of the soil fauna in both timesteps apparently removed significant faunal impacts on soil surface slackening and decomposition processes that favour soil particles to be detached and washed away during rainfall events. Furthermore, soil mesofauna and macrofauna are a prominent factor in litter fragmentation and decomposition and thus the reduction of protecting litter covers. Over all ROPs in this experiment, the presence of soil fauna led to a slight increase in mean leaf litter decomposition rates compared to no fauna treatments though non-significant. Several of the more abundant taxa found in the ROPs, such as Acari, Collembola and Oligochaeta, play an important role in litter fragmentation and decomposition (Hättenschwiler and Gasser 2005; Swift et al. 1979). As a consequence, effects of this fauna group on sediment discharge have to be considered in soil erosion experiments.

Results demonstrated that a leaf litter cover generally protects soil from erosion by rainfall as sediment discharge on leaf covered plots was reduced by 82 % (Figure 9). This protection was removed with ongoing litter decomposition as erosion rates were considerably higher in September than in May. Leaf litter covers of more than about 40 % of the ground surface can prevent soil erosion effectively. Runoff volume was significantly lower on leaf covered plots than on bare plots in timestep 1. Gerrits et al. (2007) illustrated that forest floor interception e.g. by short vegetation and litter can double the amount of intercepted rainfall in forest ecosystems and therefore, lead to lower surface runoff.

3.3. Influences on throughfall kinetic energy (manuscripts 3 and 4)

Manuscripts 3 and 4 focused on how TKE is influenced by tree species richness and how it is spatially variable under young subtropical forest canopies. Furthermore, effects of species-specific leaf and tree traits on TKE were investigated. Therefore, an experiment with splash cups was carried out on 40 VIPs at the main experimental site A.

Tree species richness

This experiment showed that TKE was not influenced by tree species richness at the plot level (Figure 12), although monocultures showed slightly lower TKE (-6 %) than species-mixtures.

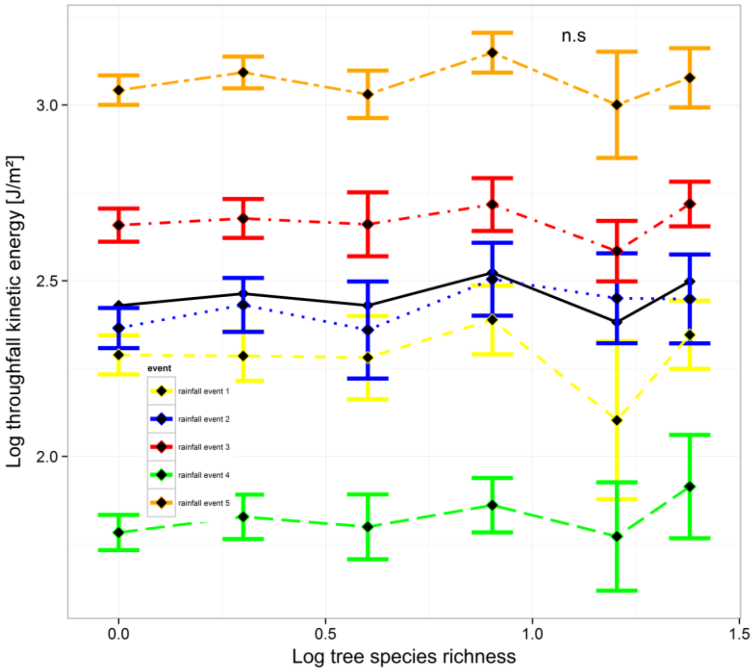


Figure 12: Throughfall kinetic energy (log10-transformed) and tree species richness (log10-transformed) during five rainfall events. Black solid line symbolizes mean of all events. Black diamonds show mean and error bars show two times standard error. Different colours show different rainfall events and dashed lines connect mean values of each tree species richness level for each rainfall event (n.s.: not significant).

This finding contradicts other studies (Geißler et al. 2013; Martin et al. 2010), which showed significant effects of biodiversity on TKE and sediment discharge. This is likely due to the young age of the experimental forest where a dense and high tree canopy has not yet been developed. At the time of this study, tree characteristics, such as crown area, were evolved to approximately 10 % compared to mature trees in the study region (Bruelheide et al. 2014). It is likely that tree species richness effects on TKE develop over time, which was shown in grassland biodiversity experiments (Reich et al. 2012; Marquard et al. 2009). We assume that five years of growing a tree plantation in subtropical forests is not enough to induce tree species richness effects on TKE.

However, TKE was positively influenced by neighbourhood tree species richness indicating that tree species richness only affected TKE on a small spatial scale within the direct neighbourhood in young forests (Figure 13). Thus, tree species richness in young forests seems not to be beneficial to ecosystem functioning due to enhanced soil erosion potential. However, this neighbourhood tree species richness effect is rather weak. The effect can be attributed to the fact that local neighbourhood interactions strongly influence tree growth and architecture (cf. Lang et al. 2012; Schröter et al. 2012). In this study, we found significantly taller trees, larger crown areas and a higher number of branches in more diverse neighbourhoods. These differences in vegetation structure as a response to the local neighbourhood species richness might influence TKE.

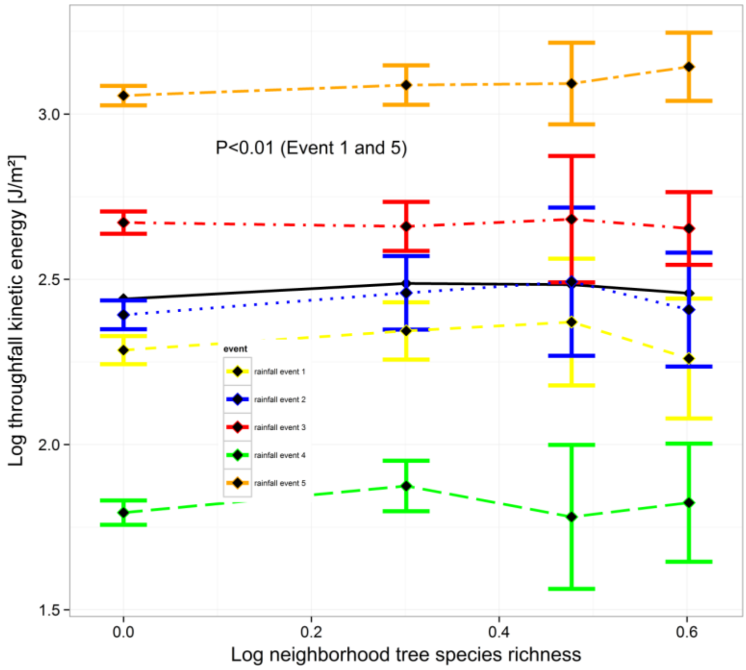


Figure 13: Throughfall kinetic energy (log10-transformed) and neighbourhood tree species richness (log10-transformed) of five rainfall events. Black solid line symbolizes mean of all events. Black diamonds show mean and bars show two times standard error. Different colors show different rainfall events and dashed lines connect mean values of each tree species richness level for each rainfall event. p < 0.01 (Event 1 and 5) indicates P-value is lower than 0.01 for analyses of rainfall event 1 and 5.

Spatial variability

TKE showed distinct spatial variability ($p < 0.1$, Figure 14). Investigating each rainfall event separately, this pattern was strong at rainfall event 1 ($p < 0.05$) and 5 ($p < 0.05$), whereas no significant spatial variability of TKE was found at the rainfall events 2, 3 and 4. Directly below the first branch of the tree individuals TKE was lowest (430 J m^{-2}) while it was highest in the middle of four tree individuals (556 J m^{-2}). Mean FKE was 480 J m^{-2} and TKE was only higher than FKE below trees exceeding 3.3 m.

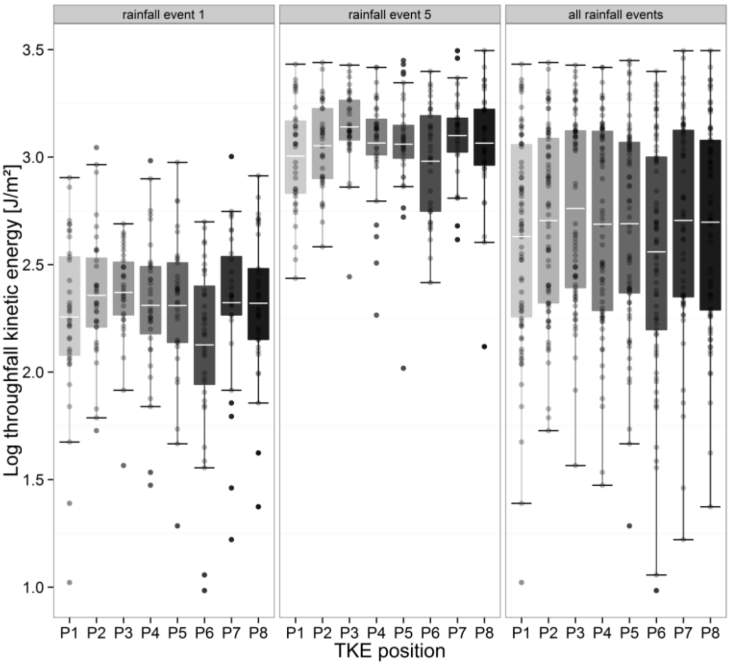


Figure 14: Throughfall kinetic energy (TKE, log10-transformed) and spatial position of TKE measurement at rainfall event 1 (left), at rainfall event 5 (middle) and in all five rainfall events (right). Different colors indicate different positions and white lines represent means.

Lower TKE below the first branch than FKE can be attributed to three components: (i) splitting of drops into smaller ones when hitting the first branches; (ii) a shorter falling height leading to lower rain drop velocity; (iii) interception of throughfall water and transfer to tree stems without release (Herwitz 1987). Despite these characteristics, the first branch is the last barrier for drops before reaching the soil surface (Nanko et al. 2008). Hence, rain drop velocities are generally lower resulting in low TKE. Higher TKE in the middle of four tree individuals can be ascribed to a high number of branches, a low LAI, a large crown area and most importantly high throughfall at this position. At last, the numbers of influencing tree individuals, that affected one measurement position, positively affected TKE due to increased throughfall and low LAI.

Species specific leaf and tree traits

Results showed that TKE was species-specific in this young successional stage as three out of 11 species showed distinct differences (Figure 15). Species-specific differences of throughfall amount or interception have been reported (Levia and Frost 2006; Aston 1979) and moreover, drop size distribution as an important driver of TKE is also known to be species-specific (Nanko et al. 2013). In this study, highest TKE was found below *Choerospondias axillaris* and *Sapindus saponaria*, while *Schima superba* showed lowest TKE, which has already been reported by Geißler et al. (2012b). *Schima superba* is well-known to show high values of canopy interception during rainfall (Guo et al. 2006), which could be partially attributed to high LAI and reinterception of rainfall by lower canopy layers (Nanko et al. 2008; Brandt 1988). These findings are as much more relevant as *Schima superba* represents one of the dominant tree species in the regional species pool (Bruelheide et al. 2011).

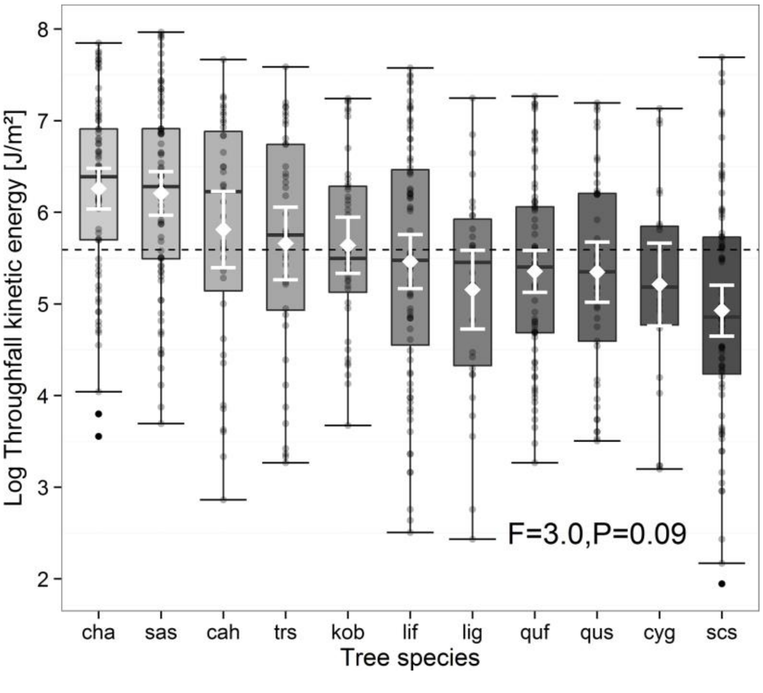


Figure 15: Throughfall kinetic energy (TKE, log-transformed) of 11 species. Dotted line represents the total mean TKE. Inside the boxplots white rectangles represent mean and white bars standard deviation. Species names from left to right: Ch. axillaris, S. saponaria, C. henryi, T. sebifera, K. bipinnata, L. formosana, L. glaber, Q. fabri, Q. serrata, C. glauca, S. suberba.

These species-specific effects were mediated by leaf habit, LA, leaf pinnation, leaf margin, stem diameter, CBH, tree height, number of branches, LAI and throughfall. Among these, LA, leaf habit and tree height showed the highest effect sizes on TKE and can be considered as major drivers of TKE. TKE was positively influenced by LA, stem diameter, CBH, tree height and LAI, while it was negatively influenced by the number of branches. TKE was lower in evergreen, simple leaved and dentate leaved than in deciduous, pinnated or entire leaved species (Figure 16).

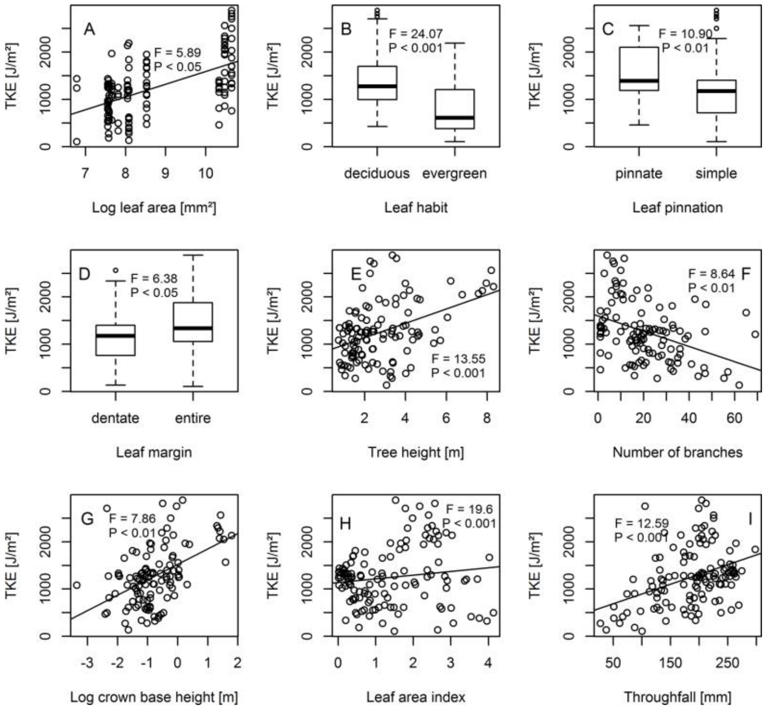


Figure 16: Throughfall kinetic energy (TKE) versus leaf traits (A-E), tree architectural traits (F-H) and abiotic covariates (I). Black solid lines indicate linear trend.

Results clearly showed that soil erosion in forest plantations can be mitigated by the appropriate choice of tree species. Besides the significant effect of plant traits on TKE, throughfall amount was also highly correlated with TKE. In several studies, throughfall amount was found to be the major driver of spatial variability of TKE (Geißler et al. 2012a; Scholten et al. 2011). However, the species comparison revealed that shifts in drop formation and drop velocity within a specific rainfall event might have a higher impact on TKE than the total amounts of rainfall. Thus, higher throughfall amounts do not necessarily lead to higher TKE at rainfall event level.

3.4. Additional study: Interrill erosion on agricultural land (manuscript 5)

The technical setup with micro-scale ROPs and a portable rainfall simulator was further tested and improved in an agricultural trial in Rümlang, Switzerland. Therefore, a field experiment was performed with the portable Tübingen rainfall simulator and micro-scale ROPs at the FArming System and Tillage experiment Agroscope (FAST) in order to investigate the influence of organic farming (Gomiero et al. 2011a; Gomiero et al. 2011b; Mäder et al. 2002) in combination with different tillage systems (Erhart and Hartl 2009; Teasdale et al. 2007) on soil erosion (Arnhold et al. 2014; Reganold et al. 1987). The field methods and the statistical approach were similar to previous studies in subtropical China. A randomized block design was established with four replications and a total of 32 ROPs. The experimental factors were the [1] organic (org) and [2] conventional (conv) farming system, as well as a tillage treatment with [3] ploughed plots (till) and plots under [4] reduced or no tillage (red-till, no-till).

Results showed that organic farming in combination with reduced-tillage practices protected agricultural land best against soil erosion. Organic farming practices significantly decreased soil erosion rates compared to conventional farming systems (Figure 17). Sediment discharge was 59 % higher ($p=0.018$) on conventional treatments (31.8 g m^{-2} h^{-1}) than on organic treatments (20.0 g m^{-2} h^{-1}). This effect was outperformed by the applied tillage system, as no-tillage had the most significant decreasing influence on erosion rates. Under both farming systems, ploughed treatments showed higher sediment discharge (conventional farming: 104 %, organic farming: 133 %, $p=0.004$) than treatments with reduced or no-tillage. This findings support assumptions which underline the potential of organic farming to reduce the risk of soil erosion (Gomiero et al. 2011a; Erhart and Hartl 2009). Furthermore, reduced-tillage practices seem to be a major soil-protecting improvement to organic farming and an efficient contribution to soil conservation.

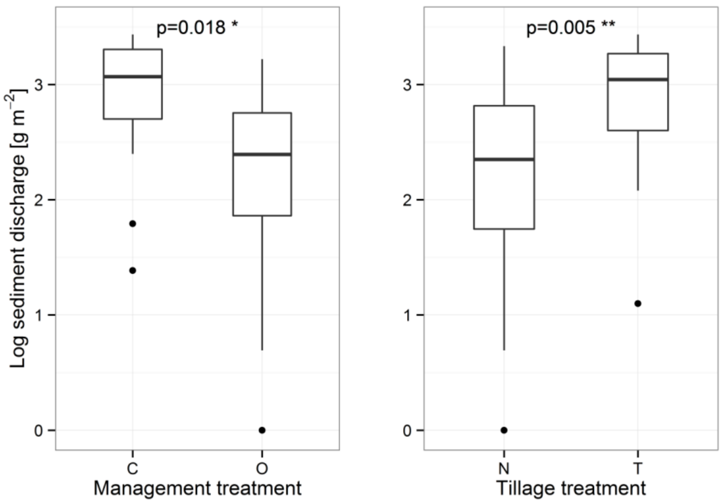


Figure 17: Log sediment discharge in management treatments (conventional and organic) and in tillage treatments (reduced-/no-tillage and tillage) at the FAST trial, Switzerland (n=32).

The combination of organic farming and reduced-tillage showed the lowest sediment discharge (7.5 g m^{-2} h^{-1}), whereas tilled conventional plots showed the highest rates (42.6 g m^{-2} h^{-1}) as shown in figure 18. Both were significantly different form each other (0.01). Conventional but no-tillage treatments performed better (20.8 g m^{-2} h^{-1}) and showed nearly the same erosion rates as tilled organic plots (21.5 g m^{-2} h^{-1}). They did not show different means (0.99), but were different from both organic farming with reduced tillage (0.05) and conventional tilled plots (0.38).

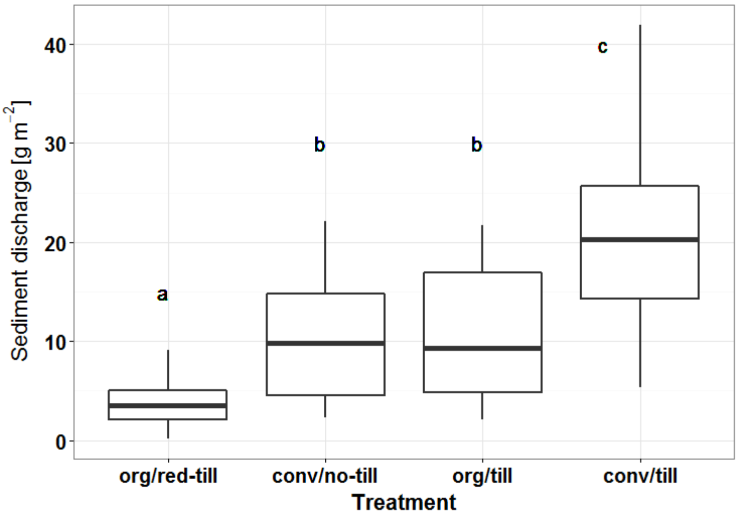


Figure 18: Sediment discharge on combined treatments (conventional/tillage, conventional/no-tillage, organic/tillage, organic/reduced-tillage, n=32).

This study was the first to compare the influences of organic and conventional farming in combination with different tillage systems on soil erosion rates in situ with an experimental field design. We could show that a well replicated and highly portable field setup with micro-scale ROPs and a lightweight rainfall simulator is a major improvement in order to investigate single vegetation and faunal effects on initial soil erosion processes. Moreover, we were able to gather reliable soil erosion data in situ and in short term without external parameterisation.

4. Summary and outlook

Experiments were carried out in a subtropical Chinese forest ecosystem in order to investigate the influences of tree species richness, tree species identity and species-specific functional traits on interrill erosion and TKE. Furthermore, the role of the leaf litter cover and soil mesofauna and macrofauna during initial soil erosion processes was studied. Therefore, a highly replicated field setup with micro-scale ROPs and splash cups was used.

Tree species richness did not affect sediment discharge, runoff and TKE in this experiment. Although a negative trend was visible on sediment discharge from diversity level 1 to 8, higher tree species richness was not mitigating soil erosion in this subtropical forest ecosystem. Nevertheless, neighbourhood diversity effects on TKE were present. 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 and a little developed canopy stratification. Nevertheless, mixed tree stands showed a more balanced and more homogenous vegetation development than monocultures. Thus, tree plantations with multiple species seem to ensure a more advanced soil cover than plantations with only one species. Furthermore, results showed that leaf litter species diversity did not influence leaf litter cover, sediment discharge and thus soil erosion. Hence, better overlap and gap-filling or different decomposition rates in highly diverse litter mixtures seemed not to be major parameters for soil erosion control, although a non-significant negative trend was visible.

Nevertheless, all studies conducted in the framework of this thesis showed that tree species identity influences initial soil erosion processes in subtropical forest ecosystems. Therefore, the appropriate choice of tree species plays a major role in erosion control. This effect becomes already considerable in an early successional stage and thus can be of importance during the establishment of tree plantations. It could be shown that *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* were related to decreasing soil erosion rates in young forest stands. Regarding the leaf litter cover on the forest floor, single leaf species showed variable influences on sediment discharge when compared among themselves. In this experiment, ROPs with leaf litter from *Machilus thunbergii* showed the highest sediment discharge, whereas plots with *Cyclobalanopsis glauca* showed the smallest rates. This can be related to variable leaf habitus, different decomposition rates and food preferences of litter decomposing fauna. The example of *Cyclobalanopsis glauca* further showed that different tree and leaf traits of one given species may have converse effects on soil erosion rates.

Furthermore, species-specific functional traits affected soil erosion rates. High crown cover and leaf area index reduced soil erosion, whereas it was slightly increased by increasing tree height. For instance, *Chorespondeas axillaris* was by far the fastest growing and tallest tree species with a nearly closed canopy and at the same time this species caused the highest amount of sediment discharge in this study. Thus, low tree stands with high canopy cover are effectively counteracting soil loss in initial forest ecosystem, before a full leaf litter layer on the forest floor has developed. At the same time, investigations on the kinetic energy of raindrops showed that low LAI, low tree height, simple pinnate leaves, dentate leaf margins, a high number of branches and a low crown base height effectively minimize TKE. Furthermore, evergreen species showed lower TKE than deciduous ones. However, traits, such as LAI or tree height, change with growing tree individuals while binary leaf morphological traits are believed not to change notably during tree growth. These implications need to be considered, when transferring the results to grown forest systems with a fully developed crown cover. Furthermore, it became clear that TKE distribution among different species is much more complex than throughfall distribution solely. Focusing on the spatial distribution of TKE, distinct positions that decrease TKE (below the first branch) or increase it (in the middle of four tree individuals) could be shown and finally, TKE was only higher than FKE for spatial spots below trees higher than 3.3 m.

Results further showed that the presence of soil mesofauna and macrofauna increased initial soil erosion. It is assumed that this faunal effect arises from arthropods slackening and processing the soil surface in subtropical forest ecosystems. Furthermore, soil mesofauna and macrofauna were a prominent factor in litter fragmentation and decomposition and thus the reduction of protecting litter covers. Effects of this fauna group on sediment discharge have to be considered in soil erosion experiments.

The field methodology with micro-scale ROPs and a portable rainfall simulator proofed its suitability and quick applicability in rough terrain conditions. Further tests and improvements in an agricultural environment in Switzerland showed its general reliability under different conditions. Focussing on interrill erosion processes with micro-scale runoff plots in combination with a high number of replications and the quick setup of the field devices will enhance further research on basic vegetation and faunal influences on soil erosion. Nevertheless, the additional determination of erosion rates from rill processes with larger ROPs is also of interest in the BEF China experiment.

Future research should concentrate on how erosion rates change with increasing stand age. Therefore, long-term monitoring of soil erosion under closing tree canopies is necessary. At the same time, further investigations on TKE below mature trees and dense canopies are needed to confirm the findings for later successional stages. Moreover, a wider range of functional tree traits should be taken into consideration. Especially studies on leaf traits (Nanko et al. 2013) as well as belowground stratification (Stokes et al. 2009; Gyssels et al. 2005) showed a potential to influence soil loss and pointed out the complexity of factors mitigating soil erosion in forest ecosystems. Investigations on the influence of biological soil crusts, topsoil microbial communities and their impact on organic matter accumulation will open the way to new insights on soil erosion processes. Another topic to be further investigated in the BEF China experiment is the nutrient cycling and transportation of SOM, nitrogen and phosphorus during the erosion processes under different tree species mixtures.

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Publications

Manuscript 1

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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 t ha^{-1} 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 conditions (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 relation 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., accepted_b).

The experimental area has been used as a commercial forest plantation (*Cunninghamia lancelota* and *Pinus massoniana*) 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 (Figure 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.

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.

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.

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.

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

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 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 (Figure 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 (g m-^{2}) and runoff volume (l m^{-2}) in relation to diversity levels were high (Figure 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 %.

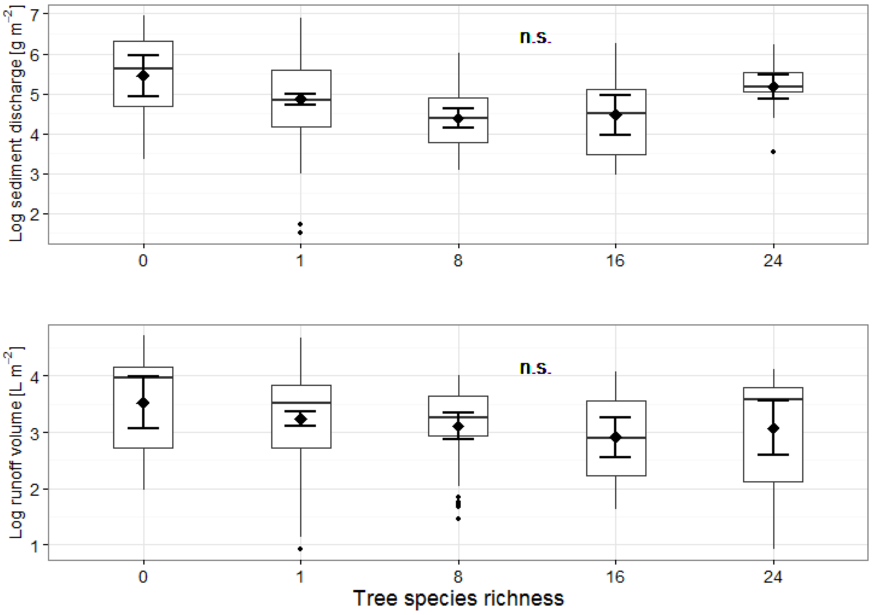

Figure 2: Sediment discharge and runoff volume at five diversity levels based on four rainfall events in May and June 2013 in Xingangshan, Jiangxi Province, PR China (n.s.: not significant, n=334). Horizontal line within boxplot represents median and diamond represents mean.

3.2 Species identity effects on interrill erosion processes

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

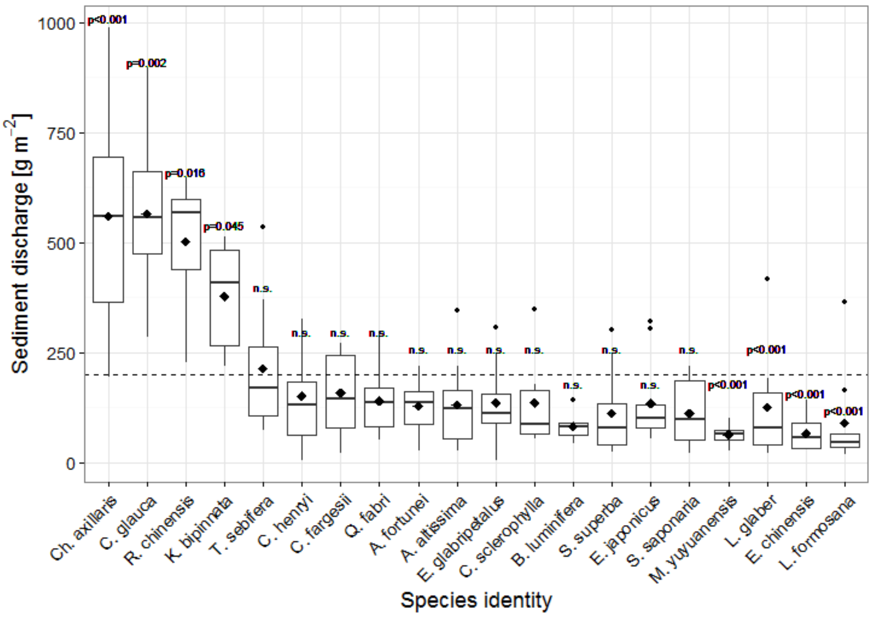

Figure 3: Sediment discharge under 20 tree species in monocultures based on four rainfall events in May and June 2013 in Xingangshan, Jiangxi Province, PR China. Dashed line indicates mean sediment discharge of all 20 species. Horizontal lines within boxplot represent median and diamonds represent mean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

The mean sediment discharge is 199 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).

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

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 (Figure 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 (Figure 4, Appendix Table A2).

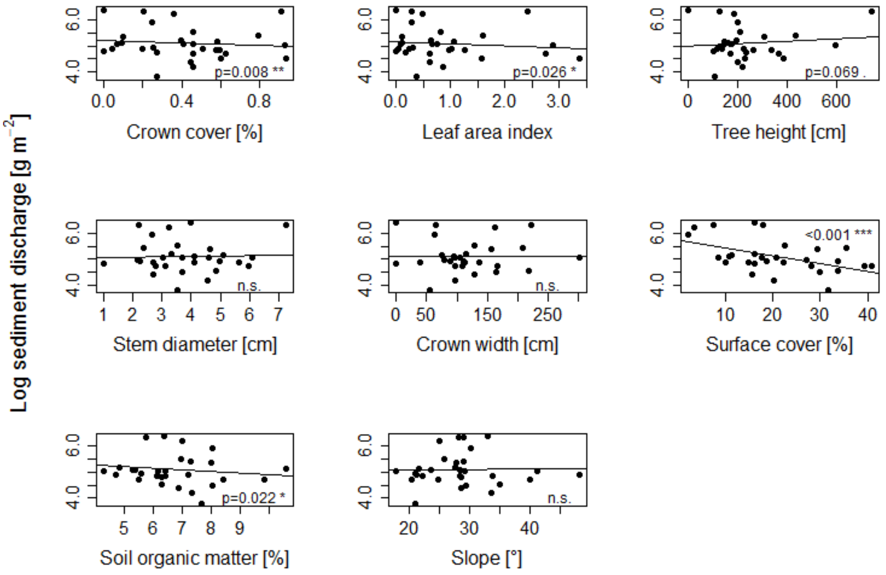

Figure 4: Effects of tree functional traits and site characteristics on sediment discharge. Analyses were based on four rainfall events in May and June 2013 in Xingangshan, Jiangxi Province, PR China. Black lines symbolize linear trends.

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).

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

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.

4 Discussions

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., accepted_b; 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. (accepted_a) 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., accepted_a). 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., accepted_a). 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., accepted_a). 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 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 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 t ha^{-1} a^{-1}) in a study based on small-scale and field ROPs. Pimentel (1993) reported an average rate of 36 t ha^{-1} a^{-1} for the same area. Zheng et al. (2007) stated an average soil loss rate of 31 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 m^2 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 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.

Appendices

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 *

Table A2: Results of the basic linear mixed effect model for sediment discharge (* : p<0.001, ** : p<0.01, * : p<0.05, . : p<0.1, n.s. : not significant; n=334). 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

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)

Author contribution

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

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The influence of leaf litter diversity and soil fauna on initial soil erosion in subtropical forests

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Abstract

Although the protective role of leaf litter cover against soil erosion is known for a long time, little research has been conducted on the processes involved. Moreover, the impact of soil meso- and macrofauna within the litter layer on erosion control is not clear. To investigate how leaf litter cover and diversity as well as meso- and macrofauna influence sediment discharge in subtropical forest ecosystems, a field experiment has been carried out in Southeast China. A full-factorial random design with 96 micro-scale runoff plots and 7 domestic leaf species was established and erosion was triggered by a rainfall simulator. Our results demonstrate that leaf litter cover protects soil from erosion (−82 % sediment discharge on leaf covered plots) by rainfall and this protection is removed as litter decomposes. The protective effect is influenced by the presence or absence of soil meso- and macrofauna. Fauna presence increases soil erosion rates significantly by 58 %, while leaf species diversity shows a non-significant negative trend. We assume that the faunal effect arises from arthropods slackening and processing the soil surface as well as fragmenting and decomposing the protecting leaf litter covers. Even though the diversity level did not show a significant influence, single leaf species in monocultures show rather different impacts on sediment discharge and thus, erosion control. In our experiment, runoff plots with leaf litter from *Machilus thunbergii* showed the highest sediment discharge (68.0 g m^{-2}) whereas plots with *Cyclobalanopsis glauca* showed the smallest rates (7.9 g m^{-2}).

Introduction

Soil erosion is one of the principal environmental problems worldwide, especially in regions with high human impacts on the ecosystems (Morgan 2005). In subtropical parts of China, for example, high rainfall intensities cause severe and continuous losses of soil and thus, important disservices in ecosystems (Wang et al., 2005; Shi et al., 2008). Soil erosion reduces soil organic matter (SOM), relocates nutrients, changes the water-holding capacity and can reduce the biodiversity of soil flora and fauna (Durán-Zuazo and Rodríguez-Pleguezuelo 2008). The degradation of soils will remain one of the principal environmental problems as an augmentation of extreme weather events and changes in mean precipitation are stated in future (Intergovernmental Panel on Climate Change (IPCC) 2007; Xu et al., 2007).

Besides soil erodibility, rainfall erosivity and inclination, vegetation is regarded as a main attribute to describe the intensity and kind of erosion caused by water (Thornes 1990). Forest in general reduces the risk of considerable soil loss and especially the litter cover is known to reduce the erosivity of rainfall by absorbing the impact of raindrops, leading to a lower sediment discharge and runoff volume compared to bare ground (Morgan 2005). Nevertheless, leaf exudates and low sunlight in dense forest plantations can lead to a declining shrub and herb layer and so to bare ground, thus increasing runoff and sediment discharge (Onda et al., 2010). Geißler et al. (2012) showed that the erosive power of raindrops is significantly higher under subtropical forest than under fallow land due to changes in drop sizes. It has been demonstrated that leaf litter quality is an important ecosystem factor in temperate climates with direct impact on soil characteristics (Kooijman and Cammeraat 2010). Leaves of different species vary in their sizes, shapes, decomposition rates (Cornelissen 1996) and water storage capacities within a litter layer (Kim et al., 2014), which has an important influence on ground coverage and surface runoff. Sidle et al. (2007) described a shallow preferential flow in soil-overlaying organic horizons (‘biomat flow’), which is not initiating soil erosion. It is suggested that plant species and functional diversity as well as a highly structured and diverse litter cover is an important factor for soil erosion control on mountain slopes (Körner and Spehn 2002; Martin et al., 2010). Although the protective role of litter cover against soil erosion has been known for a long time (e.g. Smith 1914), only little research has been conducted on the processes involved.

In general, quality and quantity of litter determines decomposer communities ranging from microbes and fungi to animals of different size classes (Hättenschwiler et al., 2005). Leaf litter provides habitats, maintains a favourable microclimate for soil fauna and further on, it is an important food source (Sayer 2006). Although the main part of litter decomposition is performed by microbes (Bardgett 2005), the mesofauna and macrofauna constitutes the dominant physical litter transformers. It is consequently essential for promoting both, litter decomposition (Hättenschwiler and Gasser 2005) and physical–chemical soil parameters (Gabet et al., 2003). By that, these organisms have the potential to influence geomorphological processes like soil erosion (Butler 1995; Wheaton et al., 2011; Butler and Sawyer 2012). Allen et al. (2014) recently linked ecological principles to geomorphological aspects in general. However, our knowledge about how these principles affect soil erosion is still rudimentary. Cammeraat and Kooijman (2009) consider both, soil fauna and tree type as important ecosystem engineers. At the same time, most of the relevant studies on faunal effects deal with bioturbation and illustrate the role of one or few functional groups of soil mesofauna and macrofauna as geomorphic agents (Viles 1988; Hupp et al., 1995). For instance, the crucial role of earthworms influencing soil structure and related soil physical properties is relatively well studied (Blanchart et al., 2004). Earthworms can decrease surface water runoff by increasing soil porosity (Edwards and Bohlen 1996). Porosity is indirectly influenced by leaf litter and SOM, as they are a major food supply to earthworms, but also arthropods (Sayer 2006). Ants and termites can also have an effect on soil turnover (Butler 1995). For example, Cerdà and Jurgensen (2011) indicated that the presence of ant nests decreases surface runoff but increases sediment loss owing to unconsolidated soil mounds. Nevertheless, the impact of other soil mesofauna and macrofauna on soil erosion, especially in ecosystems lacking the important influence of earthworms, is not yet clear.

Focusing on initial soil erosion which is triggered by rain splash striking the soil surface (e.g. Goebes et al., 2014) and shallow overland flow transporting soil particles (e.g. Shi et al., 2010), our research concentrates on how litter cover and litter diversity as well as mesofauna and macrofauna influence sediment discharge. From a hypothetical point of view, full leaf covered soil surfaces might be better protected against soil erosion by greater overlap and gap-filling in highly diverse leaf litter mixtures. At the same time, highly diverse leaf litter mixtures may lead to faster leaf decomposition caused by more active decomposer communities and thus, soil surfaces will get bared quicker. As a consequence, this effect could lead to an inferior conservation of soil surfaces and therefore, to higher erosion rates. At last, we hypothesize that soil mesofauna and macrofauna will have an impact on leaf litter cover and sediment discharge.

Based on these hypotheses, our research questions are:

1. Does higher leaf litter species diversity lead to decreasing sediment discharge, when soil surfaces are fully covered and to increasing sediment discharge, when soil surfaces are partly covered?
2. Does the presence of soil mesofauna and macrofauna lead to a faster decomposition of leaf litter cover and thus, increasing sediment discharge?

To examine the influence of leaf litter in forest ecosystems, experimental approaches are widely used, but mostly performed in temperate climates (Sayer 2006). Runoff plots are a common mean in soil erosion research to investigate sediment discharge and surface runoff (Richter 1998). To standardize rainfall conditions for those erosion measurements, rainfall simulators are widely used (Iserloh et al., 2013b). In order to investigate the role of litter diversity and mesofauna and macrofauna decomposers on soil erosion, we established a full factorial replicated field experiment that manipulated the amount and diversity of leaf litter and decomposer abundances in a subtropical forest system in Southeast China.

Material and Methods

Study site

The study took place in a *Castanea mollissima* plantation (established in 1996) near Xingangshan, Jiangxi Province, P.R. China [N 29°05.193′ and E 117°55.533′, 125 m above sea level (a.s.l.), Supporting Information Figure S1], which is part of the Biodiversity and Ecosystem Functioning project “BEF China” (Bruelheide et al., 2011, 2014). The climate in Xingangshan is characteristic for subtropical summer monsoon regions with a mean annual temperature of 17.4 °C and a mean annual rainfall of 1635 mm (Köppen–Geiger classification: Cwa). In summer 2012 meteorological data showed typical monsoon patterns with increasing mean temperature from May (21.8 °C) to August (27.4 °C) and a mean monthly precipitation of 200 mm. In September temperature decreased (22.5 °C) and the end of the monsoon season led to considerably lower precipitation (116 mm). The plantation covers an area of 1200 m^2 with an elevation difference of 11 m and is characterized by moderately steep to steep slopes (14°–38°) without an ectorganic layer. Within a distance of 5–10 m, it is surrounded by region-specific secondary forest. The soil is an Endoleptic Cambisol (cf. IUSS Working Group WRB 2006) developed on saprolite weathered to silt loam (sand = 24%, silt = 50%, clay = 26%). It does not change considerably within the study site. The bedrock consists of non-calcareous slates. Mean bulk soil density is low (1.15 g cm^{-3}).

Runoff plots (ROP)

Sediment discharge and runoff volume were measured with micro-scale runoff plots (ROPs) (0.4 m × 0.4 m) to determine initial soil erosion. The ROPs were made of stainless steel panels and connected to a covered runoff-gutter at the lower end. In this particular experiment we incorporated four holes to the side panels (circular, diameter 50 mm). Those holes were equipped with mesh (size 20 mm) to exclude bigger animals (e.g. mice and toads), but allow access to litter decomposing mesofauna and macrofauna. Half of the plots were additionally equipped with pitfall traps (diameter 0.09 m, depth 0.15 m, capacity 0.55 l) and fine mesh (size < 0.1 mm) to exclude or at least strongly reduce soil mesofauna and macrofauna from ROPs (fauna treatment). The traps were filled with 0.15 l of preserving solution (40% ethanol, 30% water, 20% glycerol, 10% acetic acid, some drops of detergent to reduce surface tension), which has shown its reliability in subtropical climates (Schuldt et al., 2011). In between the measuring campaigns all ROPs have been provided with 20-mm-mesh to fix the experimental leaves against moving downslope and to exclude falling leaves from *Castanea mollissima* (Figure 1).

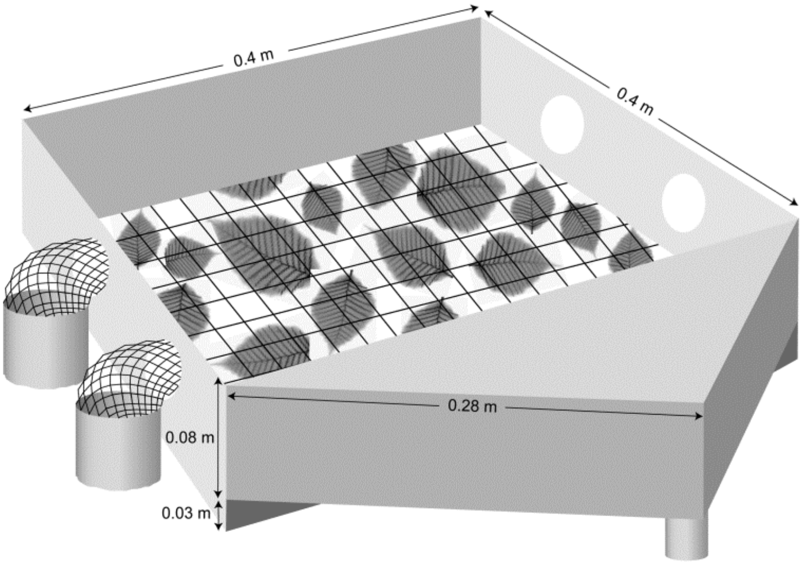

Figure 1. Runoff plot (ROP) with fauna treatment. ROPs (0.4 m × 0.4 m) consist of stainless-steel side panels and a triangular covered runoff gutter. Falling leaves are separated from experimental leaf mixtures by a 20-mm-mesh. Pitfall traps (two at each side) are installed to exclude soil mesofauna and macrofauna.

Micro-scale ROPs were chosen to establish a maximum number of replicates within a factorial random design and to assure a high level of maintenance and control (c.f.

Hudson 1993). Compared to small-scale ROPs and field plots, they are easier to handle and monitor in greater number, especially in rough terrain and climate conditions. Although ROP measurements in general are strongly influenced by arbitrary natural and measurement variability (Wendt et al., 1986), micro-scale ROPs allow focusing on initial soil erosion. Rilling processes do not occur on such short distances and the parameters of the erosion process can be controlled closer than on larger plots (Mutchler et al., 1994; Stroosnijder 2005). The stability of soil aggregates influenced e.g. by SOM or mesofauna and macrofauna are of major importance at this scale (Morgan 2005). Nevertheless, the use of such short ROPs raises certain constraints. Rim and boundary effects like the movement of splashed particles in and out of the plot have to be considered (Mutchler et al., 1994). The average splash distance for silt loam at every plot is approximately one quarter of the ROP-length (0.12 m, cf. Legout et al., 2005). The length of the ROPs is another aspect, which affects erosion measurements (Bagarello and Ferro 2004). Several studies showed that short slopes lead to higher runoff volume per surface area than longer slopes (e.g. Van de Giesen et al., 2000; Joel et al., 2002), which further leads to higher sediment discharge. Yet, Thomaz and Vestena (2012) compared micro-scale plots (1 m^2) with larger plots (10 m^2) in a subtropical environment and found that runoff volume was higher in smaller plots, but soil loss was similar. We assume that our experimental erosion rates are variational compared to natural erosion rates and values should be extrapolated with caution. Nevertheless, interplot comparison is reasonable.

Experimental design

Leaf litter of seven domestic tree species [*Liquidambar formosana* (Hance, species A), *Machilus thunbergii* (Sieb. et Zucc., species B), *Quercus serrata* (Thunb., species C), *Schima superba* (Gardn. et Champ., species D), *Castanopsis eyrei* (Champ. et Benth., species E), *Castanea henryi* (Rehd. et Wils., species F), *Cyclobalanopsis glauca* (Thunb., species G), author and abbreviation for given species in parentheses] was collected in Gutianshan National Nature Reserve close to our study site (Geißler et al., 2013). Species were organized in two pools with one overlapping species (pool I: species A, B, C, and D; pool II: species D, E, F, and G) and distributed into the ROPs. Using one, two and four species mixtures as well as a bare ground feature (diversity 0), we obtained 24 ROPs (12 ROPs for each species pool). To investigate the mesofaunal and macrofaunal influence, we applied a fauna exclusion feature using two different types of ROPs (see earlier) which doubled the number to 48 ROPs. Moreover, we replicated all plots and hence, achieved a total number of 96 ROPs (Figure 2). ROPs were placed randomly on the study site in four blocks (species pool I + replication and species pool II + replication).

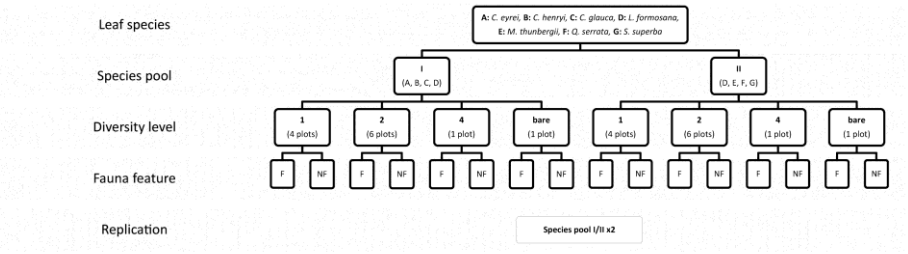

Figure 2. Experimental design with seven leaf species, two species pools (pool I and pool II), four litter diversity levels (bare ground, 1, 2, 4 species), fauna feature (F = fauna, NF = no fauna) and replication (96 runoff plots in total).

Leaf litter blends were mixed before deployment to avoid local patches of individual species (Burghouts et al., 1998) and distributed randomly among the ROPs (60 g for each ROP). Prior to this, alien leaves, twigs and loose stones (>2 mm) have been removed from the plot surfaces. Remaining stone cover was recorded photogrammetrically. The distribution was conducted six weeks before the first measurement to minimize plot installation effects and allow local leaf decomposer communities to adapt to the used leaf species. Soil mesofauna and macrofauna are mostly food generalists and can adapt quickly to changed food sources (Hättenschwiler et al., 2005). Leaf decomposition in general is largely influenced by litter quality determined by nutrient concentrations and the carbon/nitrogen (C/N) and lignin/N ratio (cf. Wang et al., 2009; Purahong et al., 2014). Dependent on the quality, the adaptation by microbes and bacteria can be fast, which is also true for our study plots (Seidelmann and Scherer-Lorenzen, personal communication, 2014).

Rainfall simulation and sediment treatment

The rainfall simulation was carried out at two timesteps (May and September 2012) with the portable Tübingen rainfall simulator (Iserloh et al., 2013a). It was modified by a Lechler 460.788.30 nozzle and adjusted to a falling height of 3.5 m (pressure at nozzle: 150 hPa). The sprinkle area was 1 m^2 and protected from outer influences by a light frame tent (16 m^3). Rainfall intensity was set to 60 mm h^{-1} for all simulation runs according to a typical regional rain storm event, as monitored in Xingangshan (2009–2012) about 5 km northeast of our experimental site. Drop size spectrum and intensity of the simulator were calibrated using a ‘Laser Precipitation Monitor’ by Thies (Lanzinger et al., 2006) to obtain homogeneous characteristics. Rainfall was simulated for 20 minutes at each ROP. To consider the effect of different water saturations, we applied a first 20 minute run on actual field water saturated ground and a second run 15 minutes later, when soils had higher water saturation. Sediment discharge was separated from runoff by vacuum filtration (fibreglass filters). The solid subsample was oven dried (40 °C) before weighing. Measurements were carried out when ROPs were fully covered (May 2012, 128 mm of natural rainfall during timestep 1) and after decomposition had led to reduced leaf litter coverage (September 2012, no natural rainfall during timestep 2). Leaf covered ROPs have not laid bare completely in the second timestep. No rills were present in the experimental plots.

Soil properties

We recorded 48 soil profiles to assure that conditions are equal for every pair of ROPs. Bulk soil density (depth 0.05 m) was measured with the mass-per-volume method (100 cm^3 intact core). The pH was determined in potassium chloride (KCl) (10 g soil on 25 ml 1 M KCl) with a WTW pH-meter and Sentix 81 electrodes. Soil organic carbon was measured with an elemental analyser (purge and trap chromatography, Elementar vario EL III). During the experiment we assessed soil moisture at both timesteps and every ROP with a Delta-T Device Wet2-Sensor (TDR). Leaf litter cover of the plots was recorded photogrammetrically at the beginning and at the end of the experiment. Mesofauna and macrofauna from pitfall traps were counted, identified and classified to higher taxonomic levels.

Data analysis

We used linear mixed effects (LME) models with species pool and block fitted first as fixed effects, followed by litter diversity, fauna treatment and their interactions. Species composition and plot as well as their interactions with the faunal exclusion treatment were fitted as random effects. For the analysis of the repeated measures, we also included the interactions of composition with the replicated rainfall simulations and the timestep of the rainfall event. These random terms ensure that effects of diversity and of diversity with the respective interactions are tested based on the correct number of replicates.

Since bare ground is different in quality from the other diversity levels, a binary contrast variable ‘litter cover’ was fitted before litter diversity. When there were indications of trends, we tested whether specific diversity levels were different from others using linear contrasts (e.g. whether monocultures were different from mixtures). Functional diversity can sometimes be a more important determinant of ecosystem functions than the number of species (Scherer-Lorenzen 2005). We determined functional diversity (sensu Petchey and Gaston 2006) using leaf traits (leaf area, leaf mass per area, C/N ratio and plant family). Leaf traits were recorded on green leaves sampled in the Gutianshan National Nature Reserve (Kröber et al., 2012). Leaf decomposition rates were calculated as remaining leaf mass at timestep 2 compared to timestep 1. Influences of site conditions were tested using the covariables slope, pH, soil moisture, soil organic carbon and bulk density. Runoff volume was used as covariable when fitting sediment discharge. All data (except runoff volume and leaf decomposition) were log-transformed to achieve normal residual distributions. All mixed models were fitted using R 2.15.3 (R Development Core Team 2013) and ASReml (Butler 2009).

Results

Effects on initial soil erosion

The result of the LME modelling for sediment discharge is shown in Table 1. Mean sediment discharge and runoff volume were calculated per square metre and 20 minutes run (Supporting Information Table S1).

*Table 1. Results of linear mixed effects modelling for sediment discharge with covariables and design effects (*** : p<0.001, ** : p<0.01, * : p<0.05, n.s. : not significant, - : not tested).*

	timestep 1 & 2	timestep 1	timestep 2
diversity level	n.s.	n.s.	n.s.
functional diversity	n.s.	n.s.	n.s.
monocultures vs. mixtures	n.s.	n.s.	n.s.
species identity vs. mixtures	n.s.	n.s.	n.s.
fauna treatment	***	*	***
fauna x diversity level	n.s.	n.s.	n.s.
covariables			
bulk density	n.s.	n.s.	*
litter cover	***	**	***
moisture	*	*	*
soil organic carbon	n.s.	n.s.	n.s.
pH	n.s.	n.s.	n.s.
slope	n.s.	n.s.	n.s.
design effects			
block	***	*	***
run	**	**	***
species pool	***	*	***
timestep	***	-	-

Mean sediment discharge was 60% higher in September than in May (Figure 3). In May mean sediment discharge decreased from bare plots (diversity level 0) to leaf covered plots by 82% ($p < 0.001$). A slight decrease was still visible in September but no longer significant with reduced litter cover (Table S1). Mean runoff volume was higher in the rainy season (timestep 1) than at the end of summer (timestep 2) and bare plots showed 28% ($p < 0.01$) more runoff in May and 13% [not significant (n.s.)] more runoff in September than leaf covered plots (Table S1). Sediment discharge was on average lower at high than at low diversity levels, but at both timesteps no significant effect of litter diversity on sediment discharge could be found. Likewise, functional diversity had no significant effect. Furthermore, no effect of mono-species plots against mixed species plots and single species within their species pool on sediment discharge could be found.

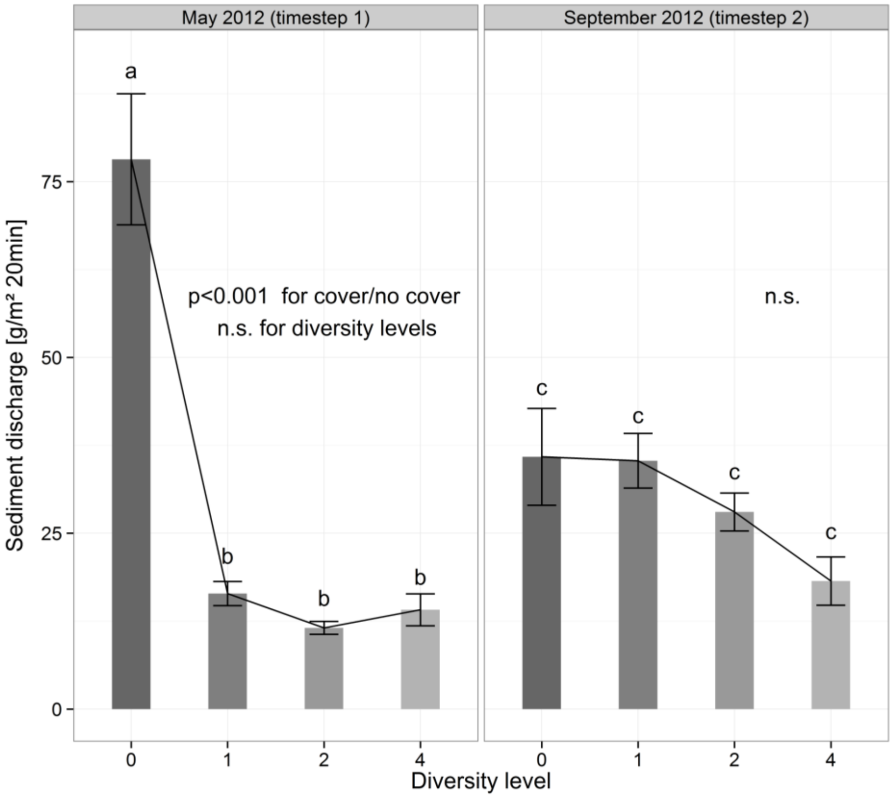

Figure 3. Effect of diversity levels and bare ground (diversity = 0) on sediment discharge at timestep 1 (May 2012) and timestep 2 (September 2012). Small letters indicate significant mean differences.

A clear difference between fauna and no-fauna treatment was evident at both timesteps (Figure 4) and sediment discharge was significantly influenced by the fauna treatment in May ($p < 0.001$) and September ($p < 0.05$). Discharge was 58% higher with fauna than without. This effect is decreasing from timestep 1 (60%) to timestep 2 (56%).

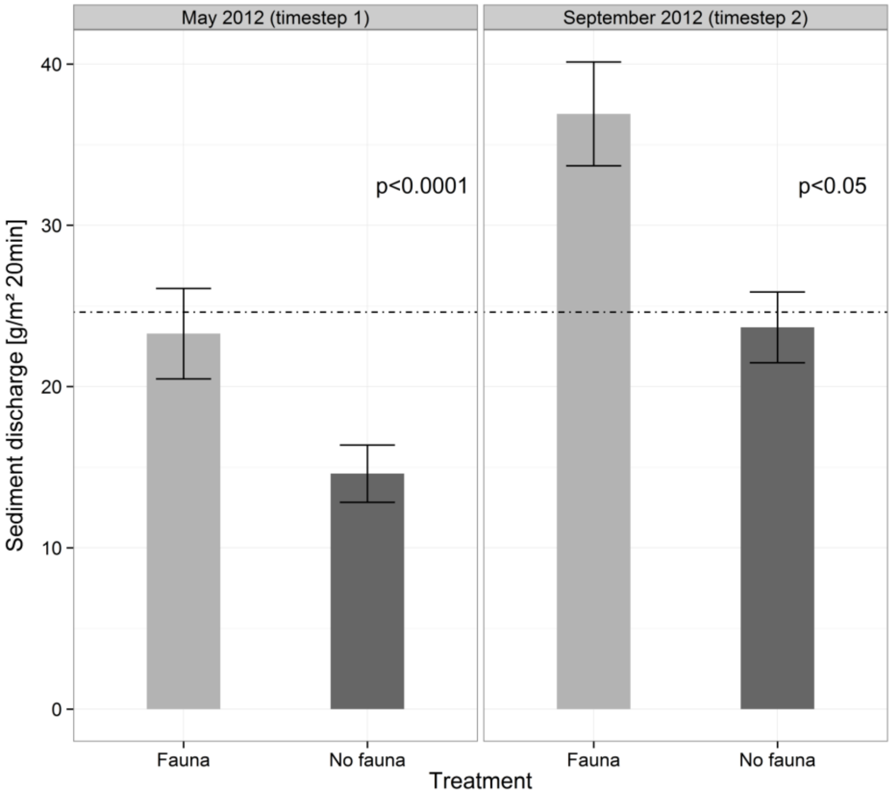

Figure 4. Effect of the fauna treatment on sediment discharge at timestep 1 (May 2012) and timestep 2 (September 2012).

All monocultures showed slightly different effects on sediment discharge and several species (species A and B: $p < 0.001$, species G: $p < 0.01$) showed significant deviation compared to the overall mean in timestep 2 (Figure 5). These differences did only occur in September, where influences of different leaf litter species were more heterogeneous than in May.

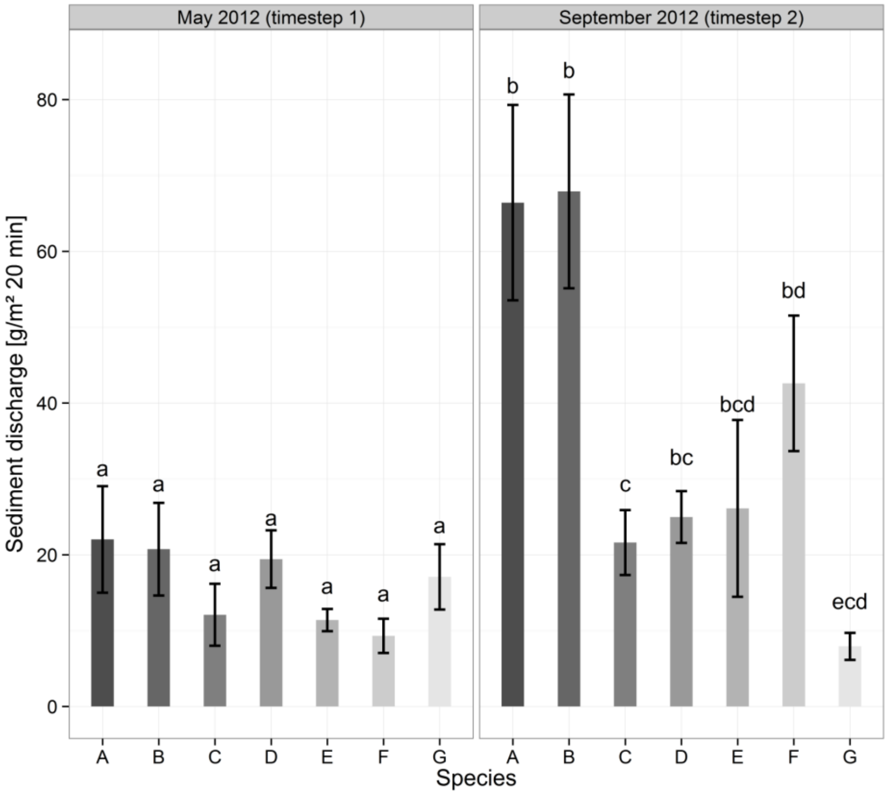

Figure 5. Relation between sediment discharge and monocultures of leaf litter species at timesteps 1 and 2. Small letters indicate significant mean differences.

Slope was not affecting sediment discharge at any time. Litter cover had a negative effect on sediment discharge ($p < 0.001$) and bulk soil density became significant in timestep 2 ($p < 0.05$). Moreover, soil moisture showed a negative effect on sediment discharge ($p < 0.001$) and mean moisture was 45% in the rainy season (timestep 1) compared to 26% at the end of summer (timestep 2). Mean moisture at the second rainfall simulation run (41%) was 24% higher than at the first run (31%). Mean soil organic carbon content was low (1.6%) and mean pH value was acidic (3.7). Both parameters did not affect soil erosion. Surface sealing and crusting could be seen on all plots during the rainfall simulations, but conditions were equal for all plot surfaces. No signs for a significant role of water repellency have been found. Basic data on soil parameters and soil moisture is presented in Supporting Information Table S2.

Mesofauna and macrofauna abundance

In the ROPs with fauna exclusion we found 2101 individuals belonging to the mesofaunal and macrofaunal size class. Springtails (Collembola) were the dominating taxon in the catches on the study site with 1038 individuals. Mites (Acari, 496 individuals), were the second largest group. Further taxa, which have a proved influence on soil structure and particles, were ants (Formicidae, 300 individuals), worms (Oligochaeta, 145 individuals) and beetles (Coleoptera, 49 individuals). Additionally, 73 larvae of unidentified insects were found. No ant mounds could be identified within the ROPs. Anecic earthworms, which are known to build vertical passages, were not found in the pitfall traps and are rare in the given study area (own observation) and under the acidic soil conditions.

Leaf litter cover and decomposition

The time between measurements was 19 weeks and none of the litter covered plots had laid bare completely within this timespan. Mean litter cover was 91% (minimum = 80%, maximum = 100%) in May, when litter was brought out and 38% (minimum = 4%, maximum = 65%) in September, when coverage was reduced due to litter decomposition. Leaf litter mass was 60 g in every ROP in May and 8.2 g (minimum = 1 g, maximum = 17 g) in September. On average, one tenth of leaf litter mass brought out in timestep 1 was still remaining in timestep 2. Differences in soil surface coverage due to different diversity levels could not be detected ($p > 0.1$).

Nevertheless, different leaf species showed rather different decomposition rates and thus, developments of surface litter cover. Species A (*Liquidambar formosana*) showed the fastest decomposition and low surface cover [2% remaining leaf mass (RLM), 10% surface cover (SC)] and species D (*Schima superba*) the slowest decomposition and high surface cover (22% RLM, 56% SC). Species B (12% RLM, 34% SC) and G (11% RLM, 27% SC) had comparable values near to the overall mean (12% RLM, 38% SC). The four species mixtures showed a mean of 11% RLM and 31% SC.

ROPs with fauna (11.2 %) lost 2% more of their leaf coverage than plots without fauna (12.9%). A slightly significant effect between fauna treatment and leaf mass loss was found ($p < 0.07$), but there is no clear directional pattern. Depending on the species and mixture, fauna shows very heterogeneous impacts on decomposition in both, positive and negative directions.

Discussion

Our results demonstrate that leaf litter cover protects soil from erosion by rainfall and this protection is removed as litter decomposes. However, this protective effect is not influenced by leaf litter species diversity, but by the presence or absence of soil mesofauna and macrofauna.

Effect of litter cover on initial soil erosion

An important difference was found between leaf-covered and bare plots. On the latter, sediment discharge was up to 1.8 times higher in timestep 1. Furthermore, erosion rates were considerably higher in September with reduced leaf litter coverage than in May. This finding emphasizes the importance of a protecting litter cover in forest ecosystems against soil erosion. A surface litter cover of nearly 100% decreased soil loss by 82% compared to bare ground. Benkobi et al., (1993) indicated that a 60% surface litter cover can reduce soil loss by 62%. Moreover, erosion rates did not differ significantly between plots with a litter cover of up to 38% (in September) and bare ground plots. Runoff volume was significantly lower on leaf covered plots than on bare plots in timestep 1. Gerrits et al., (2007) illustrated that forest floor interception e.g. by short vegetation and litter can double the amount of intercepted rainfall in forest ecosystems and therefore, lead to lower surface runoff. Runoff volume has been 18% higher during wet runs compared to dry runs due to higher initial soil water saturation. Nevertheless, sediment discharge has been 13% lower during wet runs, which we ascribe to the wash-out of loose sediments during the first run.

Effect of leaf litter diversity on initial soil erosion

Leaf species diversity did not have an impact on initial soil erosion. Even if rising leaf litter diversity seems to lead to a slightly smaller sediment discharge, these effects were not statistically significant. Moreover, no change in the influence of leaf litter diversity could be detected from May (full leaf coverage) to September (reduced leaf coverage). Functional diversity of leaf traits did not influence erosion parameters. However, single leaf litter species differ in their influence on soil erosion as a result of their different sizes, shapes and decomposition rates. This may not be detectable in the functional diversity index, as different leaf traits are merged. The large range of litter cover percentages indicates important differences in the development of soil cover between single leaf species. Nevertheless, a positive effect of more diverse leaf mixtures cannot be confirmed nor is one species driving a diversity effect through the mixtures. The overlap and gap-filling within the leaf mixtures seems not to differ between the diversity levels. Different positive and negative feedbacks of leaf litter species are levelled within their mixtures and no directional pattern could be detected (cf. Hättenschwiler et al., 2005). In addition to the latter, different decomposition rates of single leaf species resulted in a more heterogeneous influence on soil erosion in September. Sediment discharge from ROPs with *Liquidambar formosana* (species A), *Machilus thunbergii* (species B) and *Cyclobalanopsis glauca* (species G) – each in monocultures – differed significantly from the mean sediment discharge. We found that species A

(66.1 g m^{-2}) and B (68.0 g m^{-2}) showed significantly higher erosion rates, while ROPs with species G (7.9 g m^{-2}) had the lowest sediment discharge in September. Species A showed the fastest decomposition and low surface cover (2% RLM, 10% SC), whereas species B (12% RLM, 34% SC) and G (11% RLM, 27% SC) had comparable values next to the overall mean. While the effect of species A and G on soil erosion can therefore be explained by altered soil protection caused by different decomposition rates, the effect of species B has to be interpreted by other processes.

Effect of soil fauna on initial soil erosion

The presence of soil fauna has a significant impulse on sediment discharge. If fauna was present in our plots, erosion rates were higher. Macropore-building soil fauna as well as bigger ground-living animals like mice were not present in the experimental setup. However, the activity of soil-dwelling and surface-active mesofaunal and macrofaunal organisms may lead to the loosening and translocation of soil particles within the first centimetres of the soil surface. While springtails and mites were particularly abundant and might have contributed directly or indirectly to this process (by promoting the decomposition of the soil-protecting litter layer; Hättenschwiler et al., 2005), larger-sized and highly active ants and beetles can be assumed to have contributed significantly to the modification of the soil surface. Many species of these macrofauna groups influence soil processes in terrestrial ecosystems (Lavelle et al., 1997) and can show burrowing behaviour when searching for food or when trying to evade short- or long-term unsuitable microclimatic conditions on the soil surface (Swift et al., 1979; Gabet et al., 2003; Dostál et al., 2005). When leaf cover declines by decomposition, soil mesofauna and macrofauna attempt to protect themselves against transpirational water loss. This, in turn, can help to explain the fact that the impact of soil fauna slightly lost power in September. At this time, litter cover on the plots was reduced and microclimatic conditions were less optimal than earlier in the season, so the soil faunal activity was on the decline. Plots were not completely bare and thus, still offered resources for decomposers and associated predators. Exclusion of the soil fauna apparently removed significant faunal impacts on soil surface slackening and decomposition processes that favour soil particles to be detached and washed away during rainfall events. Longer-term changes in microclimatic conditions after litter is completely removed might lead to the disappearance of many soil organisms (e.g. Gill 1969). Therefore, we assume that the erosion-supporting faunal effect disappears when ROPs get completely bare and do not provide habitats for decomposing animals over a longer period.

Our results further show that the soil mesofauna and macrofauna has an influence on litter decomposition and thus, on the reduction of the protecting litter cover. Several of the more abundant taxa found in our plots, such as Acari, Collembola and Oligochaeta, play an important role in litter fragmentation and decomposition (Swift et al., 1979; Hättenschwiler et al., 2005). Over all ROPs the presence of soil fauna led to a slight increase in mean leaf litter decomposition rate compared to no fauna treatments. Nevertheless, a general pattern did not appear and results are contrasting when single leaf litter mixtures come into account. This effect is based on positive and negative feedbacks of soil fauna on different food availability and habitat conditions (Hättenschwiler et al., 2005). In our experiment positive feedbacks were slightly dominant and, as discussed earlier, might have strengthened the faunal effects on soil erosion.

Design of experimental erosion research

No significant influence of slope could be found. This may be caused by the small size of the applied ROPs with a short slope line of 0.4 m and a small length-slope factor (cf. Renard et al., 1987). Rilling processes due to low shear velocities of running water do not occur (cf. Chaplot and Le Bissonnais 2000). This result emphasizes the utility of small-scale ROPs for experimental erosion measurements where single effects have to be considered within a complicated system of parameters. Therefore, it is advisable to take out dominant parameters or to level them for every measurement. An example of this approach is the use of a portable rainfall simulator with constant rainfall intensity for every measurement.

Conclusions

We studied the influence of leaf litter cover and litter diversity as well as soil mesofauna and macrofauna on initial soil erosion processes in subtropical forest ecosystems. The general importance of leaf litter cover to protect soil against erosion can be underpinned by this study. Sediment discharge rates on bare ground plots exceeded tremendously the rates on covered plots, regardless of their diversity level. Leaf litter covers of more than about 40% of the ground surface can prevent soil erosion effectively.

Furthermore, the experimental investigation leads to the following conclusions:

1. This study provides evidence, that neither leaf litter species diversity nor functional diversity influence leaf litter cover, sediment discharge and thus, soil erosion. Hence, better overlap and gap-filling or different decomposition rates in highly diverse litter mixtures seem not to be considerable parameters for soil erosion control, although a non-significant negative trend is visible. Nevertheless, single leaf species show variable influences on sediment discharge if compared among each other. In our experiment, ROPs with leaf litter from *Machilus thunbergii* showed the highest sediment discharge whereas plots with *Cyclobalanopsis glauca* showed the smallest rates. This can be related to variable leaf habitus, different decomposition rates and food preferences of litter decomposing fauna.
2. Our results show that the presence of soil mesofauna and macrofauna increases initial soil erosion. We assume that this faunal effect arises from arthropods slackening and processing the soil surface in subtropical forest ecosystems. Furthermore, soil mesofauna and macrofauna are a prominent factor in litter fragmentation and decomposition and thus, the reduction of protecting litter covers. Effects of this fauna group on sediment discharge have to be considered in soil erosion experiments.

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Supporting information

Figure S1. Study site near Xingangshan.

Table S1. Mean sediment discharge and surface runoff volume.

	sediment discharge *(g m^{-2} / 20 min)*	runoff volume *(l m^{-2} / 20 min)*		by timesteps	sediment discharge *(g m^{-2} / 20 min)*	runoff volume *(l m^{-2} / 20 min)*
total	24.6	18.5	timestep 1	fauna tr.	23.3	21.0
timestep 1	18.9	19.7		no-fauna tr.	14.6	18.4
timestep 2	30.3	17.4		diversity 0	78.2	27.0
fauna	30.1	19.6		diversity 1	16.4	19.7
no fauna	19.1	17.4		diversity 2	11.6	18.4
1st run	26.3	17.0		diversity 4	14.1	20.4
2nd run	23.0	20.1	timestep 2	fauna tr.	36.9	18.3
diversity 0	57.0	23.5		no-fauna tr.	23.7	16.5
diversity 1	25.9	18.9		diversity 0	35.9	20.0
diversity 2	19.8	17.4		diversity 1	35.3	18.2
diversity 4	16.2	19.0		diversity 2	28.0	16.4
				diversity 4	18.2	17.6

Table S2. Basic site and soil parameters.

	mean	min	max
bulk density (g cm $^{-3}$)	1.15	1.07	1.22
pH	3.7	3.6	3.8
soil organic carbon (%)	1.6	1.1	2.0
slope (°)	24	14	38
soil texture (%)			
sand	24	21.4	26.3
silt	50	49.1	50.9
clay	26	23.8	30.9
soil moisture (%)			
in May	45	31	60
in September	26	11	42
before dry run	31	11	49
before wet run	41	22	60

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Manuscript 3

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Throughfall kinetic energy in young subtropical forests: Investigation on tree species richness effects and spatial variability

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Abstract

Soil erosion threatens ecosystem functioning by reducing soil organic carbon stocks or relocating nutrients. A common measure to protect soil against erosion is afforestation. There is growing evidence that mixed-species forest stands have beneficial effects on ecosystem functions (growth rates, nutrient cycling). In addition, species-rich forests tend to have higher and denser crown cover and thus might affect soil erosion. This study investigated the role of tree species richness on throughfall kinetic energy (TKE) as an important part of the soil erosion process and examined the spatial variability of TKE in mixed-species forest stands.

The research was conducted within BEF-China, a large-scale forest biodiversity experiment in subtropical China. In summer 2013, 1,800 TKE measurements were carried out during five rainfall events. TKE was measured using splash cups and related to tree height, crown base height, number of branches, leaf area index, stem ground diameter and crown area.

Our experiment showed that TKE was not influenced by tree species richness at the plot level. This is likely due to the young age of the experimental forest where a dense and high tree canopy has not yet been developed. However, TKE was influenced by neighborhood tree species richness indicating that tree species richness only affected TKE on a small spatial scale within the direct neighborhood in young forests.

TKE showed distinct spatial variability. Directly below the first branch of the tree individuals TKE was lowest (430 J/m²) while it was highest in the middle of four tree individuals (556 J/m²). Mean freefall kinetic energy (FKE) was 480 J/m². Lower TKE below the first branch than FKE can be attributed to low rain drop velocities due to short falling heights. Higher TKE in the middle of four tree individuals than FKE can be ascribed to a larger crown area on which drops can confluence resulting in an increase of drop mass. Furthermore, TKE was positively affected by the number of influencing tree individuals with a 13 % increase of TKE from one to four influencing tree individuals. However, further investigation on TKE below mature trees and dense canopies is needed to confirm our findings for later successional stages.

Highlights

- We measured throughfall kinetic energy in a young forest using splash cups.
- Throughfall kinetic energy was not affected by tree species richness at plot-level.
- Throughfall kinetic energy was affected by neighborhood tree species richness.
- Throughfall kinetic energy was lowest below the first branch of a tree individual.
- Throughfall kinetic energy was highest in the middle of four tree individuals.

1 Introduction

Soil erosion threatens ecosystem functioning by reducing soil organic carbon stocks, relocating nutrients and reducing the species diversity of plants, animals and microbes (Pimentel, 2006; Pimentel and Kounang, 1998). A common measure to overcome severe soil loss is afforestation, which increases ground cover and stabilizes soil aggregates by roots. In particular, rainfall erosivity is influenced by forest canopies and crown architecture. However, little research has been conducted on this part of the soil erosion process (Geißler et al., 2013). Rainfall erosivity is the product of kinetic energy and rainfall intensity (Renard and Freimund, 1994). In forests, throughfall kinetic energy (TKE) is a widely used measure to describe the power of raindrops to erode soil. Main determinants controlling TKE in forests are abiotic factors such as rainfall amount, drop size distribution and drop fall velocity as well as biotic factors such as tree height and density of the canopy, crown cover, leaf traits, leaf area index and branch architecture (Cao et al., 2008; Geißler et al., 2012b; Gómez et al., 2001; Hall and Calder, 1993; Herwitz, 1987; Nanko et al., 2008; Staelens et al., 2008; Tsukamoto, 1966). For instance, higher crown cover and density can alter TKE by creating slower, but larger rain drops and additionally change throughfall amount. These transformation processes can lead to higher TKE in forests than in open field (Geißler et al., 2012a; Nanko et al., 2004), yet this might be detected only in advanced succession (Geißler et al., 2010). A higher TKE than freefall kinetic energy (FKE) can be attributed to a drop size increase by confluence of drops on leafs and branches with different drop growth mechanisms under varying vegetation types (Nanko et al., 2013). However, the influence of drop velocity on an increasing TKE varies. While small drops can reach their terminal velocity at heights of only 0.3 m, large drops (>3 mm) need a detachment height of more than 13 m (Frasson and Krajewski, 2011). This effect on drop velocity means young forests (< 13 m) can alternatively have lower TKE than FKE due to slower large drops.

Most often, afforestation takes place by the establishment of easily manageable monocultures. Besides the increase of timber, fuel and pulp wood production, the reduction of atmospheric CO_2 by carbon sequestration in forest soils, or the stabilization of regional climate conditions (Dixon and Wisniewski, 1995; Houghton et al., 2012), erosion control is also a driving factor. However, TKE as main driver of the erosion process is highly species-specific. For instance, TKE below *Castanea henryi* and *Quercus serrata* was higher than that of *Schima superba*, *Elaeocarpus decipiens* and their mixture (Geißler et al., 2012b). These distinctions can be attributed to species-specific differences regarding the number of branches and the angle of the first branch. Branches transfer throughfall water laterally and release it as indirect throughfall (André et al., 2011). In addition, even within a monoculture below a single tree, TKE is spatially variable (Clements, 1971; Staelens et al., 2008; Stout and McMahon, 1961). Nanko et al. (2011) showed a distance to stem effect, where TKE increased below a single Japanese cypress *Chamaecyparis obtusa* with increasing distance to the stem. In addition, TKE under this tree species was negatively affected by canopy thickness (Nanko et al., 2008).

In recent years, monocultures have increasingly come under criticism due to their greater susceptibility to adverse environmental conditions, pathogens (Hantsch et al., 2013) or herbivores (Jactel and Brockerhoff, 2007), and their negative long-term impacts on soil fertility (Puettmann et al., 2008). There is growing evidence that mixed-species forest stands have beneficial effects on ecosystem functions and services (like growth rates, biomass production, nutrient cycling, light harvesting, plant nutrition, crown cover; Forrester et al., 2006; Forrester, 2014; Gamfeldt et al., 2013; Kelty et al., 1992; Loreau et al., 2001; Richards et al., 2010). Species-specific differences in growth and biomass allocation patterns as well as plant architecture are due to niche separation. As a result, stratified canopies with a high degree of crown overlap and, thus an increased mean vegetation cover and greater biomass density can be found in mixtures compared to monocultures (Lang et al., 2010; Lang et al., 2012; Menalled et al., 1998; Pretzsch, 2014). Concerning soil erosion, TKE in particular reacts strongly to these tree characteristics (Geißler et al., 2013; Nanko et al., 2006; Nanko et al., 2008). As a consequence, tree species richness might affect soil erosion processes. However, very few studies have reported tree species richness effects on TKE. In tropical conditions, a negative correlation between plant diversity and soil erosion was found (Shrestha et al., 2010). In contrast, a positive tree diversity effect on TKE was observed in a subtropical secondary forest along a range of successional stages (Geißler et al., 2013). This effect was attributed to an increase of tree species richness and canopy height with increasing stand age. Throughfall drops are more likely to be re-intercepted by a thicker vegetation layer of different heights resulting in decreasing TKE (Geißler et al., 2013). However, TKE might be unaffected if structural biodiversity effects are leveled out by an increase of TKE with increasing stand height (Brandt, 1988; Wiersum, 1985).

Such biodiversity effects on rainfall erosivity were mostly assessed by using a plot-level approach (Geißler et al., 2013). However, in mixed-species stands it has been shown that local neighborhood interactions strongly influence individual-tree growth and architecture (Biging and Dobbertin, 1992; Getzin et al., 2008) and hence might affect TKE. As a consequence, spatially explicit approaches are needed to analyze the effect of biodiversity on TKE in structurally complex forest stands with a heterogeneous mixture of species at the subplot-level. As biodiversity effects become more pronounced with time (Jonsson, 2006; Marquard et al., 2009; Reich et al., 2012), most studies analyzed the relationship between biodiversity and ecosystem functions in late succession stands (Geißler et al., 2013). Hence, research on possible biodiversity effects on ecosystem functions in early succession stands is limited and it is not clear at which time those effects set in.

In this study we present a new approach for assessing TKE in subtropical forests at an early successional stage. We investigated the influence of tree species richness as a proxy for biodiversity on TKE by making use of a high number of replicates at the plot-level. Local neighborhood tree species richness was examined at the subplot-level. Furthermore, investigations of several spatially-specific effects on TKE were combined to analyze these effects among a wide range of species. We developed a sampling design that can assess the influence of tree characteristics on TKE, for instance the distance to stem effect and the first branch effect. We analyzed how the spatial distribution of TKE is affected by different numbers of tree individuals. As TKE is higher than FKE in mature forests (Geißler et al., 2012a), we additionally compared TKE to FKE in a young forest. Our experiment includes a large species richness gradient from 1 to 24 species and thus provides more general, non-species-specific results.

With this approach, we tested three hypotheses:

(1) TKE increases with increasing tree species richness at plot and local neighborhood scale.

(2) TKE is spatially variable due to differences in tree characteristics.

(3) TKE is higher than FKE.

2 Materials and methods

2.1 Study site

The study site is located at Xingangshan, Jiangxi Province, PR China (29°08-011, E117°90-93) and is part of the “Biodiversity and Ecosystem Functioning (BEF-) China” project (Bruelheide et al., 2014) with a total size of 26.7 ha (Yang et al., 2013). Elevation of the study site ranges from 108 m to 250 m with a mean of 190 m a.s.l. Slopes range from 0 to 45 degrees. The climate in Xingangshan is typical of subtropical summer monsoon regions with an average annual temperature of 17.4 °C and a mean annual rainfall of 1635 mm (own measurements of three years, Figure A.1). The wet season lasts from April to August whereas winters are relatively dry.

The experimental area holds 261 plots with seven richness levels of 0, 1, 2, 4, 8, 16 and 24 tree species. Trees were planted after harvest of the previous stand in 2008. The plot size is 25.8 m x 25.8 m and 400 tree individuals were planted with a horizontal distance of 1.29 m. Species were randomly assigned to individual planting positions within the plots, and treatments (i.e. species richness) were randomly assigned to the plots. Further details on the general design, establishment and species of the BEF-China experiment are provided by Yang et al. (2013) and Bruelheide et al. (2014).

2.2 Experimental design

A total of 40 plots were used for TKE measurements during five rainfall events in 2013. 17 monocultures, 10 2-species mixtures, six 4-species mixtures, four 8-species mixtures, one 16-species mixtures and two 24-species mixtures plots have been sampled. Each plot had a core area, which consists of the central 6 x 6 tree individuals. The core area with these 36 tree individuals was divided into 9 sections (Figure 1). In eight out of the nine sections, a specific TKE measurement was located. In each plot, these positions were randomly assigned to the eight sections and trees inside a section were randomly selected for measurements. The data from the measurements on the eight positions within the plot core area were used for tree species richness and spatial analysis, and the control was used to determine FKE to be compared with TKE. This sampling design allowed for replicating each tree species richness level at least four times (except for tree species richness level 24) at the plot-level, with 8 measurement positions per subplot-level to test for spatial variability of TKE. This design allowed investigating TKE on a gradient of the number of influencing neighboring trees, on different distances to tree stems and by a variation of structures of influencing neighboring trees.

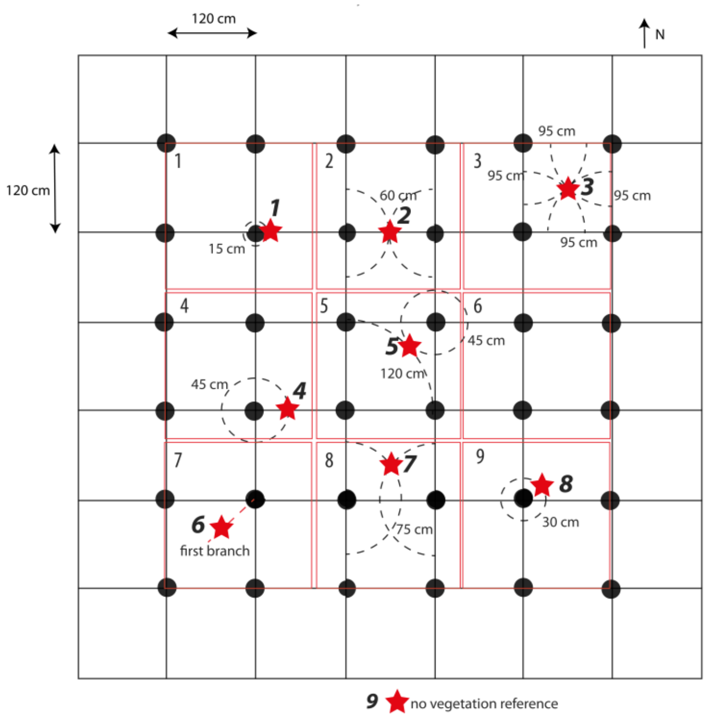

Figure 1: Sampling design with nine measurement positions. Positions (1), (4), (6) and (8) were influenced by one tree individual (1, 15 cm from the stem; 4, 45 cm from the stem; 6, first branch; 8, 30 cm from the stem), (2), (5) and (7) were influenced by two tree individuals (2, middle of two; 5, 45 x 120 cm intersection; 7, 75 x 75 cm intersection) and (3) was influenced by four tree individuals (3, middle of four). (9) was used outside the plot as control (no-vegetation reference). Black dots symbolize tree individuals and red stars throughfall kinetic energy measurement position with rainfall gauge. Dashed lines indicate radius around tree stems.

2.3 Throughfall kinetic energy measurement

Kinetic energy of rain- and throughfall was measured using Tübingen Splash Cups (diameter 4.6 cm) designed by Scholten et al. (2011). Splash cups were filled with uniform fine sand (diameter 0.125 mm). We calculated the detached sand by subtracting the weight of the dry sand inside the cup after the rainfall event from the weight of the dry sand in the full-filled splash cup before measurements. The detached sand (ds) per splash cup (sc) can be converted into kinetic energy of rainfall (KE_{rf}; for each rainfall event) using the function provided by Scholten et al. (2011) with a modified slope and standardization to 1 m^2:

$$KE_{rf}\left[\frac{J}{m^2}\right] = ds_{sc}[g] * 0.1455 * \left(\frac{10{,}000cm^2}{\pi r_{sc}{}^2}\right)$$

In general, splash cups allow a high number of replications with low costs and an easy handling in the field. Their suitability to measure TKE and FKE was shown in many studies (Geißler et al., 2010; Geißler et al., 2012b; Geißler et al., 2012a; Geißler et al., 2013). In total, 1,800 splash cups were used and measured from May to July 2013 (i.e. 40 plots x 9 measurement positions x 5 rainfall events).

2.4 Measurement of rainfall

In total, five rainfall events were registered (Table 1). Rainfall amount, intensity, duration and peak intensity were determined based on 5 minutes-interval measurements at the BEF-China climate station. For a higher spatial resolution, rainfall gauges (diameter 4.6 cm) have been placed next to each splash cup to measure the throughfall amount. In 2013, our climate station registered 33 erosive events (threshold 12.7 mm; Renard et al., 1997; Wischmeier and Smith, 1978) ranging from 13 mm to 185 mm with a total rainfall amount of 1205 mm. In 2012, 49 erosive events ranging from 13 mm to 211 mm were measured. Mean rainfall amount per event was 40 mm in 2012 and 30 mm in 2013. Hence, rainfall events were representative for this region.

Table 1: Characteristics of five rainfall events registered from May to July 2013 with information on throughfall, freefall kinetic energy (FKE) and throughfall kinetic energy (TKE). Rainfall amount, rainfall intensities and rainfall duration refer to measurement of climate station while mean rainfall amount in open field, mean throughfall amount, FKE and TKE refer to field measurements using rainfall gauges and splash cups.

Rainfall events	Rainfall amount (mm)	Rainfall duration (h)	Mean rainfall amount in open field (mm)	Mean throughfall amount (mm)	Rainfall intensity (5min peak intensity; mm/h)	Rainfall intensity of (total event; (mm/h)	FKE (J/m²)	TKE (J/m²)
Event 1	23.3	10.16	32.4	28.3	12.1	2.29	217.9	255.3
Event 2	39.3	11.50	56.4	47.9	22.8	3.42	252.2	354.7
Event 3	61.2	14.50	83.0	73.8	44.4	4.25	497.0	553.7
Event 4	6.6	2.33	4.5	5.0	25.2	2.83	85.9	78.8
Event 5	185.7	30.58	241.9	192.7	127.2	6.07	1348.4	1292.8

2.5 Measurements of tree characteristics

To specify the influence of vegetation on TKE, tree height, stem diameter at 5 cm above ground (hereafter ground diameter, GD), crown diameters, crown base height and the number of branches of individual trees were measured or counted as co-variables (Table 2). Tree height was measured with a measuring pole as the length from stem base to the apical meristem. GD was measured with a caliper to the nearest millimeter along two directions (north-south and east-west), the mean of which was used for further analyses. Crown diameters were determined with a linear tape along two directions (north-south and east-west). Based on the measured crown diameters, crown area was calculated as an ellipse form. Crown base height was measured as the distance up to bifurcation point of the lowest living crown branch. In addition, LAI was determined above every splash cup position using a Nikon F50 with a Nikon AF G DX 180° and HemiView V8 (Kundela, 2009). Photos were taken under diffuse radiation conditions. In cases when more than one tree individual had an impact on the splash cup measurement, mean values of the tree individuals involved were calculated.

Table 2: Descriptive statistics of tree characteristics used in this study.

Tree characteristics	Mean	Standard deviation	Median	Minimum	Maximum
Tree height [m]	2.72	1.56	2.35	0.30	8.31
Stem diameter [cm]	3.9	2.0	3.4	0.7	12.6
Crown area [cm^2]	24132	26462	15802	192	173590
Crown base height [m]	0.57	0.75	0.38	0.01	6.04
Number of branches	21	16	18	1	110
LAI	1.43	1.07	1.25	0.01	4.56

2.6 Data analyses

The effects of tree species richness on TKE and the spatial variability of TKE below vegetation were tested by fitting linear mixed effect models with maximum likelihood. For the tree species richness analyses, we used the complete dataset except for the control measurements outside the plot (n = 1,600 splash cups). Out of these, 189 splash cups were not covered by trees and these splash cups were omitted for the analyses of the spatial variability of TKE, independent of plot-level (n = 1,411 splash cups).

Models concerning plot-level tree species richness effects included rainfall event, measurement position, species richness and the interaction of species richness with rainfall event as fixed effects. Species richness was split into a log-linear and a factorial term to test for a linear trend and deviations from this trend. Species composition, plot and the interaction of composition with rainfall event and position were used as random factors. The interaction of species richness with rainfall event allowed us to test whether species richness affects TKE differently during different rainfall events. The interaction between species richness and position eliminates spatial differences within one plot. To test the TKE differences across the different tree species richness levels, we introduced contrasts in our models that were fitted before species richness (monocultures against mixtures, each tree species richness levels against mean of all tree species richness levels).

To test the tree species richness effect at the neighborhood level, we used rainfall event, plot, number of influencing tree individuals and a log-linear and factorial neighborhood species richness term as fixed effects. Composition of tree neighbors and its interaction with rainfall event were used as random factors. We included plot as a fixed effect to remove the variance between plots. Thus, the species richness effect of neighboring trees was tested within plots. To test for neighborhood tree species richness, only splash cups affected by two or more tree individuals on mixed-species plots were used (n = 436).

The spatial variability of TKE was analyzed using species composition and plot as random factors. Rainfall event and position as well as their interaction were used as fixed effects. Further, the eight splash cup positions per plot were assigned to three classes (further referred to as “number of influencing tree individuals”) consisting of splash cups influenced by one (position 1, 4, 6, 8), two (position 2, 5, 7) and four tree individuals (position 3). We introduced contrasts to compare one position to the mean of all other positions to detect position that yield higher or lower TKE than all other.

To investigate the mechanisms underlying spatial variability and tree species richness treatments, we constructed linear mixed effect models consisting of the tree characteristics as dependent variable and the position treatment, the number of influencing tree individuals or the tree species richness levels as explanatory variables. This allowed us to detect differences in tree characteristics among spatial positions, number of influencing tree individuals as well as tree species richness levels. In these models, composition and plot were used as random factors.

Pearson's correlation coefficient (r) was used to avoid multicollinearity (dismissing covariates with $r > 0.8$). We applied multivariate linear mixed effect model regression with six tree characteristics and rainfall event as fixed effects and plot, species composition and position as random effects. As crown area and tree height were correlated ($r > 0.8$), we fitted two multivariate models.

The distinction between FKE and TKE was investigated using Student's t-test.

TKE was transformed based on log10 before modeling. The TKE/FKE ratio as dependent variable was tested as well but did not show different results. Model residuals did not show any violation of model assumptions (normality and homogeneity of variances). All analyses were conducted using R 2.15.3 (R Core Team, 2013) together with the "asreml" package (Butler, 2009).

3 Results

Mean TKE was 503 ± 535 J/m² and mean FKE was 480 ± 485 J/m² across all rainfall events. TKE was 79 ± 53 J /m² for rainfall event 4 characterizing the lowest rainfall amounts and 1293 ± 567 J / m² for rainfall event 5 with highest rainfall amounts. Rainfall/throughfall ratio was 1.27 across all rainfall events.

3.1 TKE, FKE and rainfall characteristics

FKE did not significantly differ from TKE across all rainfall events. TKE was only significantly higher than FKE ($t(465) = 2.19$, $P < 0.05$) for trees exceeding 330 cm height. The characteristics of the five rainfall events are given in Table 1. Peak rainfall intensities (5 min) and rainfall amount increased from event 1 to 5 with the exception of rainfall event 4. All rainfall characteristics (Table 1) were highly positively interrelated ($r > 0.7$) and correlated with FKE. TKE increases with increasing rainfall amount ($r = 0.99$, $P < 0.001$) and rainfall intensity ($r = 0.95$, $P < 0.05$).

3.2 How tree species richness affects TKE

Our results showed that TKE was not affected by plot-level tree species richness ($F_{1,21} = 0.2$, $P = 0.94$) (Figure 2). No statistical difference was found between tree species richness levels for the 8-species mixtures (with highest TKE) and 16-species mixtures (with lowest TKE; $F_{1,1} = 0.68$, $P = 0.56$). Tree species richness did not affect TKE at each rainfall event ($F_{1,75}$ = 0.30, P = 0.99) or by focusing on one measurement position ($F_{1,114}$ = 1.00, P = 0.51). TKE did not differ between monocultures and species-mixtures ($F_{1,21}$ = 0.04, P = 0.55), although monocultures showed slightly lower TKE (- 6 %) than species-mixtures.

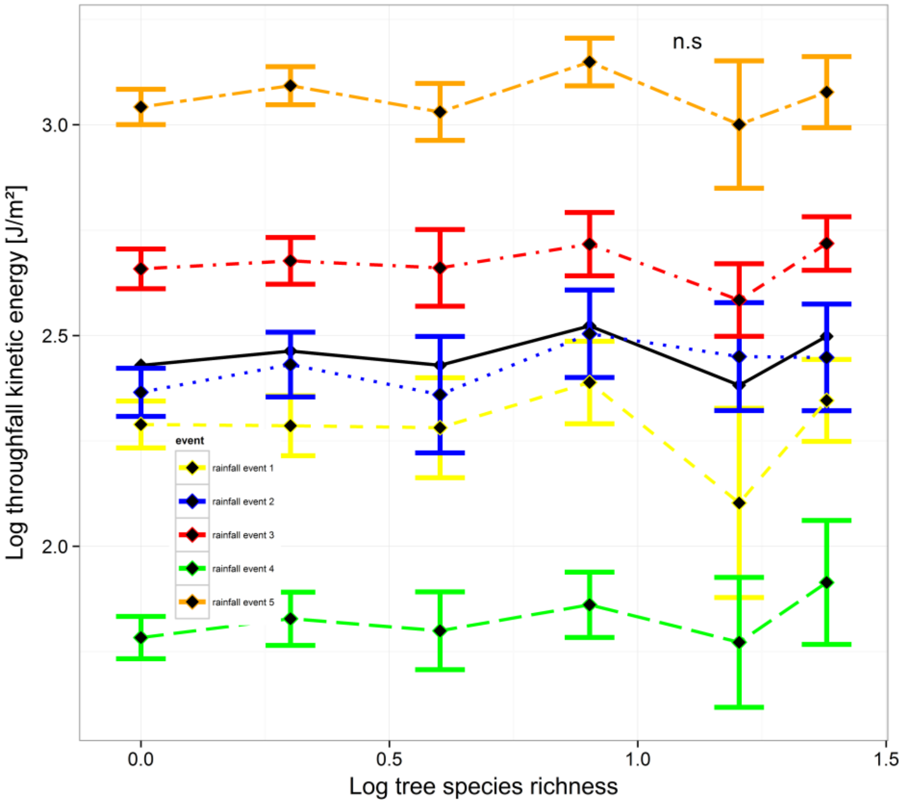

Figure 2: Throughfall kinetic energy (log10-transformed) and tree species richness (log10-transformed) during five rainfall events. Black solid line symbolizes mean of all events. Black diamonds show mean and error bars show two times standard error. Different colors show different rainfall events and dashed lines connect mean values of each tree species richness level for each rainfall event. n.s. = not significant.

On the neighborhood level, tree species richness positively affected TKE ($F_{1,45}$ = 4.30, $P < 0.05$; Figure 3). However, neighborhood species richness level 1 did not differ from neighborhood tree species richness > 1. However, after accounting for these mono-mixture differences, TKE increased with tree species richness levels 2, 3 and 4 ($F_{1,50}$ = 3.00, $P < 0.1$). Nevertheless, a distinction between position effects and neighborhood tree species richness effects within plot could not be realized as neighborhood tree species richness levels 3 and 4 were only measured at one position within one plot.

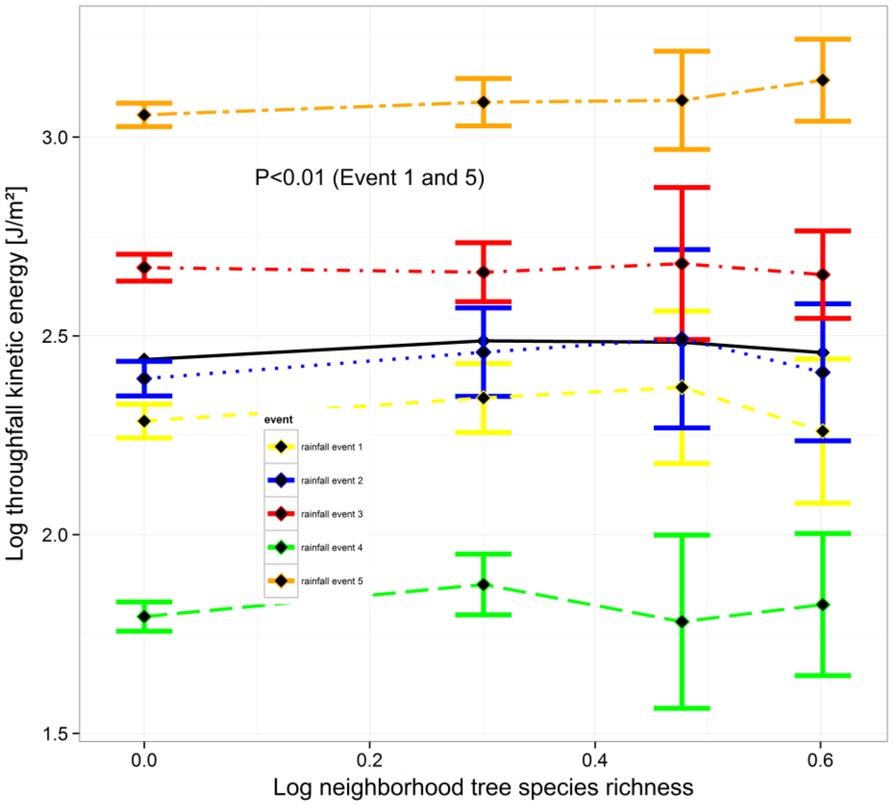

Figure 3: Throughfall kinetic energy (log10-transformed) and neighborhood tree species richness (log10-transformed) of five rainfall events. Black solid line symbolizes mean of all events. Black diamonds show mean and bars show two times standard error. Different colors show different rainfall events and dashed lines connect mean values of each tree species richness level for each rainfall event. P < 0.01 (Event 1 and 5) indicates P-value is lower than 0.01 for analyses of rainfall event 1 and 5.

3.3 How tree individuals affect spatial distribution of TKE

TKE showed spatial variability (Figure 4; $F_{1,119} = 1.90$, $P < 0.1$). Investigating each rainfall event separately, this pattern was strong at rainfall event 1 ($F_{1,237} = 2.44$, $P < 0.05$) and 5 ($F_{1,237} = 2.00$, $P < 0.05$), whereas no significant spatial variability of TKE was found at the rainfall events 2, 3 and 4. The coefficient of variation (CV) of TKE was 107 compared to 101 of FKE across all rainfall events. At rainfall event 1, 2, 3, 4 and 5, CV of TKE and FKE were 72 and 36, 85 and 73, 57 and 32, 70 and 42 as well as 46 and 20, respectively.

At each event, splash cups placed below the first branch showed on average 20 % less TKE than the mean of all other position ($F_{1,237} = 7.90$, $P < 0.01$; P6 in Figure 4). TKE was significantly higher by 13 % in the middle of four tree individuals (P3 in Figure 4) than the mean of all other positions ($F_{1,237} = 4.30$, $P < 0.05$). This effect was strongest at rainfall event 5 ($F_{1,237} = 8.00$, $P < 0.01$).

TKE was positively affected by the number of influencing tree individuals ($F_{1,119} = 6.20$, $P < 0.05$; Figure 5). TKE increased on average by 13 % from one to four influencing tree individuals. This effect was only significant at rainfall event 1 ($F_{1,237} = 3.04$, $P < 0.05$) and rainfall event 5 ($F_{1,243} = 4.80$, $P < 0.01$).

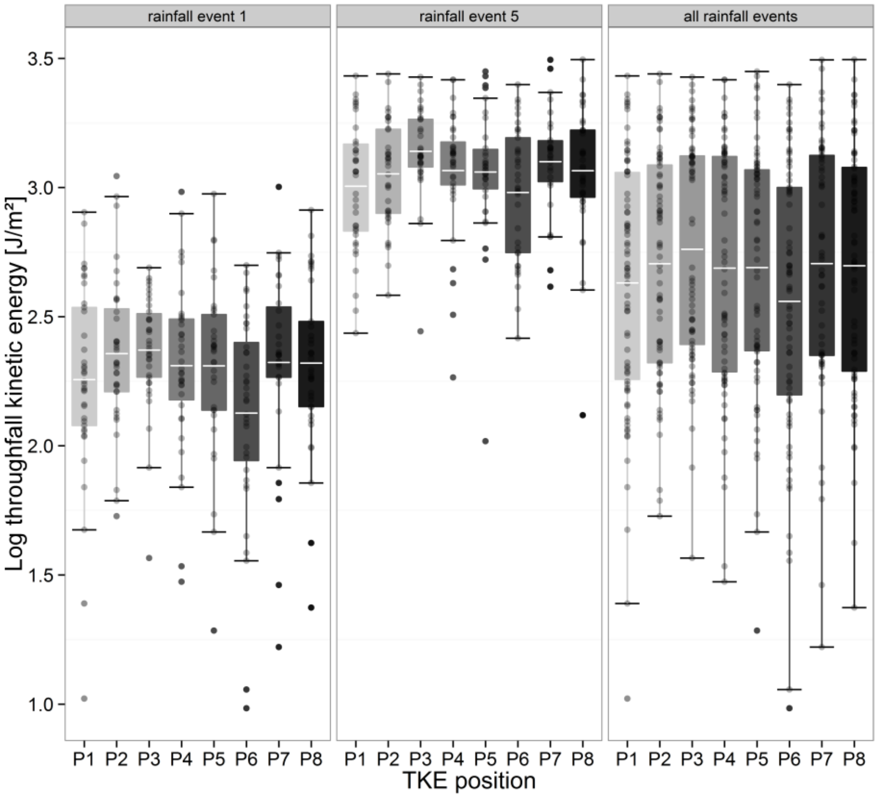

Figure 4: Throughfall kinetic energy (TKE, log10-transformed) and spatial position of TKE measurement at rainfall event 1 (left), at rainfall event 5 (middle) and in all five rainfall events (right). Different colors indicate different positions (see Figure 1) and white lines represent means.

A positive trend though non-significant ($F_{1,107} = 0.40$, $P > 0.1$) for a distance to stem effect was found when considering only splash cups influenced by one tree individual at distances of 15 cm, 30 cm, 45 cm and 60 cm away from the tree stem. Here, TKE increased with a mean of 4 % higher TKE per 15 cm distance increment towards the crown ending.

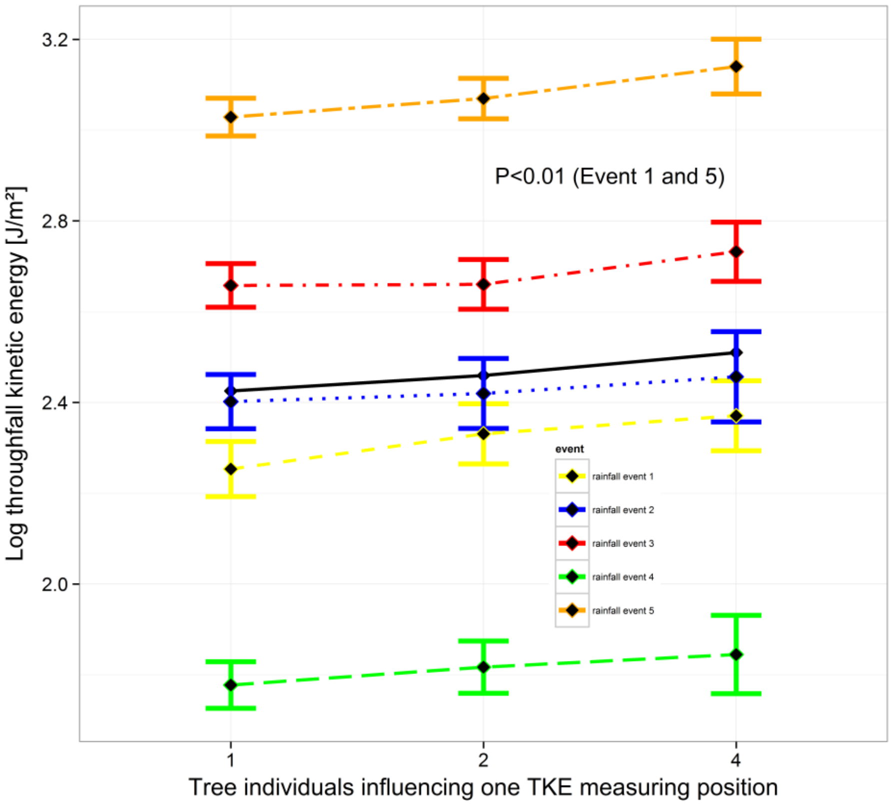

Figure 5: Throughfall kinetic energy (log10-transformed) and “number of influencing tree individuals” treatment of five rainfall events. The eight splash cup positions per plot that were covered by trees (Figure 1) were assigned to three classes: 1: splash cup is influenced directly by one tree individual; 2: the splash cup is influenced by two tree individuals; 4: the splash cup is influenced by four tree individuals. Black solid line symbolizes mean of all events. Black diamonds show mean and error bars show two times standard error. Different colors show different rainfall events and dashed lines connect mean values of each treatment level for each rainfall event. $P < 0.01$ (Event 1 and 5) indicates P-value is lower than 0.01 for analyses of rainfall event 1 and 5.

3.4 TKE and tree characteristics

TKE was positively affected by crown base height ($F_{1,101} = 7.4$, $P < 0.01$), tree height ($F_{1,93} = 13.3$, $P < 0.001$), LAI ($F_{1,215} = 8.9$, $P < 0.01$), crown area ($F_{1,208} = 7.7$, $P < 0.01$), and throughfall ($F_{1,1154} = 6.3$, $P < 0.05$), and negatively affected by the number of branches ($F_{1,211} = 23.8$, $P < 0.001$) at all rainfall events. In contrast, GD did not affect TKE ($F_{1,211} = 1.2$, $P > 0.1$).

Neither crown base height ($F_{1,19} = 0.2$, $P > 0.1$), tree height ($F_{1,18} = 0.1$, $P > 0.1$), LAI ($F_{1,19} = 0.2$, $P > 0.1$), crown area ($F_{1,18} = 0.1$, $P > 0.1$), number of branches ($F_{1,18} =0.2$, $P > 0.1$) nor throughfall ($F_{1,20} = 0.5$, $P > 0.1$) did vary among tree species richness levels. Crown area ($F_{1,238} = 2.2$, $P < 0.05$), number of branches ($F_{1,237} = 3.0$, $P < 0.01$), LAI

($F_{1,231}$ = 3.1, P < 0.01) and throughfall ($F_{1,272}$ = 2.7, P < 0.05) varied among different spatial positions while number of branches ($F_{1,242}$ = 8.2, P < 0.001), LAI ($F_{1,237}$ = 7.0, P < 0.01) and throughfall ($F_{1,277}$ = 7.8, P < 0.001) were affected by the number of influencing tree individuals.

4 Discussion

4.1 Effects of tree species richness on TKE

No significant relationship between TKE and plot-level tree species richness was found and thus hypothesis 1 has to be rejected for plot-level. However, this contradicts other studies (Geißler et al., 2013; Martin et al., 2010), which showed significant effects of biodiversity on TKE and sediment discharge. The main reason for the absence of a tree species richness effect in this study is the young age of the plantation. It is likely that tree species richness effects on TKE as an example of biodiversity effects on ecosystem functions develop over time which was shown in grassland biodiversity experiments (Jonsson, 2006; Marquard et al., 2009; Reich et al., 2012). Our study showed that five years of growing a tree plantation in subtropical forests is not enough to induce tree species richness effects on TKE and thus on soil erosion at the plot-level. In such an early successional stage, crown cover, canopy or vegetation layering have not yet fully developed. At plot-level the average crown cover and height were 0.58 (range: 0.22 – 0.95) and 2.56 m (range: 0.65 – 7.40 m), respectively showing highly heterogeneous characteristics. Furthermore, tree community structures did not vary along the plot-level tree species richness gradient and thus TKE effects based on these forest structures were not (yet) detectable. At the time of this study, tree characteristics such as crown area only were developed to approximately 10 % compared to mature trees in the study region (Bruelheide et al., 2014).

In addition, structural effects within the stand at the individual tree level (e.g. LAI, number of branches, crown base height, tree height, and throughfall) may overlay plot-level biodiversity effects (Brandt, 1988; Wiersum, 1985), in particular in young forest stands. Based on our eight TKE measurement positions, we assume that tree species richness effects on TKE did not yet occur at all positions within plot, which resulted in unaffected TKE by tree species richness at the plot-level. However, we expect that the impacts of tree species richness on TKE will change with the further development of the experimental plantation (Geißler et al., 2010), mainly due to the creation of more variable drop fall velocities through more complex forest structure in a more mature forest. For instance, mean crown base height at the study time was 0.57 m allowing drops larger than 0.5 mm to reach only about one third of the terminal velocity (Wang and Pruppacher, 1977).

Tree species richness effects on TKE were only found at the subplot-level because tree individuals interact only at the neighborhood scale in young stands. According to that, the second aspect of hypothesis 1 can be confirmed. We observed a positive tree species richness effect on TKE at the local neighborhood level (Figure 3) indicating that tree species richness in young forests is not beneficial to ecosystem functioning due to enhanced soil erosion potential. However, the neighborhood tree species richness effect on TKE is rather weak. A main reason is that the local neighborhood interactions strongly influence tree growth (in this study tree height) and tree architecture (crown area and number of branches) (Biging and Dobbertin, 1992; Getzin et al., 2008; Lang et al., 2012; Schröter et al., 2012).

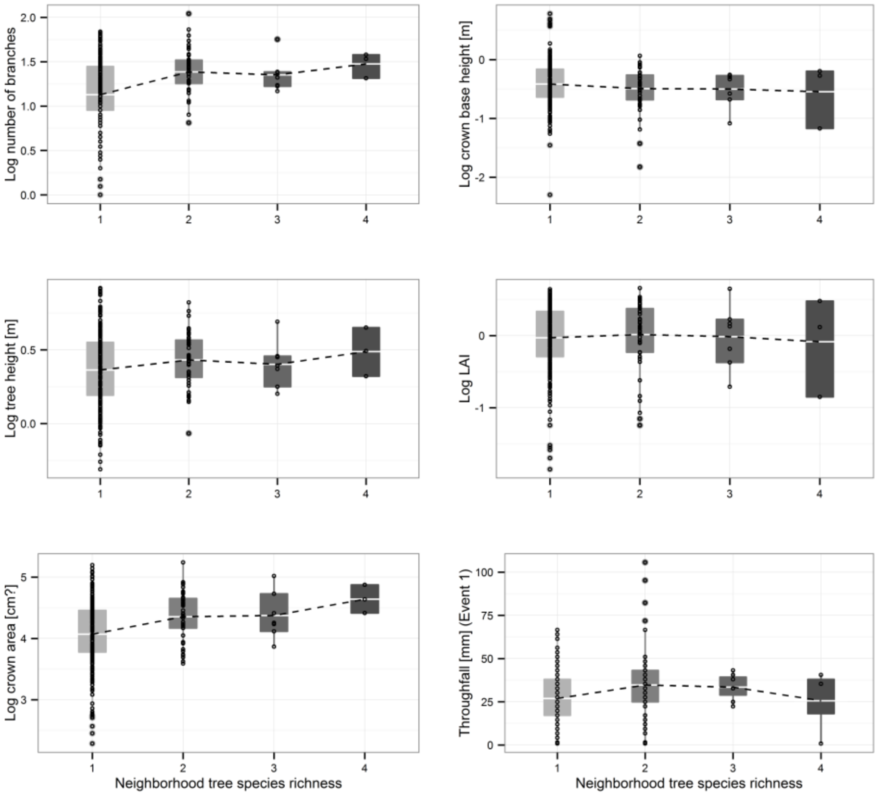

Figure 6: Tree characteristics, throughfall and neighborhood tree species richness. White lines indicate mean values and dashed black lines connect mean values of each tree species richness level.

In our study, we found significantly taller trees, larger crown areas and more branches in more diverse neighborhoods (Figure 6). These differences in vegetation structure as a response to the local neighborhood species richness might influence TKE. For instance, higher tree height in more diverse neighborhoods leads to faster drop velocities (Geißler et al., 2013; Nanko et al., 2004). However, a higher canopy may increase the space for vegetation layering, especially in diverse neighborhoods which can mask the height effect (Wiersum, 1985). It can be assumed that stand height seems to be only a very important factor in young forest plantations (Geißler et al., 2013). If the sufficient stand height for rain drops of all sizes reaching terminal velocity has been achieved, further height gain will not affect rain drop velocities. A higher crown area leads to higher TKE by creating bigger rain drops due to confluence and in addition, by creating more dripping points. However, in our study the effects of the local neighborhood species richness cannot be completely separated from the plot-level species richness effects. Higher local species richness is more likely to occur in plots with higher tree species richness. In addition, plot-level tree species richness interacts with local neighborhood species richness, if both are fitted in one model. Nevertheless, TKE of specific neighborhood richness (e.g. 2) did not vary among different plot tree species richness levels. To improve the investigation of TKE on plot- and subplot-level, changes of the sampling design have to be made. For analyzing the effects of local neighborhood diversity, only one position (P3) was used at which tree species richness levels of 3 and 4 could be realized. More investigations at the subplot-level have to be carried out to strictly isolate the local neighborhood diversity effects from the plot-level effects in the future.

4.2 Spatial variability of TKE

TKE was spatially differently distributed below forests and thus hypothesis 2 can be confirmed. In particular, the positions directly below the first branch of a tree and in the middle of four tree individuals showed distinct differences in TKE. The CV of TKE across all rainfall events was 107. Considering rainfall events 1 and 5, the CV of TKE was twofold higher below the forest than the CV of FKE outside the forest. These findings strengthen the assumption that different mechanisms in forests cause this high spatial variability of TKE. Furthermore, we observed a trend towards a positive distance to stem effect (Bochet et al., 2002; Nanko et al., 2008), which appears at all rainfall events with rainfall amounts > 20 mm. This increase of TKE towards the crown perimeter is caused by concentrated dripping like that from a peaked roof (Clements, 1971).

First branch effect

The first branch effect resulted in significantly lower rainfall erosivity (Figure 4) than all other measurement positions and the rainfall erosivity outside the forest. This can be explained by three crucial components: (i) splitting of drops into smaller ones when they hit the first branches; (ii) a shorter falling height leading to lower rain drop velocity, especially by comparing the falling heights to those of freefall outside the forest; (iii) interception of throughfall water and transfer to tree stems without release (Herwitz, 1987). The least number of branches, high LAI, and low throughfall compared to all other measurement positions in our study additionally mediate this effect (Figure A.2). High LAI values can be the reason for higher re-interception and thereafter, transpiration from leaves (Brandt, 1988). This leads to decreasing throughfall amount and thus lower TKE (Aston, 1979). In addition, few branches prevent drops from confluence resulting in lower TKE due to lighter drop mass. Despite these tree characteristics, the first branch is the last barrier for drops before reaching the soil surface (Nanko et al., 2008). Hence, low rain drop velocities occur more often resulting in low TKE.

Position of splash cup in the middle of four tree individuals

The middle position between four tree individuals showed the highest TKE. In our study, we attribute this effect to a high number of branches, a low LAI, a large crown area, and most importantly high throughfall at this position (Figure A.2). Throughfall as abiotic factor is positively related to TKE which has been found in other studies (e.g. Scholten et al., 2011). The branches of the tree individuals at this spatial position might also function differently than under the first branch. A high number of branches increase the number of dripping points on the sheltered underside. It generates drop sizes of almost the same volume as coalescence drops from leaves and hence is responsible for a higher TKE (Herwitz, 1987). Low LAI leads to high throughfall due to less interception resulting in high TKE (Geißler et al., 2013). Since FKE did not differ from TKE, the latter is particularly true for our study. In addition, larger crown areas strengthen the important role of drop confluence on leaf tips by creating larger drops (Geißler et al., 2012a). Interestingly, only horizontal tree characteristics (number of branches, crown area) and not vertical tree characteristics (such as tree height or crown base height seem to differ between the positions (Figure A.2). In such young forest stands horizontal tree characteristics seem to be of greater influence than vertical ones. This finding can support the key role of raindrop mass for rainfall erosivity over raindrop velocity being very low for small trees (Foot and Morgan, 2005; Goebes et al., 2014). Below small trees, however, small raindrop sizes can reach terminal velocity (Frasson and Krajewski, 2011).

This study is the first to show that the number of influencing tree individuals (and their different composition, see chapter 4.1) affected TKE positively. Increasing throughfall, a greater number of branches and lower LAI with increasing influencing tree individuals directly beneath the measurement position (Figure A.3) mediate this effect. However, we assume that if LAI - as prominent controlling factor of throughfall (Levia and Frost, 2006) - increases below the positions between tree individuals with further succession of a young forest due to total plant cover growth, more re-interception and transpiration will occur resulting in less TKE.

The effect power of the spatial, number of influencing tree individuals and neighborhood tree species richness effects on TKE was strongest at very low (rainfall event 1) and very high rainfall intensities (rainfall event 5) registered at study site-scale. Consistent to that, throughfall volume effects follow similar abiotic factors (Staelens et al., 2008).

Focusing on tree characteristics and their effects on TKE independent of rainfall events, spatial and tree species richness treatments, our study showed a positive effect of LAI and a negative effect of the number of branches on TKE. The positive effect of LAI on TKE occurs because LAI values were low with a mean of 1.43. In this range, LAI is closely connected to canopy gap closing and not yet to increasing vegetation layers. Thus, the throughfall/rainfall ratio is low. This is important for low- to medium-intensity rainfall (rainfall events 1, 2, 3 and 4), in which drops are small and increase on the canopy surface due to confluence resulting in higher TKE.

At first sight, this effect is contradictory to the results discussed above and recent publications (Geißler et al., 2013; Park and Cameron, 2008). However, we assume that during high-intensity rainfall (rainfall event 5, at which spatial effects were strongest), associated with larger drops and larger mean volume diameter (Marshall and Palmer, 1948), a low LAI increases the throughfall/rainfall ratio. This leads to higher and faster throughfall as if LAI would have been high and hence a higher interception would have occurred resulting in lower throughfall. As a second part of this mechanism, the confluence in the canopy would not result in increased drop sizes because the high-intensity rainfall event has already produced large drop sizes.

4.3 TKE, FKE and rain characteristics

In the young forest plantation, a weak difference between FKE and TKE was found compared to previous studies (Geißler et al., 2010) and hence we have to reject hypothesis 3. The forest was in an early successional stage during our experiment as the afforestation of the forest plantation started in 2008 allowing most trees to reach only about 10 % of their height and canopy density. If the experimental forest grows higher and denser, results might be different. Drops re-intercepted by the canopy have higher falling heights, and higher velocities, in particular of large drops, will lead to increasing kinetic energy while drop size will stay constant (Geißler et al., 2013; Nanko et al., 2006; Nanko et al., 2008). The difference between FKE and TKE for trees exceeding 330 cm in height might be used as an indication for this process. Nevertheless, throughfall below vegetation was reduced by 11-18 % compared to open field. This reduction is low compared to previous published results (Brandt, 1988; Nanko et al., 2004; Reid and Lewis, 2009; Ziegler et al., 2009) and hence is another indicator for the weak difference between FKE and TKE.

5 Conclusions

Our research focused on tree species richness effects as a measure of biodiversity on TKE at plot and subplot scale and on the spatial distribution of TKE in young forests. Measurements were carried out in 40 communities varying in species richness from 1 to 24. Results showed that plot-level tree species richness did not affect TKE, at least not in such an early successional stage. This can be attributed to a typical absence of a species richness effects on canopy characteristics in early succession stages and to the overemphasizing of side effects such as crown area or LAI on TKE. However, indications of a positive tree species richness effect on TKE were found due to taller trees, larger crown areas and more branches in more diverse local neighborhoods. This suggests that in young forests, tree species richness is affecting TKE only at local scale rather than at plot-level, which emphasizes the need to examine the erosion process at local neighborhood scale.

Focusing on the spatial distribution of TKE, we can see distinct positions that decrease TKE (below the first branch) or increase it (in the middle of four tree individuals). In addition, the numbers of influencing tree individuals, that affect one measurement position, positively affected TKE.

Independent of tree species richness and spatial treatments, number of branches, crown base height, tree height, LAI, crown area, and throughfall at each measuring position influenced the amount of TKE. Finally, TKE was only higher than FKE for spatial spots below trees exceeding 330 cm.

For future research, it will be beneficial to further improve the sampling design in order to give more weight to the positions influenced by four tree individuals. This will allow for more detailed investigations on the local neighborhood diversity effects and on the mechanisms leading to the highest TKE in the middle of four tree individuals. In addition, further investigation on TKE in this experimental forest under full-grown trees and dense canopies is needed to investigate plot-level biodiversity effects on TKE.

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Appendix:

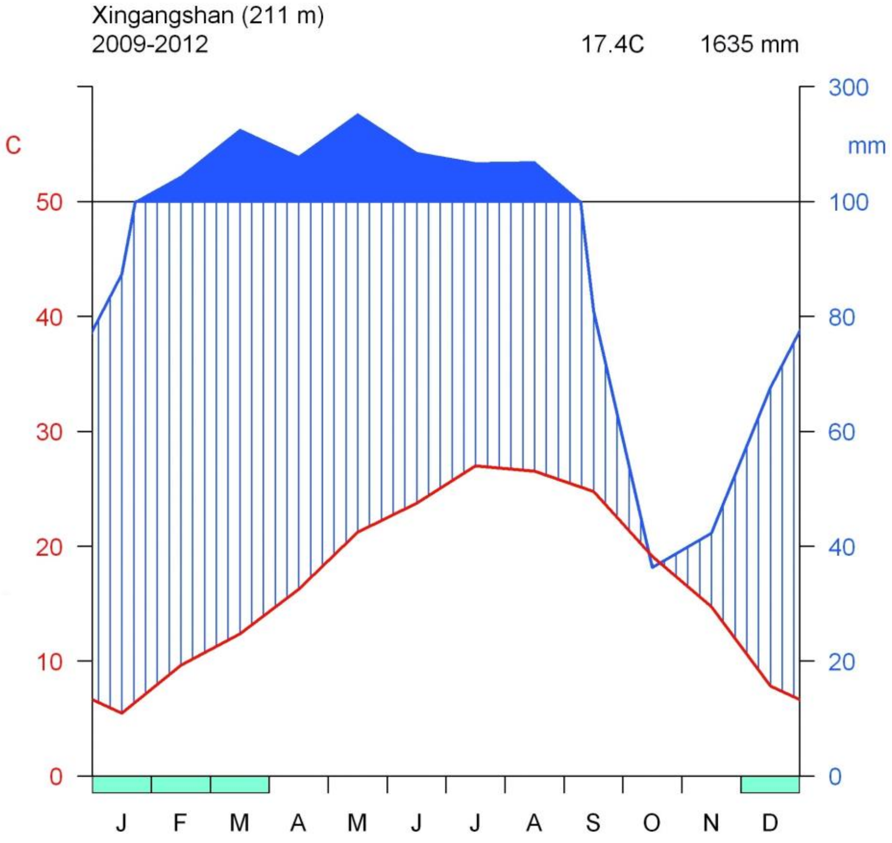

Figure A.1: Walter and Lieth climate diagram of the monthly average temperature (red solid line) and precipitation (blue solid line) at Xingangshan, PR China (2009 - 2012).

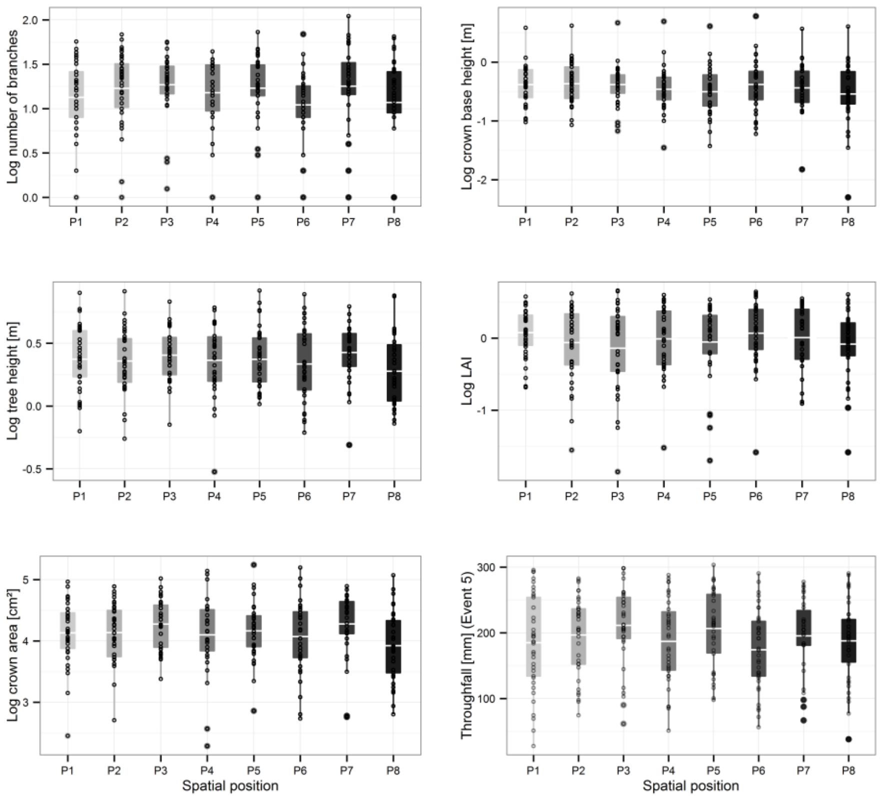

Figure A.2: Tree characteristics and throughfall of the spatial positions of the throughfall kinetic energy measurements. White lines indicate mean values.

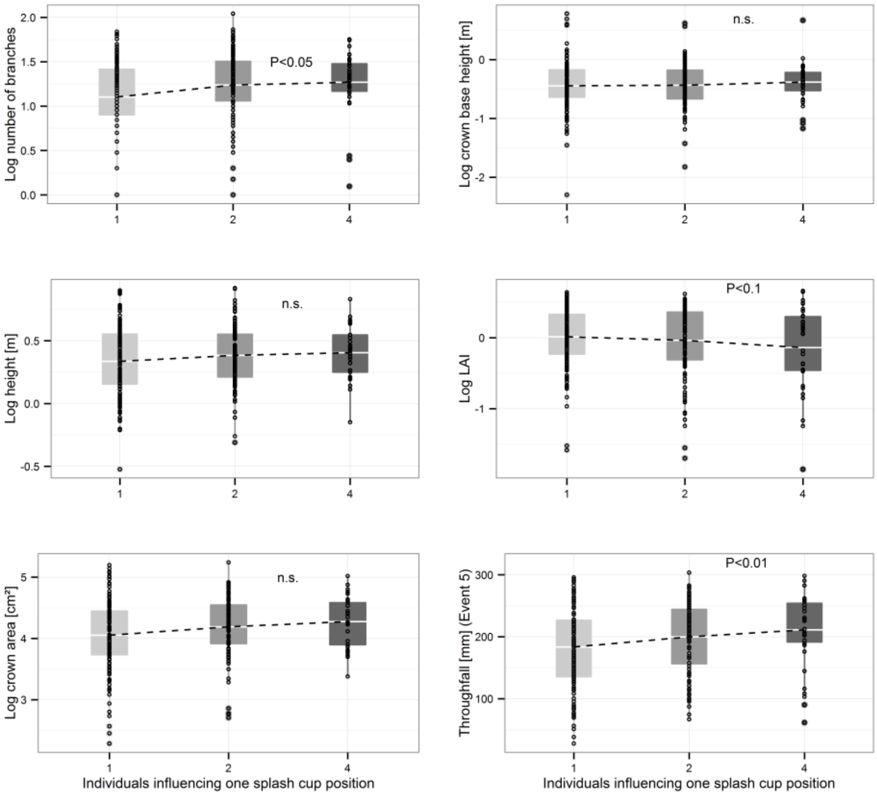

Figure A.3: Tree characteristics and throughfall of the “number of tree individuals” treatment. White lines indicate mean values and dashed black lines connect mean values.

Manuscript 4

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Species-specific effects on throughfall kinetic energy in subtropical forest plantations are related to leaf traits and tree architecture

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Abstract

Soil erosion is a key threat to many ecosystems, especially in subtropical China where high erosion rates occur. While the mechanisms that induce soil erosion on agricultural land are well understood, soil erosion processes in forests have rarely been studied. Throughfall kinetic energy (TKE) is influenced in manifold ways and often determined by the tree's leaf and architectural traits. We investigated the role of species identity in mono-specific stands on TKE by asking to what extent TKE is species-specific and which leaf and architectural traits account for variation in TKE.

We measured TKE of 11 different tree species planted in monocultures in a biodiversity-ecosystem-functioning experiment in subtropical China, using sand-filled splash cups during five natural rainfall events in summer 2013. In addition, 14 leaf and tree architectural traits were measured and linked to TKE.

Our results showed that TKE was highly species-specific. Highest TKE was found below *Choerospondias axillaris* and *Sapindus saponaria*, while *Schima superba* showed lowest TKE. These species-specific effects were mediated by leaf habit, leaf area (LA), leaf pinnation, leaf margin, stem diameter at ground level (GD), crown base height (CBH), tree height, number of branches and leaf area index (LAI) as biotic factors and throughfall as abiotic factor. Among these, leaf habit, tree height and LA showed the highest effect sizes on TKE and can be considered as major drivers of TKE. TKE was positively influenced by LA, GD, CBH, tree height, LAI, and throughfall amount while it was negatively influenced by the number of branches. TKE was lower in evergreen, simple leaved and dentate leaved than in deciduous, pinnated or entire leaved species. Our results clearly showed that soil erosion in forest plantations can be mitigated by the appropriate choice of tree species.

Introduction

Soil erosion negatively influences ecosystems widely, especially in regions with high erosion rates such as subtropical China [1]. Soil erosion brings about high economic costs due to declining agricultural productivity, reduced soil organic matter, relocation of nutrients, and off-site effects that influence human safety and food security [2–4]. Therefore, soil erosion plays an important ecological and economic role [5]. Reducing soil erosion is often achieved by afforestation [6], due to a high surface cover and stabilized soil aggregates in forests [7]. Afforestation in subtropical regions is dominated by mono-specific stands [8], primarily in order to optimize wood production in terms of quantity and quality by planting fast-growing species and to allow for a simple and standardized management [9]. Afforestations are well acknowledged for their great contribution in meeting the increasing demand for wood products and in carbon sequestration, thus having strong implications for climate change mitigation [10].

Soil erosion in forests is highly influenced by throughfall kinetic energy (TKE) [11]. TKE is a combination of the drop size distribution and drop velocity of throughfall. It is known that forests highly influence the kinetic energy of rainfall as first step towards erosion occurrence by their structure and species composition [5,12,13]. Many studies on throughfall have been conducted [14,15], but there can be different mechanisms if TKE is examined. Even though soil erosion is generally reduced in forests [7], TKE can be higher in forests than in open fields [16,17]. In particular, with a sparse understory vegetation and leaf litter cover TKE can strongly increase soil erosion under forest.

In open field sites, kinetic energy of rainfall is only affected by abiotic factors (i.e., rainfall intensity and amount; [18,19]. Below forest canopies, however, biotic factors come into play with the potential to alter throughfall and TKE considerably. As a result, large species-specific differences have been found [16,20–23]. Species-specific effects on TKE are evoked by plant traits such as leaf area index [13,21,24], leaf habit [5], tree height [12], canopy thickness [18,25], branch characteristics [26] and the first branch of a tree individual [5,25]. The mechanism of the latter is that rain is channeled by leaves and branches to drop at specific spatial points above the ground surface resulting in smaller or larger rain drops [26]. Hence, this is one of the reasons for an increased TKE, and thus soil erosion potential, on spatially confined soil patches at the micro scale. As an example of species-specific differences evoked by leaf traits, species with broad leaves and a rough cuticle produced larger drops than species with smaller and wax-coated leaves and might, thus, increase TKE [27]. For this reason, TKE under the canopy of *Schima superba* with evergreen leaves was found to be lower than under that of *Castanea henryi* and *Quercus serrata* with deciduous leaves [5]. However, most preceding studies have only dealt with at maximum four different species [5,15,16], precluding cross-species comparisons of TKE-trait relationships. One exception is the study of [27], who investigated nine different species with regard to their leaf drips but without considering plant traits. However, several studies have disregarded both the mediating effects of most biotic [28–31] and abiotic factors such as rainfall intensity and throughfall amount on TKE [18,19].

As a consequence, little is known about how and to what extent species-specific leaf and tree architectural traits mediate soil erosion processes under tree canopies. This in turn means that a broader set of species (covering a wide range of leaf and tree architectural traits) needs to be analyzed to reveal species identity effects on TKE. Therefore, it is essential to study TKE under a multitude of species that vary in leaf traits and morphology. Moreover, the investigation of several traits allows the identification of the major drivers for variations in TKE, independent of species identity. Major drivers for variations in TKE have been identified in intraspecific comparisons [32] and could be tested for their interspecific validity. However, literature reporting on TKE distribution under forest canopies remains scarce (both generally and in subtropical regions), underlining the need to further investigate TKE variation below broad-leaved tree species.

We set out to close this knowledge gap by quantifying relationships between TKE and leaf and tree architectural traits of 11 different tree species typical of subtropical broad-leaved forest ecosystems of China. Trees were grown in monocultures that were established in the context of a large-scale biodiversity-ecosystem functioning experiment (henceforth referred to as BEF-China; [33]). Specifically, we tested the following hypotheses:

H1: TKE below forest canopies is highly species-specific.

H2: Leaf and tree architectural traits mediate species-specific effects on TKE.

Materials and Methods

Study site

The BEF-China experiment is located near Xingangshan Township, Jiangxi Province (N29°08-11, E117°90-93), P.R. China. The mean annual temperature is 17.4 °C and mean annual rainfall is 1635 mm. The climate of the study area is characterized by subtropical summer monsoon with a wet season from May to July and a dry winter. After the clear-cut of a *Cunninghamia lanceolata* plantation in 2008, an experimental forest was planted on a plot-level based approach with 400 tree individuals per plot (25.8 m x 25.8 m; planted in 20 rows of 20 tree individuals each), using a planting distance of 1.29 m and including a total of 24 tree species on 261 plots to investigate biodiversity effects on ecosystem functions (see [33] and [34] for detailed explanations). This study focuses only on monoculture plots of trees that ranged in mean height from 1.10 m to 5.76 m in 2013. At the time of study, the trees were five years old. No specific permissions were required for these locations and activities. The field studies did not involve endangered or protected species.

Experimental design and data sampling

TKE was measured during five rainfall events with an event-based approach for a total of 11 species in 17 monoculture plots in 2013. Within the central part of each plot (including 6 x 6 trees), eight randomly assigned positions with distinct distances to the tree stems were used to measure TKE (1) 15 cm away from tree stem, (2) in the middle of two tree individuals, (3) in the middle of four individuals, (4) 45 cm away from tree stem, (5) at the 45 cm x 120 cm intersection between two individuals, (6) below the first branch of a tree individual, (7) at the 75 cm x 75 cm intersection between two individuals, and (8) 30 cm away from tree stem). TKE was measured using splash cups [35] and representative values of J/m² were obtained by using a modified version of the function provided by [35]. Next to each splash cup, a rainfall collector was installed to quantify throughfall with a high spatial resolution. Rainfall events were registered by the BEF-China climate stations and classified by rainfall intensity, duration and total amount (S3 Table). A total of eight leaf traits and six architectural traits were analyzed. Leaf traits included leaf area (LA), specific leaf area (SLA), leaf pinnation (simple or pinnate), leaf margin (entire or dentate), trichome cover of upper leaf surface, leaf thickness, leaf toughness and leaf habit (deciduous or evergreen). These traits were measured on individuals planted in the experiment [36,37]. Architectural traits examined at each tree individual were total height, elliptic crown area, number of branches, stem diameter at ground level (GD) and crown base height (CBH) [38]. Leaf area index (LAI) was registered at each TKE measuring point under diffuse radiation conditions, using a Nikon D100 with a Nikon AF G DX 180° and HemiView V8 (Delta-T) [39]. Table 1 gives an overview of all tree species with leaf and architectural traits influencing TKE.

Table 1. Leaf and tree architectural traits of the tree species included in the present study according to a significant influence on throughfall kinetic energy. Values represent means of the variables measured. Abbreviations: D = deciduous, E = evergreen, S = simple, P = pinnate, D = dentate, E = entire. Mean throughfall refers to the mean across all rainfall events.

Species name	Abbrev. of species name	Leaf area index (LAI)	Leaf area (LA) [mm²]	Leaf habit	Leaf pinnation	Leaf margin	Tree height [m]	Number of branches	Crown base height (CBH) [m]	Mean Through-fall [mm]
Castanea henryi Rhed. & Wils.	cah	2.77	3,128	D	S	D	4.88	19	0.82	55.7
Choerospondias axillaris (Roxb.) Burtt & Hill	cha	2.31	35,484	D	P	D	5.76	12	2.69	74.6
Cyclobalanopsis glauca (Thunb.) Oerst.	cyg	0.29	2,474	E	S	D	1.43	16	0.29	66.7
Koelreuteria bipinnata Franch.	kob	0.27	30,727	D	P	D	1.19	1	0.59	78.3
Liquidambar formosana Hance	lif	1.06	5,051	D	S	D	2.25	32	0.24	69.8
Lithocarpus glaber (Thunb.) Nakai	lig	0.77	1,956	E	S	E	1.92	27	0.30	50.4
Quercus fabri Hance	quf	0.55	1,912	D	S	D	1.66	24	0.36	65.7
Quercus serrata Murray	qus	0.41	1,972	D	S	D	1.10	23	0.16	78.2
Sapindus saponaria Linn.	sas	1.13	42,231	D	S	E	2.33	5	0.68	75.1
Triadica sebifera Small	trs	1.20	2,108	D	S	E	2.65	19	0.33	70.5
Schima superba Garder & Champion	scs	3.06	3,230	E	S	D	3.38	47	0.42	39.8

Data analyses

Species-specific variation of TKE was investigated using linear mixed-effect models fitted by restricted maximum likelihood. Rainfall event, species identity and the interaction of species identity with rainfall event were included as fixed factors. The interaction of species identity with rainfall event allowed detecting whether differences among species only played a role at certain rainfall events. Plot, measurement position within each plot, interaction of plot with rainfall event and interaction of plot with position entered the model as random effects. For testing effects within each rainfall event, rainfall event was not used as fixed factor. Contrasts were fitted before species identity to detect species which had significantly higher or lower TKE than mean of all others. Significant effects were detected using Wald Test statistics with Type I SS ANOVAs. In total, 625 data points entered the analyses (5 events x 17 plots x 8 positions – 55 failed measurements).

To specify possible effects of species identity, mediation analysis were constructed by fitting mediation trait variables before the species identity term. Mediation variables were detected as such, if significance of species identity was changed from significance to non-significance and if the mediation variable itself significantly influenced TKE. To identify the most important mediation variable, categorical levels were predicted and ranked by their magnitude of TKE differences (effect size). For continuous mediation variables, the difference in TKE was evaluated when increasing mediation variable by one standard deviation. Each model was only fitted with a single mediation variable to avoid multicollinearity among traits and overparameterization.

Additionally, a model was constructed consisting only of leaf traits (fitted first to avoid underrepresentation by larger effects), tree architectural traits and throughfall to test for their influences on TKE. In this model, plot and rainfall event were considered as random effects. Model simplification was done using step-wise backward selection with the maximum likelihood approach [40]. Hence, the final model only contained significant effects ($P<0.05$). Prior to the analyses, all covariates have been checked for collinearity (correlations were not allowed to exceed $R = \pm 0.7$). Hence, leaf toughness, leaf thickness and crown area were omitted in the final model due to multicollinearity. Predictions were used to identify the effect size according to the method described above.

If a measuring position was influenced by more than one tree individuals, mean values of leaf and tree architectural traits of surrounding individuals have been calculated.

TKE data was log-transformed to ensure normal distribution. Model residuals did not show violation of model assumptions (normality and homogeneity of variances). Analyses were conducted using R 2.14.1 [41]. Linear mixed effects models were analyzed with R package “asreml” [42] and “lme4” [43].

Results

Across all rainfall events, species and plots, TKE was highly variable ranging from 7 J/m² to 2882 J/m². Mean TKE was 494 ± 536 J/m² and differences between rainfall events were considerable. Rainfall event 4 and 5 yielded the lowest (74 ± 54 J/m²) and the highest mean TKE (1247 ± 617 J/m²), respectively. In all models, TKE was strongly positively correlated with rainfall event (F4,22= 731, P < 0.001).

TKE was species-specific but independent of a specific rainfall event. However, the species-specific effect size strongly depended on the rainfall event (Fig. 1, S1 Table). Species identity significantly affected TKE at rainfall events 1 and 2 (F1,6 = 6.3, P < 0.05 and F1,6 = 4.6, P < 0.05, respectively), whereas it was not significantly related to TKE at rainfall events 3, 4, and 5. TKE below the canopy of *Choerospondias axillaris* and *Sapindus saponaria* were significantly higher (58 %, F1,6 = 11.89, P = 0.013, and 62 %, F1,6 = 10.11, P = 0.019, respectively) and TKE below the canopy of *Schima superba* was significantly lower (42 %, F1,6 = 8.63, P = 0.026) than the mean TKE of all other species.

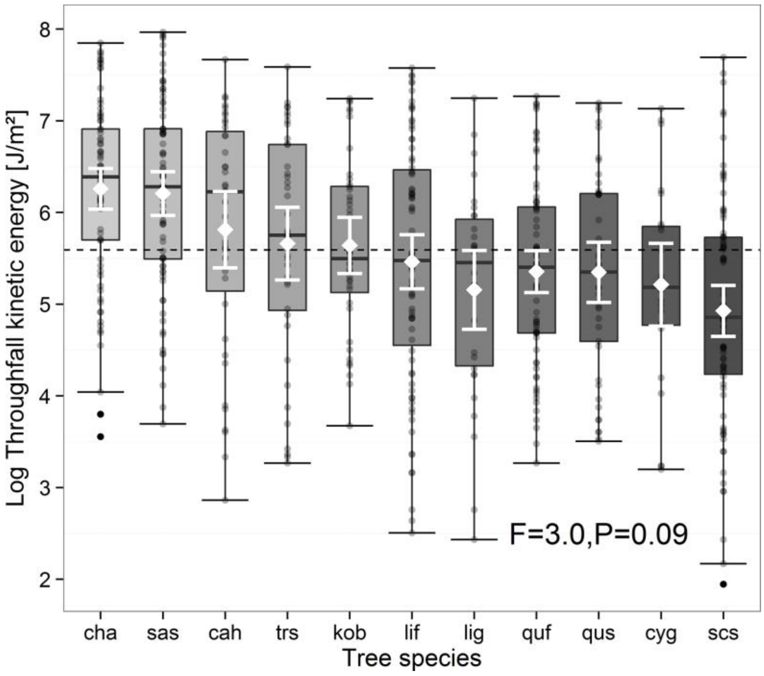

Fig. 1. Throughfall kinetic energy (TKE, log-transformed) of the 11 species analyzed. Dotted line represents the total mean TKE. Inside the boxplots white rectangles represent mean and white bars standard deviation. For abbreviations of species names see Table 1.

The effect of species identity on TKE was mediated by leaf habit, leaf pinnation, LA, tree height, LAI, SLA, throughfall, CBH, GD, crown area and number of branches (Table 2). Regarding categorical traits, the highest difference between factor levels occurred between different leaf habits (with a 92 % increase of mean TKE from evergreen to deciduous). Increase of mean TKE for pinnated leaves was 60 %. Considering vegetation continuous traits, high effect sizes were found for LA (+ 92 %), tree height (+ 33 %), LAI (- 25 %), SLA (+ 17 %), throughfall and CBH (each + 16 %). Effect sizes were small for GD and the number of branches (all < 7 %).

Table 2. Effect sizes of mediation variables (leaf and tree architectural traits). Values are predicted from mixed effect models for throughfall kinetic energy (TKE) with basic design structure (not shown, see S1 Table). For abbreviations of traits see Table 1.

		Change in TKE [J/m²] by changing mediation variable by one SD
Mediation variables	Leaf area	+ 199 **
	Leaf habit	+ 146 **
	Leaf pinnation	+ 141 *
	Height	+ 91 ***
	Leaf area index	- 65 **
	Crown base height	+ 46 **
	Throughfall amount	+ 42 ***
	Ground diameter	+ 16 .
	Number of branches	- 13 **

In general, TKE was significantly positively related to LA, CBH, height, and throughfall but negatively influenced by LAI and the number of branches. Moreover, deciduous species (+ 13 J/m²), species with pinnate (+ 32 J/m²) and entire margined (20 J/m²) leaves displayed higher TKE than evergreen species, species with simple leaves and species with dentate leave margins, respectively (Fig. 2, S2 Table).

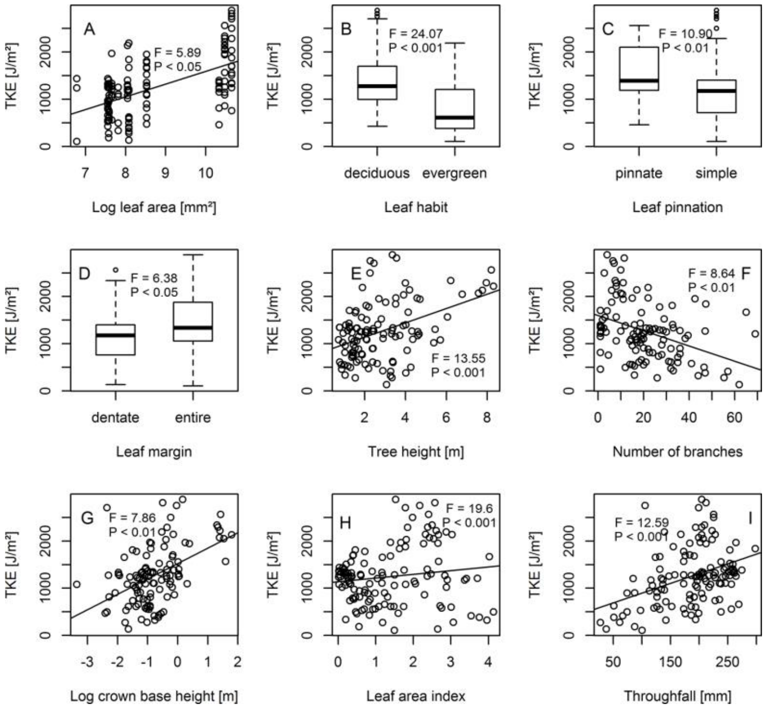

Fig. 2. Throughfall kinetic energy (TKE) versus leaf traits (A-E), tree architectural traits (F-H) and abiotic covariates (I). Black solid lines indicate linear trend.

Discussion

This study compared TKE of 11 tree species in monocultures typical of subtropical forest ecosystems in China. Further, it complemented former analyses by deepening our understanding of leaf traits and tree architecture effects on TKE [5,27]. Mediation analyses linked leaf and tree architectural traits to varying TKE induced by different tree species, and major biotic drivers of TKE variability were detected by comparing effect sizes. Finally, this study aimed to support the selection of appropriate tree species for tree plantation in order to minimize TKE and thus to counteract soil erosion in subtropical regions resulting from high monsoon precipitation (particularly in areas with steep terrain).

Species-specific TKE variation (H1)

Our first hypothesis was confirmed by the significant influence of species identity on TKE. However, only three out of 11 species showed distinct differences in TKE compared to overall means. Among these, two species positively (*Choerospondias axillaris* and *Sapindus saponaria*) and one species negatively (*Schima superba*) affected TKE (Fig. 1). Species-specific differences of throughfall amount or interception have been frequently reported [18,44]. Moreover, drop size distribution as an important driver of TKE has been found to be species-specific [27]. Nevertheless, preceding studies found no significant difference in TKE among certain species [5,16,21,44,45], which is in line with our findings. Furthermore, the highly significant interaction of species identity with rainfall event emphasizes the importance of abiotic characteristics in TKE distribution [18]. An influence of species identity on TKE was found at low-peak intensity rainfall events, whereas TKE at high rainfall intensities was not species-specific. An exception occurred at rainfall event 4 where species identity did not affect TKE although rainfall intensity was low. Higher intensity rainfall usually results in considerable canopy vibration, through which the drop sizes are reduced [20]. Therefore, the variation of TKE at high intensity rainfall could be much less than that at low intensity rainfall leading to no species-specific differences. However, this effect often is superimposed by an increase of total throughfall amount with higher rainfall intensities.

The species-specific effects of canopies of *Choerospondias axillaris*, *Sapindus saponaria* and *Schima superba* have strong implication for managing TKE. Planting *Schima superba*, which negatively affected TKE, has the potential to decrease soil erosion in early successional stages. *Schima superba* is also well-known for high values of canopy interception during rainfall [46]. This could be partially attributed to the high LAI and re-interception of rainfall by lower canopy layers [25,30]. Low TKE below *Schima superba* was also reported by [5]. These findings are as much more relevant as *Schima superba* represents one of the dominant tree species in the regional species pool [47,48]. *Choerospondias axillaris* increased TKE which is consistent with high runoff volumes found for this species in comparison to peanut crops [49]. However, despite a TKE increase, higher soil loss with *Choerospondias axillaris* can be counteracted by an intact litter cover [29].

Leaf and tree architectural traits mediate species-specific variation (H2)

First, the strong impact of rainfall event on TKE suggests that the TKE variation is predetermined by the characteristics of rainfall events, such as duration, total rain amount, wind speed and rainfall intensity [18]. However, our study could not confirm a significant effect of rainfall duration and wind speed on TKE. Within a specific rainfall event, TKE differed among tree species as species responded differently to different rainfall intensities [50]. Independent of a specific rainfall event, the species identity effects on TKE were mediated by leaf and tree architectural traits. LA, leaf habit, leaf pinnation, GD, CBH, tree height, number of branches and LAI as biotic factors were found to be responsible in mediating species-specific TKE (Table 2). Moreover, the significant effect of throughfall measured at each splash cup position on TKE showed the influence of biotic and abiotic factors on TKE [18].

In our study, species-specific changes of TKE were induced most by leaf area. A higher leaf area can increase the gathering of rain water and thus may cause larger drops resulting in higher TKE [13,20,26]. In contrast, many studies have reported on the positive influence of leaf area on interception [44], which leads to decreasing throughfall amount and decreasing TKE. Therefore, in our study variation of interception might only play a minor role in explaining species-specific differences in TKE, since all rain events lasted long enough to compensate the effect of canopy storage at the beginning of each event. However, the high effect size of LA in our study might be an overestimation, since leaf areas of *Choerospondias axillaris* and *Sapindus saponaria*, both with largest TKE, were almost twice of the standard deviation above the mean. This is due to the fact that for measurements of leaf area, the leaflets of pinnate leaves are traditionally added up to a total value per pinnate leaf [51]. Furthermore, water might gather at the branch, where each leaflet splits, which in turn may result in increased drop size and thus TKE.

TKE varied second-most between deciduous and evergreen species where deciduous species showed higher TKE. Similarly, [5] found that *Castanea henryi* and *Quercus serrata* as examples of deciduous species yielded higher TKE than the evergreen species *Schima superba*. Leaf habit represents a dominant segregation for many leaf traits and has been found to influence core functional and physiological processes specifically in the study species [47] as well as globally [52–54]. Deciduous species tend to have leaves with higher SLA [55], which we found to positively affect TKE. In addition, evergreen species tended to have a larger crown length ratio (ratio of crown length to the total tree height). Two mechanisms might elucidate the great variation between deciduous and evergreen species: (i) A lower tree height decreases falling height of raindrops and thus, results in lower TKE; (ii) a larger crown area with lower tree height (higher crown length ration) may increase LAI which results in higher interception, leading to decreasing throughfall. Moreover, leaf pinnation (pinnate or simple) can alter drop sizes. On the basis of higher margin circumference in relation to total leaf area [56], pinnate leaves create more dripping points. In addition, pinnate leaves showed the highest leaf area (see Table 1) with the exception of *Sapindus saponaria*. Corresponding to the above, a higher leaf area increases TKE. However, different leaf margins contributed only marginally to species-specific changes in TKE. As demonstrated in former studies [13,28], tree height was the most important tree architectural parameter to describe species-specific differences in TKE. Increasing tree height can contribute to higher TKE by several processes: (i) higher drop velocity due to higher falling heights [57], (ii) larger crown width [38] that increases drop size through increased confluence, and (iii) larger crown width is associated with higher LAI, which creates more dripping points [26,28]. The species-specific differences in TKE were mediated by LAI which negatively affected TKE. It is known that high canopy thickness increases drop splitting by dripping on branches and leaves [20,25], which in turn may decrease raindrop sizes. Moreover, higher canopy thickness in young forest stands might decrease space between vegetation and surface resulting in lower rain drop velocities [5]. Additionally, with denser and thicker crown cover water storage in the canopy increases, but this effect can be neglected with regard to rainfall durations longer than a day.

CBH (with half of the effect size of tree height) contributed to species variances as indirect factor, as it is usually related to tree height. CBH may constitute “the last barrier” in releasing throughfall drops, determines the falling height and thus drop velocity. Yet, our data showed that CBH contributed to interspecific TKE much less as compared to tree height. One reason might be that the same CBH might occur at trees along a large range of tree height.

GD positively and branch number negatively mediated species identity of TKE, but only to a smaller extent (< 7 % difference). This suggests that GD mediated species-specific changes as an indirect effect of tree growth characteristics (tree height and LAI). Furthermore, the number of branches might affect TKE through an indirect effect via LAI. Branches gather throughfall and release it at any random position or transfer it directly to the stem, which decreases throughfall and increases stemflow [26]. However, our results indicated that this was a weak effect and water might both, be distributed along the branches and transferred to the stem in equal proportions [26]. Moreover, the greater effect size of LAI demonstrated that leaves are much more important than the branches as regards the impacts of species-specific TKE.

Besides the significant effect of plant traits on TKE, throughfall amount was also highly correlated with TKE, but showed smaller effect sizes than the findings from other studies [19]. In most studies, throughfall amount was found to be the major driver of spatial variability of TKE [16,35]. However, our species comparison revealed that shifts in drop formation and drop velocity within a specific rainfall event might have a higher impact on TKE than the total amounts of rainfall. Thus higher throughfall amounts do not necessarily lead to higher TKE at rainfall event level.

Conclusion

This study aimed to contribute to a better understanding of mechanisms underlying the relationships between TKE and leaf and tree architectural traits, taking 11 tree species of subtropical forests in China as example. In conclusion, the optimal trait combination a tree should have to minimize TKE would be a low leaf area index and leaf area, simple pinnated leaves, dentated leaf margins, low tree height, high number of branches and a low crown base height. Furthermore, evergreen species showed lower TKE than deciduous ones. However, traits such as tree height, stem diameter and LAI will change with growing tree individuals while other traits such as SLA and all binary leaf morphological traits are believed not to change drastically during tree growth. These implications need to be considered when transferring our results to other systems with a fully developed crown cover.

Our results showed that TKE distribution among different species is much more complex than throughfall distribution solely. TKE sensitively responded to the amount of throughfall, but also to the transformation of throughfall amount (in terms of drop size and drop velocity) by leaf and tree architectural traits (Fig. 2). Thus, this study helps to understand the interaction between these vegetation characteristics, species identity and TKE as a basis for erosion modeling and the mitigation of soil erosion by means of an optimized selection of appropriate tree species in the context of afforestation programs.

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Supporting Information

S1 Table. Results from the basic mixed-effects model for throughfall kinetic energy response.

		Df	DenDF	F	P
Fixed Effects	Intercept	1	6.2	17760	$1.2*10^{-11}$ ***
	Event	4	22.4	731.4	$2.2*10^{-16}$ ***
	Position	7	101.3	1.3	0.2510
	Species identity	11	6.3	3.0	0.0936 .
	Species identity: event	44	21.5	3.0	0.0037 **
		Gamma	Component	Std.error	
Random effects	Plot	0.4120	0.0797	0.0588	
	Plot:Event	0.0013	0.0003	0.0081	
	Plot:Position	0.6095	0.1180	0.0224	

S2 Table. Effect sizes of leaf and tree architectural traits. Values are predicted from the full multivariate mixed effect models for throughfall kinetic energy (TKE) with basic design structure (not shown, see S1 Table). For abbreviations of traits see Table 1.

		Change in TKE [J/m²] by changing mediation variable by one SD
Full model	Height	+ 114 ***
	LA	+ 34 *
	Leaf pinnation	+ 31 **
	LAI	- 24 ***
	Throughfall amount	+ 21 ***
	Leaf margin	+ 20 *
	Leaf habit	+ 13 ***
	Branch number	- 12 **
	CBH	+ 3 **

S3 Table. Characteristics of the five rainfall events.

Rainfall events	Rainfall amount (mm)	Rainfall 5-min peak intensity (mm/h)	Rainfall intensity of total event (mm/h)
Event 1	23.3	12.1	2.29
Event 2	39.3	22.8	3.42
Event 3	61.2	44.4	4.25
Event 4	6.6	25.2	2.83
Event 5	185.7	127.2	6.07

Manuscript 5

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Interrill erosion under organic and conventional farming and different tillage systems at a farming trial in Switzerland

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Abstract

Conventional farming strategies have come into criticism due to diminishing topsoil depths, degradation of soil structures, compaction, loss of soil organic matter (SOM) and nutrient depletion. The use of alternative low input strategies like organic farming is therefore of rising interest. Organic farming has the potential to reduce soil erosion due to important impacts on soil characteristics and soil surface cover. At the same time, tillage operations appear to be necessary in organic farming for weed control. Tillage practices generally lead to increasing soil erosion and thus reduced-tillage appears to be an improvement for soil conservation. Different studies carried out on the influence of organic farming and reduced-tillage on soil erosion vary greatly in their methodologies and results and are mainly based on modelling approaches. Therefore, a field experiment was performed with a portable rainfall simulator and micro-scale ROPs on a farming system trial to investigate the influence of organic farming in combination with different tillage systems on soil erosion.

Our results show that organic farming significantly decreases soil erosion rates compared to conventional farming systems. Sediment discharge is 59 % higher ($p=0.018$) on conventional treatments (31.8 g m^{-2} h^{-1}) than on organic treatments (20.0 g m^{-2} h^{-1}). This effect is outperformed by the applied tillage system, as no-tillage has the most significant decreasing influence on erosion rates. Under both farming systems, ploughed treatments show higher sediment discharge (conventional farming: 104 %, organic farming: 133 %, $p=0.004$) than treatments with reduced or no-tillage. Reduced-tillage practices are a major soil-protecting improvement to organic farming and an efficient contribution to soil conservation. Soil surface cover is the main factor in the prevention of soil erosion, but mostly controlled by tillage than the farming system. With the help of a well-replicated micro-scale runoff plot design and a portable rainfall simulator we were able to gather reliable soil erosion data in situ in short term and without external parameterization. Our field assessment shows that organic farming in combination with reduced-tillage practices protect agricultural land best against soil erosion.

1. Introduction

Soil erosion is a major environmental problem of our time with severe impacts on terrestrial and fluvial ecosystems (Toy et al., 2002). Verheijen et al. (2009) indicate that 3 to 40 t ha^{-1} of soil material is eroded in Europe every year, whereas mean soil formation rates do not exceed 0.3 to 1.4 t ha^{-1} a^{-1}. It is well established that agricultural practices have a high influence on soil erosion (Pimentel et al., 1995) and especially the intensification of cultivation over the past 50 years led to increasing soil losses (Matson, 1997). Pimentel and Kounang (1998) illustrated that the world agricultural areas account for approximately three-quarter of the global soil erosion. In this context, conventional farming strategies have come into criticism (Pimentel et al., 1995; Gomiero, 2013). Those methods often lead to diminishing topsoil depths, degradation of soil structures, compaction, loss of soil organic matter (SOM) and nutrient depletion (Morgan, 2005). Thus, crop yields can be reduced and fields rendered unproductive in the long term (Lal et al., 2000). The use of alternative low input strategies like organic farming instead of conventional high-input farming systems is therefore of rising interest (Gomiero et al., 2011b).

Organic farming relies on crop rotation, absence of agrochemicals, green manure and weed control without herbicides (Gomiero et al., 2011a). As a consequence, this practice leads to higher soil fertility as well as biodiversity, as long-term field trials showed (Reganold et al., 2001; Mäder et al., 2002). Further, organic farming has the potential to reduce soil erosion due to important impacts on soil characteristics and soil surface cover (Lockeretz et al., 1981; Reganold et al., 1987). Soil aggregate stability is typically positively affected by organic farming practices (Erhart and Hartl, 2009). Several studies showed that SOM contents in the top soil layer are usually higher on arable land under organic farming than under conventional land use (Pimentel et al., 2005; Marriott and Wander, 2006). SOM provides organic binding agents, which enhance and stabilize soil aggregation (Morgan, 2005). Those binding agents (e.g. polysaccharides) are applied by microorganisms breaking down organic matter (Watson et al., 2002). Further, grass roots and fungal hyphae enmesh and stabilize soil aggregates (Erhart and Hartl, 2009) and a well-developed and undisturbed root system increases the internal stabilization of soils (Pohl et al., 2009). Additionally, soil aggregates can be braced by arbuscular mycorrhizal fungi (AMF) through glomalin (Nichols and Wright, 2004). Moreover, soil surfaces in organic farming are better protected against particle detachment by rain splash and particle transportation by overland flow due to a continuous soil surface cover (Blanco-Canqui and Lal, 2008). Although, there are evidences for a positive effect of organic farming on soil erosion (Erhart and Hartl, 2009), studies carried out on this topic come to varying conclusions (c.f. Arnhold et al., 2014). This is mostly due to different site conditions, varying crops and rotation systems, but also to affiliated practices like the tillage system (Gomiero et al., 2011a).

Erhart and Hartl (2009) indicated that frequent soil disturbance by mechanical cultivation increase soil erosion. At once, tillage operations appear to be necessary in organic farming for weed control as herbicides are not used (Erhart and Hartl, 2009). Goh (2011) assumed that conventional but no-tillage farming systems do not show lower performances regarding soil carbon sequestration, than regularly tilled organic systems. Green et al. (2005) found that the potential loss of micro-aggregates through sediment transport is higher on tilled organic treatments than on no-tillage treatments. Teasdale et al. (2007) showed that reduced-tillage in organic systems could improve soil quality compared to conventional systems, but stated that only few comparative studies have been conducted on this topic.

At last, different studies carried out on the influence of organic farming on soil erosion vary greatly in their methodologies (Arnhold et al., 2014). Most of them used empirical (e.g. USLE) or process-based (e.g. WEPP) models to calculate soil erosion rates (e.g. Lockeretz et al., 1981; Reganold et al., 1987; Auerswald et al., 2003; Pacini et al., 2003; Arnhold et al., 2014). Some studies applied combined field and laboratory approaches to study soil erodibility (Fleming et al., 1997; Siegrist et al., 1998; Kuhn et al., 2012). Eltun et al. (2002) studied nutrients in runoff of a farm drainage system. Reganold and his team (Reganold, 1988) recorded rill erosion with the Alutin rill erosion method. To our knowledge, direct field assessment of sediment discharge rates is rarely done in this context, especially at single sites with equal conditions for both, organic and conventional treatments (Auerswald et al., 2003). At the same time, a greater number of field experimental approaches should be conducted to compare the sustainability of different farming systems (Gomiero, 2013). Our study is the first to compare the influences of organic and conventional farming in combination with different tillage systems on soil erosion rates in situ with a well-replicated experimental field design. Therefore, we are using a set of micro-scale runoff plots and a portable rainfall simulator, which have recently shown their reliability in rough subtropical conditions.

We assume that the overall benefits of organic farming leads to lower soil erosion rates compared to conventional farming, but the applied tillage system has a more important impact on sediment discharge. Further, we believe that reduced-tillage can substantially improve organic farming regarding soil erosion and thus, is a major improvement for soil conservation.

Thus, we present the following hypotheses:

1. Soil erosion is lower under organic than conventional farming practices, but this effect is outperformed by reduced and no-tillage practices
2. A reduced-tillage system decreases soil erosion in organic farming due to higher soil surface cover and aggregate stability

2. Material and methods

This study took place on an experimental farming trial in Rümlang, Switzerland (Farming System and Tillage experiment Agroscope, 47°26’20’’N, 8°31’40”E). A randomized block design was setup with four replications. The experimental factors are the organic (org) and conventional (conv) farming system, as well as a tillage treatment with ploughed plots (till) and plots under reduced or no tillage (red-till, no-till). In the organic treatment weed control is performed mechanically (hoeing, raking), whereas herbicides are used in the conventional treatment. In the tillage treatment, ploughing is carried out to a depth of 20 cm with a rotary harrow. Reduced tillage is done by a disk harrow and a rotary harrow (depth 5 cm). Further, mineral fertilizer is used in the conventional treatment and slurry is used in the organic treatment. Four cover crop treatments (brassica, legume, mixture and control) have been established. The soil type is loam (clay 24 %, silt 34 %, sand 42 %). The climate is humid and continental (Köppen-Geiger classification: Dfb) with a mean temperature of 9.7° C and an annual precipitation of 1000 mm.

Measurements took place one week after harvesting (field beans) in August 2014. For the erosion experiment the control feature with farming (conv, org) and tillage (till, red-till, no-till) treatments was used. A portable single nozzle rainfall simulator and a light weight tent (Iserloh et al., 2013; Seitz et al., 2015b) were used with micro-scale runoff plots (0.4 x 0.4 m, ROP, cf. Seitz et al., 2015a). Rainfall was applied for 30 min at every plot with an intensity of 60 mm h^{-1}. Four treatments (conv/till, conv/no-till, org/till, org/red-till) were included with 8 replications each for a total of 32 ROPs and all ROPs were distributed randomly within the different treatments. Runoff and sediment discharge were collected in 2 l bottles and filtrated on fibre-glass filters. The sediment was oven-dried (40° C) before weighing. Slope was measured at every ROP with a clinometer, soil surface cover was determined photogrammetrically and soil moisture content was measured with the thermo-gravimetric method.

Linear mixed effects modelling with restricted maximum likelihood was used with the R-package “asreml” (www.vsni.co.uk; c.f. Gilmour et al., 1995). Surface cover, runoff volume, soil moisture, slope, management treatment and tillage treatment were fitted as fixed effects and rainfall event, block and plot as random effect. The data was log-transformed and the residuals did not show irregularities in normality or homogeneity of variances. The maximum likelihood approach was used to obtain model simplification by step-wise backward selection. Tukey's honest significance test was used to investigate differences in the means of the treatments. All analysis were performed with R 3.1.2 (R Development Core Team, 2014).

3. Results

3.1 The influence of organic farming on soil erosion and the effect of tillage systems

Sediment discharge was affected by the farming system (p=0.018) and 59 % higher on conventional treatments (31.8 g m^{-2} h^{-1}) than on organic treatments (20.0 g m^{-2} h^{-1}) (Table 1 and Figure 1). Likewise, tillage systems affected sediment discharge (p=0.005) and ploughed treatments showed 164 % higher soil losses (37.5 g m^{-2} h^{-1}) than treatments with reduced- or no-tillage (14.2 g m^{-2} h^{-1}) (Table 1 and Figure 1). An interaction between management and tillage treatment could not be found.

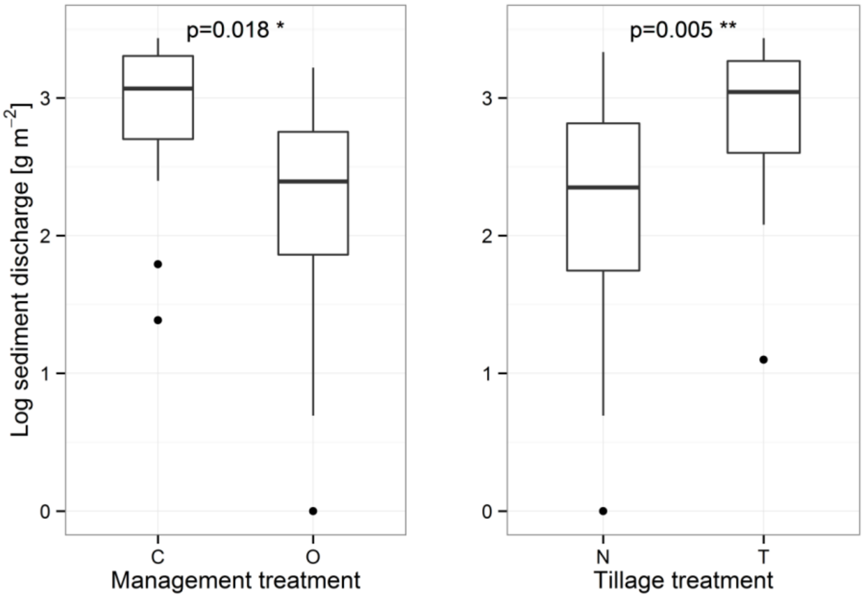

Figure 1: Log sediment discharge in management treatments (Conventional and Organic farming) and in tillage treatments (Reduced- and No-Tillage and Tillage) at the FAST trial, Switzerland (n=32).

Table 1: Results of the basic linear mixed effects model for sediment discharge (* : p<0.001, ** : p<0.01, * : p<0.05, n.s. : not significant, n=32).**

		denDF	F	Pr	
Fixed effects	Surface cover	19.0	47.8	1.359e-6	***
	Runoff volume	19.0	37.0	7.515e-6	***
	Soil moisture	19.1	2.2	0.157	n.s.
	Slope	19.0	4.3	0.053	n.s.
	Management treatment	19.4	6.4	0.018	*
	Tillage treatment	19.2	9.6	0.005	**
		DF	Variance		
Random effect	rainfall event	25	1.09		
	Block				
	Plot				

Runoff volume showed an important influence on sediment discharge (p<0.001), but was not affected by the management or tillage treatment. Furthermore, the soil surface cover affected soil loss considerably (p<0.001, Table 1). Initial soil moisture and slope angle did not show any influence.

3.2 The influence of reduced tillage on soil erosion under organic farming

The combination of organic farming and reduced-tillage showed the lowest sediment discharge (7.5 g m^{-2} h^{-1}), whereas tilled conventional plots showed the highest rates (42.6 g m^{-2} h^{-1}). Both were significantly different form each other (0.01) (Figure 2). Conventional but no-tillage treatments performed better (20.8 g m^{-2} h^{-1}) and showed nearly the same erosion rates as tilled organic plots (21.5 g m^{-2} h^{-1}). They did not show different means (0.99), but were different from both organic farming with reduced tillage (0.05) and conventional tilled plots (0.38).

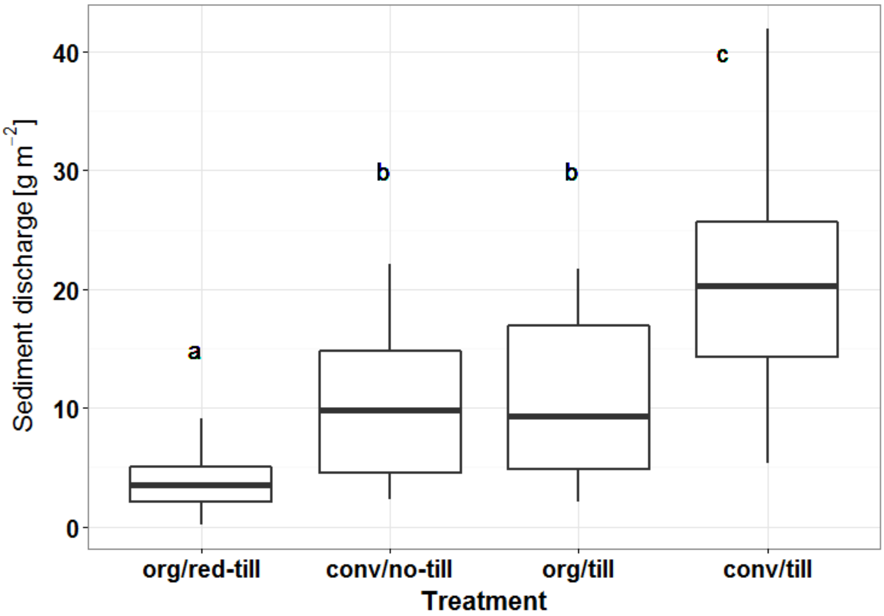

Figure 2: Sediment discharge on combined treatments (conventional/tillage, conventional/no-tillage, organic/tillage, organic/reduced-tillage, n=32).

Tillage affected the surface cover (p<0.001) whereas the management treatment did not show significance. Ploughed treatments negatively affected soil surface cover and thus led to higher sediment discharges (Figure 3).

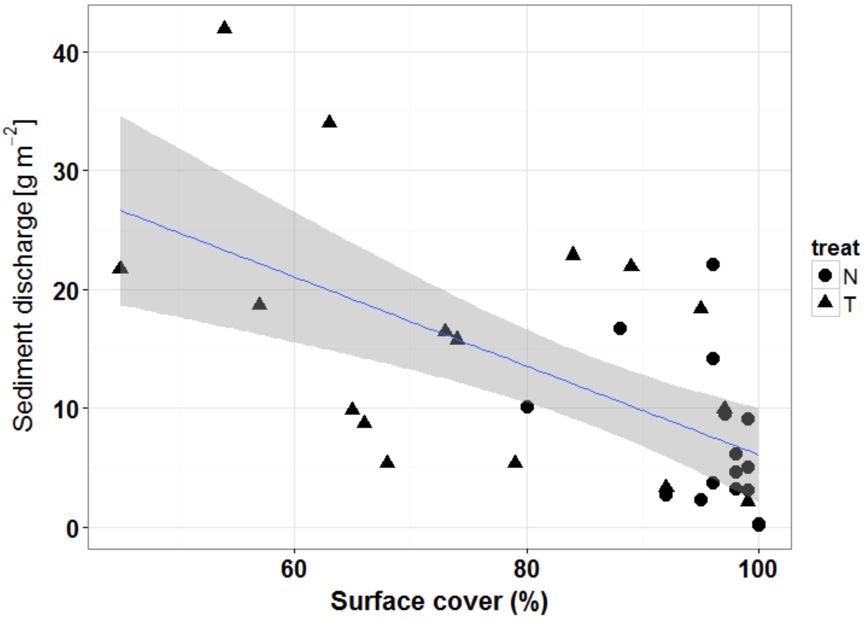

Figure 3: Surface cover affects sediment discharge on tilled and no-tilled treatments at the FAST trial, Switzerland (n=32).

4. Discussion

Micro-scale ROPs are a useful tool to compare the influences of different treatments on soil erosion under homogenous site conditions (Wainwright et al., 2000). Furthermore, they offer the possibility to raise the number of replications (c.f. Rüttimann et al., 1995) and thus, improve the statistical outcome of field measurements (Hudson, 1993). Nevertheless, those measurements show high natural and measurement variability (Bagarello and Ferro, 1998) as well as physical disturbances, such as boundary effects (Mutchler et al., 1994). The methodological limitations have to be considered when comparing results to other studies (Boix-Fayos et al., 2006) and direct extrapolation from plot to hill slope or catchment scale is not suitable in most cases (Cammeraat, 2002). The application of a rainfall simulator is especially useful to create standardized rain spectra and compare specific treatments among each other (Seeger, 2007).

4.1 The influence of organic farming on soil erosion and the effect of tillage systems

Results show that organic farming reduces soil erosion compared to conventional farming in our field trial. This finding supports assumptions which underline the potential of organic farming to reduce the risk of soil erosion (Erhart and Hartl, 2009; Gomiero et al., 2011a). In the study of Reganold et al. (1987) organic farming had a long-term erosion-reducing effect and at the same time, soils showed higher SOM contents and thicker topsoil layers. Auerswald et al. (2003) compared soil erosion on organic and conventional farms in 2050 districts of Bavaria and found erosion rates slightly lower on organic treatments. Nevertheless, due to a large variability between different farming systems, the authors concluded that no general effect can be stated and the implementation of best management practices could also improve conventional farming systems. Alike, Arnhold et al. (2014) found soil erosion rates 18 % lower in organic compared to conventional radish treatments but at once, soil erosion was 25 % higher on organic potato treatments. This difference was mainly explained by varying weed and crop covers, respectively. Those findings show that even if organic farming has the potential to reduce sediment loss, this effect can be outperformed by other agricultural practices. Our study provides evidence that the applied tillage system has a higher influence on soil erosion than the farming system and the use of ploughs down to a depth of 20 cm favours sediment discharge most. Conservation tillage (red-till and no-till) has the highest impact on the reduction of sediment discharge in our study. Bradford and Huang (1994) ascribed this effect to a continuous soil surface cover as well as greater soil strength and resistance against detachment. Nevertheless, the missing interaction between the management treatment and the tillage system strengthens the importance of both, farming and tillage practice. Even if higher runoff volume causes higher sediment discharge, there is no influence of the management practice or tillage system on this parameter. This finding has to be explained by other processes.

4.2 The influence of reduced tillage on soil erosion under organic farming

Organic farming in combination with reduced-tillage shows the best performance regarding soil conservation in our experiment. The opposite is true for conventional farming with conventional tillage practices. It has been shown that conservation tillage systems decrease sediment loss and thus seem to be a major contribution to soil conservation in general (Gebhardt et al., 1985; Montgomery, 2007). Those practices leave a protecting cover from the previous year's residue (Bradford and Huang, 1994), as well as soil disturbances and downslope-movements of soil particles are reduced (Richter, 1998). Further, our results show that conventional farming without plough operations can importantly reduce soil erosion. At the same time, conventional tillage, which is to a certain extent necessary in organic farming for weed control, can raise soil erosion rates to the level of well managed conventional farming. Pacini et al. (2003) investigated the influence of organic farming on soil erosion at three Italian farms and reported increasing soil losses on one farm that is partially hilly. This finding was explained by a greater number of tillage-requiring crops at this particular farm. Arnhold et al. (2014) recently assumed that organic farming requires additional conservation measures to provide an effective control of soil erosion. We conclude that reduced-tillage is a major improvement to the latter and can help to decrease soil erosion rates in organic farming considerably.

Our study underlines that a protecting soil surface cover is the most important parameter for soil erosion control, as it has already shown by many authors (c.f. Thornes, 1990; Richter, 1998; Morgan, 2005; Blanco-Canqui and Lal, 2008). Additionally, a continuous or longer lasting soil surface coverage throughout the year is one of the main contributions of conservation tillage for soil protection (Blanco-Canqui and Lal, 2008). Our study underpins the latter and shows that the effect of organic or conventional farming on soil surface cover is less important than the effect of tillage operations. Another aspect of tillage erosion is the destruction of the natural soil structure by ploughing together with a decrease in aggregation (Mikha and Rice, 2004).

5. Conclusions

A field experiment was performed with a portable rainfall simulator and micro-scale ROPs on a farming system trial to investigate the influence of organic farming and tillage systems on soil erosion. Our results show that:

1. Organic farming decreases soil erosion rates compared to conventional farming. This effect is outperformed by the applied tillage system, as no-tillage has the most significant decreasing influence on erosion rates.
2. Reduced tillage practices are a major soil-protecting improvement to organic farming and an efficient contribution to soil conservation. Soil surface cover is the main factor in the prevention of soil erosion, but mostly controlled by tillage than the farming system.

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- SEITZ S, GOEBES P, KÄPPELER K, NEBEL M, KÜHN P, SCHOLTEN T. (2014): Biological soil crusts in subtropical China and their influence on initial soil erosion. European Geosciences Union (EGU) General Assembly 2014, Vienna, Austria. [Poster]
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