On the Influence of Afforestation on Soil Erosion and Soil Carbon in a Subtropical Chinese Forest Ecosystem

Results from a tree diversity experiment

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Abbreviations

AGB	Aboveground Biomass
ANOVA	One-way Analysis of Variance
BD	Soil Bulk Density
BEF	Biodiversity and Ecosystem Functioning
BGB	Belowground Biomass
BSCs	Biological Soil Crusts
C	Carbon
DBH	Diameter at Breast Height
ER	Enrichment Ratio
LAI	Leaf Area Index
LSD	Least Significant Difference
N	Nitrogen
ROP	RunOff Plot
SOC	Soil Organic Carbon
SOCD	Soil Organic Carbon Density
TLS	Terrestrial Laser Scanner
TKE	Throughfall Kinetic Energy
TSR	Tree Species Richness
VIFs	Variance Inflation Factors

Abstract

Forests in subtropical China were undergoing great changes in the last decades, mainly caused by extensive deforestation. Afforestation in turn can help not only to increase the production of timber but also to enhance forest ecosystem services such as soil erosion control, soil properties, carbon storage and thus help mitigating climate change. However, even after long-term afforestation projects the hilly red soil region in southern China is still facing serious soil erosion. This might result from structural shortcomings of the tree species chosen and tree species richness planed for afforestation. Therefore, it is urgent to answer the question how tree species and tree diversity and especially the relationship between diversity and ecosystem functioning affect soil erosion. In addition, little research addresses the role of afforestation for carbon (C) and nitrogen (N) turnover and transport by soil erosion under forest, which is important for soil fertility and the assessment of carbon and nitrogen fluxes from soil to adjacent aquatic ecosystems as well as to the atmosphere. Moreover, in the earlier stage of afforestation after deforestation, soil organic carbon (SOC) dynamics are still unclear, especially in subtropical areas with intensive human impacts on forest ecosystems.

Based on a biodiversity and ecosystem functioning project in China (BEF China), this dissertation firstly used point cloud data from terrestrial laser scanners (TLS) and splash cups to analyze spatial leaf area index (LAI) and to predict the potential of splash erosion in subtropical forests. Measurements of sediment delivery were conducted during the rainy seasons from 2013 to 2015 to detected temporal changes of soil erosion and soil carbon and nitrogen fluxes and investigate the influences of tree species and diversity. Finally, 132 soil profiles at five increments (0-5 cm, 5-10 cm, 10-20 cm, 20-30 cm, 30-50 cm) were sampled in 2010 and 2014 to assess changes of SOC stocks.

Results showed that lognormal and exponential linear models were suitable to describe the vertical and horizontal LAI distribution of selected tree species, respectively. Vertical distributions of LAI and throughfall kinetic energy (TKE) of different tree species were significantly different. BEF China is still suffering from severe soil erosion even after 6 years of tree growth. Leaf area index (LAI) and biological soil crusts (BSCs) were the two main factors driving soil erosion within tree stands of different species richness. Higher tree species richness lead to decreasing soil erosion by positive effects on tree canopies and surface covering BSCs. Sediment C and N concentrations increased while annual soil C and N fluxes significantly decreased at a rate of 50% in the observed three years together with sediment delivery. Soil C and N fluxes in the study were as high as in deforestation areas even after 6 years of tree growth. Earlier afforestation in BEF China resulted in a reduction of approximately 274 Mg SOC from 2010 to 2014 in total. The reduction of SOC is mainly from the 0-20 cm topsoil. Afforested areas with higher original SOC stock showed higher losses. Tree growth and litter fall as an important carbon input to soil could not compensate SOC stock reduction in the earlier stage of the afforestation.

Zusammenfassung

Die Wälder im subtropischen China erfuhren in den letzten Jahrzehnten große Veränderungen durch umfassende Entwaldungen. Aufforstungen können nicht nur dazu beitragen, die Holzproduktion zu steigern, sondern auch Dienstleistungen von Waldökosystemen wie z.B. Schutz gegen Bodenerosion, Verbesserung von Bodeneigenschaften oder Kohlenstoffspeicherung sicherzustellen und damit einen Beitrag zur Eindämmung des Klimawandels zu leisten. Die hügelige „Red Soil“ Region in Südchina ist auch trotz zahlreicher, langfristiger Aufforstungsprojekte nach wie vor mit hohen Bodenerosionsraten konfrontiert. Dies kann u.a. auf funktionelle Mängel der ausgewählten Baumarten und der für die Aufforstung geplanten Baumartendiversität zurückzuführen sein. Es ist daher von großem Interesse, wie Baumarten und Baumartenvielfalt, sowie insbesondere der Zusammenhang zwischen Artenvielfalt und Ökosystemfunktionen die Bodenerosion beeinflussen. Darüber hinaus befassen sich nur wenige Studien mit der Rolle der Aufforstung für den Kohlenstoff- und Stickstoff-Haushalt, sowie deren Transport durch Bodenerosionsprozesse unter Wald. Diese Fragestellungen sind für die Bodenfruchtbarkeit und die Bewertung von Kohlenstoff- und Stickstoffflüssen vom Boden zu angrenzenden aquatischen Ökosystemen sowie zur Atmosphäre von großer Wichtigkeit. Weiterhin ist bisher auch nur wenig über die Dynamik des organischen Kohlenstoffs im Boden in frühen Phasen der Aufforstung bekannt. Dies gilt insbesondere für subtropische Waldökosysteme unter intensiver menschlicher Nutzung.

Im Rahmen eines Biodiversitätsprojekts innerhalb eines subtropischen chinesischen Waldgebietes (BEF China) wurden in dieser Arbeit zunächst Punktwolkendaten von terrestrischen Laserscannern (TLS) und Splash Cups verwendet, um den räumlichen Blattflächenindex (LAI) zu analysieren und das Potenzial der Splash-Erosion im Bestandsniederschlag (TKE) vorherzusagen. Während der Regenzeiten von 2013 bis 2015 wurden Messungen der Sedimentfracht mit Erosionsmessplots durchgeführt, um zeitliche Veränderungen der Erosionsraten und der Kohlenstoff- und Stickstoffflüsse im Boden zu erfassen und die Auswirkungen von Baumarten und Baumartendiversität auf diese zu untersuchen. In den Jahren 2010 und 2014 wurden 132 Bodenprofile in fünf Tiefenstufen (0-5 cm, 5-10 cm, 10-20 cm, 20-30 cm, 30-50 cm) untersucht, um die Veränderungen der Kohlenstoff-Bestände im Boden zu beurteilen.

Die Ergebnisse zeigen, dass lognormale und exponentielle lineare Modelle geeignet sind, die vertikale und horizontale LAI-Verteilung ausgewählter Baumarten zu beschreiben. Die vertikalen Verteilungen von LAI und TKE verschiedener Baumarten waren signifikant unterschiedlich. Innerhalb des BEF China Projektes lassen sich auch nach 6 Jahren Baumwachstum noch immer starke Erosionsraten nachweisen. LAI und biologische Bodenkrusten waren die beiden Haupteinflussfaktoren auf Bodenerosionsprozesse in Baumbeständen mit unterschiedlichem Artenreichtum. Eine höherer Baumartendiversität führte zu einer abnehmenden Bodenerosion durch positive Auswirkungen der Kronendächer und flächendeckender biologischer Bodenkrusten. Die Konzentrationen von C und N im Sedimentabtrag stiegen im Untersuchungszeitraum an, während die jährlichen Abflüsse von C und N in den beobachteten drei Jahren zusammen mit der Sedimentabgabe signifikant um 50 % zurückgingen. Die C- und N-Flüsse waren auch nach 6 Jahren Baumwachstum so hoch wie in Entwaldungsgebieten. Die junge Aufforstung im BEF China Experiment führte zu einer Reduktion von insgesamt ca. 274 Mg Bodenkohlenstoff von 2010 bis 2014. Die Reduktion des Kohlenstoffs erfolgte hauptsächlich im Oberboden (0-20 cm). Aufgeforstete Flächen mit höheren ursprünglichen C-Beständen zeigten höhere Verluste. Baumwachstum und Streufall als wichtiger Kohlenstoffeintrag in den Boden konnten die Reduzierung des C-Bestands in der frühen Phase der Aufforstung nicht kompensieren.

List of publications and personal contribution

Accepted manuscripts

(1) Song Z, Seitz S, Li J, Goebes P, Schmidt K, Kühn P, Shi X, Scholten T. (2019): Tree diversity reduced soil erosion by affecting tree canopy and biological soil crust development in a subtropical forest experiment. *Forest Ecology and Management* 444: 69-77. doi: 10.1016/j.foreco.2019.04.015

(2) Song Z, Seitz S, Zhu P, Goebes P, Shi X, Xu S, Schmidt K, Kühn P, Shi X, Scholten T. (2018): Spatial distribution of LAI and its relationship with throughfall kinetic energy of common tree species in a Chinese subtropical forest plantation. *Forest Ecology and Management*, 425, 189-195. doi: https://doi.org/10.1016/j.foreco.2018.05.046

(3) Seitz S, Nebel M, Goebes P, Käppeler K, Schmidt K, Shi X, Song Z, Webber C.L, Weber B, Scholten T. (2017): Bryophyte-dominated biological soil crusts mitigate soil erosion in an early successional Chinese subtropical forest. *Biogeosciences* 14(24): 5775-5788. doi: 10.5194/bg-14-5775-2017

(4) Seitz S, Goebes P, Song Z, Bruelheide H, Härdtle W, Kühn P, Li Y, Scholten T. (2016): Tree species and functional traits but not species richness affect interrill erosion processes in young subtropical forests. *Soil*, 2, 49-61. doi: https://doi.org/10.5194/soil-2-49-2016

(5) Trogisch S, Schuldt A, Bauhus J, Blum JA, Both S, Buscot F, Castro-Izaguirre N, Chesters D, Durka W, Eichenberg D, Erfmeier A, Fischer M, Geißler C, Germany MS, Goebes P, Gutknecht J, Zacharias Hahn C, Haider S, Härdtle W, He J-S, Hector A, Hönig L, Huang Y, Klein A-M, Kühn P, Kunz M, Leppert KN, Li Y, Liu X, Niklaus PA, Pei Z, Pietsch KA, Prinz R, Proß T, Scherer-Lorenzen M, Schmidt K, Scholten T, Seitz S, Song Z, Staab M, von Oheimb G, Weißbecker C, Welk E, Wirth C, Wubet T, Yang B, Yang X, Zhu CD, Schmid B, Ma K, Bruelheide H. (2017): Toward a methodical framework for comprehensively assessing forest multifunctionality. Ecology and Evolution 7(24), 10652-10674. doi: https://doi.org/10.1002/-ece3.3488

(6) Scholten T, Goebes P, Kühn P, Seitz S, Assmann T, Bauhus J, Bruelheide H, Buscot F, Erfmeier A, Fischer M, Haerdtle W, He J, Ma K, Niklaus PA, Scherer-Lorenzen M, Schmid B, Shi X, Song Z, vonOheimb G, Wirth C, Wubet T, Schmidt K. (2017): On the combined effect of soil fertility and topography on tree growth in subtropical forest ecosystems-a study from SE China. *Journal of Plant Ecology*, 10, 111-127. doi: https://doi.org/10.1093/jpe/rtw065

Submitted manuscripts

(7) SONG Z, SEITZ S, LI J, SCHMIDT K, KÜHN P, SHI X, SCHOLTEN T. Soil Carbon and Nitrogen Fluxes by Water Erosion in Young Forest Biodiversity Function Ecosystem of Subtropical China.

In preparation

(8) SONG Z, SEITZ S, LI J, GOEBES P, SCHMIDT K, KÜHN P, SHI X, SCHOLTEN T. Soil Organic Carbon Changes in the Earlier Stage of a Chinese Subtropical Tree Diversity Experiment.

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	80	70	80	80
(2)	yes	8	1	50	90	80	80
(3)	yes	7	3	10	15	15	15
(4)	yes	10	7	5	10	10	10
(5)	yes	23	19	<5	<5	<5	<5
(6)	yes	51	39	<5	<5	<5	<5
(7)	no	7	1	80	80	80	80

1. Introduction and objectives

Forests are undergoing great changes globally (Bonan, 2008; Smith *et al.*, 2016). According to the data of FAO (2015), the natural forest area declined about 240 Mha between 1990 and 2015 while planted forest increased by 110 Mha (Keenan *et al.*, 2015). Many countries make great endeavors to afforest (Paul *et al.*, 2002; Korkanç, 2014; Yosef *et al.*, 2018). China is one of the largest cultivators of forest plantations in the world and its forested area was increasing by 1.5 Mha a^{-1} between 2010 and 2015 (FAO, 2015; Keenan *et al.*, 2015). In subtropical China, the ecosystems are dominated by evergreen broad-leaved forests without human disturbance (Wang *et al.*, 2007; Bruelheide *et al.*, 2014a). However, in the last decades these areas were mostly cleared and have been converted into monospecific conifer stands for many reasons (Zhao, 2006; Wang *et al.*, 2007; Li *et al.*, 2014a). For instance, the two most important tree species for forest resources and ecological services in subtropical China, *Chinese fir* and *Pinus massoniana*, cover approximately 12.39×10^6 ha or 10% of the forest area and 6.78×10^5 ha or 27% of the forest area in the Three Gorges Reservoir area, respectively (Wang *et al.*, 2012a; Huang *et al.*, 2013; Wang, 2014). Apparently, afforested areas attracted scientists' attentions due to the impacts on forest ecosystem services such as soil erosion control, soil properties improvement, C storage and mitigation of climate change (Piao *et al.*, 2009; Assefa *et al.*, 2017; de Araújo Filho *et al.*, 2018; Hong *et al.*, 2018; Li *et al.*, 2018b).

Tree species structures and their contribution to splash erosion

Soil erosion is a serious environmental hazard of global scale (Lal, 2003) and vegetation cover of the soil surface is one key factor in controlling soil erosion (Stednick, 1996; Cao *et al.*, 2008; Shi *et al.*, 2009; Chen *et al.*, 2011; Filoso *et al.*, 2017; Feng *et al.*, 2018). Forest vegetation cover affect splash erosion at the ground surface by the interception process from its structure, such as modifying drop size and speed, changing rainfall amount and spatial distribution (Nanko *et al.*, 2006; Geißler *et al.*, 2012b; Geißler *et al.*, 2013; Goebes *et al.*, 2015b). It is generally accepted that soil erosion is reduced under forests (Smith, 1914). Although great endeavors have been made to restore and afforest vast areas with commercial monocultures (Zhao, 2006; Wang *et al.*, 2007; Lei *et al.*, 2009; Guo *et al.*, 2015), soil erosion commonly occurs (cf. Figure 1) and highly varies even in forested areas of subtropical China ranging from 0 to 6.32 t ha^{-1} a^{-1}. These current circumstances imply that monospecific plantations might be less suitable for soil erosion control. One reason is that effects of forest cover on splash erosion are dynamic in space as the structures of tree species differ. Hence, calculating an index that describes the ability of cover plants, especially trees, is essential to analyze the splash erosion risk under forest and can help to better understand the relationship between cover plants and splash erosion. Such an index can also serve in planning and management of afforestation as part of soil and water conservation approaches, e.g. in the hilly red soil region in southern China.

One well established index that describes the plant cover is the leaf area index (LAI, (Jordan, 1969)). It is defined as projected leaf area per unit ground area (Gower and Norman, 1991). As an important biophysical parameter, LAI is often used in quantitative analyses of processes related to vegetation dynamics such as rainfall interception (Maass *et al.*, 1995), soil erosion modeling (Laflen *et al.*, 1997; Zhou *et al.*, 2008; Zhang *et al.*, 2014), land surface process models (Chen *et al.*, 2011; Tesemma *et al.*, 2015) and global climate change (Claverie *et al.*, 2016). In the subtropical part of China, studies showed that LAI has a significant effect on throughfall kinetic energy (TKE) in secondary forest (Geißler *et al.*, 2012a), on soil loss in 30-year afforestation (Sun *et al.*, 2010; Zhang *et al.*, 2011) and on sediment discharge and TKE in young afforestation (Goebes *et al.*, 2015a; Seitz *et al.*, 2016). Further vegetation factors that are correlated with TKE in forests are crown cover, leaf traits, tree height and branch architecture (Cao *et al.*, 2008; Geißler *et al.*, 2010; Geißler *et al.*, 2012b; Goebes *et al.*, 2015a; Goebes *et al.*, 2015b). Another important aspect is, that the process of free raindrops passing the tree canopies is dynamic (Nanko *et al.*, 2006) and the canopy architecture can change the drop size and spatial distribution significantly at different positions and height of the tree canopy (Nanko *et al.*, 2006; Goebes *et al.*, 2015b). Hence, the relationship between general LAI values and splash erosion is questionable since it neglects the effects of spatial distributions of LAI. Also, most studies concentrate on mature forests (Cao *et al.*, 2008; Geißler *et al.*, 2013). Regarding afforestation measures on heavily eroded soils with a low structure stability and without shrubs or litter cover, like in the hilly red soil region in southern China (Zhao, 2006; Shi *et al.*, 2009), the role of forests in their early stage of tree growth to protect the soil from erosion is of ample interest. Such research is still scarce.

The development of sediment delivery and its relationship with tree diversity after afforestation

High sediment delivery often occurs in forested catchments in subtropical regions (Marks, 1998; Molnar, 2004; Zhao, 2006). Along with soil erosion, growing concern about loss of biodiversity is emerging worldwide due to substantial contemporary declines in biodiversity at different scales (Tittensor *et al.*, 2014; Mori *et al.*, 2017). As part of the heated scientific research, recently, different researchers focused on the effect of biodiversity on soil erosion control. Pohl *et al.* (2009); Martin *et al.* (2010) and Wang *et al.* (2012b) pointed out that plant species richness negatively correlated with runoff and sediment delivery. TKE from splash cup measurements can decrease with higher tree species richness in forest stands (Geißler *et al.*, 2013), but higher neighborhood diversity can increase TKE in young forest plantations (Goebes *et al.*, 2015b). Meanwhile, TKE and sediment delivery are strongly affected by tree species, but the effect of tree species richness in early stage afforestation is not yet clear (Goebes *et al.*, 2015b; Goebes *et al.*, 2016; Seitz *et al.*, 2016). Those findings suggested a high grade of uncertainty about the relationships between tree diversity and soil erosion. Additionally, they all focus on a single point in time and to our knowledge, measurements covering a longer period have not been conducted. Different tree species have different spatial distribution patters e.g. regarding leaf areas, leading to changing TKE (Song *et al.*, 2018). Thus, with ongoing tree growth the combination of different tree species tends to not only modify the vertical vegetation structure and increase the quantity of root and litter but also improve soil properties and consequently reduce soil erosion. At the same time, highly diverse biological soil crusts (BSCs) cover important areas in young subtropical forest plantations and have a high mitigating influence on soil losses (Seitz *et al.*, 2017). Therefore, research on the temporal relationship between forest biodiversity, soil protecting vegetation patterns and soil losses is essential for understanding how biodiversity might sustain ecological services such as water erosion prevention.

Figure 1 Sediment transport in the river Ganghang nearby the BEF China experimental site, Xingangshan, Jiangxi Province, PR China after high rainfall events in June 2014 (left) and July 2015 (right).

The development of soil carbon and nitrogen fluxes and its relationship with tree diversity after afforestation

Soil erosion strongly affects the global carbon cycle as it redistributes soil and related soil C (Carpenter *et al.*, 1998; McCorkle *et al.*, 2016; García-Díaz *et al.*, 2017; Poesen, 2018; Lal, 2019). Around the world, 5.7 Pg C equivalent to 0.84% of global soil C stock (677 Pg in 0-30 cm soil depth) was displaced by soil erosion every year (Lal, 2003, 2018). In China, water erosion induced 180 ± 80 Tg C equivalent to 0.41% of national topsoil C stock (43.6 Pg in topsoil) of displacement per year between 1995 and 2015 (Song *et al.*, 2005; Ni, 2013; Yue *et al.*, 2016). These results confirm the importance of soil C transported by erosion process on the global carbon balance.

C and N displacement caused by soil erosion is a scientific research hotspot (Wang *et al.*, 2013b). Many papers address land use change as driving factor of soil C alterations by soil erosion (Jacinthe *et al.*, 2004; Martinez-Mena *et al.*, 2008; Nadeu *et al.*, 2012). This includes afforestation which increasingly spreads in many countries (Paul *et al.*, 2002; Korkanç, 2014; Keenan *et al.*, 2015; Yosef *et al.*, 2018). China as one of the largest cultivators of forest plantations in the world has approximately 90% of afforestation to its forest area expansion since the last 50 years (Piao *et al.*, 2009). Generally, increased forest cover prolongs the process of throughfall reaching soil surface by intercepting raindrops, modifying drop size and speed, and changing rainfall amount and energy (Nanko *et al.*, 2006; Geißler *et al.*, 2012b; Geißler *et al.*, 2013; Masselink *et al.*, 2016). In addition, afforestation can improve soil properties and structure such as soil water holding capacity and aggregate stability (Gol *et al.*, 2010; Korkanç, 2014) and produce litter that covers the soil surface (Seitz *et al.*, 2015). Therefore, it is accepted that afforestation is reducing soil erosion (Bonan, 2008; Zhao *et al.*, 2013; Keesstra *et al.*, 2017). However, recent examples from subtropical China show that afforestation can have an inconsistent effect on throughfall kinetic energy and sediment discharge (Goebes *et al.*, 2015a; Seitz *et al.*, 2016). Positive or negative effects of forests for soil erosion depend on many dynamic and species specific factors such as leaf area index, BSCs, tree height, spatial distribution of leafs and stand age (Goebes *et al.*, 2015a; Seitz *et al.*, 2016; Song *et al.*, 2018). Over time after afforestation, sediment delivery decreases (Song *et al.*, 2019). However, coupling of sediment and C and N fluxes during erosion events is still not well understood and studies on the carbon budget of forest ecosystems related to soil erosion are limited (Stacy *et al.*, 2015). In addition, it is not reported how tree diversity affects sediment C and N fluxes.

Soil carbon stock changes after afforestation

Soil organic carbon (SOC) as the largest pool of terrestrial organic carbon accounts for approximate 40% of the whole C stock (to 1 m soil depth) in forest (Dixon *et al.*, 1994; Jobbágy and Jackson, 2000; Pan *et al.*, 2011). It is sensitive to land use changes such as deforestation and afforestation (Jandl *et al.*, 2007; Laganiere *et al.*, 2010). Deforestation could reduce SOC stock due to the decreased organic matter inputs to soil and the increase of decomposition rate and soil erosion caused by soil disturbance while no consistent effect of afforestation on SOC exists (Veldkamp, 1994; Murty *et al.*, 2002; Assefa *et al.*, 2017; de Araújo Filho *et al.*, 2018; Lal, 2019). Many studies address that positive or negative effects of afforestation on SOC stocks largely depend on factors such as previous land use, tree species, stand age, and site management (Paul *et al.*, 2002; Laganiere *et al.*, 2010; Shi and Cui, 2010; Li *et al.*, 2012). For instance, a tendency of an initial loss in SOC are detected in the first few years of afforestation where soils are rich in original SOC (Paul *et al.*, 2002; Laganiere *et al.*, 2010; Shi and Cui, 2010). Therefore, influences of afforestation on SOC stocks in the earlier stage cannot be neglected due to the potential source of atmospheric CO_2 and its large areas around the world. However, research on this process do not attract enough attention.

BEF China

BEF China is located in Xingangshan Town, Dexing City, Jiangxi Province, PR China (29.08°–29.11° N, 117.90°–117.93° E). It is not only one of the largest forest biodiversity experiments in the world but also the first tree diversity experiment in the humid subtropics (Trogisch *et al.*, 2017). The project includes two parallel sites (Site A and Site B, which is planted in 2009 and 2010, respectively, Figure 2) with an area of 50 ha (Bruelheide *et al.*, 2014a).

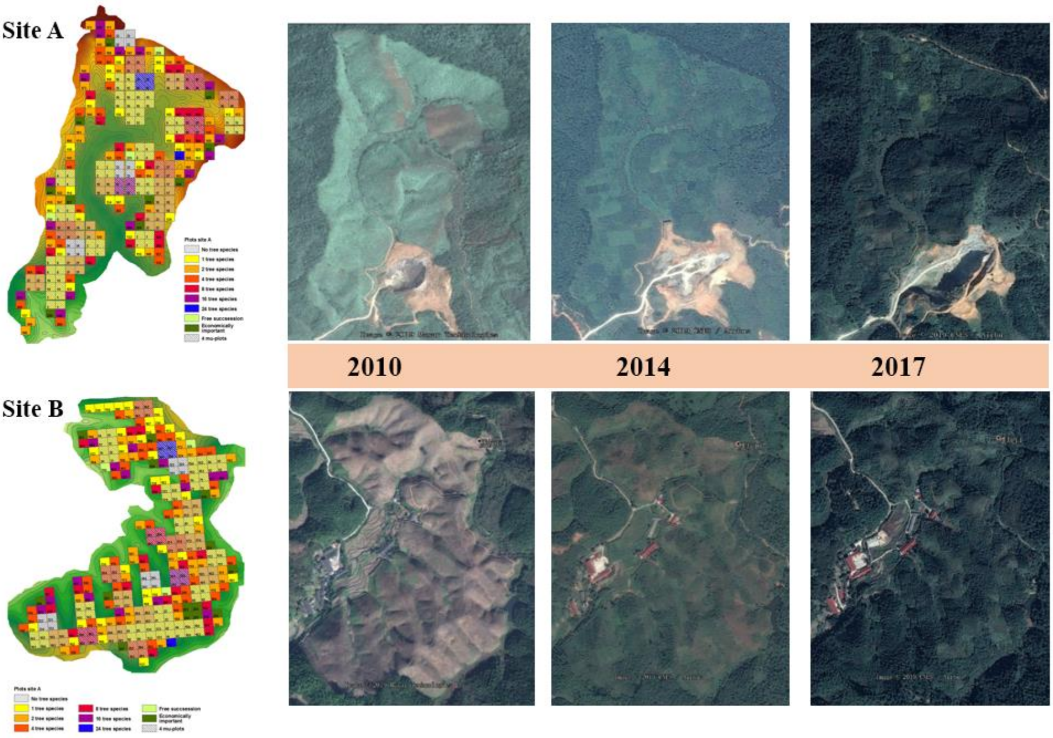
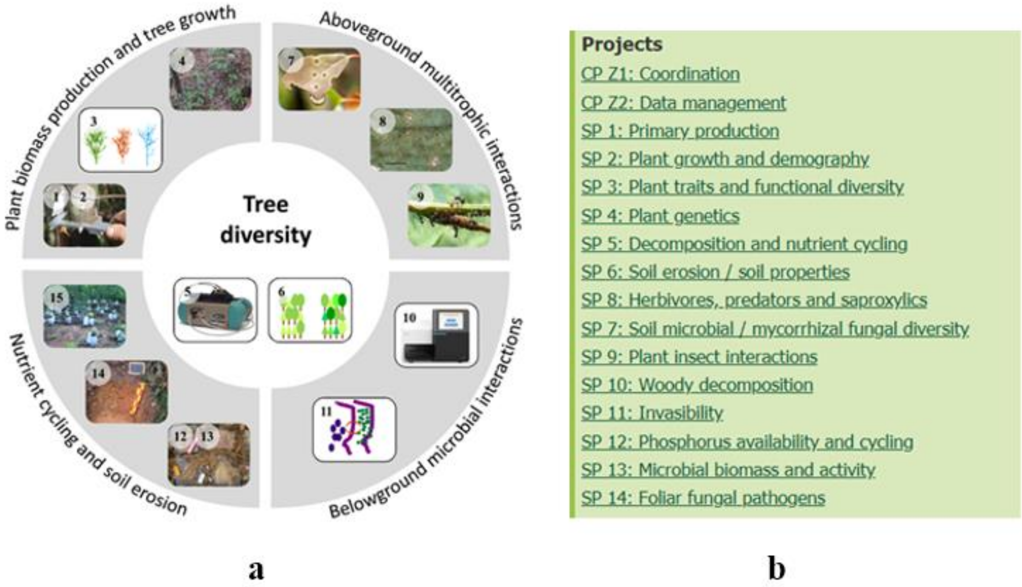


Figure 2 The distribution of plots structured on Site A and Site B in the BEF China project and Site A and B images from google earth in 2010, 2014 and 2017.

The sites were established by transplanting forty broad-leaved tree species after logoff of original forest (Bruelheide *et al.*, 2014a). The forty tree species were planted in monocultures and mixtures of 2, 4, 8, 16 and 24 species on 566 plots with each measuring 25.8 × 25.8 m (667 m^2) (Bruelheide *et al.*, 2014a). For each plot, 400 tree individuals were planted in 20 rows of 20 tree individuals with a planting distance of 1.29 m (Bruelheide *et al.*, 2014a). With its unique feature of the large range of tree species richness levels, BEF China provides a platform to various research topics, especially on finding out the effects of tree species and its richness on primary productivity, carbon and nitrogen storage, and soil erosion. To our knowledge, at least fifteen studies and fourteen projects are conducted (Figure 3) (Trogisch *et al.*, 2017). As an important research project in BEF China, Subproject 6 is mainly concerning on soil properties and soil erosion under afforestation and the role of biodiversity for soil erosion in forest ecosystems (Figure 4).



Projects
CP Z1: Coordination
CP Z2: Data management
SP 1: Primary production
SP 2: Plant growth and demography
SP 3: Plant traits and functional diversity
SP 4: Plant genetics
SP 5: Decomposition and nutrient cycling
SP 6: Soil erosion / soil properties
SP 8: Herbivores, predators and saproxylics
SP 7: Soil microbial / mycorrhizal fungal diversity
SP 9: Plant insect interactions
SP 10: Woody decomposition
SP 11: Invasibility
SP 12: Phosphorus availability and cycling
SP 13: Microbial biomass and activity
SP 14: Foliar fungal pathogens

a b

Figure 3 a) Range of methodical approaches applied in BEF-China to study effects of tree diversity including leaf functional trait diversity (5) and genetic diversity (6) on plant biomass production and tree growth (1+2=aboveground and belowground tree biomass and productivity, 3=tree growth and canopy architecture, 4=herb-layer biomass and diversity), aboveground multitrophic interactions (7=herbivory, 8=plant-fungal pathogens interactions, 9=trophobiosis), belowground microbial interactions (10=microbial diversity, 11=microbial biomass and activity), nutrient cycling and soil erosion (12+13=leaf litter and deadwood decomposition, 14=soil fertility and C storage, 15=soil erosion) (Trogisch et al., 2017) and b) the list of projects involved in BEF China (http://www.bef-china.de).

The objectives

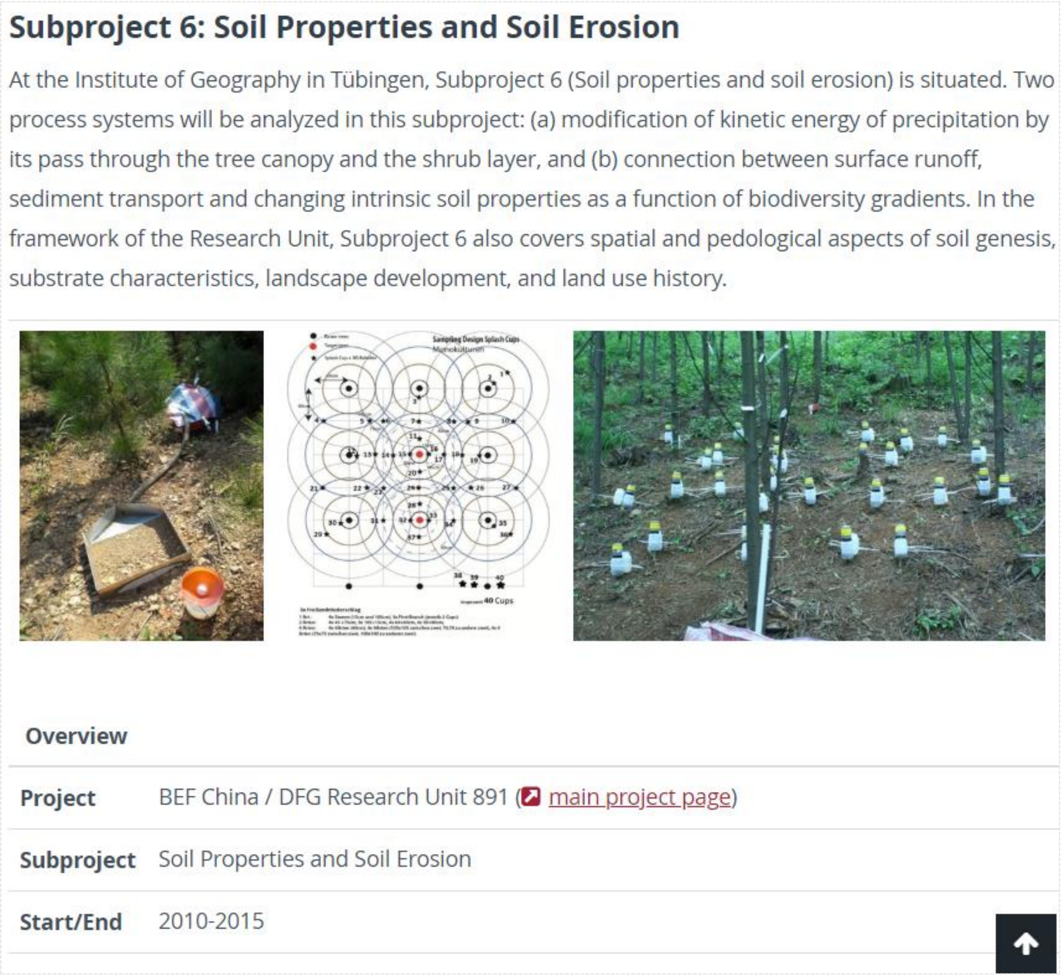
Previous research in our group have investigated the effects of species diversity, species Identity, functional traits on sediment discharge as well as on TKE (Goebes, 2015; Seitz, 2015). Therefore, based on these findings, the objectives of this dissertation were to:

1) build up the spatial distributions of LAI in common tree species and find out its relationship with throughfall kinetic energy (Objective 1).

2) investigate temporal changes of sediment delivery and its driving factors, as well as the underlying mechanism of tree diversity influences on sediment delivery after afforestation (Objective 2).

3) detect temporal changes of soil C and N fluxes by water erosion and its driving factors, as well as its relationship with tree diversity after afforestation (Objective 3).

4) monitor SOC changes and the driving factors (Objective 4).



Subproject 6: Soil Properties and Soil Erosion

At the Institute of Geography in Tübingen, Subproject 6 (Soil properties and soil erosion) is situated. Two process systems will be analyzed in this subproject: (a) modification of kinetic energy of precipitation by its pass through the tree canopy and the shrub layer, and (b) connection between surface runoff, sediment transport and changing intrinsic soil properties as a function of biodiversity gradients. In the framework of the Research Unit, Subproject 6 also covers spatial and pedological aspects of soil genesis, substrate characteristics, landscape development, and land use history.

Overview

Project	BEF China / DFG Research Unit 891 (main project page)
Subproject	Soil Properties and Soil Erosion
Start/End	2010-2015

Figure 4 Subproject 6 conducted within BEF China (http://www.bef-china.de)

2. Materials and methods

2.1. Study site

All the research was conducted in the framework of the BEF China project, which is located in Xingangshan Town, Dexing City, Jiangxi Province, PR China (29.08°–29.11° N, 117.90°–117.93° E). The climate is dominated by subtropical monsoon with a mean annual temperature of 17.4 ^{o}C and a mean annual precipitation of 1635 mm with half of it falling from May to August (Goebes *et al.*, 2015b). The natural vegetation is dominated by broadleaved forest with evergreen species (Trogisch *et al.*, 2017). The area shows mainly hills at elevations from 105 m to 200 m with slopes from 15° to 41° (Scholten *et al.*, 2017). Soils in the project are mainly Cambisols, with Anthrosols in downslope positions and Gleysols in valleys and the bedrock is non-calcareous slates weathered to saprolite (Scholten *et al.*, 2017).

The BEF China project includes two parallel sites (A and B) on which commercial monocultures were originally planted that were cut down in 2009 and 2010, respectively (Bruelheide *et al.*, 2014a). Holes of 0.5 m (length) × 0.5 m (width) × > 0.2 m (depth) were dug for seedlings (Yang *et al.*, 2013). Forty local tree species were replanted in monocultures and mixtures of 2, 4, 8, 16 and 24 species on 566 plots with each measuring 25.8 × 25.8 m (667 m^2) (Yang *et al.*, 2013). For each plot, 400 tree individuals were planted in 20 rows of 20 tree individuals with a planting distance of 1.29 m (Bruelheide *et al.*, 2014a)

2.2. Objective 1 (Selected afforested tree species structures and their contributions to splash erosion)

Tree parameters retrieval

In this study, three subtropical tree species were selected, including evergreen broadleaved species (*Lithocarpus glaber and Schima superba*) and a deciduous broadleaved species (*Sapindus saponaria*). These three species are the recommended species for the afforestation project of water and soil conservation in the subtropical region of China (The Ministry of Water Resources, 2013). For each tree species, three tree individuals were randomly selected. LAI measurements were carried out in October 2013 and point cloud data for each tree was obtained using a Terrestrial Laser Scanner (RIEGL VZ-400, Horn, Austria) (Figure 5). For each tree, 3 to 5 measurement positions were set at different directions with a horizontal distance ranging from 1.5 m to 8 m. The view zenith angle from the center of the scanner to the canopy was set to 60 degrees. Before the measurement, high reflectance sheets were stuck on pegs around the trees at various distances, heights and directions, to guarantee that more than 6 common sheets were scanned for each two adjacent stations, which provided reference points to convert all data in the same coordinate. For the parameters of RIEGL VZ-400, scanning angle resolution is 0.01° and measurement rate is 122000 points s^{-1}.

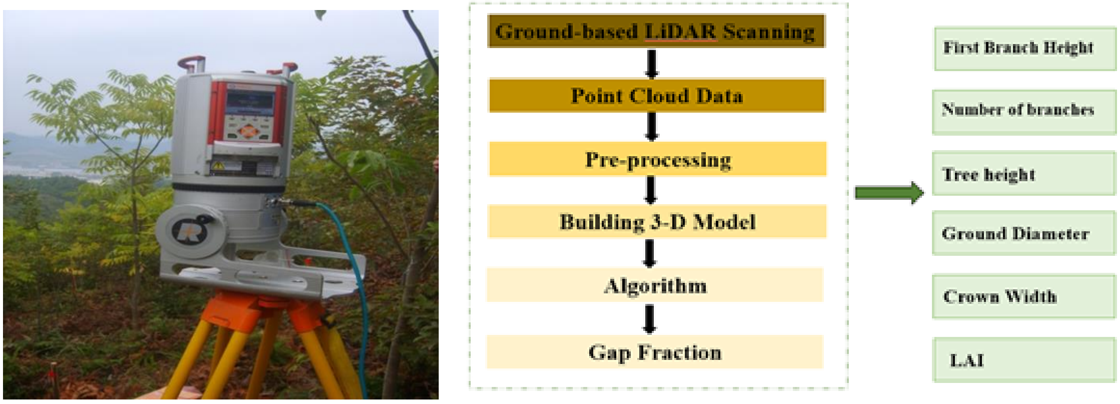


Figure 5 Terrestrial laser scanner (RIEGL VZ-400, Horn, Austria) and the flow chart of cloud point data process for tree parameters.

TKE measurement

TKE was measured using Tübingen Splash Cups (T-Cup, (Scholten *et al.*, 2011)). The cup has a diameter of 4.6 cm and a height of 4 cm (Scholten *et al.*, 2011). It is filled with uniform fine sand (0.125 mm) (Figure 6). The detached sand is calculated by the weight difference between the dry sand in the full-filled splash cup before measurements and the dry sand inside the cup after the rainfall event. Then kinetic energy of rainfall (KE_{rf}) is calculated by the detached sand (*ds*) per splash cup (*sc*) using the equation (Eq.1) below with a modified slope and standardization to 1 m^2 (Goebes *et al.*, 2015b).

$$KE_{rf}\left(Jm^{-2}\right) = ds_{sc}\left(g\right) \times 0.1455 \times \left\{1000\left(cm^{2}\right) \div \pi r_{sc}^{2}\right\} \qquad \text{Eq.1}$$

Its application was approved in field studies in subtropical China (Geißler *et al.*, 2012a; Geißler *et al.*, 2012b; Goebes *et al.*, 2015b). Five monoculture plots of *Lithocarpus glaber* (1 plot), *Schima superba* (2 plots) and *Sapindus saponaria* (2 plots) were selected to install splash cups under different tree individuals using the design of (Goebes *et al.*, 2015b). The cup positions were 15 cm, 30 cm, 45 cm, 60 cm, 75 cm, and 95 cm from the stem respectively (Figure 6, six splash cups per plot). Five rainfall events from May to July in 2013 were measured (Table 1). In total, data from 150 splash cups were collected.

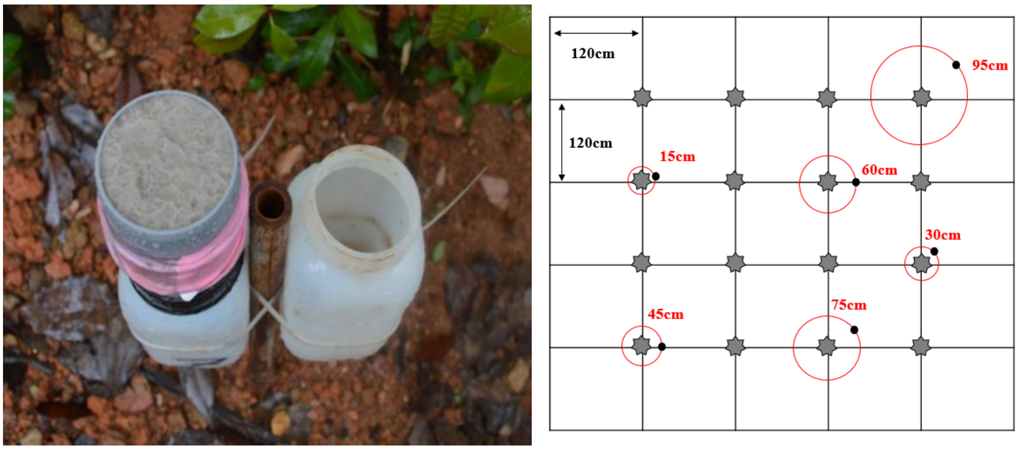


Figure 6 Splash cup measurement design with six positions according to (Goebes et al., 2015b). Gray stars, black dots and red circle lines represent tree individuals, splash cup position and radius around tree stems, respectively.

Table 1 Characteristics of the five captured rainfall events (Goebes et al., 2015b)

Rainfall events	Event 1	Event 2	Event 3	Event 4	Event 5
Rainfall amount (mm)	6.6	23.3	39.3	61.2	185.7
Rainfall duration (h)	2.33	10.16	11.5	14.5	30.58
Mean throughfall amount (mm)	5	28.3	47.9	73.8	192.7
TKE (Jm^{-2})	78.8	255.3	354.7	553.7	1292.8

Data analysis

LAI was estimated using volume element model from the point cloud data (Hosoi and Omasa, 2006; Zheng and Moskal, 2012) and was calculated with Matlab 2010b (The Mathworks Inc., Natick, MA, USA). Basic parameters of trees (ground diameter, tree height, first branch height, crown width, branch number and crown cover) were measured with the laser scanner software RiScan Pro (http://www.riegl.com). A one-way analysis of variance (ANOVA) was conducted to compare the mean value of canopy structure parameters. Skewness of LAI vertical distribution and Pearson correlation analyses to test LAI effects on TKE were conducted with IBM SPSS Statistics for Windows Version 19.0 (IBM Corp., Armonk, NY, USA). Before the Pearson correlation analyses, TKE was log10 transformed to normal distribution and tested by Kolmogorov–Smirnov (Significance = 0.2). Graph and curves fitting were processed in Origin 8.0 (Origin Lab Corporation, Northampton, MA, USA).

2.3. Objective 2 and 3 (The development of sediment delivery and soil carbon and nitrogen fluxes)

Research plot

For Objective 2 and 3, 45 of these plots were selected, 23 on Site A and 22 on Site B with no tree planting and four tree species richness levels: monocultures, 8 tree species, 16 tree species and 24 tree species stands (Table 2).

Table 2 Tree, topography and soil data (0-5 cm) of 45 selected research plots in the BEF China project. (TSR: tree species richness; Soil BD: soil bulk density; SOC: soil organic carbon)

Plot	TSR	Tree species	Site	Slope	Aspect	Altitude	BD	pH	SOC
				(°)		(m)	(g m^{-3})		(%)
L20	0	/	A	24	W	229	0.86	3.68	3.96
Q23	0	/	B	23	N	153	0.78	3.39	3.47
D29	1	M. flexuosa	B	31	N	159	0.90	3.68	2.77
I25	1	M. yuyuanensis	B	29	N	152	0.96	3.47	2.90
M07	1	B. luminifera	B	31	S	129	0.89	3.55	2.52
N02	1	M. flexuosa	B	41	S	129	0.89	3.61	3.01
N05	1	A. altissima	B	32	N	119	0.89	3.63	3.17
N28	1	I. polycarpa	B	19	E	167	0.97	3.56	2.21
Q27	1	A. fortunei	B	35	S	160	0.97	3.66	2.78
Q29	1	M. leptophylla	B	33	E	144	0.90	3.74	2.42
R29	1	C. fargesii	B	33	S	146	0.91	3.65	2.48
T13	1	M. thunbergii	B	21	W	133	0.96	3.44	2.59
U16	1	E. japonicus	B	20	W	147	0.94	3.44	2.65
V24	1	E. chinensis	B	32	E	137	0.94	3.71	3.01
W10	1	Ph. bournei	B	27	E	147	0.92	3.45	2.35
W11	1	E. glabripetalus	B	19	S	148	1.04	3.25	2.72
X21	1	M. grijsii	B	24	N	132	0.91	3.65	2.54
Y09	1	C. biondii	B	32	E	126	1.10	3.71	1.93
E31	1	Q. fabri	A	22	S	144	0.95	3.86	2.48
E33	1	L. glaber	A	19	S	144	1.12	3.94	2.18
E34	1	C. henryi	A	21	S	125	1.06	4.09	2.84
G33	1	Q. serrata	A	18	S	127	0.85	3.92	3.45
I28	1	L. formosana	A	26	S	163	0.90	3.81	3.29
K19	1	S. superba	A	24	N	199	0.80	3.70	4.18
L10	1	C. eyrie	A	34	S	211	0.92	3.92	2.81
L11	1	C. sclerophylla	A	28	S	201	1.04	3.87	2.95
N11	1	S. saponaria	A	26	S	203	0.82	3.63	3.93
N13	1	S. sebiferum	A	31	S	182	0.78	3.78	3.62
N17	1	R. chinensis	A	28	W	221	0.91	3.79	3.39
O22	1	C. myrsinaefolia	A	21	W	229	0.86	3.80	3.54
O27	1	Ch. axillaris	A	21	W	185	1.07	4.12	2.41
Q13	1	K. bipinnata	A	30	W	215	0.90	3.86	3.84
R14	1	C. glauca	A	30	N	228	0.82	3.80	4.25
J29	8	1*	B	31	N	182	0.81	3.39	4.85
Q17	8	2*	B	22	N	131	0.99	3.52	2.91
S10	8	3*	A	36	S	220	0.96	3.79	3.04
T15	8	4*	A	30	N	244	0.87	3.67	3.42

I22	16	5*	B	28	S	119	1.07	3.58	2.28
S22	16	5*	B	33	W	145	1.00	3.61	3.26
L22	16	6*	A	21	W	180	0.80	3.79	3.48
M22	16	6*	A	23	W	221	0.95	3.79	3.48
U10	16	6*	A	40	S	231	0.96	3.86	3.22
R30	24	7*	B	27	S	136	0.95	3.67	2.46
N09	24	8*	A	33	S	218	0.86	3.58	3.60
R18	24	8*	A	36	W	215	0.91	3.82	3.50

1*: 8 tree species from monocultures N05, Y9, W11, U16, N28, X21, D29 and W10.

2*: 8 tree species from monocultures Q27, M07, R29, V24, Q29, T13, I25 and *Q. phillyreoides*.

3*: 8 tree species from monocultures E34, L11, O27, I28, G33, N11, N13 and *N. sinensis*.

4*: 8 tree species from monocultures E33, E31, N17, K19, L10, R14, O22 and Q13.

5*: 16 tree species from 1* and 2*.

6*: 16 tree species from 3* and 4*.

7*: 24 tree species from 5*, from monocultures E33, K19, R14, L10, L11, and *Cinnamomum camphora*, *Daphniphyllum oldhamii* and *Diospyros glaucifolia*.

8*: 24 tree species from 6* and *Cinnamomum camphora*, *Daphniphyllum oldhamii*, *Diospyros glaucifolia*, *Acer davidii*, *Castanopsis carlesii*, *Melia azedarach*, *Quercus acutissima* and *Sapium discolor*.

Soil erosion and soil carbon and nitrogen fluxes measurement

Based on the design of BEF China and considering the various research topics investigated, a selected area of each research plot was used for soil erosion measurements (Bruelheide *et al.*, 2014a; Trogisch *et al.*, 2017). Five micro-scale runoff plots (ROP) (0.4 m length × 0.4 m width × 0.1 m height) were randomly installed in 2013 and connected to 20 L reservoirs to collect runoff and sediment delivery (Figure 7) (Seitz *et al.*, 2016). The runoff plots were operated from May to July during the rainy season in 2013, 2014 and 2015. Runoff volume was collected in situ and sediment delivery was calculated after sampling. Dried sediment was carefully collected and grounded on a ball mill for C and N analysis. Sediment C and N were measured with a CN-analyzer (VARIO EL III, Elementar, Hanau, Germany). In total, 550 valid measurements from 215 runoff plots were captured (182 in 2013, 158 in 2014 and 210 in 2015).

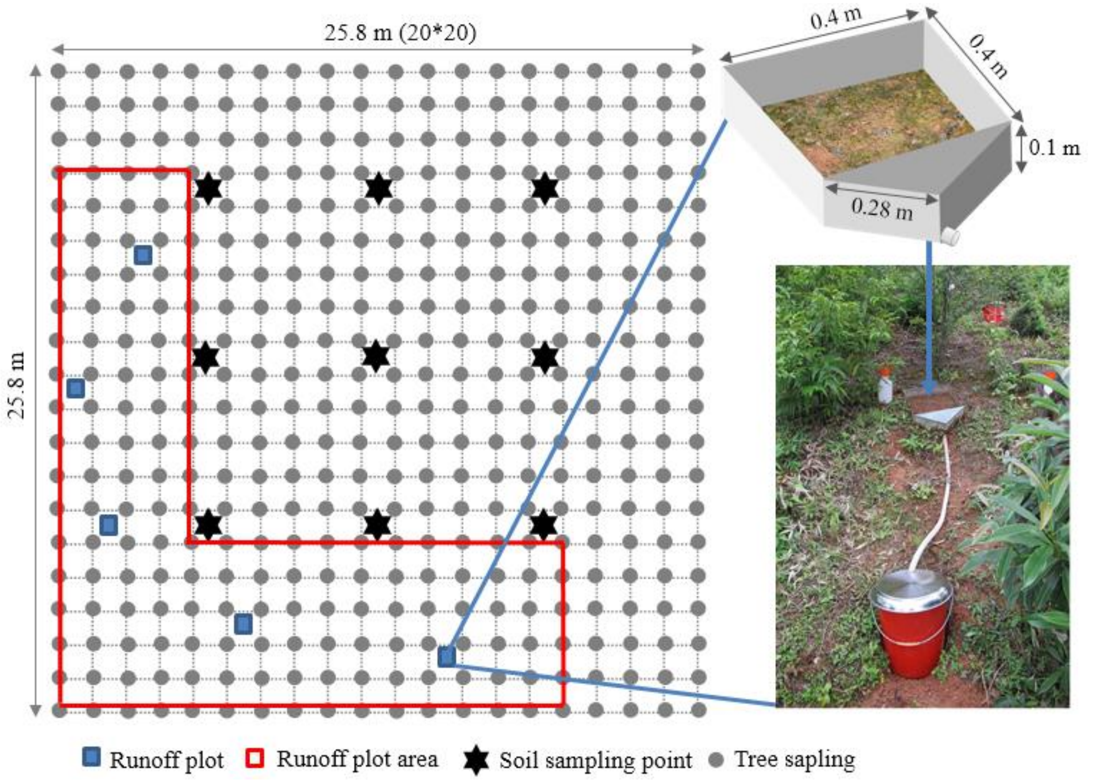


Figure 7 Random positions of runoff plots for soil erosion measurements and soil samples in one research plot of the BEF China project (0.4 m length × 0.4 m width × 0.1 m height).

Rainfall

Rainfall data during the soil erosion measurements was captured by climate stations on both sites (ecoTech data logger with Vaisala weather transmitter and ecoTech tipping bucket balance, Bonn, Germany). Daily accumulated precipitation curves and the ten largest daily rainfall events during the three years were shown in Figure 8. Further data on regional precipitation was used from the National Meteorological Information Center (NMIC) of China and China Meteorological Administration (CMA).

Tree parameters

Tree measurements with laser scanning (FARO Laser Scanner Photon 120, FARO Technologies Inc., FL, USA) at all plots started in September 2010 for both experimental sites on a yearly base, which were determined by the central 6 × 6 trees (36 trees) in the monocultures and the central 12 × 12 trees (144 trees) in the 8, 16 and 24 tree species stands (Li *et al.*, 2014a; Li *et al.*, 2017).

Crown cover and LAI were measured each May from 2013 to 2015 at the ROP scale using a fish-eye camera system (Seitz *et al.*, 2016). 552 valid pictures of forest canopy at runoff plot scale were captured.

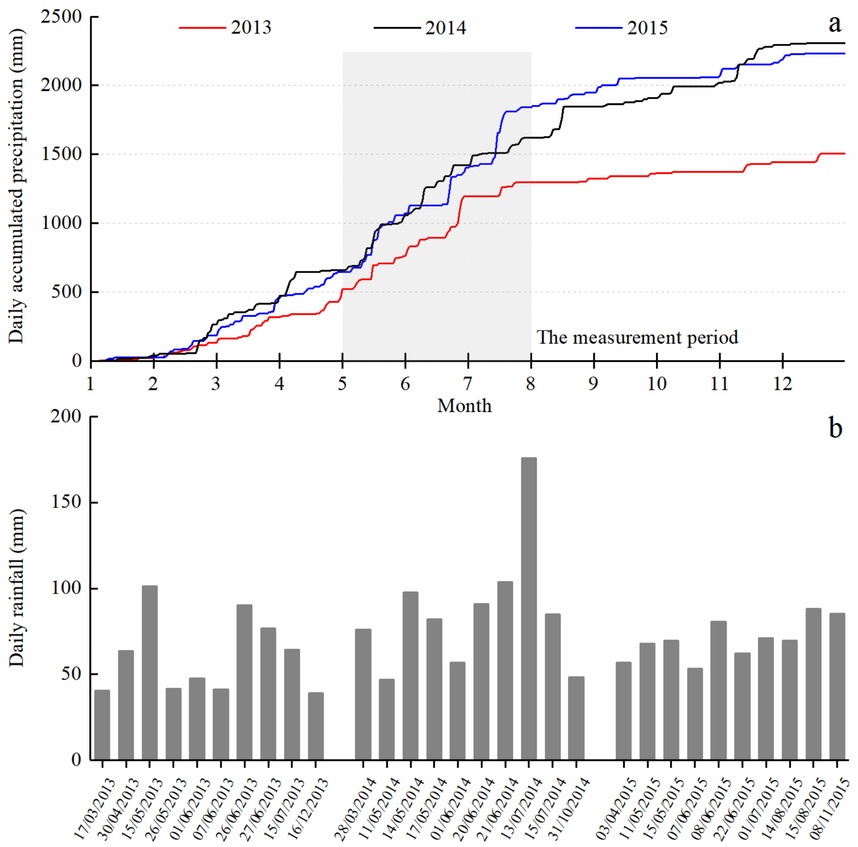


Figure 8 (a) Daily accumulated precipitation and (b) The ten largest daily rainfall events in BEF China from 2013 to 2015.

Soil surface cover and soil properties

Soil surface cover including BSCs and stone cover was surveyed yearly. BSCs were measured photogrammetrically during the rainy seasons from 2013 to 2015 within the runoff plots. Perpendicular images for each runoff plot were taken by a camera system (Canon 350D, Tokio, Japan). The images were processed by the grid quadrat method with 10 × 10 subdivisions of a digital grid in GIMP 3.0. BSCs and stone cover were separated by hue distinction. Further soil surface cover by shrubs was not present due to weeding according to the experimental design and a continuous leaf litter layer could not be recorded during the first years of this early successional afforestation.

The soil sampling was conducted in 2014. Soil cores with 6 cm in diameter were taken to a depth of 50 cm and then divided into five depth increments (0-5 cm, 5-10 cm, 10-20 cm, 20-30 cm and 30-50 cm) (Scholten et al., 2017). For this study, soil properties of the 0-5 cm increment were used (Table 2). For each plot, nine soil cores were collected (Figure 7) and mixed. Soil samples were air-dried, sieved through a 2 mm mesh, handpicked to remove plant and animal residuals and then grounded for soil analyses. For total soil carbon analyses, about 40 mg of ground sample material was weighed into tin foil and analyzed using oxidative heat combustion at 1150 °C in a helium atmosphere in a Vario EL III elemental analyzer (Elementar Analysensysteme GmbH, Hanau, Germany). Soil pH was determined with a 1 M KCl solution (soil-to-solution ratio 1:2.5) by a WTW pH meter pH 340 (WTW GmbH, Weilheim, Germany) using a Sentix 81 electrode according to DIN EN15933 (2012). Since pH is < 6.7 for all samples, total soil carbon equals SOC. In addition, five replicates of bulk density samples were obtained for each plot at the same depth increments in 2015. The soil BD was gravimetrically determined from the five replicate volumetric samples per depth increment (samples dried at 105 °C).

Data analysis

Annual sediment delivery and soil carbon and nitrogen fluxes

A rainfall threshold of 12.7 mm was applied to distinguish erosive rainfall amounts after Wischmeier and Smith (1978a). Annual erosive rainfall amount (AER, mm) and erosive rainfall during the runoff plot measurements (ERM, mm) in the rainy seasons were calculated based on precipitation curves from climate stations (Figure 8). Then, with sediment delivery acquired during the runoff plot measurements (SE, Mg ha^{-1}), an annual sediment delivery (ASD, Mg ha^{-1}) was calculated (Eq. 2). Then, with sediment C and N concentrations (SCC and SNC, %) and annual sediment delivery (ASD, Mg ha^{-1}), annual soil C and N fluxes (ASC and ASN, Mg ha^{-1}) were calculated by Eq. 3. To illustrate C and N differences between sediment and soil, the enrichment ratio (ER) of sediment C (N) concentration to soil C (N) concentration was calculated as given in Eq. 4.

$$ASD = \frac{AER}{ERM} \times SE \qquad \text{Eq.2}$$

$$ASC(N) = ASD \times SC\left(N\right)C \qquad \text{Eq.3}$$

$$ER = \frac{SC\left(N\right)C}{SoilC\left(N\right)} \qquad \text{Eq.4}$$

Statisstical analysis

For Objective 2, ANOVA and least significant difference (LSD) tests were conducted to assess temporal changes of sediment delivery. Linear mixed effects (LME) models with restricted maximum likelihood were used to detect driving factors on sediment delivery changes. Before modelling, all factors were tested on normal distribution. Sediment delivery was twice squared root transformed ($\sqrt{\sqrt{y}}$) to achieve normal distribution. Tree height, stem diameter, crown width, crown cover, LAI, BSCs, surface cover and soil properties were fitted as fixed factors, while site, runoff plots nested in plot and tree species composition were fitted as random factors. If multi-collinearity (correlation index > 0.7) was detected among the fixed factors, correlated factors were fitted individually in exchange to the counterpart.

Then, for detecting the effects of tree species richness on sediment delivery, we used linear mixed effects models to (I) analyse the temporal development of annual sediment delivery under changing tree species richness and to (II) investigate driving factors on annual sediment delivery and how in turn those factors are influenced by tree species richness. Before modelling, annual soil erosion rates from 2013 to 2015 were twice square root transformed to fit normal distribution. A first model was calculated with tree species richness, year and the interaction of tree species richness with year as fixed factors, while site, plot, runoff plot nested in plot and tree species composition were fitted as random factors. Finally, models were used to analyse the effects of tree species richness on identified main influencing factors of soil erosion by using tree species richness, year and the interaction of tree species richness with year as fixed factors, while site, plot, runoff plot nested in plot and tree species composition were used as random factors.

For Objective 3, ANOVA and LSD tests were conducted to assess temporal changes of sediment C and N concentrations and annual soil C and N fluxes as well as the effect of tree species richness. Multiple regression was used to detect significant predictors. For each multiple linear regression model, all independent variables (terrain parameters, soil properties, sediment delivery, surface cover, plant traits) were tested on normal distribution and transformed by square root when needed, and then z-scored (zero-mean normalization). Potential collinearity between independent variables was detected by the Pearson correlation coefficient. One independent variable was fitted individually in exchange to the other when their correlation coefficient was higher than |±0.7|. Dependent variables (sediment C concentrations, sediment N concentrations, annual soil C flux and annual soil N flux) were tested on normalized distribution and annual soil C flux, annual soil N flux were square root transformed. Beta value as standard regression coefficient from multiple regression models was used to illustrate the importance of independent variables on dependent variables.

All statistical analyses were performed with R 3.4.3 (R Foundation for Statistical Computing, Vienna, Austria) and SPSS 13.0 (SPSS Inc., Chicago, Illinois, USA). Graph and curve fittings were processed in Origin 8.0 (OriginLab Corporation, Northampton, USA).

2.4. Objective 4 (Soil carbon stock changes after afforestation)

Soil sampling

In this study, 132 of plots were selected for soil sampling (Table 3). Soil sampling was conducted in September and October in 2010 and 2014, respectively. Soil core with 6 cm in diameter was taken at a depth of 50 cm and then divided into five depth increments (0-5 cm, 5-10 cm, 10-20 cm, 20-30 cm and 30-50 cm) (Scholten *et al.*, 2017). For each plot, nine soil cores were collected (Figure 9) and mixed resulting in five soil samples. Soil samples were air-dried, sieved through a 2 mm mesh, handpicked to remove plant and animal residuals and then grounded for soil properties analysis. SOC and soil N was determined by a CN-analyzer (VARIO EL III, Elementar, Hanau, Germany) (Scholten *et al.*, 2017). Soil pH was determined in 1M KCl (Scholten *et al.*, 2017). In addition, five replicates of BD sample for each plot were obtained at the same depth increments as soil sampling in 2015 for soil BD determination. Soil organic carbon density (SOCD, kg m^{-2}) of five depth increments and SOC stock (0-50 cm) were calculated as given Eq. 5 and 6 (Don et al. 2009):

$$SOCD_i = T_i \times BD_i \times SOC_i \times \left(100 - C_i\right)\% \times 0.1 \qquad \text{Eq. 5}$$

$$SOC\ stock_{0-50cm} = \sum_{0}^{n} SOCD_i \qquad \text{Eq. 6}$$

SOCD represents soil organic carbon density (kg m^{-2}); i represents different five depth increments of 0-5 cm, 5-10 cm, 10-20 cm, 20-30 cm, 30-50 cm; T represents soil layer (cm); BD represents soil bulk density (g cm^{-3}); SOC represents soil organic content (%); C represents stone percentage (%).

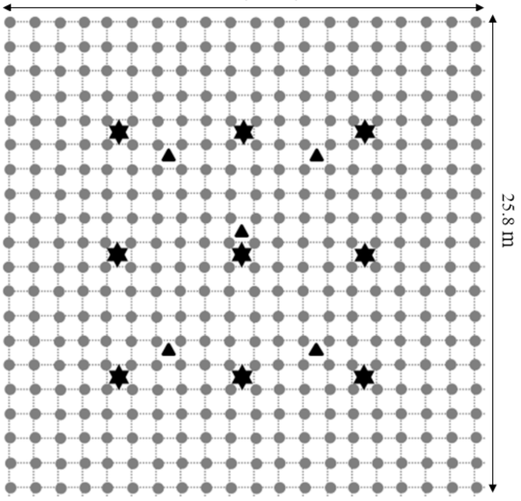


Figure 9 Positions of soil sampling for soil properties and bulk density on one plot. Grey dot means tree saplings. Black stars and triangles means the positions of soil samples (n = 9, subsamples) and bulk density (n = 5, subsamples), respectively.

Tree and litter measurement

Tree height and diameter at breast height (DBH) as two important parameters for biomass estimation were determined by the central 6 × 6 trees (36 trees) in the monocultures and 2 species plots and the central 12 × 12 trees (144 trees) in the 4, 8, 16 and 24 species mixtures (Li *et al.*, 2014a; Li *et al.*, 2017). Tree measurement of all plots started in September and October 2010 for Site A and in 2011 for Site B on a yearly base. Aboveground biomass (AGB) (kg dry mass) and belowground biomass (BGB) (kg dry mass) were calculated as given (Eq. 7 and 8) which were developed from 147 trees and 41 species in subtropical China (Xu *et al.*, 2015).

For litter measurement, 56 plots on Site A and 45 plots on Site B (Table 3) with 1, 2, 4, 8 and 16 species were selected and equipped with litter traps (Huang, 2017). Litter trap of 0.75 × 0.75 m was made of nylon nets (1 mm mesh) and fixed over a PVC frame at a height from 1 m to 1.5 m aboveground under tree canopy (Huang, 2017). For each plot, three litter traps were installed in the central area. The measurement started in March 2014 for Site A and March 2015 for Site B. Litter collection was done every month. Litter sample was put into oven and dried for 48 hours at 60 ℃ and weighed. The data was summed up for annual litter fall production. More details please see Huang (2017). Then, biomass values (AGB, BGB and litter fall) were calculated to carbon stock using carbon conversion factor of 0.47 (IPCC) (Martin and Thomas, 2011).

$$AGB = \exp\left[-2.334 + 2.118 \times \ln(D) + 0.5436 \times \ln(H) + 0.5953 \times \ln(WD)\right] \quad \text{Eq. 7}$$

$$BGB = \exp\left[-2.80346 + 2.004 \times \ln(D)\right] \quad \text{Eq. 8}$$

D represents diameter at breast height (cm), H represents tree height (m), WD represents wood density (g cm^{-3}*).*

Soil erosion

Sediment delivery was determined as described in section 2.3.

Topography

Altitude, slope, terrain ruggedness index (TRI), Monte-Carlo based flow accumulation (MCCA) were calculated (Scholten *et al.*, 2017). Moreover, 10 geomorphological units (geomorphons: flat, footslope, valley, peak, shoulder, ridge, spur, slope, pit and hollow) was computed in our study area according to the concept of openness and geomorphons (Yokoyama *et al.*, 2002; Jasiewicz and Stepinski, 2013; Scholten *et al.*, 2017). Depression, flat and valley summarized as valley while shoulder, peak and ridge summarized as ridge for further processing (Scholten *et al.*, 2017).

Table 3 Plots information of soil survey, litter collection and soil erosion measurement

		Soil samples		Soil erosion		Litter collection	
Tree species richness		Site A	Site B	Site A	Site B	Site A	Site B
Bare plot		3	2	1	1	/	/
Afforested plot	1	31	20	15	16	31	19
	2	16	15	/	/	16	15
	4	8	7	/	/	8	8
	8	4	4	2	2	4	4
	16	2	2	2	2	2	2
	24	2	2	2	2	/	/
	In all	66	52	22	23	61	48
Failed afforested		1	13	/	/	/	/

Statistical analysis

A one-sided, paired t-test was applied to determine the differences of SOCD, soil C/N ratio and soil pH between 2010 and 2014 at different soil depth increments. Before statistical analysis, normal distributions of variables were tested by Q-Q plot and SOCD$_{2010}$ and SOCD$_{2014}$ were log transformed. All the factors applied were scaled. Then, multiple regression was applied to detect the predictors of SOCD$_{2010}$, soil erosion, tree species richness, tree species, aboveground and belowground biomass, litter fall, aspect, elevation, TRI, MAAC and geomorphy on changes of SOCD$_{2014-2010}$. In the multiple regression, tree species of monocultures were set as dummy variables while geomorphy of summit ridge, spur, slope, hollow and valley in the study was set as 1, 2, 3, 4, 5 and 6, respectively. Variance inflation factors (VIFs) for each covariate in each model were calculated and lower than 3 lower (Chen *et al.*, 2017). All statistical analyses were performed with R 3.4.3 (R Foundation for Statistical Computing, Vienna, Austria) and SPSS 13.0 (SPSS Inc., Chicago, Illinois, USA). Graph and curve fittings were conducted in Origin 8.0 (OriginLab Corporation, Northampton, USA).

3. Results and discussion

3.1. Selected afforested tree species structures and their contributions to splash erosion

Spatial distribution of LAI between different tree species

In the vertical direction (Figure 10), high LAI of *Lithocarpus glaber* and *Schima superba* was mainly located at the middle-lower part of the trees (Skewness = 0.13 and -0.22, respectively) while at *Sapindus saponaria* it was mainly found at the middle-upper part (Skewness = 1.24). Lognormal equations were suitable to describe the vertical distribution LAI of *Lithocarpus glaber*, *Schima superba* (Figure 10, $R^2 > 0.9$) and *Sapindus saponaria* ($R^2 = 0.7$). For LAI radial distribution, remarkable exponential decreasing trends were observed from the tree stems to the edge of the canopy with the highest value at the stems (Figure 11, $R^2 > 0.9$).

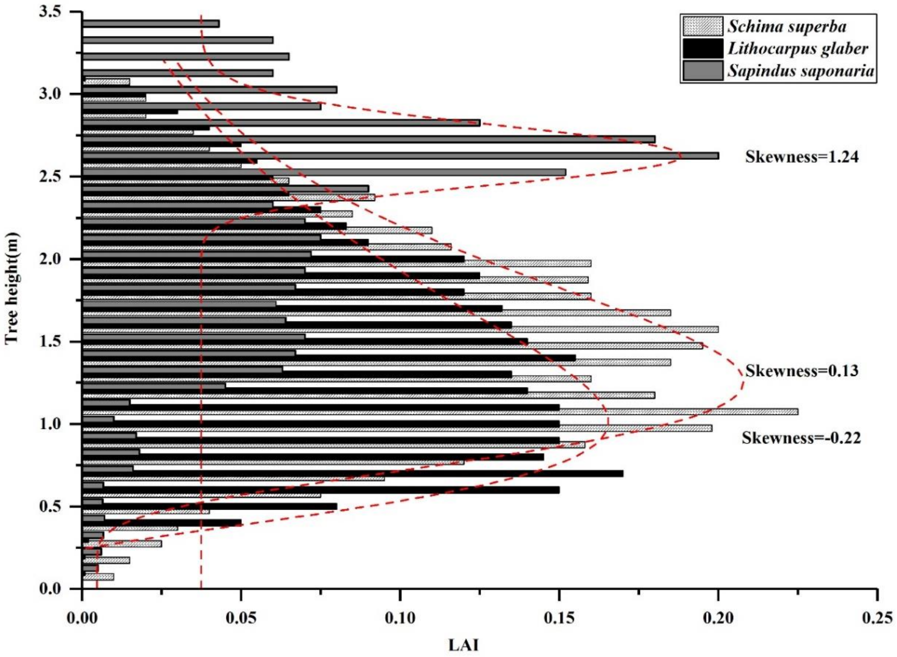
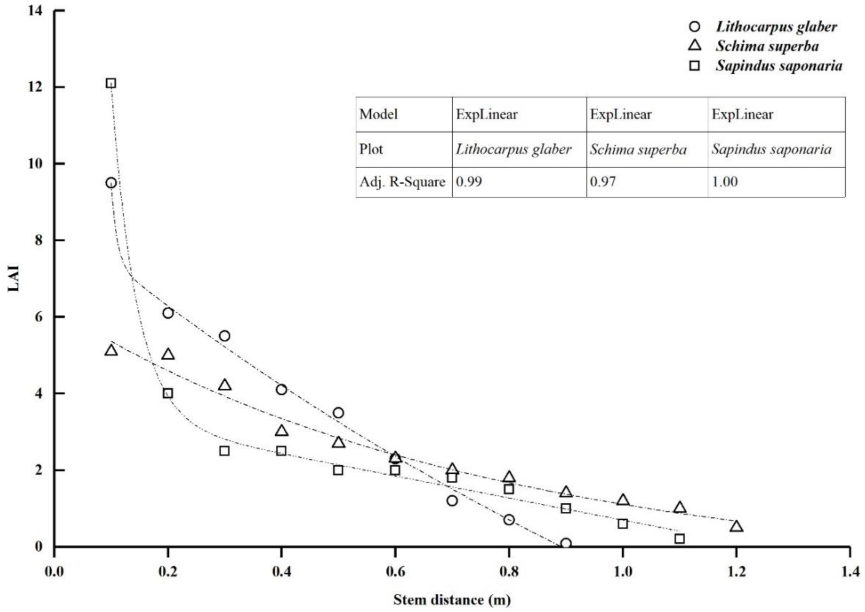


Figure 10 LAI vertical distribution pattern of three tree species.



Model	ExpLinear	ExpLinear	ExpLinear
Plot	*Lithocarpus glaber*	*Schima superba*	*Sapindus saponaria*
Adj. R-Square	0.99	0.97	1.00

Figure 11 LAI radial distribution patterns of three tree species.

Tree saplings growth and tree shape were significantly species-specific (Figure 10 and Figure 11), indicating an interspecific variation in certain functional traits in BEF China as described in detail by (Li *et al.*, 2014a). In our research, the different leaf traits of trees showed different vertical distribution patterns of LAI, while same leaf trait of trees had similar distribution patterns, as it was shown that little difference between *Lithocarpus glaber* and *Schima superba* occurred (Figure 10 and Figure 12, Table 4). Deciduous trees like *Sapindus saponaria* allocate more photosynthetic products in height and branch growth with simple crown architecture, while evergreen trees such as *Lithocarpus glaber* and *Schima superba* would consume more energy to branch construction and leaves with complex crown architecture (Chave *et al.*, 2009; Kang, 2010). Meanwhile, our results showed that a lognormal model can be used to predict the vertical distribution of LAI for broadleaved species (Figure 11). This result was in accordance with (Lu, 2011; Zhao *et al.*, 2015), who found that foliage distribution of major broadleaved species in secondary forest in northern China had the lognormal patters. In our study, LAI vertical distribution and its skewness reflected the difference in height of the first branch, number of branches and crown cover among the three species, for *Lithocarpus glaber* and *Schima superba* both having lower skewness with lower height of first branch, more branches and high crown cover comparing to *Sapindus saponaria* (Table 4). Moreover, the skewness could also account for the heterogeneity in horizontal and vertical leaf area distribution, which may provide a better way to understand the species-specific relationship between LAI and canopy water storage (Llorens and Gallart, 2000; Keim and Link, 2018) and tree diversity effects on TKE (Geißler *et al.*, 2013). Therefore, skewness of LAI vertical distribution might be a promising index comprehensively describing tree function in ecosystem, especially the process of hydrology.

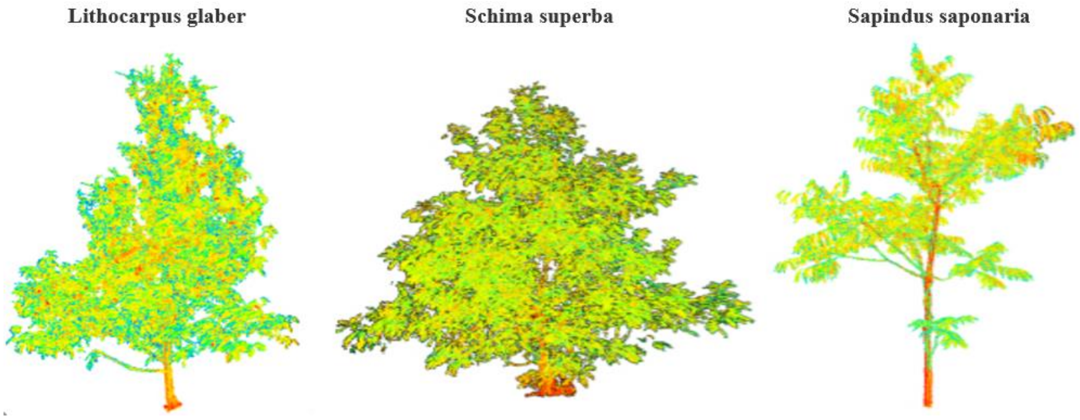


Figure 12 Three single tree species images from point cloud data measured with the laser scanner software RiScan Pro in Xingangshan, Jiangxi Province, PR China.

Table 4 Comparison of basic parameters of three investigated tree species in Xingangshan, Jiangxi Province, PR China (n = 9).

Tree species	*Lithocarpus glaber*	*Schima superba*	*Sapindus saponaria*
Leaf habit	E	E	D
Ground Diameter (m)	0.037±0.015 a	0.071±0.037 a	0.049±0.007 a
First branch height (m)	0.30±0.04 b	0.15±0.13 b	1.13±0.45 a
Tree height (m)	2.98±0.21 a	3.14±0.79 a	3.48±0.07 a
Crown Width (m)	1.70±0.23 a	2.12±0.36 a	1.91±0.28 a
Number branches	33±4 a	40±5 a	7±5 b
Crown cover	0.38±0.03 a	0.42±0.09 a	0.28±0.02 b

E represents evergreen broadleaved species; D represents deciduous broadleaved species

Different lower letters in the same tree basic parameters denote significant difference at $p < 0.05$

Potential of splash erosion under different tree species

Sapindus saponaria had the highest values of TKE among the observed species and events. Compared to TKE of open-field rainfalls, TKE was enhanced under *Sapindus saponaria* and reduced under *Lithocarpus glaber* and *Schima superba* (Figure 13). For all measured rainfall events, *Sapindus saponaria* increased by 60-80% compared to TKE in open-field, while *Lithocarpus glaber* and *Schima superba* decreased approximately 60% and 30% to 80%, respectively. With increasing radial distance from the stem, TKE of all three species was generally increasing during different rainfall events although significance of positive correlation was only detected with *Lithocarpus glaber* (Table 5).

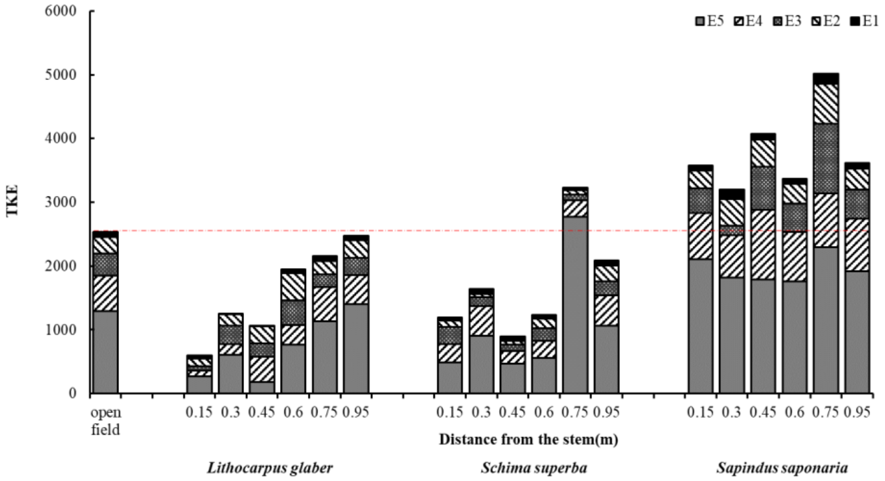


Figure 13 Throughfall kinetic energy (TKE) changes with the distances from the stem under different tree species in Xingangshan, Jiangxi Province, PR China.

Our results demonstrated that TKE was species-specific, with lower TKE of *Lithocarpus glaber* and *Schima superba* than *Sapindus saponaria*. On the one hand, it is assumed that if *Lithocarpus glaber* and *Schima superba* have higher LAI, they also show high rainfall interception. This is because canopy water storage increases with increasing LAI with a higher vertical distribution of foliage and canopy roughness (Aston, 1979; Marin *et al.*, 2000; Fleischbein *et al.*, 2005) and falling drops and drop sizes are more likely to be re-modified and split by lower parts of the canopy (Wiersum, 1985). On the other hand, for *Sapindus saponaria* and in the radial direction, high LAI is mainly located at the top height (Skewness = 1.24), which may lead to less interception and higher speed of falling drops reaching the soil surface and thus contribute to higher kinetic energy. Lognormal distribution LAI of *Lithocarpus glaber* and *Schima superba* with lower Skewness values indicated that the two species may have higher rainfall interception and lower speed of falling drops which contribute to reducing TKE.

Table 5 Pearson Correlation between distances from the stem and throughfall kinetic energy (TKE).

	Lithocarpus glaber	*Schima superba*	*Sapindus saponaria*
Event 1	0.73⁎	0.02	0.14
Event 2	0.51	0.67	0.28
Event 3	0.44	-0.18	0.46
Event 4	0.87⁎	0.23	0.25
Event 5	0.88⁎	0.49	0.10

⁎ Significant level $p < 0.05$, ⁎⁎ Significant level $p < 0.01$.

3.2. Sediment delivery development after afforestation

Temporal changes of sediment delivery

In our afforested study area, the soil erosion rate was 47.5 Mg ha^{-1} a^{-1} in 2013 and then decreased to 24.5 Mg ha^{-1} a^{-1} in 2014 and 9.6 Mg ha^{-1} a^{-1} in 2015 with the annual mean of 27.2 Mg ha^{-1} a^{-1} in the observed three years (Figure 14). Those rates are importantly higher than generally assumed for forests in the south of China (Guo et al., 2015: 1.89 Mg ha^{-1} a^{-1}). In Europe, the annual soil erosion rate under forest was given as 0.7 Mg ha^{-1} a^{-1}(Maetens *et al.*, 2012) while in Australia it ranged from 0 to 8 Mg ha^{-1} a^{-1}(Cerdan *et al.*, 2010b). From these comparisons, it can be concluded that the BEF China experiment is still suffering from severely high soil erosion even after six years of forest restoration. Considering thresholds for soil erosion rates assumed to be tolerable in general of 1 Mg ha^{-1} a^{-1}(Verheijen *et al.*, 2009) and specifically under undisturbed forest of 0.12 to 0.25 Mg ha^{-1} a^{-1} (Patric, 1976), we assume that BEF China will need five and nine more years under carefully managed forest practices, respectively to reach a tolerable soil erosion rate (Figure 14).

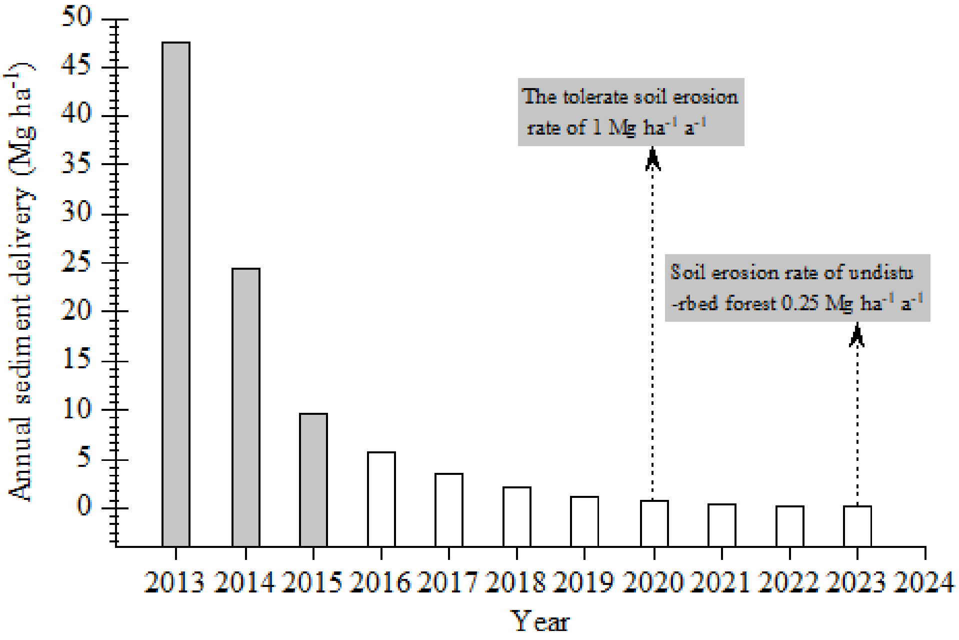


Figure 14 Annual sediment delivery in BEF China. From 2013 to 2015 (gray bars), the values were calculated from the field observation. From 2016 to 2023 (white bars), the values were calculated by 60% of one year earlier based on the ratio of the three years field observation.

Driving factors on sediment delivery changes

Regarding the linear mixed effects model (Table 6), results showed tree canopy parameters and BSCs were the two main factors driving soil erosion. Vegetation cover of the soil surface is a key control for soil erosion (Stednick, 1996; Zhou *et al.*, 2008) and afforestation is a common measure to reduce sediment delivery (Zhang and Song, 2006; Zheng *et al.*, 2008; Huang *et al.*, 2017a). This point is proved by our results, which show that bare plots consistently had higher sediment delivery than the afforested plots (Figure 15).

Table 6 Linear mixed effects models for annual soil erosion (n = 550). (LAI: leaf area index; BSCs: biological soil crusts; soil BD: soil bulk density; SOC: soil organic carbon. ddf mean denominator degree of freedom; F and P mean F-ratio and P-value of the significance test.)

Fixed effect	*ddf*	*F*	*P*	*Estimate*
LAI	340	22.49	0.000	-0.19
BSCs	523	198.28	0.000	-0.55
Soil BD	32	15.01	0.079	0.06
Soil pH	32	2.92	0.097	0.08
SOC	32	0.09	0.650	0.02
Slope	32	5.50	0.025	0.01
Altitude	31	0.08	0.228	0.04
Crown cover	400	20.675	0.011	-0.20
Tree height	61	2.021	0.001	-0.09
Crown width	61	2.034	0.001	-0.06
Stem diameter	64	9.959	0.002	-0.08

Random effects

Groups	Variance	S.D.
Plot	0.021	0.145
Tree composition	0.000	0.000
Site	0.000	0.000
Residual	0.066	0.258

Notes: Fixed effects were fitted sequentially as shown in the table while random effects are site, plot, runoff plots nested in plot and tree species composition. As multicollinearity of fixed factors (correlation index > 0.7) was detected among LAI, crown cover, tree height, crown width, and stem diameter, one factor was fitted individually in exchange to the other in the linear mixed effects model. All variables were tested on normal distribution. Annual soil erosion was twice squared root scaled while BSC was square-root transformed with arcsign reconstruction. Then all variables were scaled before modelling. Fixed effects were fitted sequentially as shown in the table while random effects are site, plot, runoff plots nested in plot and tree species composition.

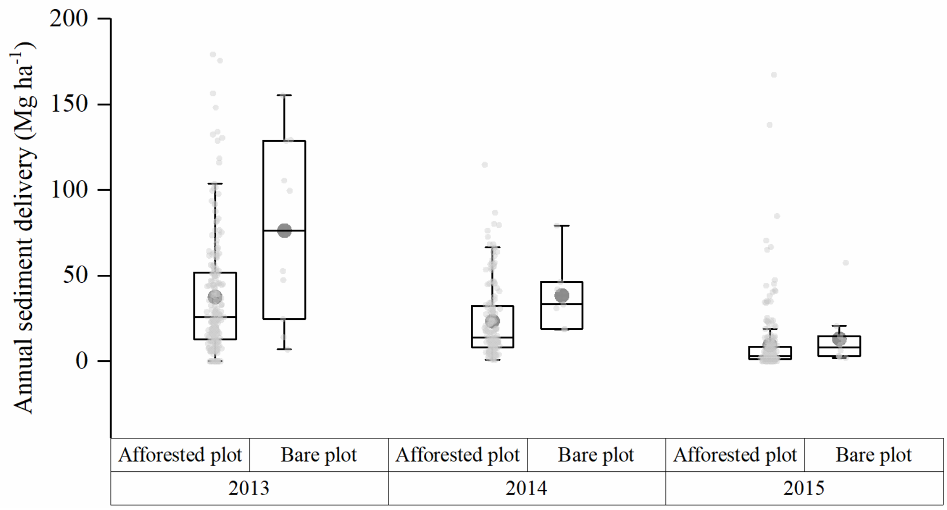


Figure 15 Boxplot of annual sediment delivery under afforested plot and bare plot in 2013, 2014 and 2015. Black dot means the mean values; Black middle line in the box means the median values; Gray dots mean measurements from ROPs. The box boundaries indicate the 75% and 25% quartiles; the whisker caps indicate the 90% and 10% quartiles.

Forest canopies influence soil erosion mainly by intercepting the rainfall (Goebes *et al.*, 2015b). Before hitting the soil surface, raindrops are modified by the forest canopy while rainfall amount is reduced by vertical distribution of foliage and canopy roughness. There is no doubt that trees with high LAI have high rainfall interception (Aston, 1979; Marin *et al.*, 2000; Fleischbein *et al.*, 2005) and several studies showed that LAI has a significant negative effect on throughfall kinetic energy and soil erosion at different forest stages in the subtropical part of China (Zhang *et al.*, 2011; Geißler *et al.*, 2013; Seitz *et al.*, 2016; Song *et al.*, 2018). These results are in line with our finding that increasing crown cover and LAI over time reduce sediment delivery (Table 6). Beside LAI, tree height is considered as an important biotic factor on soil erosion (Cao *et al.*, 2008; Geißler *et al.*, 2013; Goebes *et al.*, 2015b). Higher tree height tends to produce faster velocities of falling drops regaining high kinetic energy before reaching the soil surface and thus causing more soil erosion (Cao *et al.*, 2008; Geißler *et al.*, 2013). However, tree height was detected to negatively influence soil erosion in our research. It is assumed that positive effects from tree growth such as fast increase of LAI might overweight its negative effects on soil erosion in this early stage. This finding indicates that with ongoing tree growth vegetation parameters change and thus alter their erosion-influencing characteristics.

In addition to the crown layer, the vegetation directly covering the forest floor is of great importance for soil erosion control. In this context, BSCs were extensively occurring in our experimental areas. They use the new habitat created by deforestation and spread as pioneer vegetation in the resulting vegetation gap. These aggregations of biotic components including bacteria, fungi, mosses, lichens, algae and bryophytes in the topsoil (Schulten, 1985b; Eldridge, 1993) are closely dependent on surrounding trees and both their growth is closely linked. This study confirms findings from 2013 (Seitz *et al.*, 2017) for a now longer period and shows that BSCs are still competitive six years after tree replantation and have a significant influence on soil erosion rates in early-successional forests. BSCs absorb raindrop impacts on the soil surface (Eldridge, 1993; Eldridge and Greene, 1994), aggregate soil particles and stabilize the upper soil surface (Rodríguez-Caballero *et al.*, 2012; Gao *et al.*, 2017), reduce the surface water flow by providing high infiltration and water storage capacity (Kidron *et al.*, 1999; Gaur and Mathur, 2003; Liu and Singh, 2004; Belnap, 2006) and consequently mitigate sediment delivery (Seitz *et al.*, 2017). Our study substantiates that this is also true for mesic forest environments over several years of tree growth, where BSCs play an even more important role than LAI (Table 6). In plots without trees, 67% of the variability of sediment delivery could be explained by BSCs (Figure 16A). In addition, the nonlinear relationship between sediment delivery and BSCs cover (Figure 16A) implies that an approximately 40% coverage of BSCs will be a sufficient threshold for soil erosion control on bare land. On the other hand, sediment delivery in afforested plots with BSCs cover lower than 40% was even higher than in bareplot (Figure 16B). Further studies need to concentrate on functional mechanisms of surface-covering vegetation and how they influence sediment delivery.

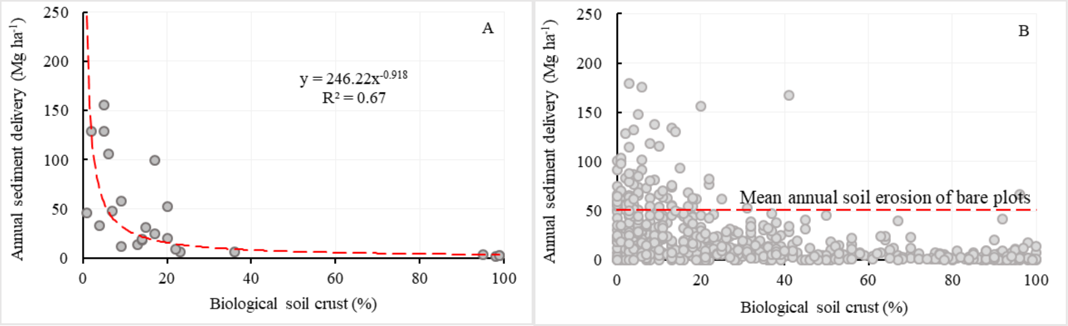


Figure 16 Relationships between annual sediment delivery and biological soil crusts (BSCs) in bare plots (A) and afforested plots (B) in BEF China.

Sediment delivery under different tree species richness

In 2013, monocultures and 24 tree species stands had similar mean annual sediment delivery, whereas 16 tree species stands showed lower rates and 8 tree species stands showed the lowest value (Figure 17). From 2014 to 2015, the mean annual sediment delivery decreased from monocultures to the 8 tree species stands, to the 16 tree species stands and finally to the 24 tree species stands. Besides, the highest and lowest annual sediment delivery measured during 2013-2015 were all detected in monocultures (Figure 17), indicating that monocultures have a high variability regarding soil erosion. From linear mixed effects model, tree species richness reduced annual sediment delivery over the observed three years significantly (Table 7).

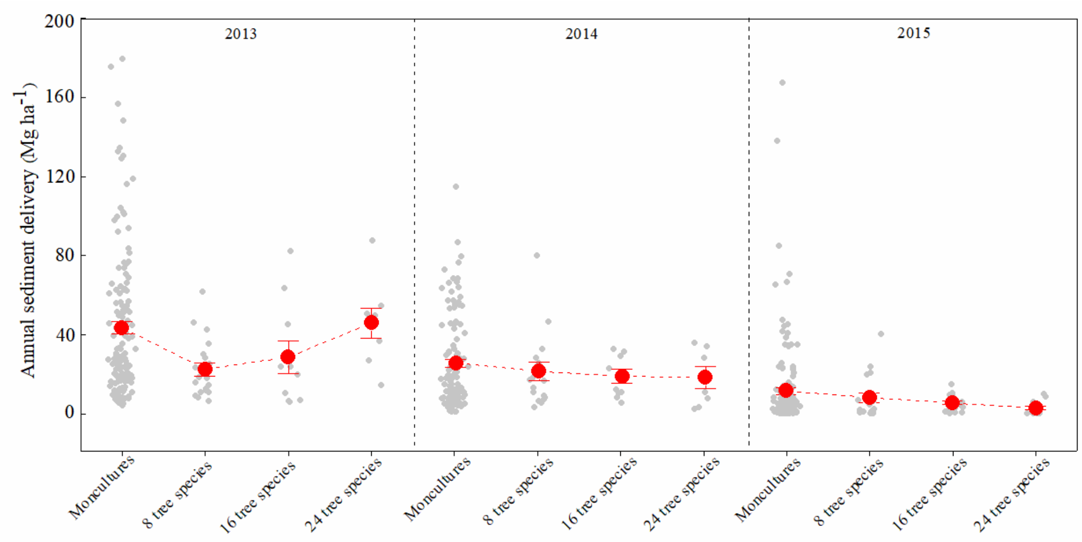


Figure 17 Annual sediment delivery (Mg ha^{-1}) in 2013, 2014 and 2015 under different tree species richness in the BEF China experiment in Xingangshan, Jiangxi Province, PR China. Gray points mean data collected from runoff plots (n = 535). Red spline dashed lines connected mean ± standard error of each tree species richness for each year.

Table 7 Linear mixed effects models for the effects of tree species richness (TSR), year and the interaction tree species richness × year on annual sediment delivery (n = 535). (Annual sediment delivery were twice squared root scaled while tree species richness and year were scaled. ddf mean denominator degree of freedom; F and P mean F-ratio and P-value of the significance test.)

Fixed effect	*ddf*	*F*	*P*
TSR	522	65.13	< 0.001
year	529	262.60	< 0.001
TSR × year	522	2.049	0.152

As one key factor on soil erosion, LAI at runoff plot scale was increasing every year from 2013 to 2015 in the research area. This increase of LAI strengthened the interception of rainfall and modified water fluxes. At the same time, tree species richness showed a significantly positive effect on LAI (Table 8 and Figure 18). Faster increases of LAI in stands of higher tree species richness resulted in higher decrease of annual soil erosion rates. In addition to this finding, previous investigations conducted in the same study area showed that tree species with different architecture and trait forms (Geißler *et al.*, 2012a; Goebes *et al.*, 2015a; Li *et al.*, 2017) such as *Sapindus* *saponaria, Lithocarpus glaber* and *Schima superba* planted in different tree species richness levels have different patterns of spatial distribution for LAI (Song *et al.*, 2018). Thus, the combination of these differing tree species planted in mixtures leading to more homogenous layering as well as the faster increase of LAI allocated from different tree species collectively optimize patterns of spatial vegetation structures and distribution (Lang *et al.*, 2012a; Lang *et al.*, 2012b; Peng *et al.*, 2016). These optimized patterns could enhance the interception of rainfall within the tree canopies, prolong the process of raindrops falling on surface soil and improve the efficiency of aboveground vegetation in reducing rainfall kinetic energy and thus leading to a faster decrease of annual soil erosion along a tree species richness gradient. Therefore, further studies on positive effects of tree species richness on LAI appear to be necessary with particular focus on underlying mechanisms within remote vegetation layers such as tree, branch and leaf traits.

Furthermore, near-surface vegetation layers such as BSC communities are of great importance for soil erosion control (Belnap and Gillette, 1997; Belnap, 2006; Belnap and Büdel, 2016). As another main factor on soil erosion in this study, BSCs showed an increasing trend in coverage with higher tree species richness, which consequently lead to a decrease in soil erosion rates (Table 8 and Figure 19). BSCs mitigate the kinetic energy of raindrop impacts on the soil surface and stabilize the upper soil surface as well as they reduce the surface water flow (Liu and Singh, 2004; Belnap, 2006; Rodríguez-Caballero *et al.*, 2012; Gao *et al.*, 2017; Seitz *et al.*, 2017; Xiao *et al.*, 2019). Further research is necessary to understand the influence of higher diversity in tree stands on the development of near-surface vegetation layers, but also on single traits within BSC community species on raindrop impacts and interrill erosion. Moreover, a positive relationship between BSCs and LAI was detected based on six years of field observations in the BEF China experiment from 2010 to 2015 (Seitz *et al.*, 2017). Higher LAI in subtropical forests might contribute to humid conditions and higher interception of light (Chang *et al.*, 1991; Yan *et al.*, 2000), which is benefit for the mosses and liverworts of BSCs (Seitz *et al.*, 2017; Zhou *et al.*, 2019). With the faster increase of LAI within stands of higher tree species richness, BSCs would be accordingly enhanced. Therefore, regarding the increase of LAI and BSCs and their correlation, we assume that forest stands with higher tree species richness will reach the tolerable soil erosion rate earlier. Thus, we recommend to consider the plantation of forest stands with higher tree diversity in this area to actively counteract soil degradation and improve ecosystem services, not only by the direct impact on the tree layer but also by influences on near-surface vegetation.

Table 8 Linear mixed-effects models for tree species richness (TSR), year and the interaction tree species richness × year on leaf area index (LAI) and biological soil crusts (BSCs) (n = 552). ddf mean denominator degree of freedom; F and P mean F-ratio and P-value of the significance test.

Fixed effect	*LAI*			*BSCs*		
	ddf	*F*	*P*	*ddf*	*F*	*P*
TSR	40	8.6	< 0.01	40	1.0	0.32
Year	529	35.6	< 0.001	535	92.5	< 0.001
TSR× year	522	12.1	< 0.01	536	1.4	0.245

Notes: Fixed effects were fitted sequentially as shown in the table while random effects are site, plot, runoff plots nested in plot and tree species composition. LAI was scaled. BSCs were square-root transformed with arcsign reconstruction and then scaled. Tree species richness and year were scaled.

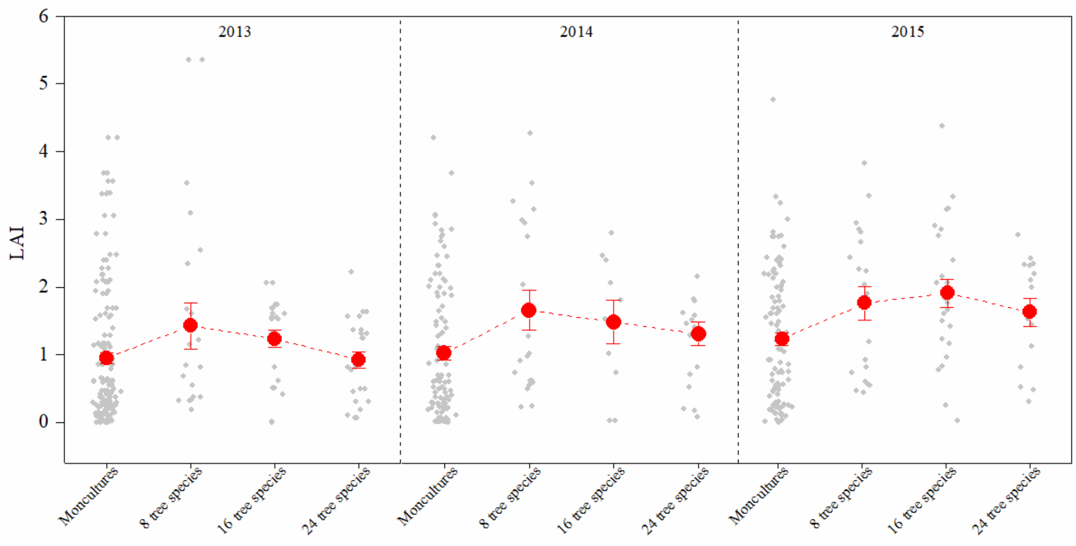


Figure 18 Leaf area index (LAI) in 2013, 2014 and 2015 under different tree species richness in the BEF China experiment in Xingangshan, Jiangxi Province, PR China. Gray points mean data collected at runoff plots scale (n = 552). Read spline dashed lines connected mean ± standard error of each tree species richness for each year.

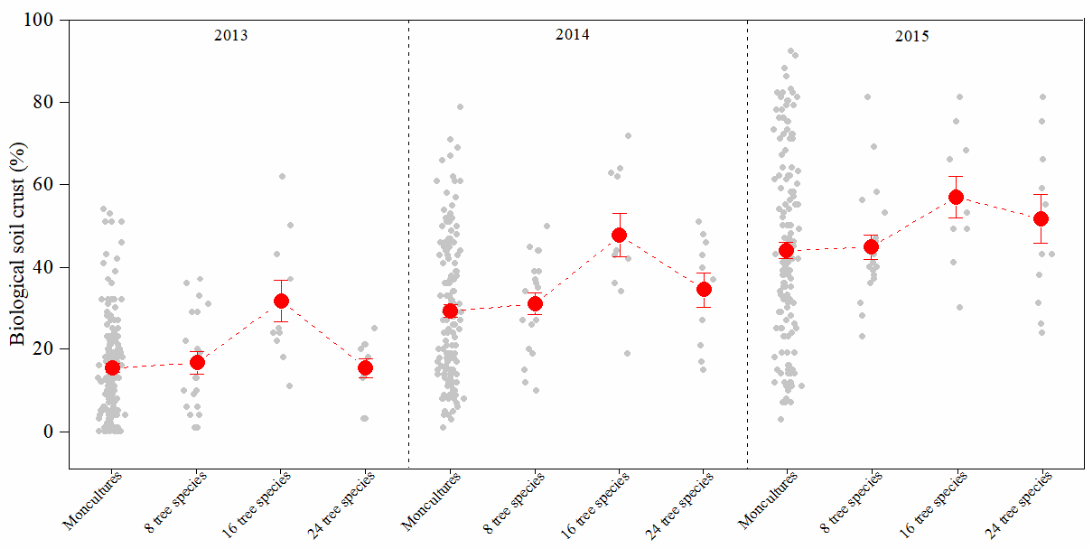


Figure 19 Biological soil crusts (BSCs) (%) within ROPs in 2013, 2014 and 2015 under different tree species richness in the BEF China experiment in Xingangshan, Jiangxi Province, PR China. Gray points mean data collected at runoff plots scale (n = 552). Read spline lines connected mean ± standard error of each tree species richness for each year.

3.3. Soil carbon and nitrogen fluxes development after afforestation

Temporal changes of soil carbon and nitrogen fluxes

Sediment C concentration increased every year ($p < 0.05$) and was significantly higher than in the topsoil over the three years ($p < 0.05$) (Figure 20 and Figure 21). The same general relations were observed for sediment N concentrations (Figure 20 and Figure 21). ER of C in sediment to soil were 1.5, 1.65 and 1.98 while for N they were 1.32, 1.73 and 2.23 in the observed three years. Topsoil C/N ratio was 13.9 and sediment C/N ratio of 2013, 2014 and 2015 were 16.4, 13.3 and 14.0, respectively. Annual soil C and N fluxes significantly decreased every year at a rate of 50% ($p < 0.05$) (Figure 22).

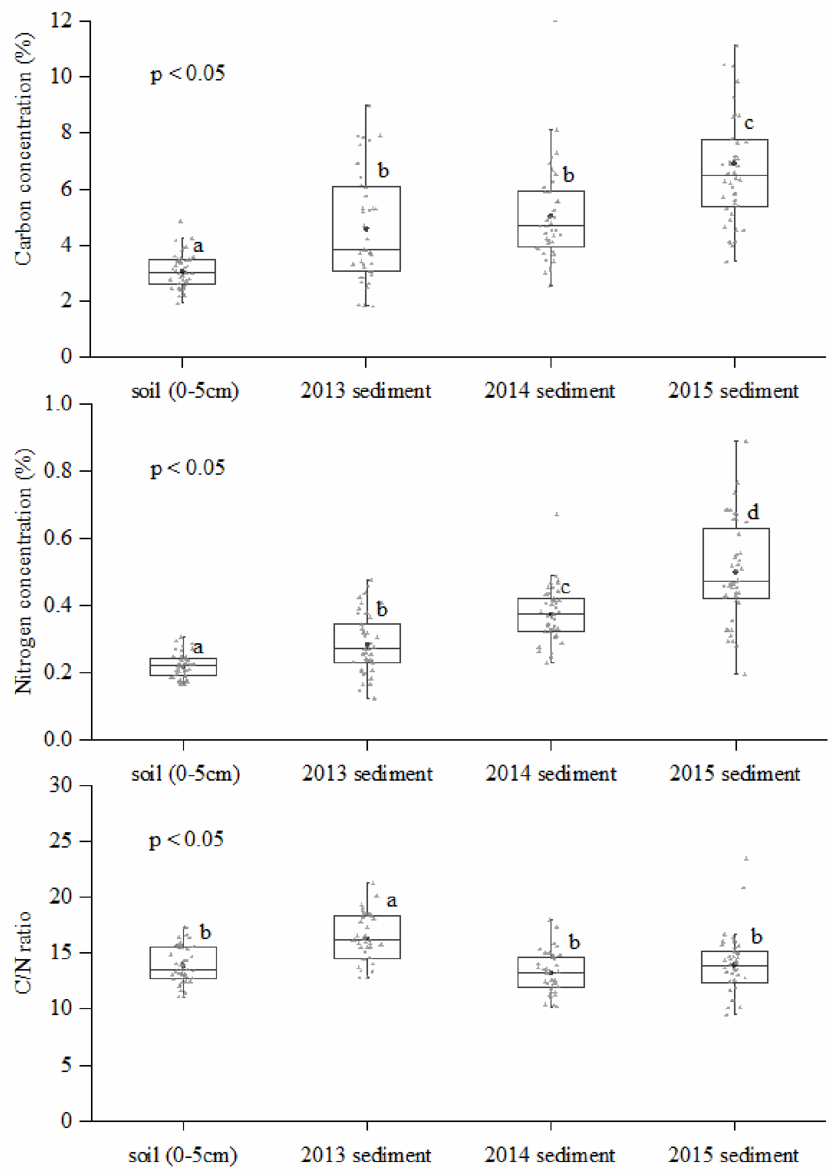


Figure 20 C and N concentration and C/N ratio of soil sampled at 0-5 cm depth in 2014 and sediment collected in 2013, 2014 and 2015 at the BEF China experiment in Xingangshan,

Jiangxi Province, PR China. Triangles represent soil C and N concentration from plots (n = 45) and sediment C and N concentration from plots (n = 45) based on 550 runoff plots measurements. Horizontal lines within boxplot represent medians and diamonds represent means. Different small letters mean significant differences at $p < 0.05$.

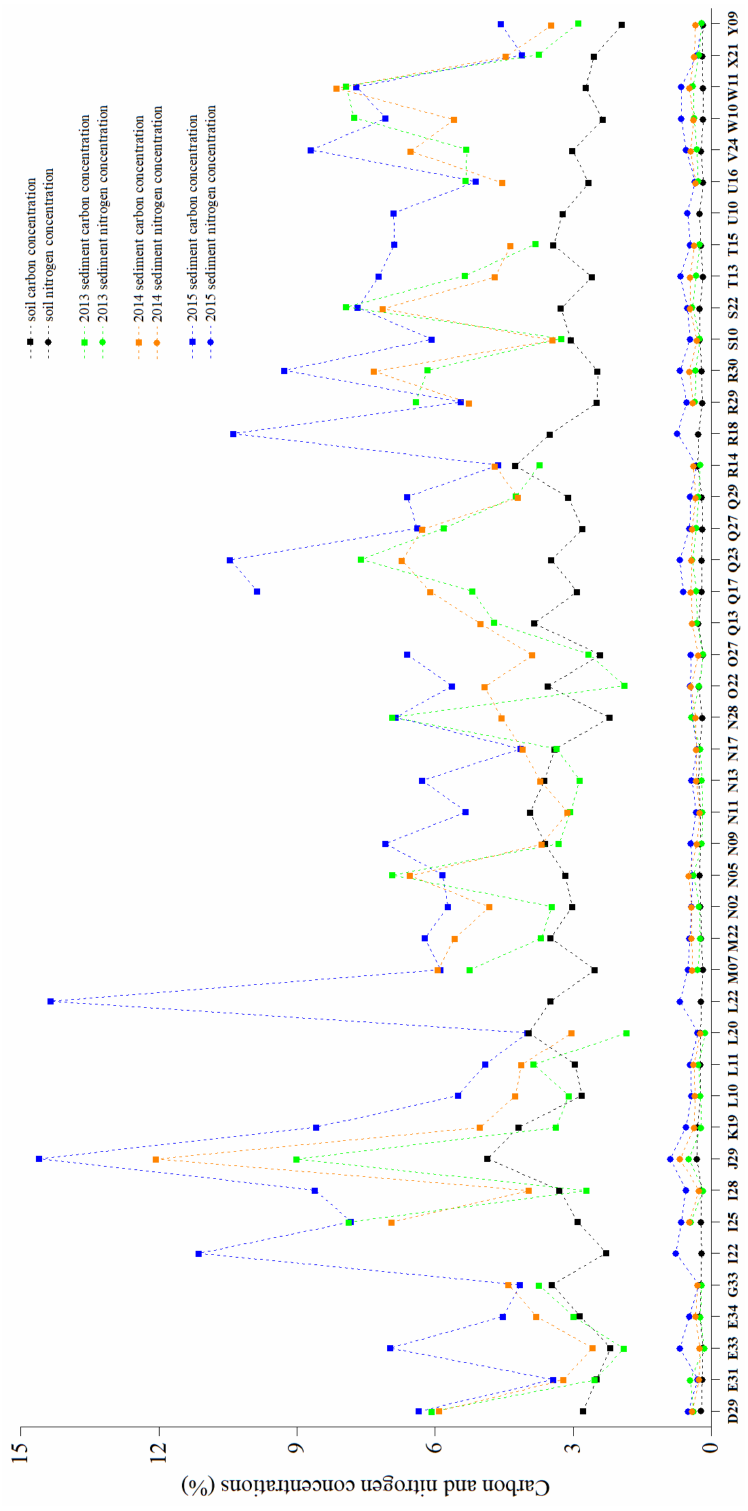


Figure 21 Means of carbon and nitrogen concentrations in soils and sediment sampled (0-5 cm) within 45 selected plots at the BEF China experiment in Xingangshan, Jiangxi Province, PR China.

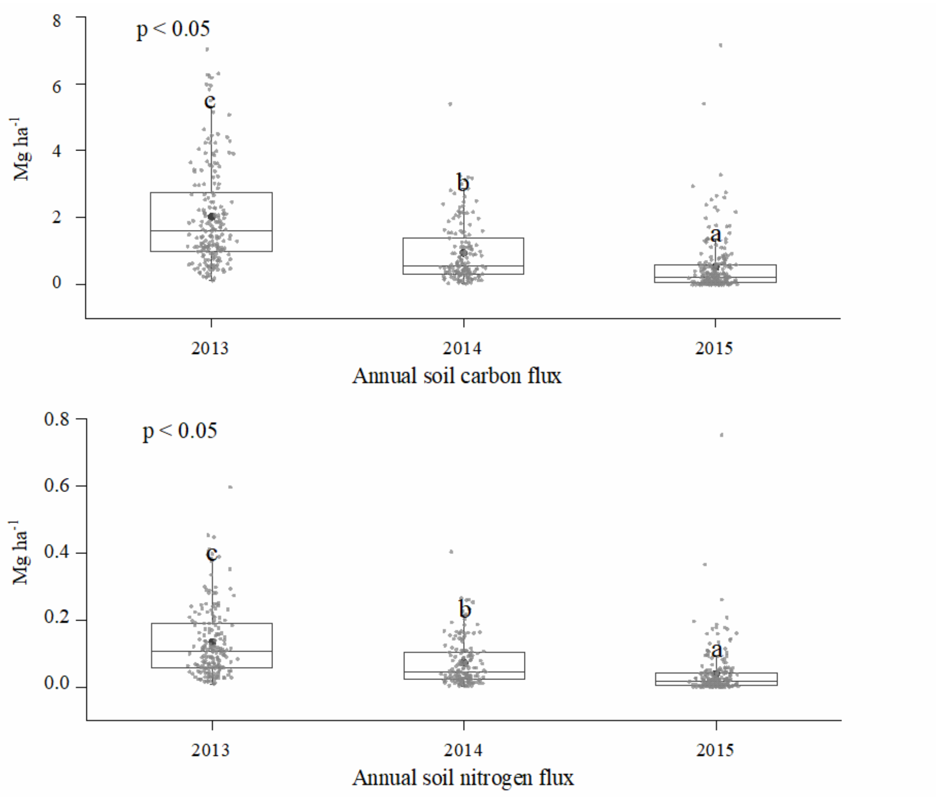


Figure 22 Annual soil C and N fluxes in 2013, 2014 and 2015 at the BEF China experiment in Xingangshan, Jiangxi Province, PR China (n = 550). Triangles represent annual soil carbon and nitrogen fluxes from runoff plots (n = 182 in 2013, n = 158 in 2014 and n = 210 in 2015). Horizontal lines within boxplot represent medians and diamonds represent means. Different small letters mean significant differences at p < 0.05.

Sediment transported by water erosion are normally enriched in C and N compared to their source soils (Wang *et al.*, 2013b). Our study confirmed these findings and showed ER of C and N in sediment to soil ranging from 1.2 to 2.0 for afforested areas. This is comparable with other land use systems which show ER varied from 1.2 to 4.0. For example, in an agricultural catchment in the Belgian Loess Belt, ER was between 1.2 and 3.0 in simulated rainfall events captured by runoff plots (Wang *et al.*, 2010). The enrichment process of C and N in eroded sediments can be attributed to the preferential removal of fine particles higher in mineral-organic complexes than coarser particles like sandy grains and micro-aggregates (Palis *et al.*, 1997; Six *et al.*, 2002; Zinn *et al.*, 2007). Another aspect is the transport of unprotected young organic material from the free and/or light fraction of organic matter in soils. This fraction is easily detached by water-induced surface erosion processes (Jacinthe *et al.*, 2004; Stacy *et al.*, 2015). Before the BEF China experiment was established and since the study area was previously covered by secondary forest, the organic horizon and topsoil is likely to contain a certain amount of unprotected organic materials with higher C/N ratio than the mineral soils (Wang *et al.*, 2014b; Stacy *et al.*, 2015; McCorkle *et al.*, 2016). Therefore, eroded sediment is not only richer in C and N but also shows a higher C/N ratio in the first year. Then, with soil erosion processes ongoing, the organic horizon depleted and more mineral soil was exposed to rainfall. Thus, eroded mineral soil was the dominate part within sediments, which resulted in C/N ratio decreasing in the following years and getting closer to topsoil ratio (Figure 20). Annual soil C and N fluxes in our study were 1.167 Mg ha^{-1} a^{-1} and 0.083 Mg ha^{-1} a^{-1} which is as much as in deforestation areas of the Canary Islands (Spain), with an annual soil C flux caused by water erosion of 1.14 Mg ha^{-1} a^{-1} (Rodrıguez *et al.*, 2004), but far higher than in forested areas (Stacy *et al.*, 2015). Severe soil C and N fluxes within BEF China in 2013 (2.03 Mg ha^{-1}) mainly resulted from high annual erosive rainfall amounts and less coverage of the soil surface (LAI and BSC) (Table 9) which caused considerable sediment delivery (Song *et al.*, 2019). Besides, soil C and N fluxes caused by water erosion accounted for approximate 24% of the 0-5 cm topsoil C (14.03 Mg ha^{-1} a^{-1}) and N (1.02 Mg ha^{-1} a^{-1}) (Li *et al.*, 2019), which occupied a considerable part of soil organic carbon stock. Therefore, the study suggests that deforestation and afforestation both should be implemented with caution as high nutrient losses and important differences between afforested areas and the undisturbed forest might occur in the earlier years, although temporal forest recovery can reduce soil C and N fluxes by controlling water erosion.

Table 9 AER, crown cover, LAI and BSC in the observed three years. (AER: annual erosive rainfall amount; LAI: leaf area index; BSCs: biological soil crusts)

Year	AER (mm)	Crown cover (%)	LAI	BSC (%)
2013	1319	47	1.04	24
2014	1885	50	1.15	36
2015	1920	62	1.45	45

Influences of topography, soil properties, surface cover and plant traits on soil carbon and nitrogen fluxes

Results from multiple linear regression models showed that topography does not play a significant role for sediment C and N concentrations as well as for annual fluxes ($p > 0.05$) (Table 10). Soil properties (C and N concentrations), surface cover (BSC and stone cover) and plant traits (diameter at breast height, crown cover, tree height, crown width, LAI) could explain 39.7% of the variability of sediment C and N concentrations. Soil C and N positively affected sediment C and N concentrations while sediment delivery showed a negative impact. BSC and plant traits had comparable positive effects on sediment C and N concentrations. Regarding annual soil C and N fluxes, 93% of the variability was explained by sediment delivery, sediment C and N concentrations, BSC and LAI.

*Table 10 Multiple linear regression of factors on sediment carbon and nitrogen concentrations and annual soil C and N fluxes. SCC: sediment carbon concentration; SNC: sediment nitrogen concentration; SC: soil carbon; SN: soil nitrogen; ASD: annual sediment delivery; BSCs: biological soil crusts; LAI: leaf area index. DBH: diameter at breast height; n.s.: no significance at p < 0.05; *: significance at p < 0.05; **: significance at p < 0.01; ***: significance at p < 0.001. /: the variable not fitted into linear regression models.*

	Sediment CN concentrations		Annual soil CN fluxes	
	C	N	C	N
Adj.R^2	0.397	0.401	0.925	0.934
Factor	Beta(Sig.)			
Slope	n.s.	n.s.	n.s.	n.s.
Altitude	n.s.	n.s.	n.s.	n.s.
SCC	/	/	0.26***	/
SNC	/	/	/	0.24***
ASD	-0.05***	-0.05***	1.02***	1.02***
SC	0.07*	/	n.s.	/
SN	/	0.08*	/	n.s.
BSC	0.18***	0.22***	-0.04***	-0.04***
Stone cover	-0.03***	-0.03***	-0.01***	-0.01***
Surface cover	0.18***	0.23***	-0.04***	-0.04***
Tree species richness	n.s.	n.s.	n.s.	n.s.
LAI	0.07***	0.05***	-0.03***	-0.03**
DBH	0.18***	0.20***	n.s.	n.s.
Crown cover	0.03***	0.03***	n.s.	n.s.
Tree height	0.10***	0.09***	n.s.	n.s.
Crown width	0.10*	0.10*	n.s.	n.s.

Topography (slope and altitude) did not play a significant role for sediment C and N concentrations and annual soil C and N fluxes. This is surprising since many studies have shown that gravity driven processes of particle movement along slopes are to a large extend a function of slope angle (Wischmeier, 1965; Martz and De Jong, 1987; Jain *et al.*, 2001; Lal, 2001; Cerdan *et al.*, 2010a; Sun *et al.*, 2014; Hancock *et al.*, 2019). One explanation is the uniform inclination ranging from 20° to 40° for all plots (Table 2). Further, the small size of our runoff plots does not allow rill formation and splash erosion is the main active process of particle detachment (Seitz, 2015). Thus, overland flow, the erosive power which is mainly controlled by slope (Wischmeier, 1965; Morgan, 2009), does transport the sediment to the collector but could not contribute significantly to erode topsoil during transport over such short transport distances of max. 0.4 m. In addition, sediment C and N concentrations was found to inversely correlate with sediment delivery, which is in accordance with other research (Lal, 1976; Owens *et al.*, 2002; Nadeu *et al.*, 2012; Wang *et al.*, 2014a; Stacy *et al.*, 2015). Given a certain slope length, more carbon-rich fine aggregates are depleted in the earlier stage of interill erosion (Lal, 1976; Polyakov and Lal, 2008; Jin *et al.*, 2009; Martínez-Mena *et* *al.*, 2012). Therefore, with the decrease of sediment delivery in BEF China every year, associated sediment C and N concentrations were increasing. Furthermore, BSC and plant traits were another two key factors on sediment C and N concentrations (Table 10). On the one hand, BSCs not only improve the labile organic carbon as they are aggregating biotic components and soil particles in the topsoil but also reduce sediment delivery (Schulten, 1985a; Eldridge, 1993; Seitz *et al.*, 2017). With increasing BSC cover in the research plots every year from 2010 to 2015 (Seitz *et al.*, 2017), once water erosion occurred and BSCs were destroyed and detached, sediment C and N concentrations would be enhanced. This also explained the increase of sediment C and N concentrations in bare plots. On the other hand, tree growth increases litter and root production in BEF China which can protect soil from splash erosion and reduce sediment delivery (Seitz *et al.*, 2015; Huang, 2017; Sun *et al.*, 2017). Hence, sediment C and N concentrations would be enhanced with sediment delivery decreasing.

Our measurements confirm that annual soil C and N fluxes in afforested areas are strongly affected by sediment delivery, which was also shown for undisturbed forest and agriculture and grassland ecosystems (Zöbisch *et al.*, 1995; Owens *et al.*, 2002; Wang *et al.*, 2013b; Stacy *et al.*, 2015). This means that almost no dilution effects could be observed during the erosive events and particulate transport is the main mechanism of the C and N fluxes during erosion. From a soil conservation perspective, the results suggest that the first years after afforestation are most important to prevent high C and N fluxes due to erosion. One possible measure is to plant shrubs and to establish BSCs in different species compositions on bare ground.

Soil carbon and nitrogen fluxes under different tree species richness

No significant effect of tree species richness but a tendency was detected reducing soil C and N fluxes (Table 10 and Figure 23). This inspired the thinking of how the effect of tree species richness is defined. As many researchers declare, it is difficult to identify the impact of plant diversity as it interacts with other plant factors and soil properties (Bezemer *et al.*, 2006; Pohl *et al.*, 2009; Shrestha *et al.*, 2010). In this study, BSC and LAI as the two significantly negative factors on soil C and N fluxes (Table 10) were detected to increase with tree species richness from 2014 (Song *et al.*, 2019), which masked the effect of tree species richness. Moreover, litter fall as a significant source of soil C and vital protection of soil surface from rainfall was reported to increase with tree species richness from 2015 (Seitz *et al.*, 2015; Huang, 2017). Considering these findings, we assume that tree species richness may reduce soil C and N fluxes in the future.

Finally, measurements of soil C and N fluxes caused by water erosion in our research area need to be adapted for a potential assessment on a regional scale due to the run-off treatment and measurement plot sizes. As we could not take the whole process of soil erosion (detachment, transport, deposition and export from the watershed) into consideration, further research is needed to accurately assess sediment export at afforested watershed scales. Furthermore, as remaining leaf litter and branches were removed from the ROP before the measurements, it has to be stated that the residuals protection on topsoil would improve erosion control by further decreasing sediment delivery and elements fluxes in ROP.

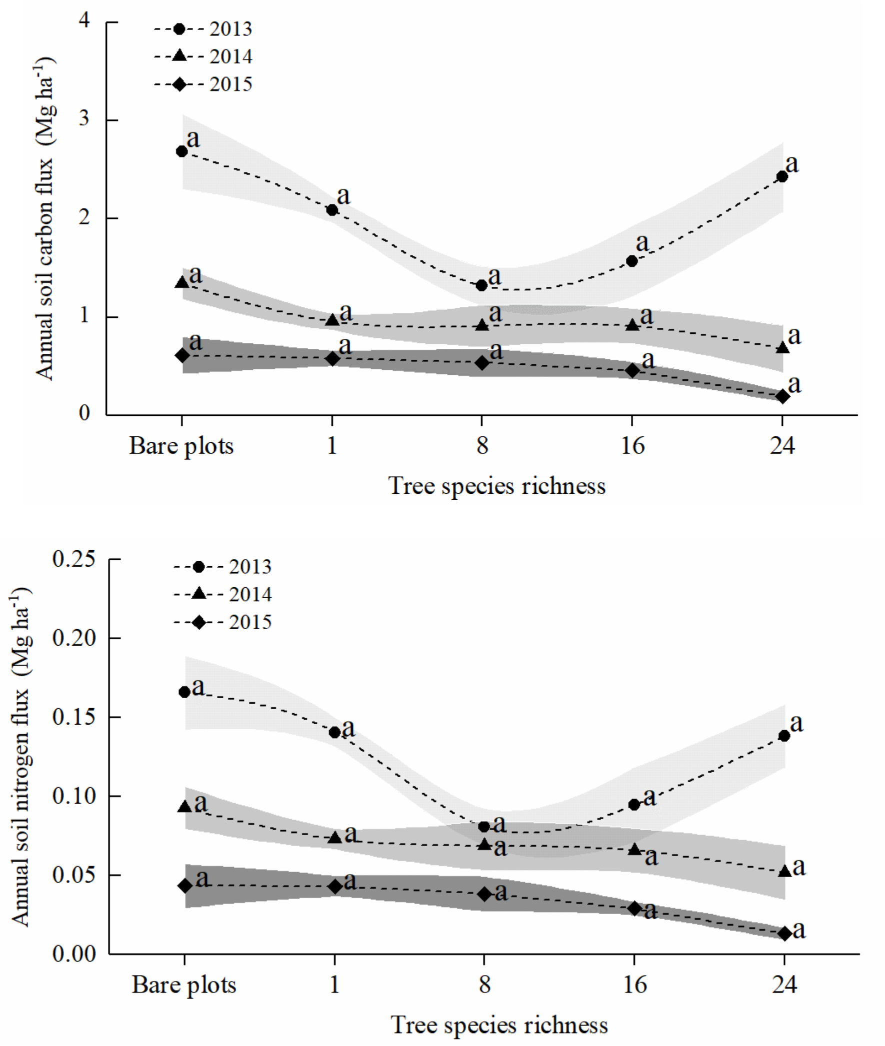


Figure 23 Annual soil carbon and nitrogen fluxes in 2013, 2014 and 2015 under different tree species richness in the BEF China experiment in Xingangshan, Jiangxi Province, PR China. Black circles, triangles and diamonds represent mean and error bars represent standard error. Spline dashed lines connect mean values of each tree species richness for each year. Different small letters mean significant differences at p < 0.05.

3.4. Soil carbon stock changes after afforestation

Changes of SOCD after five years of afforestation

A significant decrease of SOCD at topsoil depth 0-20 cm was detected across the afforested plots from 2010 to 2014 (Figure 24). Means of SOCD at 0-5 cm, 5-10 cm and 10-20 cm in 2010 and 2014 were 1.69 kg m^{-2} and 1.48 kg m^{-2}, 1.25 kg m^{-2} and1.12 kg m^{-2}, 2.02 kg m^{-2} and 1.82 kg m^{-2}, respectively. The decreasing rates of SOCD were 13%, 11%, 10% at soil depth 0-5 cm, 5-10 cm, 10-20 cm. At deeper soil depth (20-50 cm), SOCD showed no significant difference between 2010 and 2014 (Figure 24).

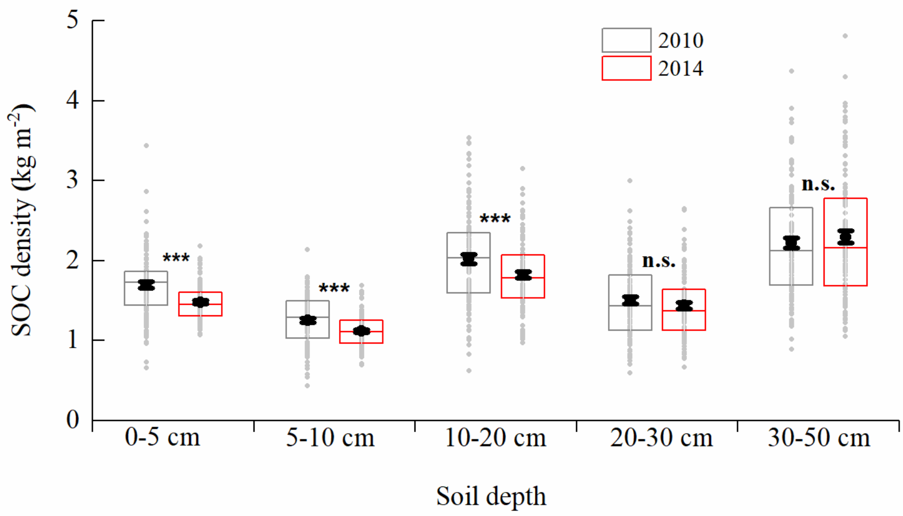


*Figure 24 SOCD at different soil depths in afforested plots in 2010 and 2014 in BEF China. Horizontal lines in boxplot represent medians and black dots represent means with standard error bars. Grey dots represent the SOCD of 113 plots and *** represent significant difference between 2010 and 2014 (paired t-tests $p < 0.001$). n.s represent no significance at $p < 0.05$.*

The changes of soil C stocks depend on the balance of C inputs and outputs (Davis and Condron, 2002). Our study showed SOCD significantly decreased in the afforested areas where were covered by secondary forest in the first five years. The result was reasonable: on the one hand, soil C decomposition was still continuing and might be accelerated during site preparation and human disturbance in the first years. On the other hand, limited C inputs from litter and fine roots due to the tree saplings were flowing into soil from in the earlier stages of afforestation (Davis and Condron, 2002; Huang, 2017; Sun *et al.*, 2017). Therefore, these processes led to the decrease of soil C stock. Additionally, in the study the decreasing rate of SOC (87 g m^{-2} a^{-1} in the 0-10 cm and 137 g m^{-2} a^{-1} in the 0-20 cm) was accordingly higher than findings from other studies (Paul *et al.*, 2002; Shi and Cui, 2010; Deng *et al.*, 2016a; Moore *et al.*, 2018). For example, in afforestation areas less than < 5 years in China soil C reduced at an average rate of 20 g m^{-2} a^{-1} at 0-20 cm soil depth based on 55 observations (Shi and Cui, 2010).

In another study, it could be shown that within the earlier ten years of a pasture converted to a tree plantation soil C at the 0-10 cm showed a decreasing rate of 60 g m^{-2} a^{-1} (Moore *et al.*, 2018). Meanwhile, at global scale, in the afforested agricultural areas less than < 5 years soil C in the < 10 cm layers generally decreased by 60.1 g m^{-2} a^{-1} based on 73 observations (Paul *et al.*, 2002). The higher decreasing rate of SOC in our research was mainly caused by denser tree plantations. In BEF China, the density of tree plantation (1.29 m × 1.29 m) implied more disturbance of soil and an accelerated decomposition of SOC during site preparation (Turner and Lambert, 2000; Guo and Gifford, 2002; Paul *et al.*, 2002; Turner *et al.*, 2005; Jandl *et al.*, 2007; Laganiere *et al.*, 2010; Tosi *et al.*, 2016). As shown in Figure 25A, bare plots without human disturbance almost kept the same amount of SOC stock as before afforestation along the whole soil profiles while failure afforested plots with soil disturbance had the decreased SOC. Moreover, tree growth could accelerate soil C mineralization for nutrients supply from soil. This point was enhanced by the finding that the afforested plots had a higher decreasing rate of SOC density than failure afforested plots (Figure 25). Besides, similar to tropical areas, subtropical areas has warm temperatures and moist soils as well as high soil microbial activity, which might facilitate high decomposition of organic matter, especially in soils without forest cover (Giongo *et al.*, 2011; Qiu *et al.*, 2015; de Araújo Filho *et al.*, 2018). Therefore, an appropriate density of tree plantation, lower soil disturbance and increased protection of soil surface should be considered during afforestation to reduce SOC depletion.

Table 11 Multiple regression analysis of key factors on SOCD changes

Factors	0-5 cm	5-10 cm	10-20 cm	20-30 cm	30-50 cm	0-50 cm
			Adj. R^2			
$SOCD_{2010}$	-0.743***	-0.625***	-0.482***	-0.424***	-0.139***	-0.285***
Soil erosion	n.s.	/	/	/	/	n.s.
TSR	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.
TS	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.
AGB	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.
BGB	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.
Litter fall	n.s.	/	/	/	/	n.s.
Aspect	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.
Elevation	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.
TRI	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.
MAAC	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.
Geomorphy	n.s.	n.s.	0.065***	0.092***	0.126***	0.050***

n.s.: no significance at $p < 0.05$; ***: significance at $p < 0.001$. $SOCD_{2010}$: soil organic carbon density of 2010. TSR: tree species richness; TS: tree species; AGB: aboveground biomass; BGB: belowground biomass; TRI: terrain ruggedness index; MCCA: Monte-Carlo based flow accumulation.

Key factors driving SOCD changes

Multiple linear regression showed that $SOCD_{2010}$ could explain 75% to 14% of the variability of SOCD changes along soil profiles ($p < 0.001$). Geomorphy had no significant effect on SOCD changes at the topsoil but subsoils. At 10-50 cm soil depth, 7% to13% of the variabilities of SOCD could be explained by geomorphy ($R^2 = 0.07_{10\text{-}20\ cm}$, $0.09_{20\text{-}30\ cm}$ and $0.13_{30\text{-}50\ cm}$, $p < 0.001$, Table 11). Tree parameters (tree species, tree species richness, AGB, BGB and litter fall), soil erosion, aspect, elevation, TRI and MAAC did not play an important role in SOCD changes. Therefore, $SOCD_{2010}$ and geomorphy were the main factors on SOCD changes at the earlier stage of afforested area in the study.

Recent studies have shown that SOC decreases in soils with high original SOC and increases in soils with lower original SOC (Garten Jr, 2002; Guo and Gifford, 2002; Paul *et al.*, 2002; Vesterdal *et al.*, 2002; Stevens and Van Wesemael, 2008; Shi and Cui, 2010; Chen *et al.*, 2017). Our results confirmed this point showing that SOC density changes in a strong negative relation with the original SOCD in 0-20 cm soil depth ($R^2 > 0.5$). One explanation might be afforestation can stimulate microbial activity and increase soil C decomposition by altering soil properties and microbial community composition (Deng *et al.*, 2016b; Pei *et al.*, 2016; Tosi *et al.*, 2016; Xu *et al.*, 2017; Hong *et al.*, 2018; Zhou *et al.*, 2018). For example, soil pH as an important index for microbial activity was found to increase after afforestation in BEF China (Figure 25). The increasing soil pH might improve total microbial biomass and the microbial activity (Pei *et al.*, 2016) and thus accelerate microbial respiration and soil C decomposition rate which led to soil C and C/N decrease. This process was illustrated in Figure 26 and Figure 27 by significant decreases of soil C/N from 2010 to 2014 and negative relationship between changes of soil C/N and $SOCD_{2010}$. In addition, the BEF China was previously covered by secondary forest and therefore in topsoil layer organic soil tended to have a high proportion of slowly decaying organic matter which mineralization rate is sensitive to temperature changes (Knorr *et al.*, 2005; Xu *et al.*, 2010; Wang *et al.*, 2013a; Li *et al.*, 2018a). Considering the elevated soil temperature after secondary forest clearance in the study area (Ma *et al.*, 2013), SOC mineralization might be accelerated and thus led to more soil C reduction. In general, topography as an important environmental factor indirectly affects SOC dynamics by soil temperature, soil moisture, soil fertility and vegetation (Raich *et al.*, 2006; Yimer *et al.*, 2006; Lybrand and Rasmussen, 2015; Tesfaye *et al.*, 2016; Tu *et al.*, 2018). In our study area, topographic heterogeneity leads to ecological gradients due to the significant relationships between geomorphological positions and soil fertility and trees survival and growth (Yang *et al.*, 2013; Scholten *et al.*, 2017). For instance, trees survival and growth increased with elevation decreasing. Therefore, hollow and valley areas with higher tree coverage might have less changes of soil temperature and moisture and then SOC was decomposed less slowly to accumulate. Moreover, our result showed topography played a stronger effect on SOC of the deeper soil layers (Table 11). This was mainly caused by: Deeper soil had less human disturbance and site preparation. And, compared with surface soils, deeper soils have a higher proportion of recalcitrant organic carbon which is sensitive to soil temperature and moisture changes caused by topography (Xu *et al.*, 2010; Wang *et al.*, 2013a).

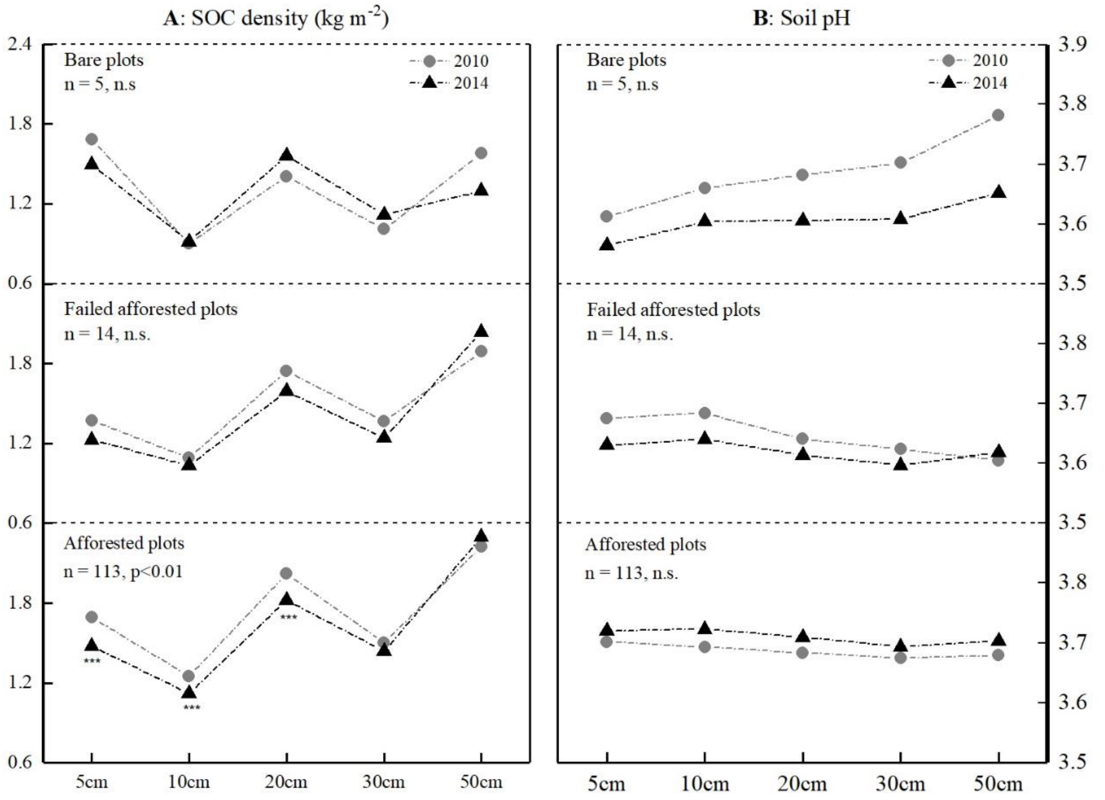


Figure 25 Means of SOC density (A) and soil pH (B) at different soil depths in bare, failed afforested and afforested plots in 2010 and 2014 BEF-China, respectively.

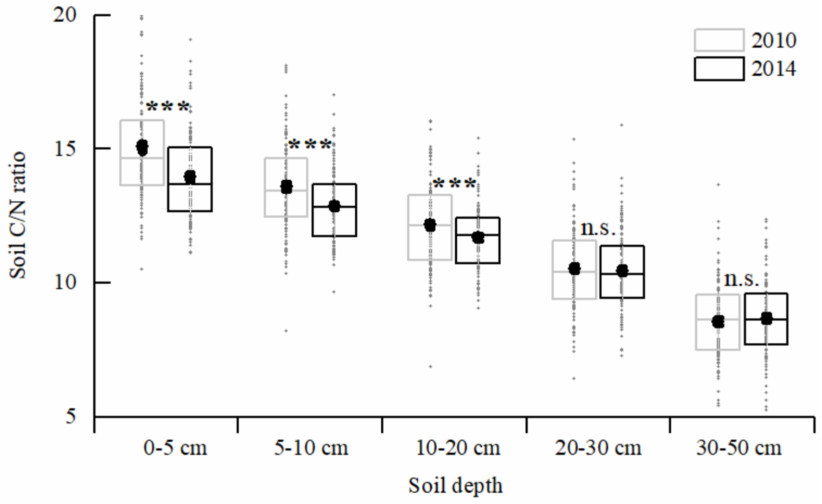


*Figure 26 Soil C/N ratio at different soil depths in afforested plots of 2010 and 2014 in BEF China. Horizontal lines in boxplot represent medians and black dots represent means with standard error bars. Grey dots represent the Soil C/N ratio of 113 plots and *** represent significant difference between 2010 and 2014 (paired t-tests p < 0.001).*

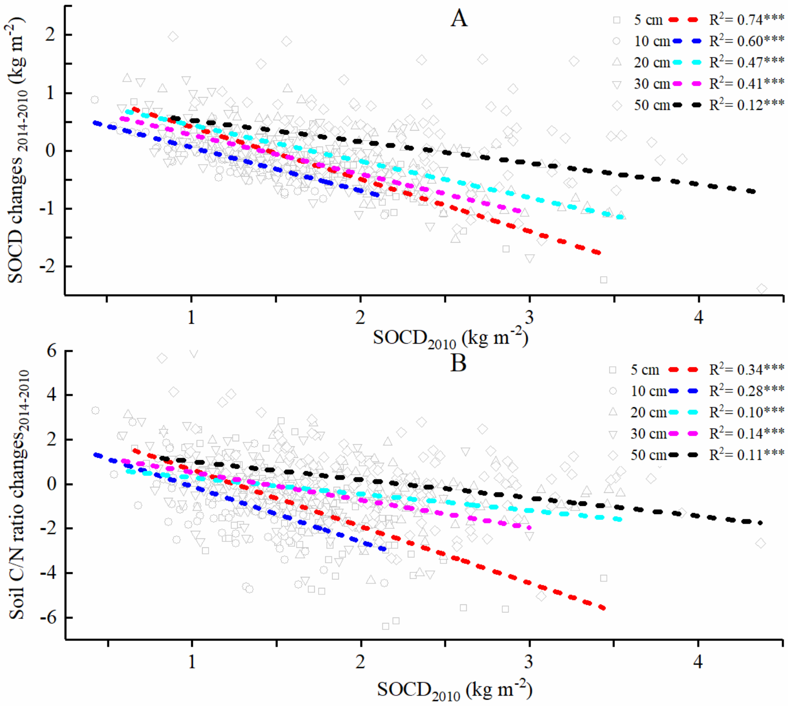


Figure 27 Relationships between SOC density changes 2014-2010 and soil C/N ratios changes 2014-2010 and 2010 SOC density in afforested plots of BEF China

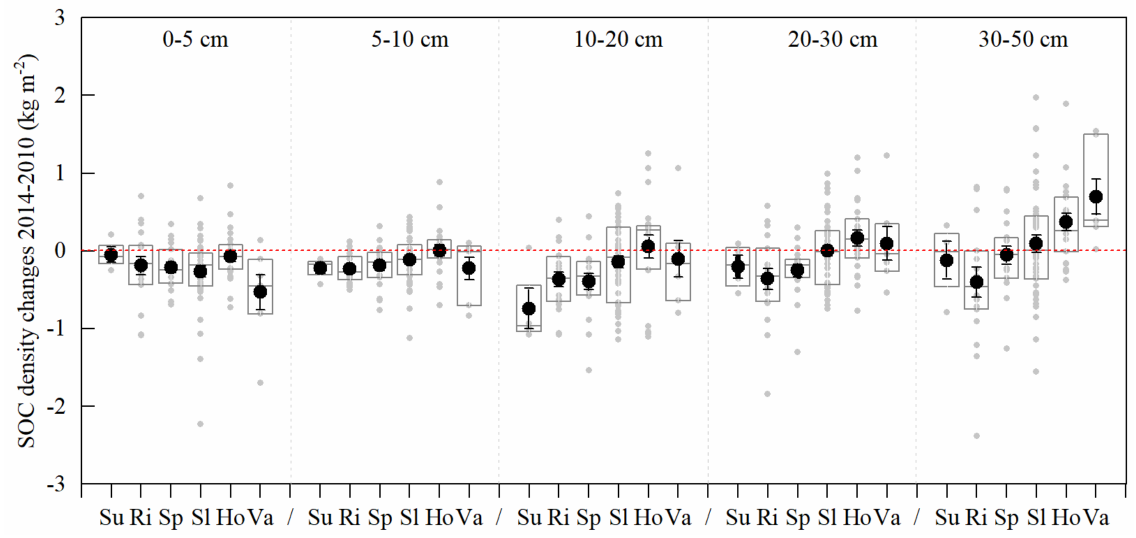


Figure 28 SOC density changes at different soil depths under six geomorphons in afforested plots of 2010 and 2014 in BEF China. Horizontal lines in boxplot represent medians and black dots represent means with standard error bars. Grey dots represent the SOC density changes of 113 plots. Su = summit (n = 4); Ri = ridge (n = 18); Sp = spur (n = 18); Sl = slope (n = 44); Ho = hollow (n = 21); Va = valley (n = 8).

Trees affect SOC mainly by C inputs from AGB and BGB such as litter and fine root (Kuzyakov and Domanski, 2000). For example, SOC in high forest productivity can be improved due to abundant C inputs from plant residence returning to soil (Dyckmans *et al.*, 2000; Kuzyakov and Domanski, 2000). Additionally, tree species and tree species richness can affect SOC by their impacts on the quality and quantity of litter production as well as on the transfer rate of litter to SOC (Vesterdal and Raulund-Rasmussen, 1998; Paul *et al.*, 2002; Huang *et al.*, 2017b). This point was proved by many studies showing that different tree species had different SOC changes in afforested area (Paul *et al.*, 2002; Laik *et al.*, 2009; Laganiere *et al.*, 2010; Shi and Cui, 2010). However, in our research no significant relationships existed between SOCD changes and tree species and tree species richness as well as forest biomass and litter fall (Table 11). It could be explained by that five years was too short for tree growth and C inputs from AGB and BGB transferring to SOC. As shown in Figure 29, limited C of AGB and BGB with a range of 0 to 2 kg C m^{-2} would flow into soil. For instance, litter fall as an important source of C input to soil only produced max. 0.3 kg C m^{-2} after five years of afforestation. Besides, tree not only allocates most of its biomass in the trunk but also has a slow turnover rate of its root biomass to soil (Cerri *et al.*, 1991; Kuzyakov and Domanski, 2000; Guo *et al.*, 2007; Laganiere *et al.*, 2010), which suggests the increased biomass C hardly contributes to SOC. Hence, in BEF China, the decrease of SOC could not be compensated by the increased C from forest biomass in the earlier stage of afforestation (Figure 29).

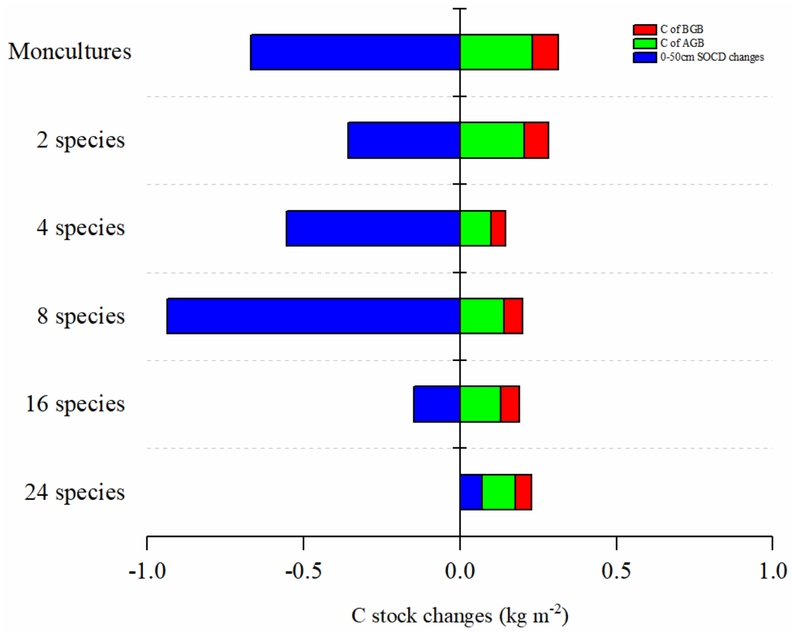


Figure 29 Carbon stocks of aboveground biomass, belowground biomass (2015) and 0-50 cm SOCD changes 2015-2010 in BEF China. C represent carbon. AGB represent aboveground biomass. BGB represent belowground biomass.

Soil erosion did not influence SOCD changes in the study. We assume this was mainly caused by our experiment settings. On the one hand, we did not take the whole process of soil erosion into consideration (splash, transport, redistribution and deposition) (Lal, 2003; Lal *et al.*, 2015; Lal, 2019). The operable measurement unit of ROPs in the experiment was 0.4 m ×0.4 m, that is small and limited for water erosion process. On the other hand, leaf litter and branches as a protective role against soil erosion (Seitz *et al.*, 2015), were removed from the ROPs, which is different from natural systems. Therefore, no certain relationship might be found when we linked sediment delivery at ROPs scale to SOC changes that were measured at plot scale (25 m × 25 m). In this respect, further research should be concentrated on water erosion influencing on SOC at the watershed scale.

4. Summary and outlook

Within the BEF China project, research was conducted to describe the influence of afforestation on soil erosion and soil carbon in a subtropical Chinese forest ecosystem:

In an early stage of afforestation, splash erosion might be accelerated because of the selected tree species. Results showed that vertical distributions of LAI and TKE of *Lithocarpus glaber*, *Schima superba* and *Sapindus saponaria* were significantly different. TKE was measured with splash cups and affected by LAI and its spatial distribution. *Sapindus saponaria* enhanced TKE while *Lithocarpus glaber and Schima superba* reduced it. Skewness of LAI vertical distribution seems to be a suitable index to comprehensively describe tree functions within hydrological systems. However, many questions are still unclear, for example, to what extent the skewness values of LAI vertical distribution have negative or positive effects on TKE under different ages of tree species not only for broadleaved species but also for needle species. Some recent measurements (Figure 30) showed that higher LAI of two needle species (*Cunninghamia lanceolata* and *Pinus massoniana*) existed in the middle and lower parts of the trees, which was similar to *Lithocarpus glaber* and *Schima superba*. Skewness of LAI vertical distribution indicated these two needle species might produce lower TKE. However, it is challenged when we consider that soil erosion is still severe, where monospecific plantations are popular in subtropical China, as we declared before. Therefore, more research should concentrate on the spatial distribution of LAI of different tree species and its related hydrological process to address underlying mechanisms of soil erosion.

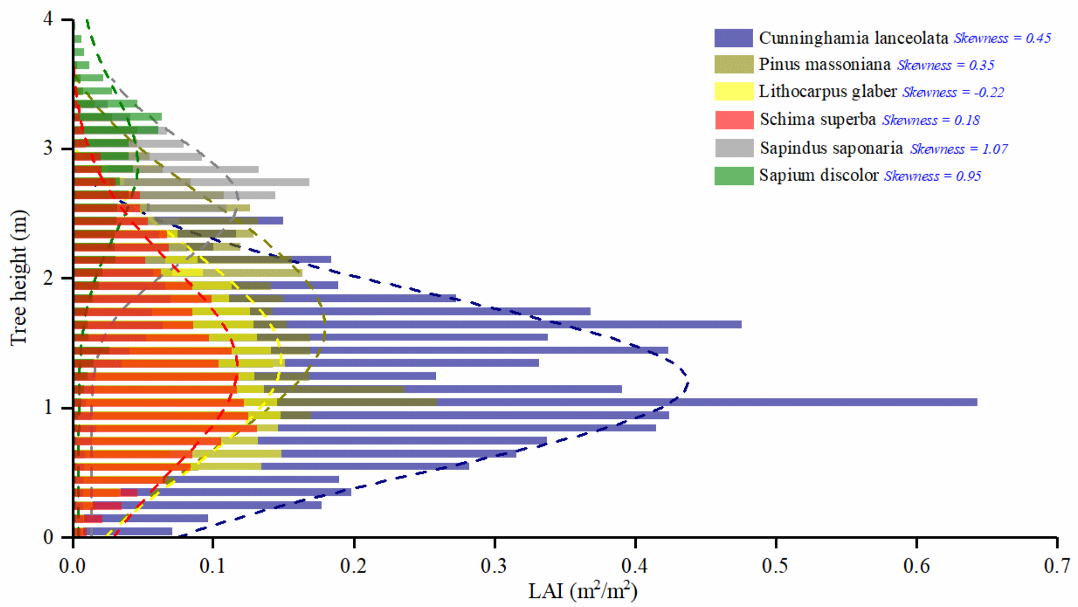


Figure 30 LAI vertical distribution pattern of six tree species.

BEF China is still suffering from severe soil erosion even after six years of tree growth. It could be shown that mean annual soil erosion rates decreased with tree species richness significantly over the observed three years. LAI and BSCs were the two main factors driving soil erosion within tree stands of different species richness. Positive effects of tree species richness on tree canopy structure and BSCs might drive the reduction of soil erosion in the earlier successional stage after afforestation of tree plantations. Therefore, the results not only highlight the importance of tree species richness on soil erosion control, but also enhance the role of LAI and BSCs and their temporal changes in the restoration of afforested ecosystems. However, research on LAI and BSCs and their temporal and spatial changes is still lacking. For example, although we found that LAI was increasing faster with higher tree species richness, the exact locations under the trees and amounts of increased LAI are still unclear (Figure 31). BSCs were abundant in every year of measurements, however, BSC dynamics, development patterns (e.g. in patches or area-wide) and species occurrence are still not known in detail (Figure 32).

Figure 31 The development of leaf area at runoff plot scale (0.4 m × 0.4 m) of Plot R30 (24 tree species richness) in 2013, 2014 and 2015 (from left to right) at the BEF China experiment.

Figure 32 The development of biological soil crusts (BSCs) in runoff plots (0.4 m × 0.4 m) of Plot N09 (24 tree species richness) in 2013, 2014 and 2015 (from left to right) at the BEF China experiment.

In BEF China, sediment C and N concentrations increased while annual soil C and N fluxes decreased with sediment delivery decreasing every year. Soil C and N fluxes in the study were as high as in deforestation areas even after six years of tree growth. Therefore, afforestation should be regarded with care, as high soil nutrient fluxes may result in the decrease of soil productivity in the earlier years. To reduce sediment delivery by increasing soil surface cover, BSC recovery is of particular interest. It is recommended as a measure to conserve soil fertility and reduce C and N transported from soil to aquatic ecosystems and the atmosphere. In addition, soil C fluxes caused by water erosion are an important and dynamic component of terrestrial carbon stocks and should be taken into consideration in the C budget of afforested area. Therefore, using the current runoff treatment and measurement plot sizes (0.4 m × 0.4 m) is not the only thinkable way, but further research is needed to accurately assess sediment and nutrient transport export in afforested areas at watershed scales, and also using DEMs with a cell size of 5 m × 5 m (Figure 33) and ArcGIS.

Afforestation significantly reduced SOC stocks in formerly deforested areas. Early-stage afforestation in BEF China resulted in a reduction of approximately 274 Mg SOC from 2010 to 2014 in total. Afforested areas with higher original SOC stock showed higher SOC losses. Tree growth and litter fall as an important carbon input to soil could not compensate SOC stock reduction in the earlier stage of afforestation. Therefore, high original SOC stocks of an area need more attention in afforestation ecosystem. Meanwhile, SOC changes within afforestation areas should be accounted for when the contribution to atmospheric CO_2 dynamics is evaluated. The results highlight that afforestation in deforested areas contribute to atmospheric carbon accumulation and the original SOC stock could be an important parameter in modelling afforested ecosystem carbon balances. Further studies should focus on how long the afforested area requires to play a role as a carbon sink as well as SOC recovery at the pre-deforested level by different models such as DNDC (Denitrification–Decomposition).

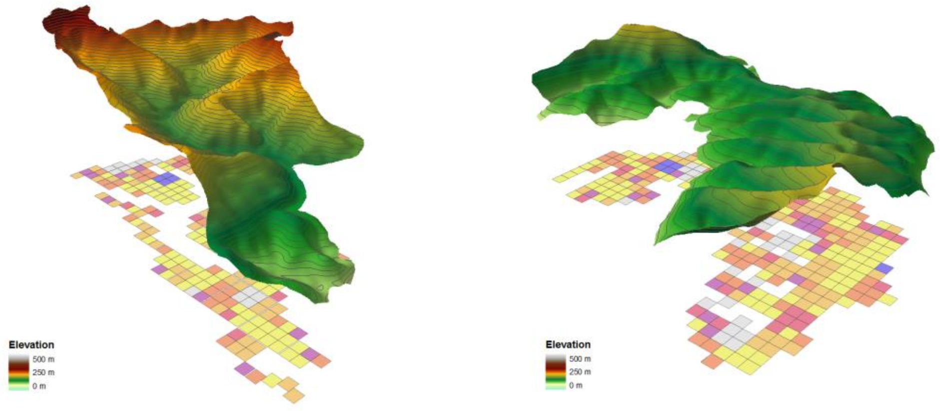


Figure 33 The elevation of Site A and Site B in the BEF China project.

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Spatial distribution of LAI and its relationship with throughfall kinetic energy of common tree species in a Chinese subtropical forest plantation.

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Abstract

The hilly red soil region in southern China is still facing serious soil erosion, even after long-term afforestation projects. This might result from structural shortcomings of the tree species chosen for afforestation. Within the Biodiversity and Ecosystem Functioning China project (BEF China), we used point cloud data from terrestrial laser scanners (TLS) and splash cups to analyze spatial leaf area index (LAI) and to predict the potential of splash erosion in subtropical forests. High LAI of *Lithocarpus glaber* and *Schima superba* was measured mainly at the middle and lower parts of the trees while for *Sapindus saponaria* it was found at the upper parts. LAI was decreasing from the tree stems to the edges of the canopy. Lognormal and exponential linear models were suitable to describe the vertical and horizontal LAI distribution of selected tree species, respectively. *Sapindus saponaria* generally had the highest values of throughfall kinetic energy (TKE) among the analyzed tree species and measured rainfall events. In the radial direction, higher LAI tended to produce lower TKE, whereas in the vertical direction, higher skewness of LAI distribution had higher TKE. LAI and its spatial distribution both were important for TKE. These findings can help to understand mechanisms of splash erosion in forest plantations related to unsuitable spatial LAI of tree species planted. It might further improve our knowledge how tree diversity may influence splash erosion by enriching the canopy layers in an early successional stage of subtropical forest plantations.

Key words: BEF China, TLS, Tree species, Splash erosion, LAI, TKE

1. Introduction

Soil erosion is a serious environmental hazard of global scale (Lal, 2003) and vegetation cover of the soil surface is one key factor in controlling soil erosion (Stednick, 1996; Cao *et al.*, 2008; Shi *et al.*, 2009; Chen *et al.*, 2011; Filoso *et al.*, 2017; Feng *et al.*, 2018). Forest vegetation cover affects splash erosion at the soil surface by intercepting rainfall and thus modifying rain patters, such as adapted drop size and speed, changing rainfall amount and spatial distribution (Nanko *et al.*, 2006; Geißler *et al.*, 2012b; Geißler *et al.*, 2013; Goebes *et al.*, 2015b). It is generally accepted that soil erosion is reduced under forests (Smith, 1914). However, research showed that high sediment delivery often occurred in forested catchments in subtropical regions (Marks, 1998; Molnar, 2004; Zhao, 2006). One reason is that effects of forest cover on splash erosion are dynamic in space as the structures of tree species differ. Hence, calculating an index that describes the ability of cover plants, especially trees, is essential to analyze the splash erosion risk under forest and can help to better understand the relationship between cover plants and splash erosion. Such an index can also serve in planning and management of afforestation as part of soil and water conservation approaches, e.g. in the hilly red soil region in southern China.

One well established index that describes the plant cover is the leaf area index (LAI, (Jordan, 1969)). It is defined as projected leaf area per unit ground area (Gower and Norman, 1991). As an important biophysical parameter, LAI is often used in quantitative analyses of processes related to vegetation dynamics such as rainfall interception (Maass *et al.*, 1995), soil erosion modeling (Laflen *et al.*, 1997; Zhou *et al.*, 2008; Zhang *et al.*, 2014), land surface process models (Chen *et al.*, 2011; Tesemma *et al.*, 2015) and global climate change (Claverie *et al.*, 2016). In the subtropical part of China, studies showed that LAI has a significant effect on throughfall kinetic energy (TKE) in secondary forest (Geißler *et al.*, 2012a), soil loss in 30-year afforestation (Sun *et al.*, 2010; Zhang *et al.*, 2011) and sediment discharge and TKE in young afforestation (Goebes *et al.*, 2015a; Seitz *et al.*, 2016). Further vegetation factors that are correlated with TKE in forests are crown cover, leaf traits, tree height and branch architecture (Cao *et al.*, 2008; Geißler *et al.*, 2010; Geißler *et al.*, 2012b; Goebes *et al.*, 2015a; Goebes *et al.*, 2015b). Another important aspect is, that the process of free raindrops passing the tree canopies is dynamic (Nanko *et al.*, 2006) and the canopy architecture can change the drop size and spatial distribution significantly at different positions and height of the tree canopy (Nanko *et al.*, 2006; Goebes *et al.*, 2015b). Hence, the relationship between general LAI values and splash erosion is questionable since it neglects the effects of spatial distributions of LAI. Also, most studies concentrate on mature forests (Cao *et al.*, 2008; Geißler *et al.*, 2013). Regarding afforestation measures on heavily eroded soils with a low structure stability and without shrubs or litter cover, like in the hilly red soil region in southern China (Zhao, 2006; Shi *et al.*, 2009), the role of forests in their early stage of tree growth to protect the soil from erosion is of ample interest. Such research is still scarce.

Generally, there are several methods to estimate LAI, such as determination from satellite images (Knyazikhin *et al.*, 1998; Deng *et al.*, 2006; Xiao *et al.*, 2014), instrumental measurements (Fassnacht *et al.*, 1994; Chen *et al.*, 1997; Gower *et al.*, 1999), and direct acquisition from destructive measures (Gower *et al.*, 1999; Nanko *et al.*, 2006). As high-resolution, non-destructive and efficient tool, terrestrial laser scanners (TLS) are increasingly applied in forest inventory for reliable three-dimensional (3D) data acquisition and comparison (Clawges *et al.*, 2007; Maas *et al.*, 2008; Fleck *et al.*, 2011; Lovell *et al.*, 2011), especially for indexes of difficult acquisition using traditional methods (Moorthy *et al.*, 2011; Li *et al.*, 2014b). In addition, TLS is a useful tool for the retrieval of LAI (Hosoi and Omasa, 2006; Moorthy *et al.*, 2008; Zheng and Moskal, 2012).

Using TLS to measure LAI also allows to precisely calculate the radial and vertical distribution of LAI for individual trees over the whole crown area.

The objectives of this study were (1) to assess the differences in the spatial distribution of LAI among different common tree species and along the tree stem and height of single species and (2) to explore the relationship between spatial distribution of LAI and TKE for different rainfall events.

2. Materials and Methods

2.1 Study area

The field experiment was conducted in the context of the joint Sino–German–Swiss Research Unit “BEF China” (BEF, biodiversity and ecosystem functioning) (Bruelheide *et al.*, 2011). The BEF China project is located in Xingangshan Town, Dexing City, Jiangxi Province, PR China (29.08°–29.11° N, 117.90°–117.93° E). The climate of the area is dominated by subtropical monsoon, with mean annual temperature of 17.4 ℃ and mean annual precipitation of 1821 mm (Yang *et al.*, 2013). The subtropical summer monsoon starts from May to July (Goebes *et al.*, 2015a; Seitz *et al.*, 2016). The area is hilly with mean elevations of 189 m a.s.l. (site A) and 137 m a.s.l. (site B) (Scholten *et al.*, 2017). Soils in the region are mainly Cambisols, Acrisols and Ferralsols (Scholten *et al.*, 2017). The BEF China project is a forest experiment on approximately 50 ha and includes two parallel sites, A and B, planted in 2009 and 2010, respectively. These two sites were established by transplanting seedlings of 40 local trees and shrubs after logging of the original secondary forest (Bruelheide *et al.*, 2014b).

2.2 Sample selection and data collection

2.2.1 Tree parameters retrieval

In this study, three subtropical tree species were selected, including evergreen broadleaved species (*Lithocarpus glaber* and *Schima superba*) and a deciduous broadleaved species (*Sapindus saponaria)*. These three species are recommended species for afforestation projects regarding water and soil conservation in the subtropical region of China (The Ministry of Water Resources, 2013). For each tree species, three tree individuals were randomly selected. LAI measurements were carried out in October 2013 and point cloud data for each tree was obtained using a Terrestrial Laser Scanner (RIEGL VZ-400, Horn, Austria). For each tree, 3 to 5 measurement positions were set at different directions with a horizontal distance ranging from 1.5 to 8 m. The view zenith angle from the center of the scanner to the canopy was set to 60 degrees. Before the measurement, high reflectance sheets were stuck on pegs around the trees at various distances, heights and directions, to guarantee that more than 6 common sheets were scanned for each two adjacent stations, which provided reference points to convert all data in the same coordinate. The scanning angle resolution of the TLS was 0.01° and measurement rate was 122000 points s^{-1}.

2.2.2 TKE measurements

TKE was measured using Tübingen Splash Cups (T-Cup, (Scholten *et al.*, 2011)). The cup has a diameter of 4.6 cm and a height of 4 cm (Scholten *et al.*, 2011). It is filled with uniform fine sand (0.125 mm). The detached sand is calculated by the weight difference between the dry sand in the filled-up splash cup before measurements and the dry sand inside the cup after the rainfall event. Then kinetic energy of rainfall (KE_{rf}) is calculated by the detached sand (*ds*) per splash cup (*sc*) using the equation (Eq.1) below with a modified slope and standardization to 1 m^2 (Goebes *et al.*, 2015b).

$$KE_{rf}(\mathrm{J\ m^{-2}}) = ds_{sc}(\mathrm{g}) \times 0.1455 \times \{1000(\mathrm{cm}^2)/\pi r_{sc}^2\} \qquad \text{Eq.1}$$

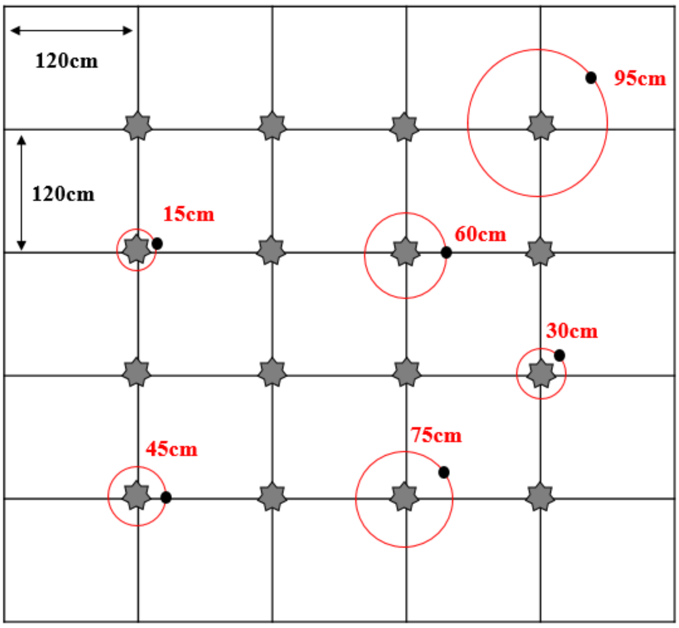


Fig. 1. Splash cup measurement design with six positions according to Goebes *et al.* (2015b). Gray stars, black dots and red circle lines represent tree individuals, splash cup position and radius around tree stems, respectively.

Its application was approved in field studies in subtropical China (Geißler *et al.*, 2012a; Geißler *et al.*, 2012b; Goebes *et al.*, 2015b). Five monoculture plots of *Lithocarpus glaber* (1 plot), *Schima superba* (2 plots) and *Sapindus saponaria* (2 plots) were selected to install splash cups under different tree individuals using the design of (Goebes *et al.*, 2015b). The cup positions were 15 cm, 30 cm, 45 cm, 60 cm, 75 cm, and 95 cm from the stem respectively (Fig 1, six splash cups per plot). Five rainfall events from May to

July in 2013 were measured (Table. 1). In total, data from 150 splash cups were collected.

Table 1 Characteristics of the five captured rainfall events (Goebes *et al.*, 2015b).

Rainfall events	Rainfall amount (mm)	Rainfall duration (h)	Mean throughfall amount (mm)	TKE (J/m2)
Event 1	6.6	2.33	5.0	78.8
Event 2	23.3	10.16	28.3	255.3
Event 3	39.3	11.5	47.9	354.7
Event 4	61.2	14.5	73.8	553.7
Event 5	185.7	30.58	192.7	1292.8

2.3 Data analysis

LAI was estimated using a volume element model from the point cloud data (Hosoi and Omasa, 2006; Zheng and Moskal, 2012) and was calculated with Matlab 2010b (The Mathworks Inc., Natick, MA, USA). Basic parameters of trees (ground diameter, tree height, first branch height, crown width, branch number and crown cover) were measured with the laser scanner software RiScan Pro (http://www.riegl.com). A one-way analysis of variance (ANOVA) was conducted to compare the mean value of canopy structure parameters. Skewness of LAI vertical distribution and Pearson correlation analyses to test LAI effects on TKE were conducted with IBM SPSS Statistics for Windows Version 19.0 (IBM Corp., Armonk, NY, USA). Before the Pearson correlation analyses, TKE was log10 transformed to normal distribution and tested with the Kolmogorov–Smirnov test (Significance = 0.2). Graphs and curve fittings were processed in Origin 8.0 (Origin Lab Corporation, Northampton, MA, USA)

3. Results

3.1 Spatial distribution of LAI between different tree species

In the vertical direction (Fig 2), high LAI of *Lithocarpus glaber* and *Schima superba* was mainly located at the middle-lower part of the trees (Skewness = 0.13 and -0.22, respectively) while at *Sapindus saponaria* it was mainly found at the middle-upper part (Skewness = 1.24). LAI of *Schima superba* and *Lithocarpus glaber* ranged from 0.05 to 0.15 at a tree height from 0.5 to 2 m, while *Sapindus saponaria* ranged from 0.02 to 0.2 at a tree height from 2 to 3 m. Lognormal equations were suitable to describe the vertical distribution LAI of *Lithocarpus glaber*, *Schima superba* (Fig 2, $R^2 > 0.9$) and *Sapindus saponaria* ($R^2 = 0.7$).

For LAI radial distribution, remarkable exponential decreasing trends were observed from the tree stems to the edge of the canopy with the highest value at the stems (Fig 3, $R^2 > 0.9$). The order of the stem LAI value was ranked as *Sapindus saponaria* > *Lithocarpus glaber* > *Schima superba*. From 0.1 m to 0.3 m along stem distance, LAI of *Sapindus saponaria* decreased drastically.

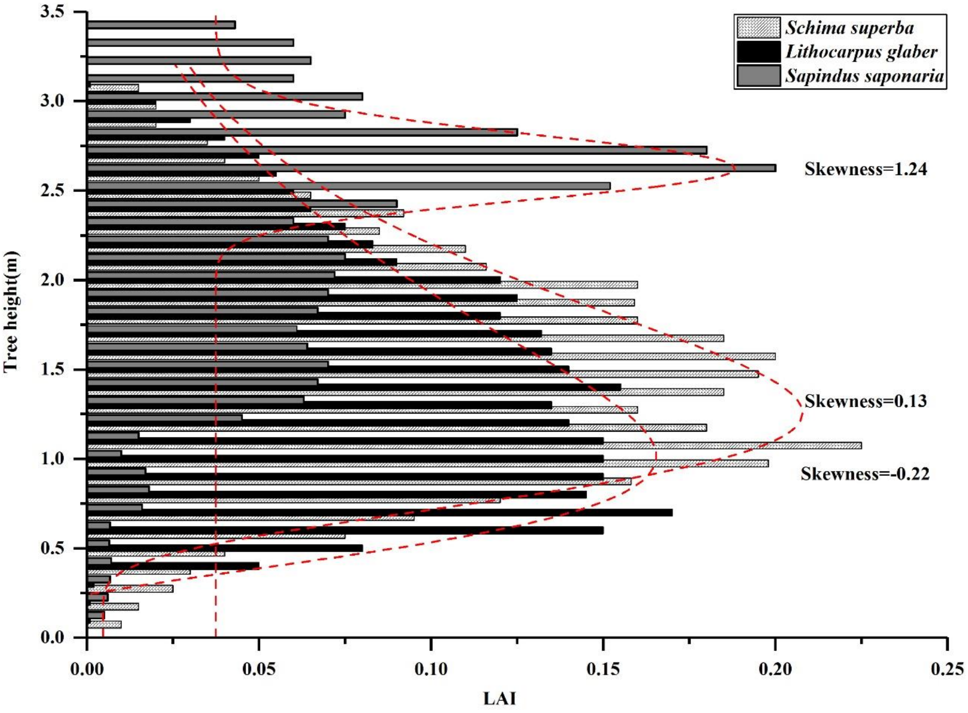
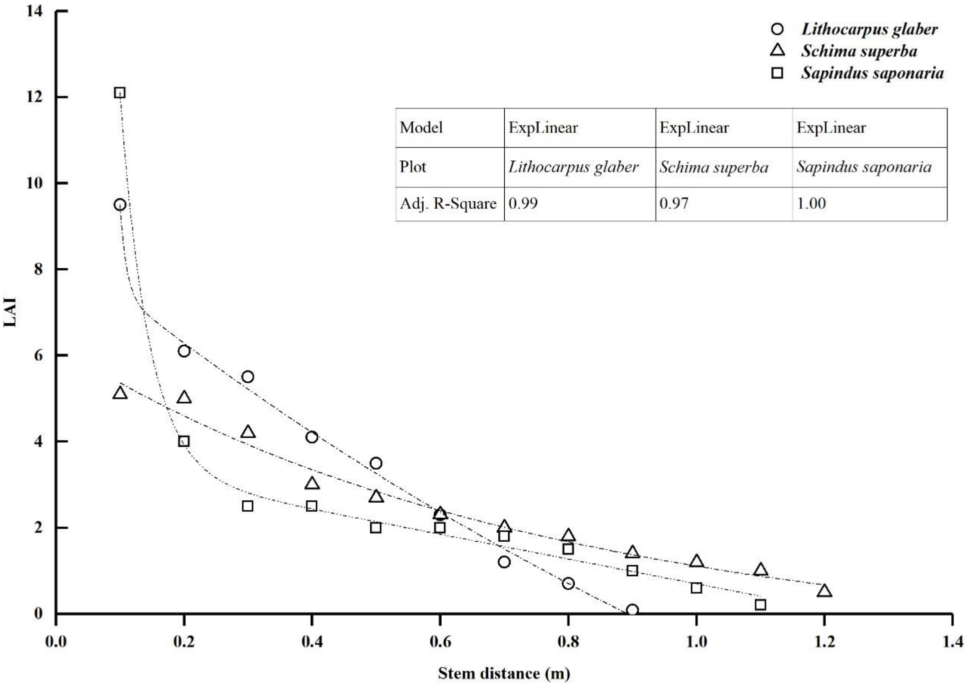


Fig. 2. LAI vertical distribution pattern of three tree species.



Model	ExpLinear	ExpLinear	ExpLinear
Plot	*Lithocarpus glaber*	*Schima superba*	*Sapindus saponaria*
Adj. R-Square	0.99	0.97	1.00

Fig. 3. LAI radial distribution patterns of three tree species.

3.2 Potential of splash erosion under different tree species

Sapindus saponaria had the highest values of TKE among the observed species and events. Compared to TKE of open-field rainfalls, TKE was enhanced under *Sapindus saponaria* and reduced under *Lithocarpus glaber* and *Schima superba* (Fig 4). For all measured rainfall events, *Sapindus saponaria* increased by 60-80% compared to TKE in open-field, while *Lithocarpus glaber* and *Schima superba* decreased approximately 60% and 30-80%, respectively. With increasing radial distance from the stem, TKE of all three species was generally increasing during different rainfall events although significance of positive correlation was only detected with *Lithocarpus glaber* (Table 2).

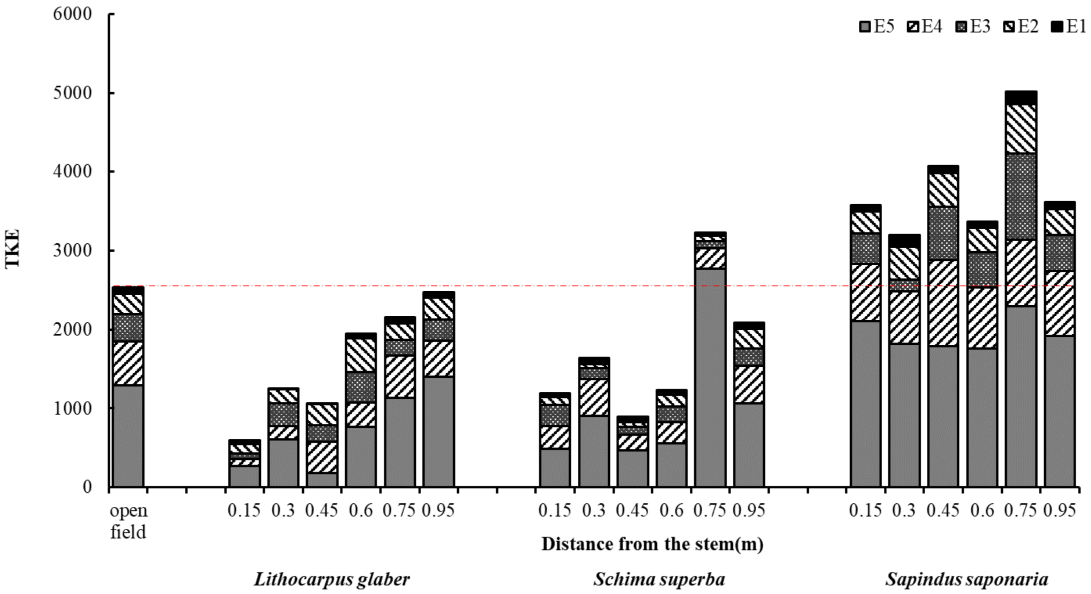


Fig. 4. Throughfall kinetic energy (TKE) changes with the distances from the stem under different tree species in Xingangshan, Jiangxi Province, PR China.

Table 2 Pearson Correlation between distance from the stem and TKE.

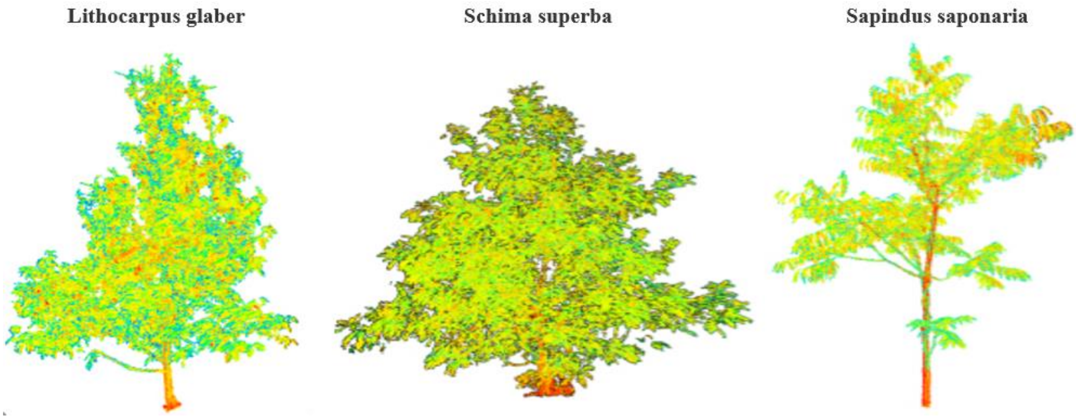
	Lithocarpus glaber	*Schima superba*	*Sapindus saponaria*
Event 1	0.73*	0.02	0.14
Event 2	0.51	0.67	0.28
Event 3	0.44	-0.18	0.46
Event 4	0.87*	0.23	0.25
Event 5	0.88*	0.49	0.10

* Significant level $p < 0.05$

4. Discussion

4.1 Difference in spatial distribution of LAI

Tree saplings growth and tree shape were significantly species-specific (Fig 2, 5), indicating an interspecific variation in certain functional traits in BEF China as described in detail by (Li *et al.*, 2014a). Different growth in tree functional traits among specific species might reflect their adaptive strategy related to the hereditary and environment. According to classical life-history theory of plants (Pianka, 1970), light-demanding tree species might be characterized by small values of leaf mass per area and maximum adult height, while shade-tolerant species show the opposite trait values (Poorter *et al.*, 2008). In our research, the different leaf traits of trees showed different vertical distribution patterns of LAI, while same leaf trait of trees had similar distribution patterns, as it was shown that little difference between *Lithocarpus glaber* and *Schima superba* occurred (Fig 2 and 5, Table 3). Accordingly, shape and size of the leaves did not only influence the LAI in general, but were also related to the spatial distribution of LAI along tree height. Deciduous trees like *Sapindus saponaria* allocate more photosynthetic products in height and branch growth with simple crown architecture, while evergreen trees such as *Lithocarpus glaber* and *Schima superba* would consume more energy to branch construction and leaves with complex crown architecture (Chave *et al.*, 2009; Kang, 2010). Leaf mass area distribution is correlated with light availability, moisture and nutrients transportation and production of trees (Ellsworth and Reich, 1993). Our results showed that a lognormal model can be used to predict the vertical distribution of LAI for broadleaved species. This result was in accordance with Lu (2011) and Zhao *et al.* (2015), who found that foliage distribution of major broadleaved species in secondary forest in northern China had the lognormal patters. As an important biophysical parameter in ecology, LAI not only quantifies effects of canopy thickness, leaf mass and branch count (Geißler *et al.*, 2013), but also its vertical distribution illustrate the distribution of tree height, first branch height and canopy thickness, indicating the process of rain drops from tree canopy to surface soil. In our study, LAI vertical distribution and its skewness reflected the difference in height of the first branch, number of branches and crown cover among the three species, for *Lithocarpus glaber* and *Schima superba* both having lower skewness with lower height of first branch, more branches and high crown cover comparing to *Sapindus saponaria* (Table 3). Moreover, the skewness could also account for the heterogeneity in horizontal and vertical leaf area distribution, which may provide a better way to understand the species-specific relationship between LAI and canopy water storage (Llorens and Gallart, 2000; Keim and Link, 2018) and tree diversity effects on TKE (Geißler *et al.*, 2013). Therefore, skewness of LAI vertical distribution might be a promising index comprehensively describing tree function in ecosystem, especially the process of hydrology. In further studies, more tree species of LAI vertical distribution will be established to prove the availability.



Lithocarpus glaber

Schima superba

Sapindus saponaria

Fig. 5. Three single tree species images from point cloud data measured with the laser scanner software RiScan Pro in Xingangshan, Jiangxi Province, PR China.

Table 3 Comparison of basic parameters of three investigated tree species in Xingangshan, Jiangxi Province, PR China (n = 9).

Tree species	Leaf habit	Ground Diameter(m)	First branch height(m)	Tree height(m)	Crown Width(m)	Number branches	Crown cover
Lithocarpus glaber	E	0.037±0.015a	0.30±0.04 b	2.98±0.21 a	1.70±0.23 a	33±4 a	0.38±0.03a
Schima superba	E	0.071±0.037a	0.15±0.13 b	3.14±0.79 a	2.12±0.36 a	40±5 a	0.42±0.09a
Sapindus saponaria	D	0.049±0.007 a	1.13±0.45 a	3.48±0.07 a	1.91±0.28 a	7±5b	0.28±0.02b

E represents evergreen broadleaved species; D represents deciduous broadleaved species. Different lower letters in the same tree basic parameters denote significant difference at $p < 0.05$

4.2 LAI effects on TKE

Leaf area index is regarded as a major influence on TKE of rainfall drops (Gómez *et al.*, 2001; Park and Cameron, 2008) and thus differences in LAI spatial distribution contribute to differences in TKE (Geißler *et al.*, 2013), although they both have seasonal temporal patterns (Levia Jr and Frost, 2006; Doughty and Goulden, 2008). TKE generally increased as the radial distance from the trunk increased with the LAI decreasing during different rainfall events. (Nanko *et al.*, 2011) found that the TKE was positively correlated with the radial distance in a 9.8-m-tall transplanted Japanese cypress tree (*Chamaecyparis obtusa*). Our results demonstrated that TKE was species-specific, with lower TKE of *Lithocarpus glaber* and *Schima superba* than *Sapindus saponaria*. On the one hand, it is assumed that if *Lithocarpus glaber* and *Schima superba* have higher LAI, they also show high rainfall interception. This is because canopy water storage increases with increasing LAI with a higher vertical distribution of foliage and canopy roughness (Aston, 1979; Marin *et al.*, 2000; Fleischbein *et al.*, 2005) and falling drops and drop sizes are more likely to be re-modified and split by lower parts of the canopy (Wiersum, 1985). On the other hand, for *Sapindus saponaria* and in the radial direction, high LAI is mainly located at the top height (Skewness = 1.24), which may lead to less interception and higher speed of falling drops reaching the soil surface and thus contribute to higher kinetic energy. Lognormal distribution LAI of *Lithocarpus glaber* and *Schima superba* with lower Skewness values indicated that the two species may have higher rainfall interception and lower speed of falling drops which contribute to reducing TKE. Research show that many factors affect TKE in forests such as crown cover and thickness, leaf traits, leaf area index, first branch height, tree height and branch architecture (Cao *et al.*, 2008; Geißler *et al.*, 2012b; Goebes *et al.*, 2015a; Goebes *et al.*, 2015b) . As we discuss above, skewness of LAI vertical distribution would be a better index to exactly indicate the process of rain drops from tree canopy to surface soil by combining the effects of many factors such as canopy thickness, leaf mass and branch count, tree height and first branch height together. Also, our result show higher skewness of LAI vertical distribution had higher TKE among the observed species and events. Therefore, the importance of LAI spatial distribution should be emphasized for TKE. Compared to open field, TKE of *Lithocarpus glaber* and *Schima superba* was reduced while for *Sapindus Saponaria* it was enhanced. The finding reminded us that afforestation in the first years might accelerate splash erosion because of structural shortcomings of tree species chosen. It also gives the suggestion to improve the protection of soil in afforested area by combining different tree species and thus apply a higher level of biodiversity. However, many questions are still unclear, for example, to what extent the skewness values of LAI vertical distribution have negative or positive effects on TKE under different ages of tree species not only for broadleaved species but also for needle species.

5. Conclusions

Spatial distributions of LAI of *Lithocarpus glaber*, *Schima superba* and *Sapindus saponaria* were determined by a terrestrial laser scanner (TLS) in a subtropical Chinese afforestation. Vertical distributions of LAI and TKE of different tree species were significantly different. TKE was measured with splash cups and affected by LAI and its spatial distribution. *Sapindus saponaria* enhanced TKE while *Lithocarpus glaber* and *Schima superba* reduced it. In an early stage of afforestation, splash erosion might be accelerated because of tree species selected. Skewness of LAI vertical distribution appears to be a suitable index to comprehensively describe tree functions within hydrological systems. Combining LAI with its spatial distribution patters using a TLS could be a promising method to evaluate potential splash erosion risk in afforested areas.

Acknowledgments

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Manuscript 2

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Tree diversity reduced soil erosion by affecting tree canopy and biological soil crust development in a subtropical forest experiment.

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Abstract

Biodiversity plays a crucial role in forest ecosystem sustainability. However, it is unclear how tree diversity and especially the relationship between diversity and ecosystem functioning affect soil erosion. Based on a forest biodiversity and ecosystem functioning experiment established in subtropical China (BEF China), we measured soil erosion at four tree species richness levels (monocultures, 8 tree species, 16 tree species and 24 species stands) during the rainy seasons from 2013 to 2015. The result showed that mean annual soil erosion rates were detected to decrease with tree species richness significantly over the observed three years. Leaf area index (LAI) and biological soil crusts (BSCs) were the two main factors driving soil erosion within tree stands of different species richness. Positive effects of tree species richness on tree canopy structure and BSCs might drive the reduction of soil erosion in the earlier successional stage after afforestation of tree plantations. Therefore, we highlight the important influence of tree species richness on soil erosion control, hydrologic processes and thus sustainable ecology services.

Key words: BEF China, Soil loss, Tree species richness, Leaf area index, Biological soil crust

1. Introduction

Growing concern about biodiversity is emerging worldwide due to substantial contemporary declines in species richness at different scales (Tittensor *et al.*, 2014; Mori *et al.*, 2017). As part of recent scientific research, forest diversity has shown to promote biomass production, carbon accumulation and nutrient cycling (Kelty, 2006; Cardinale *et al.*, 2012; Gamfeldt *et al.*, 2013; Forrester, 2014; Błońska *et al.*, 2018; Huang *et al.*, 2018). At the same time, the question how forest diversity might influence soil erosion also attracts researchers' attention. Answering this question is significant not only for designing restoration and rehabilitation programs to achieve the sustainable goals for ecosystem development but also for emphasizing the diversity of the biota in nature based solutions when they will be designed (Keesstra *et al.*, 2018a; Keesstra *et al.*, 2018b; Solomun *et al.*, 2018). Recent studies pointed out that plant diversity could enhance soil stability and reduce soil erosion in different environments such as alpine grasslands or on dyke slopes (Pohl *et al.*, 2009; Martin *et al.*, 2010; Wang *et al.*, 2012b; Geißler *et al.*, 2013; Berendse *et al.*, 2015).

Meanwhile, it was shown that in young forest plantations, soil erosion is strongly affected by tree species but a clear effect of tree diversity could not be found in early stage forests (Goebes *et al.*, 2015b; Goebes *et al.*, 2016; Seitz *et al.*, 2016). Those studies suggested an unclear relationship between tree diversity and soil erosion during forest recovery. Therefore, research on the temporal development of soil losses under different forest diversity levels is essential for understanding how biodiversity might sustain ecological services such as water erosion prevention in forest plantations.

Subtropical forests as one of the most productive ecosystems in the world are closely related with global biogeochemical cycles and climate change (Melillo *et al.*, 1993; Bala *et al.*, 2007). They are increasingly suffering from intensive human activities such as deforestation, but also human-induced afforestation with decreasing plant diversity (Durieux *et al.*, 2003; Malhi *et al.*, 2008; Barrufol *et al.*, 2013). Without human disturbance, subtropical forests in China are dominated by evergreen broad-leaved species (Wang *et al.*, 2007; Bruelheide *et al.*, 2014b). However, these areas were mostly cleared and have been converted into monospecific conifer stands in the last decades for many reasons such as economic benefits and easy management (Zhao, 2006; Wang *et al.*, 2007; Li *et al.*, 2014a). For instance, the two most important tree species for forest resources and ecological services in subtropical China, *Chinese fir* and *Pinus massoniana*, cover approximately 12.39 × 10^6 ha or 10% of the total forest area and 6.78 × 10^5 ha or 27% of the forest area in the Three Gorges Reservoir area(Wang *et al.*, 2012a; Huang *et al.*, 2013; Wang, 2014). Apparently, these monoculture plantations bring with them a higher potential of ecological insecurity because of greater susceptibility to adverse environmental conditions and negative effects on soil properties (Yang *et al.*, 2009; Li *et al.*, 2014a). Moreover, soil erosion commonly occurs in forest plantations and highly varies in forested areas of subtropical China ranging from 0 - 6.32 t ha^{-1} yr^{-1} (Hill and Peart, 1998; Guo *et al.*, 2015; Seitz, 2015; Seitz *et al.*, 2016). This results in high sediment load in rivers, although great endeavors have been made to restore and afforest vast areas with commercial monocultures (Zhao, 2006; Wang *et al.*, 2007; Lei *et al.*, 2009; Guo *et al.*, 2015). These current circumstances imply that monospecific plantations might be less suitable for soil erosion control.

Generally, tree diversity is assumed to affect soil erosion mainly by two aspects: First, it optimizes patterns of spatial vegetation structures and distribution (e.g. by better gap filling) and thus strengthens the interception of rainfall (Zheng *et al.*, 2008; Geißler *et al.*, 2013; Liu *et al.*, 2018). Second, it could reduce soil erosion by improving soil properties and soil covering vegetation on the forest floor that attributes to the reduction of raindrop impacts on the soil surface as well as enhancing soil water capacity and infiltration (Janssens *et al.*, 1998; Bezemer *et al.*, 2006; Pohl *et al.*, 2009; Martin *et al.*, 2010; Wang *et al.*, 2012b). Tree species richness has shown to increase canopy stratification and growth rates, leading to a more homogenous canopy development due to a higher number of functional traits (Lang *et al.*, 2010; Lang *et al.*, 2012a; Chisholm *et al.*, 2013). Recent research showed that different tree species have different spatial distribution patterns in monocultures but also in mixtures, e.g. regarding leaf area, leading to changing kinetic energy of raindrops and thus splash erosion rates (Goebes *et al.*, 2015a; Goebes *et al.*, 2015b). Furthermore, tree traits such as leaf area or crown cover affect the development of soil surface covering vegetation, in particular biological soil crusts (BSCs), e.g. by modifying light conditions and thus the succession sequence from cyanobacteria to bryophytes (Issa *et al.*, 1999; Belnap *et al.*, 2001; Zhang *et al.*, 2016). BSCs dominated by mosses and liverworts in turn, proved to effectively mitigate soil losses in young forest plantations in subtropical China (Seitz *et al.*, 2017).

Based on these findings, we assume that tree species richness reduces soil erosion with ongoing tree growth by not only modifying the tree canopy structure but also improving the soil surface coverage by BSCs. Thus, we hypothesize in this study that tree species richness could affect soil erosion by altering vegetation patterns in the tree canopy layer as well as in the vegetation layer directly covering the soil surface.

To answer those hypotheses, we made use of a Biodiversity and Ecosystem Functioning Experiment under forest in PR China using monocultures and three different tree species richness levels (Bruelheide *et al.*, 2014a). Soil erosion was measured as interrill sediment delivery with micro-scale runoff plots which are commonly applied in current studies (Seitz, 2015; Cerdà *et al.*, 2017; Rodrigo-Comino *et al.*, 2018) and annual soil erosion rates calculated after Wischmeier and Smith (1978a). Canopy traits, soil surface cover and soil parameters were recorded and results obtained using linear mixed effects models (Trogisch *et al.*, 2017).

2. Materials and Methods

2.1 Study area

The research was conducted in the framework of the BEF China project, which is located in Xingangshan Town, Dexing City, Jiangxi Province, PR China (29.08°–29.11° N, 117.90°–117.93° E). The climate is dominated by subtropical monsoon with a mean annual temperature of 17.4 ℃ and a mean annual precipitation of 1635 mm with half of it falling from May to August (Goebes *et al.*, 2015b). The natural vegetation is dominated by broadleaved forest with evergreen species (Trogisch *et al.*, 2017). The area shows mainly hills at elevations from 105 to 200 m with slopes from 15° to 41° (Scholten *et al.*, 2017). Soils in the project are mainly Cambisols, with Anthrosols in downslope positions and Gleysols in valleys and the bedrock is non-calcareous slates weathered to saprolite (Scholten *et al.*, 2017).

2.2 Experimental design and data collection

2.2.1 Research plots

The BEF China project includes two parallel sites (A and B) on which commercial monocultures were originally planted that were cut down in 2009 and 2010, respectively (Bruelheide *et al.*, 2014a). Holes of 0.5 m (length) × 0.5 m (width) × > 0.2 m (depth) were dug for seedlings (Yang *et al.*, 2013). Forty local tree species were replanted in monocultures and mixtures of 2, 4, 8, 16 and 24 species on 566 plots with each measuring 25.8 × 25.8 m (667 m^2) (Yang *et al.*, 2013). For each plot (Fig. 1), 400 tree individuals were planted in 20 rows of 20 tree individuals with a planting distance of 1.29 m (Bruelheide *et al.*, 2014a). For this study, 43 of these plots were selected, 22 on Site A and 21 on Site B with four tree species richness levels: monocultures, 8 tree species, 16 tree species and 24 tree species stands (Table 1).

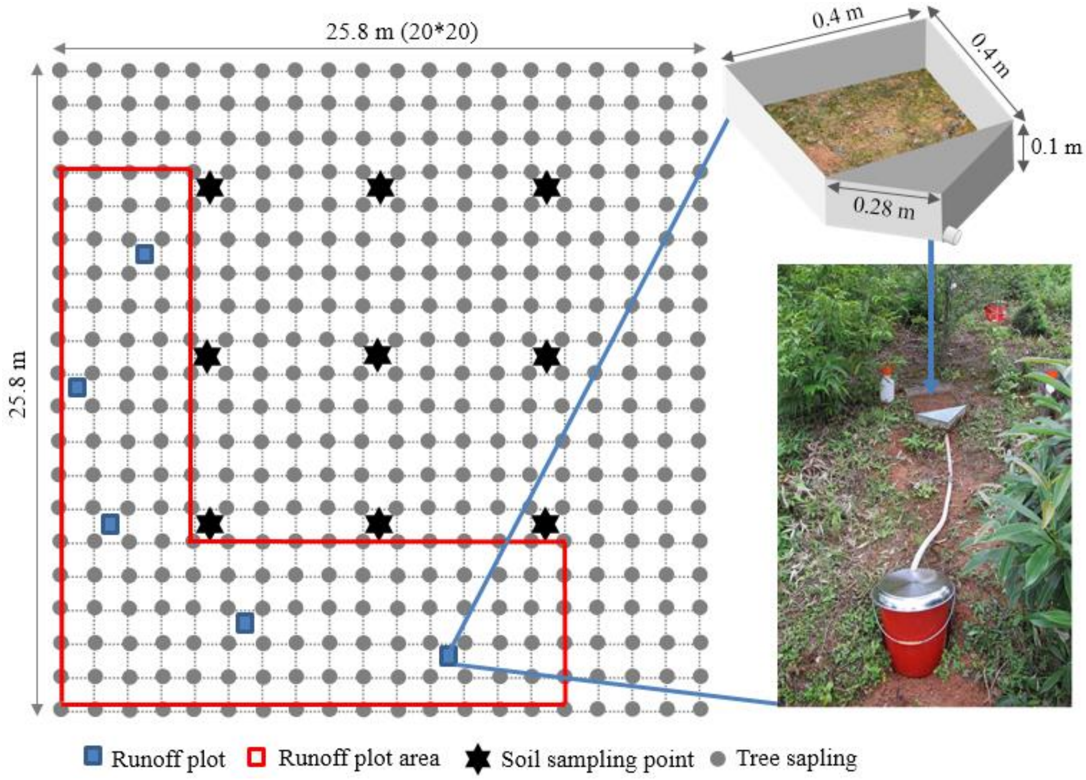

Fig. 1. Random positions of runoff plots for soil erosion measurements and soil samples in one research plot of the BEF China project (0.4 m length × 0.4 m width × 0.1 m height).

Table 1 Tree, topography and soil data (0-5 cm) of 43 selected research plots in the BEF China project. (TSR: tree species richness; Soil BD: soil bulk density; SOC: soil organic carbon)

				Topography			Soil properties		
Plot	TSR	Tree species	Site	Slope (°)	Aspect	Altitude (m)	BD (g m^{-3})	pH	SOC (%)
D29	1	*M. flexuosa*	B	31	N	159	0.90	3.68	2.77
I25	1	*M. yuyuanensis*	B	29	N	152	0.96	3.47	2.90
M07	1	*B. luminifera*	B	31	S	129	0.89	3.55	2.52

N02	1	*M. flexuosa*	B	41	S	129	0.89	3.61	3.01
N05	1	*A. altissima*	B	32	N	119	0.89	3.63	3.17
N28	1	*I. polycarpa*	B	19	E	167	0.97	3.56	2.21
Q27	1	*A. fortunei*	B	35	S	160	0.97	3.66	2.78
Q29	1	*M. leptophylla*	B	33	E	144	0.90	3.74	2.42
R29	1	*C. fargesii*	B	33	S	146	0.91	3.65	2.48
T13	1	*M. thunbergii*	B	21	W	133	0.96	3.44	2.59
U16	1	*E. japonicus*	B	20	W	147	0.94	3.44	2.65
V24	1	*E. chinensis*	B	32	E	137	0.94	3.71	3.01
W10	1	*Ph. bournei*	B	27	E	147	0.92	3.45	2.35
W11	1	*E. glabripetalus*	B	19	S	148	1.04	3.25	2.72
X21	1	*M. grijsii*	B	24	N	132	0.91	3.65	2.54
Y09	1	*C. biondii*	B	32	E	126	1.10	3.71	1.93
E31	1	*Q. fabri*	A	22	S	144	0.95	3.86	2.48
E33	1	*L. glaber*	A	19	S	144	1.12	3.94	2.18
E34	1	*C. henryi*	A	21	S	125	1.06	4.09	2.84
G33	1	*Q. serrata*	A	18	S	127	0.85	3.92	3.45
I28	1	*L. formosana*	A	26	S	163	0.90	3.81	3.29
K19	1	*S. superba*	A	24	N	199	0.80	3.70	4.18
L10	1	*C. eyrie*	A	34	S	211	0.92	3.92	2.81
L11	1	*C. sclerophylla*	A	28	S	201	1.04	3.87	2.95
N11	1	*S. saponaria*	A	26	S	203	0.82	3.63	3.93
N13	1	*S. sebiferum*	A	31	S	182	0.78	3.78	3.62
N17	1	*R. chinensis*	A	28	W	221	0.91	3.79	3.39
O22	1	*C. myrsinaefolia*	A	21	W	229	0.86	3.80	3.54
O27	1	*Ch. axillaris*	A	21	W	185	1.07	4.12	2.41
Q13	1	*K. bipinnata*	A	30	W	215	0.90	3.86	3.84
R14	1	*C. glauca*	A	30	N	228	0.82	3.80	4.25
J29	8	1*	B	31	N	182	0.81	3.39	4.85
Q17	8	2*	B	22	N	131	0.99	3.52	2.91
S10	8	3*	A	36	S	220	0.96	3.79	3.04
T15	8	4*	A	30	N	244	0.87	3.67	3.42
I22	16	5*	B	28	S	119	1.07	3.58	2.28

S22	16	5*	B	33	W	145	1.00	3.61	3.26
L22	16	6*	A	21	W	180	0.8	3.79	3.48
M22	16	6*	A	23	W	221	0.95	3.79	3.48
U10	16	6*	A	40	S	231	0.96	3.86	3.22
R30	24	7*	B	27	S	136	0.95	3.67	2.46
N09	24	8*	A	33	S	218	0.86	3.58	3.60
R18	24	8*	A	36	W	215	0.91	3.82	3.50

1*: 8 tree species from monocultures N05, Y9, W11, U16, N28, X21, D29 and W10.

2*: 8 tree species from monocultures Q27, M07, R29, V24, Q29, T13, I25 and *Q. phillyreoides*.

3*: 8 tree species from monocultures E34, L11, O27, I28, G33, N11, N13 and *N. sinensis*.

4*: 8 tree species from monocultures E33, E31, N17, K19, L10, R14, O22 and Q13.

5*: 16 tree species from 1* and 2*.

6*: 16 tree species from 3* and 4*.

7*: 24 tree species from 5*, from monocultures E33, K19, R14, L10, L11, and *Cinnamomum camphora*, *Daphniphyllum oldhamii* and *Diospyros glaucifolia*.

8*: 24 tree species from 6* and *Cinnamomum camphora*, *Daphniphyllum oldhamii*, *Diospyros glaucifolia*, *Acer davidii*, *Castanopsis carlesii*, *Melia azedarach*, *Quercus acutissima* and *Sapium discolor*.

2.2.2 Measurement of sediment delivery

Based on the design of BEF China and considering the various research topics investigated, a selected area of each research plot was used for soil erosion measurements (Bruelheide *et al.*, 2014b; Trogisch *et al.*, 2017). Five micro-scale runoff plots (0.4 m length × 0.4 m width × 0.1 m height) were randomly installed in 2013 and connected to 20 L reservoirs to collect runoff and sediment delivery (Fig. 1) (Seitz *et al.*, 2016). The runoff plots were operated from May to July during the rainy season in 2013, 2014 and 2015. Runoff volume was collected in situ and sediment delivery was calculated after sampling. In total, 535 valid measurements from 215 runoff plots were captured (180 in 2013, 152 in 2014 and 203 in 2015).

2.2.3 Tree parameters

Tree measurements with laser scanning (FARO Laser Scanner Photon 120, FARO Technologies Inc., FL, USA) at all plots started in September 2010 for both experimental sites on a yearly base, which were determined by the central 6 × 6 trees (36 trees) in the monocultures and the central 12 × 12 trees (144 trees) in the 8, 16 and 24 tree species stands (Li *et al.*, 2014a; Li *et al.*, 2017).

Crown cover and leaf area index (LAI) were measured each May from 2013 to 2015 at the ROP scale using a fish-eye camera system (Seitz *et al.*, 2016). 552 valid pictures of forest canopy at runoff plot scale were captured.

2.2.4 Soil surface cover

Soil surface cover including biological soil crusts (BSCs) and stone cover was surveyed yearly. BSCs were measured photogrammetrically during the rainy seasons from 2013 to 2015 within the runoff plots. Perpendicular images for each runoff plot were taken by a camera system (Canon 350D, Tokio, Japan). The images were processed by the grid quadrat method with 10 × 10 subdivisions of a digital grid in GIMP 3.0. BSCs and stone cover were separated by hue distinction. Further soil surface cover by shrubs was not present due to weeding according to the experimental design and a continuous leaf litter layer could not be recorded during the first years of this early successional afforestation (Seitz, 2015).

2.2.5 Soil properties

The soil sampling was conducted in 2014. Soil cores with 6 cm in diameter were taken to a depth of 50 cm and then divided into five depth increments (0-5 cm, 5-10 cm, 10-20 cm, 20-30 cm and 30-50 cm) (Scholten *et al.*, 2017). For this study, soil properties of the 0-5 cm increment were used (Table 1). For each plot, nine soil cores were collected (Fig. 1) and mixed. Soil samples were air-dried, sieved through a 2 mm mesh, handpicked to remove plant and animal residuals and then grounded for soil analyses. For total soil carbon analyses, about 40 mg of ground sample material was weighed into tin foil and analyzed using oxidative heat combustion at 1150 °C in a helium atmosphere in a Vario EL III elemental analyzer (Elementar Analysensysteme GmbH, Hanau, Germany). Soil pH was determined with a 1 M KCl solution (soil-to-solution ratio 1:2.5) by a WTW pH meter pH 340 (WTW GmbH, Weilheim, Germany) using a Sentix 81 electrode according to DIN EN15933 (2012). Since pH is < 6.7 for all samples: total soil carbon = soil organic carbon (SOC). In addition, five replicates of bulk density samples were obtained for each plot at the same depth increments in 2015. The soil bulk density (BD) was gravimetrically determined from the five replicate volumetric samples per depth increment (samples dried at 105 °C).

2.2.6 Rainfall characteristics

Rainfall data during the soil erosion measurements was captured by climate stations on both sites (ecoTech data logger with Vaisala weather transmitter and ecoTech tipping bucket balance, Bonn, Germany) (Fig. 2). Daily accumulated precipitation curves and the ten largest daily rainfall events during the three years were shown in Fig. 2. a and b.

Further data on regional precipitation was used from the National Meteorological Information Center (NMIC) of China and China Meteorological Administration (CMA).

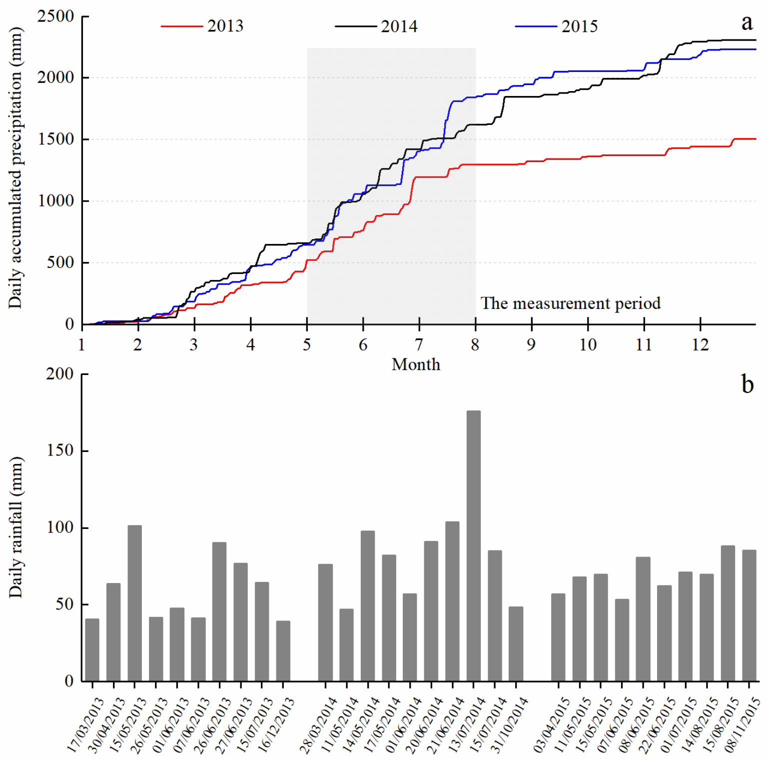

Fig. 2 (a) Daily accumulated precipitation and (b) The ten largest daily rainfall events in BEF China from 2013 to 2015.

2.3 Statistical methods

2.3.1 Annual soil erosion rate

A rainfall threshold of 12.7 mm was applied to distinguish erosive rainfall amounts after Wischmeier and Smith (1978a). Annual erosive rainfall amount (AER, mm) and erosive rainfall during the runoff plot measurements (ERM, mm) in the rainy seasons were calculated based on precipitation curves from climate stations (Fig. 2). Then, with sediment delivery acquired during the runoff plot measurements (SE, Mg ha^{-1}), an annual soil erosion rate (Mg ha^{-1}) was calculated (Eq. 1).

$$\text{Annual soil erosion} = (\text{AER} / \text{ERM}) \times \text{SDE} \qquad \text{Eq. 1}$$

2.3.2 Data analysis

We used linear mixed effects models to (I) analyze the temporal development of annual soil erosion rates under changing tree species richness and to (II) investigate driving factors on soil erosion rates and how in turn those factors are influenced by tree species richness.

Before modelling, annual soil erosion rates from 2013 to 2015 were twice square root transformed ($\sqrt{\sqrt{y}}$) to fit normal distribution. A first model was calculated with tree species richness, year and the interaction of tree species richness with year as fixed factors, while site, plot, runoff plot nested in plot and tree species composition were fitted as random factors.

Further models were compiled to identify explaining parameters on soil erosion. Tree height, stem diameter, crown width, crown cover, LAI, biological soil crusts, surface cover and soil properties were fitted as fixed factors, while site, runoff plots nested in plot and tree species composition were fitted as random factors. If multi-collinearity (correlation index > 0.7) was detected among the fixed factors, correlated factors were fitted individually in exchange to the counterpart.

Finally, models were used to analyze the effects of tree species richness on identified main influencing factors of soil erosion by using tree species richness, year and the interaction of tree species richness with year as fixed factors, while site, plot, runoff plot nested in plot and tree species composition were used as random factors.

All statistical analyses were conducted with R 3.4.3 (R Foundation for Statistical Computing, Vienna, Austria). Graph and curve fittings were processed in Origin 8.0 (OriginLab Corporation, Northampton, USA).

3. Results

3.1 Soil erosion under different tree species richness

In 2013, monocultures and 24 tree species stands had similar mean soil erosion rates with 43.5±3.1 Mg ha^{-1} and 45.8±7.7 Mg ha^{-1}, whereas 16 tree species stands showed lower rates (28.7±8.3 Mg ha^{-1}) and 8 tree species stands showed the lowest value with 22.5±3.4 Mg ha^{-1} (Fig. 3). From 2014 to 2015, the mean annual soil erosion decreased from monocultures (24.7±2.1 Mg ha^{-1}, 11.3±1.9 Mg ha^{-1}) to the 8 tree species stands (20.8±4.6 Mg ha^{-1}, 7.9±2.6 Mg ha^{-1}), to the 16 tree species stands (18.2 ±3.6 Mg ha^{-1}, 5.3±0.8 Mg ha^{-1}) and finally to the 24 tree species stands (17.5±5.6 Mg ha^{-1} and 2.7±0.8 Mg ha^{-1}). Besides, the highest and lowest annual soil erosion rates measured during 2013-2015 were all detected in monocultures (Fig. 3), indicating that monocultures have a high variability regarding soil erosion. From linear mixed effects model, tree species richness reduced annual soil erosion rates over the observed three years significantly (Table 2, $P < 0.001$, $F = 65.13$).

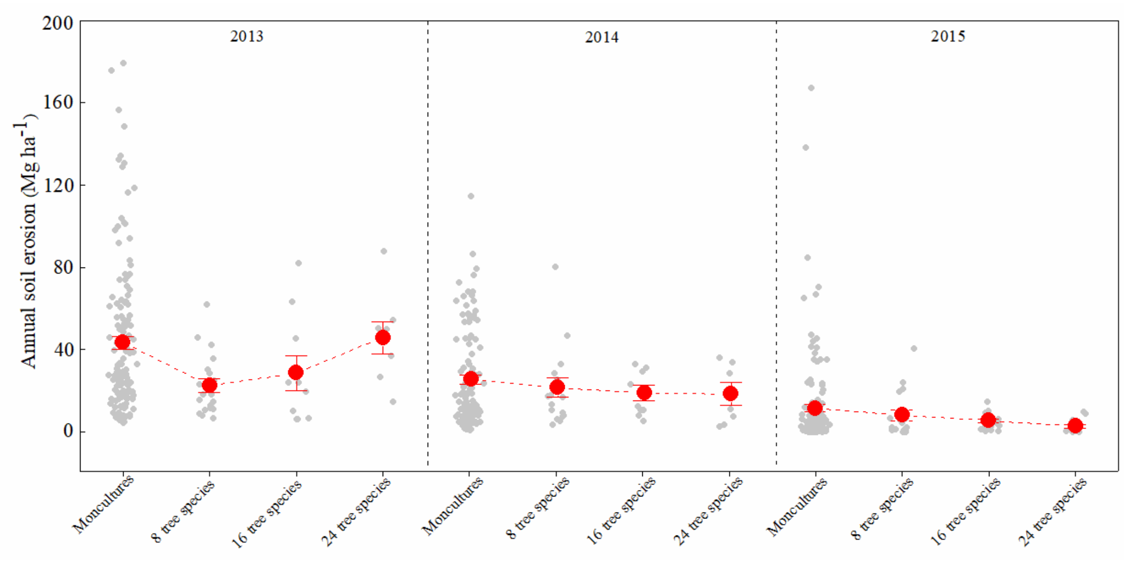

Fig. 3. Annual soil erosion (Mg ha^{-1}) in 2013, 2014 and 2015 under different tree species richness in the BEF China experiment in Xingangshan, Jiangxi Province, PR China. Gray points mean data collected from runoff plots (n = 535). Red spline dashed lines connected mean±standard error of each tree species richness for each year.

Table 2 Linear mixed effects models for the effects of tree species richness (TSR), year and the interaction tree species richness × year on annual soil erosion (n = 535). (Annual soil erosion were twice squared root scaled while tree species richness and year were scaled. *ddf* mean denominator degree of freedom; *F* and *P* mean *F*-ratio and *P*-value of the significance test.)

Fixed effect	*ddf*	*F*	*P*
TSR	522	65.13	< 0.001
year	529	262.60	< 0.001
TSR × year	522	2.049	0.152

Notes: fixed effects were fitted sequentially as shown in the table while random effects are site, plot, runoff plots nested in plot and tree species composition.

3.2 Factors driving changes of soil erosion with different tree species richness

Regarding the linear mixed effects model (Table 3), results showed that tree parameters and BSC negatively affected soil erosion ($P < 0.01$). The estimate for LAI ($P < 0.001$), crown cover ($P < 0.05$), crown width ($P < 0.01$), tree height ($P < 0.01$), stem diameter ($P < 0.01$) and BSC ($P < 0.001$) on soil erosion were -0.21, -0.20, -0.06, -0.09, -0.08 and -0.52, respectively. Soil properties including soil BD, soil pH, SOC and topography parameters including slope and altitude did not show significant effects on soil erosion. Therefore, tree canopy parameters and BSCs were the two main factors driving soil erosion.

Table 3 Linear mixed effects models for annual soil erosion (n = 535). (LAI: leaf area index; BSCs: biological soil crusts; soil BD: soil bulk density; SOC: soil organic carbon. *ddf* mean denominator degree of freedom; *F* and *P* mean *F*-ratio and *P*-value of the significance test.)

Fixed effect	*ddf*	*F*	*P*	*Estimate*
LAI	246	22.49	0.000	-0.21
BSCs	515	198.28	0.000	-0.52
Soil BD	32	15.01	0.079	0.06
Soil pH	32	2.92	0.097	0.08
SOC	32	0.09	0.650	0.02
Slope	32	5.50	0.025	0.01
Altitude	31	0.08	0.228	0.04
Crown cover	300	20.675	0.011	-0.20
Tree height	61	2.021	0.001	-0.09
Crown width	61	2.034	0.001	-0.06
Stem diameter	64	9.959	0.002	-0.08

Notes: Fixed effects were fitted sequentially as shown in the table while random effects are site, plot, runoff plots nested in plot and tree species composition. As multicollinearity of fixed factors (correlation index > 0.7) was detected among LAI, crown cover, tree height, crown width, and stem diameter, one factor was fitted individually in exchange to the other in the linear mixed effects model. All variables were tested on normal distribution. Annual soil erosion was twice squared root scaled while BSCs were square-root transformed with arcsign reconstruction. Then all variables were scaled before modelling.

A significantly positive effect of tree species richness on LAI was found and the size of this effect increased with years (Table 4 and Fig. 4; $P_{TSR} < 0.001$, $F_{TSR} = 7.6$; $P_{year} < 0.001$, $F_{year} = 24.6$; $P_{TSR : year} < 0.01$, $F_{TSR : year} = 9.1$). Monocultures had the lowest LAI, with mean values of 0.95 in 2013, 1.02 in 2014 and 1.25 in 2015 (Fig. 4). 8 tree species stands had the highest LAI in 2013 and 2014 with mean values of 1.43 and 1.66 and lower in 2015 with a mean of 1.78. LAI of 16 tree species and 24 tree species stands was increasing from 1.24 and 0.93 in 2013 to 1.93 and 1.65 in 2015 respectively, indicating that they both have faster increase rate of LAI.

Regarding BSCs, an increasing trend with tree species richness was shown, although no significant effect of tree species richness was detected (Table 4 and Fig. 5). 16 tree species stands showed the highest coverage of BSCs with 32%, 48% and 57% from 2013 to 2015 (Fig. 5). Monocultures, 8 tree species and 24 tree species stands in 2013 had a similar BSCs coverage ranging from 16%-17%. Then, the ranges of their BSCs coverage were 30%-35% in 2014 and 44%-52% in 2015 with the order of monocultures < 8 tree species stands < 24 tree species stands.

Table 4 Linear mixed-effects models for tree species richness (TSR), year and the interaction tree species richness × year on leaf area index (LAI) and biological soil crusts (BSCs) (n = 552). *ddf* mean denominator degree of freedom; *F* and *P* mean *F*-ratio and *P*-value of the significance test.

Fixed	*LAI*			*Biological soil crusts*		
effect	*ddf*	*F*	*P*	*ddf*	*F*	*P*
TSR	40	8.6	< 0.01	40	1.0	0.32
Year	529	35.6	< 0.001	535	92.5	< 0.001
TSR× year	522	12.1	< 0.01	536	1.4	0.245

Notes: Fixed effects were fitted sequentially as shown in the table while random effects are site, plot, runoff plots nested in plot and tree species composition. LAI was scaled. Biological soil crusts were square-root transformed with arcsign reconstruction and then scaled. Tree species richness and year were scaled.

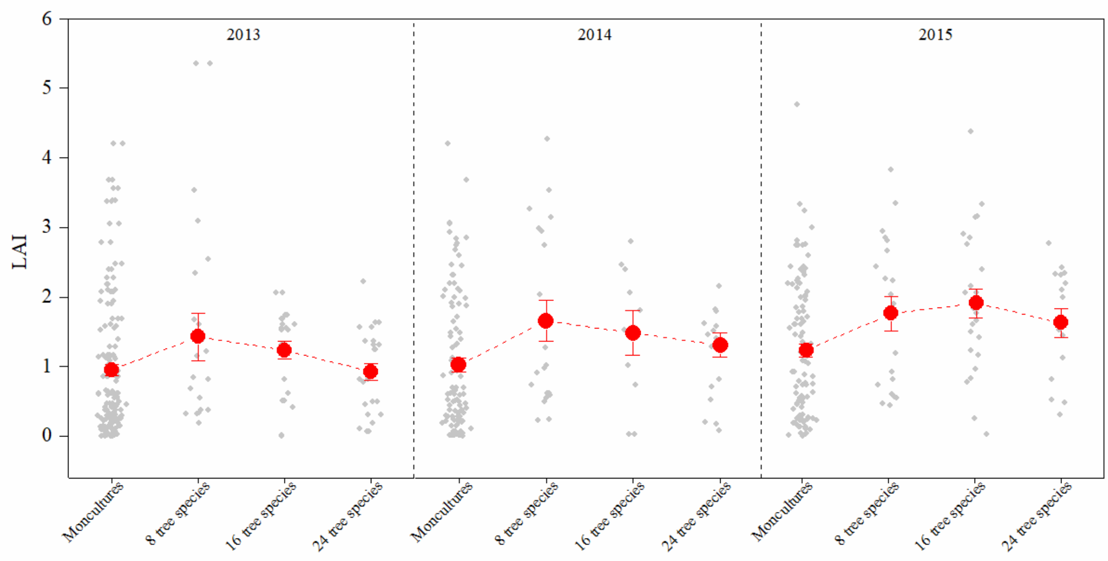

Fig. 4. Leaf area index (LAI) in 2013, 2014 and 2015 under different tree species richness in the BEF China experiment in Xingangshan, Jiangxi Province, PR China. Gray points mean data collected at runoff plots scale (n = 552). Read spline dashed lines connected mean ± standard error of each tree species richness for each year.

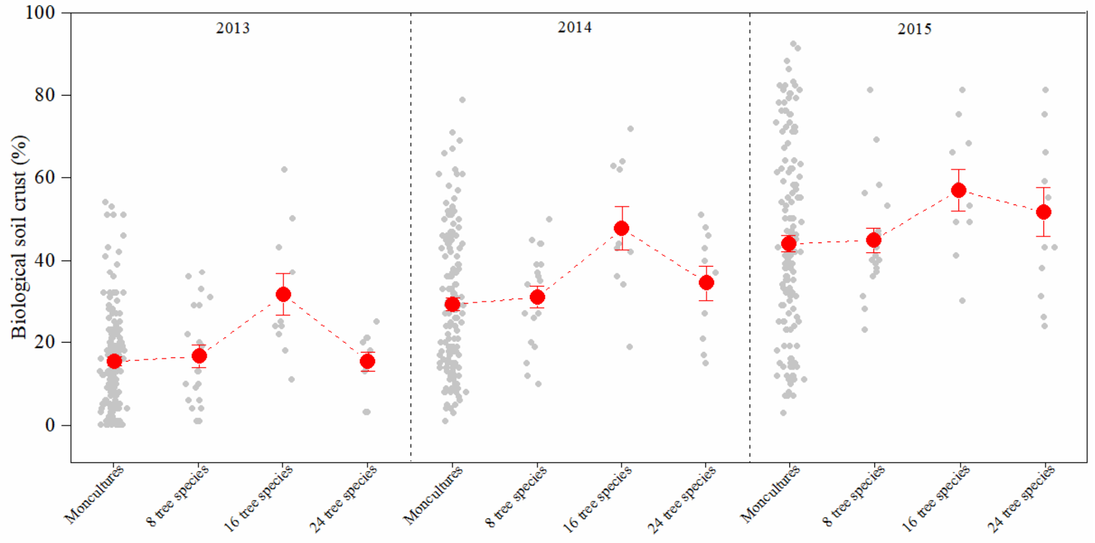

Fig. 5. Biological soil crusts (BSCs) within runoff plots (%) in 2013, 2014 and 2015 under different tree species richness in the BEF China experiment in Xingangshan, Jiangxi Province, PR China. Gray points mean data collected at runoff plots scale (n = 552). Read spline lines connected mean ± standard error of each tree species richness for each year.

4. Discussion

4.1 Soil erosion under different tree species richness

Annual soil erosion rates in the afforested study area were 40.6 Mg ha^{-1} in 2013, 23.5 Mg ha^{-1} in 2014 and 9.7 Mg ha^{-1} in 2015. Those rates are importantly higher than the 1.89 Mg ha^{-1} a^{-1} generally assumed for forests in the south of China (Guo *et al.*, 2015), even if we consider that measurements with micro-scale runoff plots were focusing on interrill erosion only. In Europe, the annual soil erosion rate under forest was given as 0.7 Mg ha^{-1} a^{-1} (Maetens *et al.*, 2012), whereas it ranged from 0 to 8 Mg ha^{-1} a^{-1} in Australia (Cerdan *et al.*, 2010b). Considering that soil erosion rates of 1 Mg ha^{-1} a^{-1} are assumed to be tolerable in general (Verheijen *et al.*, 2009) and in the range of 0.12 to 0.25 Mg ha^{-1} a^{-1} particularly under undisturbed forests (Patric, 1976), we conclude that the BEF China experiment is still suffering from severely high soil erosion even after 6 years of forest restoration.

Our study showed that higher tree species richness lead to decreasing soil erosion rates during the three years of observation. These calculated rates suggest that higher tree species richness could reduce soil erosion in subtropical Chinese forest plantations. For instance, stands of tree species richness higher than eight with annual soil erosion rate 16.1 Mg ha^{-1} a^{-1} showed to reduce soil erosion by 30% compared to monocultures (26.5 Mg ha^{-1} a^{-1}). Therefore, more diverse plantations should be considered as an important afforestation policy in this region to help conserving soil and water resources. However, in 2013, tree species richness did not show a decreasing pattern regarding soil erosion, as 24 tree species stands showed higher rates than 16 and 8 tree species stands and comparable rates to monocultures. This absence of a tree diversity effect was mainly caused by the early successional stage of the forest experiment (Seitz *et al.*, 2016). At this point, tree species richness had not yet contributed to vegetation cover development both in the canopy and the biological soil crust layer in a way that it affects soil erosion (Li *et al.*, 2014a; Goebes *et al.*, 2015b; Seitz *et al.*, 2016). In 2014, 24 tree species stands showed lowest erosion rates followed by 16 tree species stands, 8 tree species stands and monocultures. This finding indicates that tree diversity could alter soil erosion with ongoing tree growth and changing vegetation parameters, but forest ecosystems need several years of time for adaption. A similar result could be observed in land abandonment in the Mediterranean mountains of Spain based on an 11 years observation, with agricultural abandonment increasing soil erosion in the first two years due to low vegetation recovery and the development of a soil crust then resulting in lower erosion rates afterwards (Cerdà *et al.*, 2018). Therefore, it is necessary to monitor soil erosion in forest plantations for a longer term to identify the effect of plants and tree diversity in particular (Seitz *et al.*, 2016; Trogisch *et al.*, 2017).

4.2 Tree species richness reduced soil erosion by positive effects on tree canopy and biological soil crusts

In forests, natural rainfall passes through the vegetation cover before hitting the soil surface and can potentially cause soil erosion. We assume that positive effects of tree species richness on this vegetation cover above the soil surface result in a reduction of soil erosion.

Forest canopies influence soil erosion mainly by intercepting rainfall and altering rainfall patterns (Goebes *et al.*, 2015b). Raindrops are modified by tree and leaf traits within the forest canopy while rainfall amounts are reduced by vertical distribution of foliage and canopy roughness. It has been shown that trees with high LAI have high rainfall interception (Aston, 1979; Marin *et al.*, 2000; Fleischbein *et al.*, 2005) and several studies pointed out that LAI has a significant negative effect on throughfall kinetic energy and thus soil erosion at different forest stages in the subtropical part of China (Zhang *et al.*, 2011; Geißler *et al.*, 2013; Seitz *et al.*, 2016; Song *et al.*, 2018). These results are in line with our finding that increasing LAI over time significantly reduced soil erosion. As one key factor on soil erosion, LAI at runoff plot scale was increasing every year from 2013 to 2015 in the research area. This increase of LAI strengthened the interception of rainfall and modified water fluxes. At the same time, tree species richness showed a significantly positive effect on LAI. Faster increases of LAI in stands of higher tree species richness resulted in higher decrease of annual soil erosion rates. In addition to this finding, previous investigations conducted in the same study area showed that tree species with different architecture and trait forms (Geißler *et al.*, 2012a; Goebes *et al.*, 2015a; Li *et al.*, 2017) such as *Sapindus saponaria, Lithocarpus glaber* and *Schima superba* planted in different tree species richness levels have different patterns of spatial distribution for LAI (Song *et al.*, 2018). Thus, the combination of these differing tree species planted in mixtures leading to more homogenous layering as well as the faster increase of LAI allocated from different tree species collectively optimize patterns of spatial vegetation structures and distribution (Lang *et al.*, 2012a; Lang *et al.*, 2012b; Peng *et al.*, 2016). These optimized patterns could enhance the interception of rainfall within the tree canopies, prolong the process of raindrops falling on surface soil and improve the efficiency of aboveground vegetation in reducing rainfall kinetic energy and thus leading to a faster decrease of annual soil erosion along a tree species richness gradient. Therefore, further studies on positive effects of tree species richness on LAI appear to be necessary with particular focus on underlying mechanisms within remote vegetation layers such as tree, branch and leaf traits.

Furthermore, near-surface vegetation layers such as BSC communities are of great importance for soil erosion control (Belnap and Gillette, 1997; Belnap, 2006; Belnap and Büdel, 2016). They were abundant within the study area and used the new habitat created by deforestation as pioneer vegetation in the resulting vegetation gap (Seitz *et al.*, 2017). These aggregations of biotic components including bacteria, fungi, mosses, lichens, algae and bryophytes in the topsoil (Belnap and Büdel, 2016; Bowker *et al.*, 2018) were temporally dynamic. As another main factor on soil erosion in this study, BSCs showed an increasing trend in coverage with higher tree species richness, which consequently lead to a decrease in soil erosion rates. BSCs mitigate the kinetic energy of raindrop impacts on the soil surface and stabilize the upper soil surface as well as they reduce the surface water flow (Liu and Singh, 2004; Belnap, 2006; Rodríguez-Caballero *et al.*, 2012; Gao *et al.*, 2017; Seitz *et al.*, 2017; Xiao *et al.*, 2019). Further research is necessary to understand the influence of higher diversity in tree stands on the development of near-surface vegetation layers, but also on single traits within BSC community species on raindrop impacts and interrill erosion.

Moreover, a positive relationship between BSCs and LAI was detected based on six years of field observations in the BEF China experiment from 2010 to 2015 (Seitz *et al.*, 2017). Higher LAI in subtropical forests might contribute to humid conditions and higher interception of light (Chang *et al.*, 1991; Yan *et al.*, 2000), which is benefit for the mosses and liverworts of BSCs (Seitz *et al.*, 2017; Zhou *et al.*, 2019). With the faster increase of LAI within stands of higher tree species richness, BSCs would be accordingly enhanced. Therefore, regarding the increase of LAI and BSCs and their correlation, we assume that forest stands with higher tree species richness will reach the tolerable soil erosion rate earlier.

Thus, we recommend to consider the plantation of forest stands with higher tree diversity in this area to actively counteract soil degradation and improve ecosystem services, not only by the direct impact on the tree layer but also by influences on near-surface vegetation.

5. Conclusions

The study area in subtropical China is still suffering from severe soil erosion even after 6 years of tree growth. It could be shown that higher tree species richness leads to decreasing soil erosion by positive effects on tree canopies and surface covering BSCs. Compared to monocultures, stands with tree species richness higher than eight could reduce soil erosion by more than 30%. Therefore, plantations with higher diversity should be regarded as an important afforestation policy in subtropical China for soil and water conservation. Further research needs to concentrate on how tree diversity impacts soil erosion at a watershed scale, but also on how different plant traits and characteristics influence the erosion process and are in turn also influenced by diversity.

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Manuscript 3

Science of Total Enviroment
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Soil Carbon and Nitrogen Fluxes by Water Erosion in a Young Afforested Ecosystem in Subtropical China

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Abstract

Afforestation helps worldwide not only to increase the production of timber but also to enhance water and soil conservation and reduce atmospheric CO_2 levels. However, little research addresses the role of afforestation for carbon (C) and nitrogen (N) turnover and transport by soil erosion under forest, which is important for soil fertility and the assessment of carbon and nitrogen fluxes from soil to adjacent aquatic ecosystems as well as to the atmosphere. Based on the research platform of the BEF (Biodiversity and ecosystem functioning) China project, we measured the amount of C and N transported with sediment delivery from 550 runoff plots during 2013 and 2015. Our results show that 116.7 g C m^{-2} a^{-1} and 83 g N m^{-2} a^{-1} were transported by soil erosion between 2013 and 2015 in total. Although, eroded sediment C and N concentrations increased with forest age and were higher than in topsoils (enrichment ratio ranging from 1.2 to 2.0), annual soil C and N fluxes significantly decreased at a rate of 50% in the observed three years together with sediment delivery. Soil properties and biological soil crusts (BSCs) cover as well as plant traits mainly affect sediment C and N concentrations. This study indicates that afforestation in the earlier stage should be done with great caution as high nutrients loss by water erosion can occur. Further, our study highlights the importance of accurate estimation of sediment delivery and C and N fluxes under forest for assessing terrestrial C and N budget in afforested ecosystems.

Key words: Afforestation; Water erosion; Soil C and N fluxes; Runoff plots; BEF China

1. Introduction

Soil erosion significantly affects on-site soil properties like soil biodiversity, soil fertility, soil water holding capacity as well as off-site sediment-related problems like surface water pollution, muddy floods and eutrophication of shallow lakes (Quinton *et al.*, 2010; Gardi *et al.*, 2013; Adhikari and Hartemink, 2016; Scholten *et al.*, 2017). Further, soil erosion strongly affects global carbon cycle as it redistributes top soil material which is typically rich in soil carbon (C) and nitrogen (N) (Carpenter *et al.*, 1998; McCorkle *et al.*, 2016; García-Díaz *et al.*, 2017; Poesen, 2018; Lal, 2019). Around the world, 5.7 Pg C equivalent to 0.82% of global soil carbon stock (699 Pg in 0-30 cm soil depth) was estimated to be displaced by soil erosion every year (Lal, 2003, 2018). In China, water erosion induced 180 ± 80 Tg C equivalent to 0.41% of national topsoil C stock (43.6 Pg in topsoil) of displacement per year between 1995 and 2015 (Song *et al.*, 2005; Ni, 2013; Yue *et al.*, 2016). These results confirm the importance of soil C transported by erosion processes for the global carbon balance.

Many studies address land use change as driving factor of soil C alterations by soil erosion (Jacinthe *et al.*, 2004; Martinez-Mena *et al.*, 2008; Nadeu *et al.*, 2012). This includes afforestation which increasingly spreads in many countries (Paul *et al.*, 2002; Korkanç, 2014; Keenan *et al.*, 2015; Yosef *et al.*, 2018). China as one of the largest cultivators of forest plantations in the world has afforested approximately 90% of its forest area expansion since the last 50 years (Piao *et al.*, 2009; Hong *et al.*, 2018). Generally, increased forest cover prolongs the process of throughfall reaching soil surface by intercepting raindrops, modifying drop size and speed, and changing throughfall amount and energy (Nanko *et al.*, 2006; Geißler *et al.*, 2012b; Geißler *et al.*, 2013; Masselink *et al.*, 2016). In addition, afforestation can improve soil properties and structure such as soil water holding capacity and aggregate stability (Gol *et al.*, 2010; Korkanç, 2014; García-Díaz *et al.*, 2017) and produces litter that covers the soil surface and protects against soil erosion (Seitz *et al.*, 2015). Therefore, it is widely accepted that afforestation could reduce soil erosion (Bonan, 2008; Zhao *et al.*, 2013; Keesstra *et al.*, 2017). However, recent examples from subtropical China show that afforestation has an inconsistent effect on throughfall kinetic energy and sediment delivery (Goebes *et al.*, 2015a; Seitz *et al.*, 2016) and thus C and N fluxes. Positive or negative effects of forests for soil erosion depend on many dynamic and species specific factors such as leaf area index (LAI), BSC, tree height, spatial distribution of leafs, stand age and tree species richness (Goebes *et al.*, 2015a; Seitz *et al.*, 2016; Song *et al.*, 2018; Song *et al.*, 2019). Over time after afforestation, sediment delivery decreases (Song *et al.*, 2019). However, there is a lack of research on coupling of sediment and C and N fluxes during erosion events under different land use especially in afforested areas (Stacy *et al.*, 2015; Doetterl *et al.*, 2016).

Therefore, main objectives of this study are (i) to assess temporal changes of sediment C and N concentrations and annual soil C and N fluxes by water erosion in afforested areas and (ii) to determine which topographic features, soil properties and plant traits control such C and N fluxes.

2. Materials and methods

2.1 Study area

The study was carried out as part of the BEF China experiment, located close to Xingangshan Town, Dexing City, Jiangxi Province, PR China (29.08°–29.11° N, 117.90°–117.93° E). BEF China was established in 2009 (for more information see Bruelheide *et al.* (2014b)). As a forest experiment, it provides a platform to study afforestation impacts on erosion-induced C and N fluxes and the underlying mechanisms. Subtropical monsoon dominates the area with a mean annual temperature of 17.4℃ and a mean annual precipitation of 1635 mm which half of it falling from May to August (Goebes *et al.*, 2015b). Broadleaved forest with evergreen species is the natural vegetation (Seitz *et al.*, 2017). The hilly terrain with mean elevations of 189 m a.s.l. (site A) and 137 m a.s.l. (site B) has steep slopes with inclinations from 15° to 41° (Scholten *et al.*, 2017). The Middle and Upper Proterozoic slightly metamorphosed sedimentary bedrocks are covered from crest to valley by Regosols, Cambisols and Gleysols as main soils types (Scholten *et al.*, 2017).

2.2 Experiment design

Two parallel sites (A and B) are included in the BEF China project. After logging of commercial monocultures, forty local tree species were replanted in different species mixtures on 566 research plots (25.8 m × 25.8 m each) with 400 tree saplings per plot (1.29 m × 1.29 m each) in 2009 and 2010 (Bruelheide *et al.*, 2014a). In this study, 45 of those plots were selected (see Supplementary 1). Five micro-scale runoff plots (ROP, 0.4 m × 0.4 m, cleared from leaf litter and branches) with 20 L reservoirs were randomly installed on each research plot in 2013 to collect runoff and sediment (Seitz *et al.*, 2016). 550 valid ROP measurements during the rainy seasons from May to July were captured with 182 in 2013, 158 in 2014 and 210 in 2015, respectively. Runoff volume was collected in situ and sediment delivery was determined after sampling, drying at 40℃ and weighing (Seitz *et al.*, 2016). Dried sediment was carefully collected and grounded on a ball mill for carbon and nitrogen analysis. Sediment C and N were measured with a CN-analyzer (VARIO EL III, Elementar, Hanau, Germany). Soil properties (0-5 cm depth, soil bulk density, soil pH, soil C and N) and terrain parameters (slope and altitude) were measured for each plot (Song *et al.*, 2019). Soil surface cover including BSCs and stone cover as well as crown cover and LAI were measured every May from 2013 to 2015 at the ROP scale (Seitz *et al.*, 2017). Tree height, crown width, diameter at breast height were measured yearly (Li *et al.*, 2014a). Mean values of soil properties and plant traits during the observed three years were given in Supplementary 1. Rainfall amount and intensity was recorded for each erosion event by two climate stations on both sites (ecoTech data logger with Vaisala weather transmitter and ecoTech tipping bucket balance). Regional precipitation data were measured by the National Meteorological Information Center (NMIC) of China and the China Meteorological Administration.

2.3 Data analysis

2.3.1 Annual soil C and N fluxes

Erosive rainfall amounts was defined as a threshold of 12.7 mm rainfall (Wischmeier and Smith, 1978b). From precipitation curves of climate stations, erosive rainfall during the measurement of runoff plots (ERM, mm) in rainy seasons and annual erosive rainfall amount (AER, mm) were calculated. Then, with sediment delivery acquired from runoff plots (SDE, g m^{-2}) and sediment C and N concentrations (SCC and SNC, %), annual sediment delivery (ASD, g m^{-2} a^{-1}) and annual soil C and N fluxes (ASC and ASN, g m^{-2} a^{-1}) were calculated by Eq. 1 and Eq. 2, respectively. To illustrate C and N differences between sediment and soil, the enrichment ratio of sediment C (N) concentration to soil C (N) concentration (ER) was calculated as given in Eq. 3.

$$ASD = \left(\frac{SDE}{ERM}\right) \times AER \qquad \text{Eq. 1}$$

$$ASC(N) = ASD \times SC\left(N\right)C \qquad \text{Eq. 2}$$

$$ER = \frac{SC\left(N\right)C}{SoilC\left(N\right)} \qquad \text{Eq. 3}$$

2.3.2 Statistical methods

A one-way analysis of variance (ANOVA) and least significant difference (LSD) tests were conducted to assess temporal changes of sediment C and N concentrations and annual soil C and N fluxes as well as the effect of tree species richness. Multiple regression was used to detect significant predictors. For each multiple linear regression model, all independent variables (terrain parameters, soil properties, sediment delivery, surface cover, plant traits) were tested on normal distribution and transformed by square root when needed, and then z-scored (zero-mean normalization). Potential collinearity between independent variables was detected by the Pearson correlation coefficient. One independent variable was fitted individually in exchange to the other when their correlation coefficient was higher than $|\pm 0.7|$. Dependent variables (sediment C concentrations, sediment N concentrations, annual soil C flux and annual soil N flux) were tested on normalized distribution and annual soil C flux, annual soil N flux were square root transformed. Beta value as standard regression coefficient from multiple regression models was used to illustrate the importance of independent variables on dependent variables. All statistical analyses were performed with R 3.4.3 (R Foundation for Statistical Computing, Vienna, Austria) and SPSS 13.0 (SPSS Inc., Chicago, Illinois, USA). Graph and curve fittings were processed in Origin 8.0 (OriginLab Corporation, Northampton, USA).

3. Results

3.1 Soil carbon and nitrogen fluxes

Sediment C concentration increased every year ($p < 0.05$) with means of 4.62% in 2013, 5.07% in 2014 and 6.95% in 2015 and was significantly higher than in the topsoil over the three years ($p < 0.05$) (Fig. 1 and Supplementary 2). Topsoil C concentrations ranged from 1.93% to 4.85% with a mean of 3.08% while sediment C concentrations showed a higher variability ranging from 1.83% to 14.59% in the observed three years. The same general relations were observed for N concentrations. Mean value of N concentration in soil was 0.22% while in sediment it was 0.29%, 0.38% and 0.50% in 2013, 2014 and 2015. ER of C in sediment to soil were 1.5, 1.65 and 1.98 while for N they were 1.32, 1.73 and 2.23 in the observed three years. Topsoil C/N ratio was 13.9 and sediment C/N ratio of 2013, 2014 and 2015 were 16.4, 13.3 and 14.0, respectively.

Annual soil C and N fluxes significantly decreased every year at a rate of 50% ($p < 0.05$) (Fig. 2). The means of annual soil C and N fluxes were 116.7 g m^{-2} and 8.3 g m^{-2}, respectively. In 2013, the means of soil C and N fluxes were 203 g m^{-2} and 14 g m^{-2}, respectively. Then, they declined to 95 g m^{-2} and 7 g m^{-2} in 2014. In 2015, the means were 52 g m^{-2} and 4 g m^{-2} for soil C and N fluxes.

3.2 Influences of topography, soil properties, surface cover and plant traits on soil carbon and nitrogen fluxes

Results from multiple linear regression models showed that topography does not play a significant role for sediment C and N concentrations as well as for annual fluxes ($p > 0.05$). Soil properties (C and N concentrations), surface cover (BSC and stone cover) and plant traits (diameter at breast height, crown cover, tree height, crown width, LAI) could explain 39.7% of the variability of sediment C and N concentrations. Soil C and N positively affected sediment C and N concentrations (Beta = 0.07, $p < 0.05$) while sediment delivery showed a negative impact (Beta = -0.05, $p < 0.001$). BSC (Beta = 0.17, $p < 0.001$) and plant traits (Beta = 0.07 ~ 0.18, $p < 0.001$) had comparable positive effects on sediment C and N concentrations. Regarding annual soil C and N fluxes, 93% of the variability was explained by sediment delivery, sediment C and N concentrations, BSC and LAI (Beta = 1.02, 0.24, -0.04 and -0.03 in the model, respectively, $p < 0.01$).

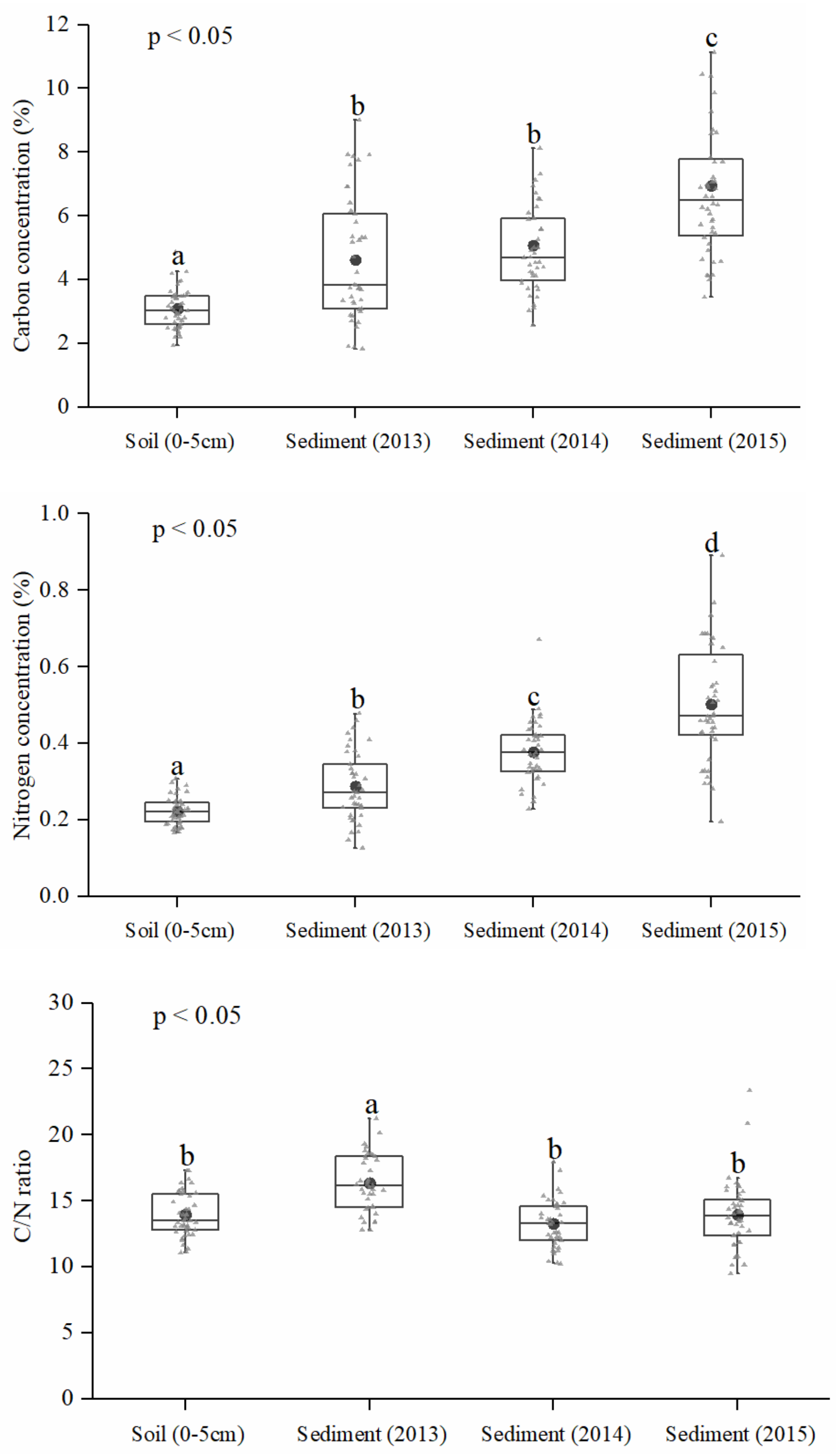

Fig. 1 C and N concentration and C/N ratio of soil sampled at 0-5 cm depth in 2014 and sediment collected in 2013, 2014 and 2015 at the BEF China experiment in Xingangshan, Jiangxi Province, PR China. Triangles represent soil C and N concentration from plots (n = 45) and sediment C and N concentration from plots (n = 45) based on 550 runoff plots measurements. Horizontal lines within boxplot represent medians and diamonds represent means. Different small letters represent mean significant differences at $p < 0.05$.

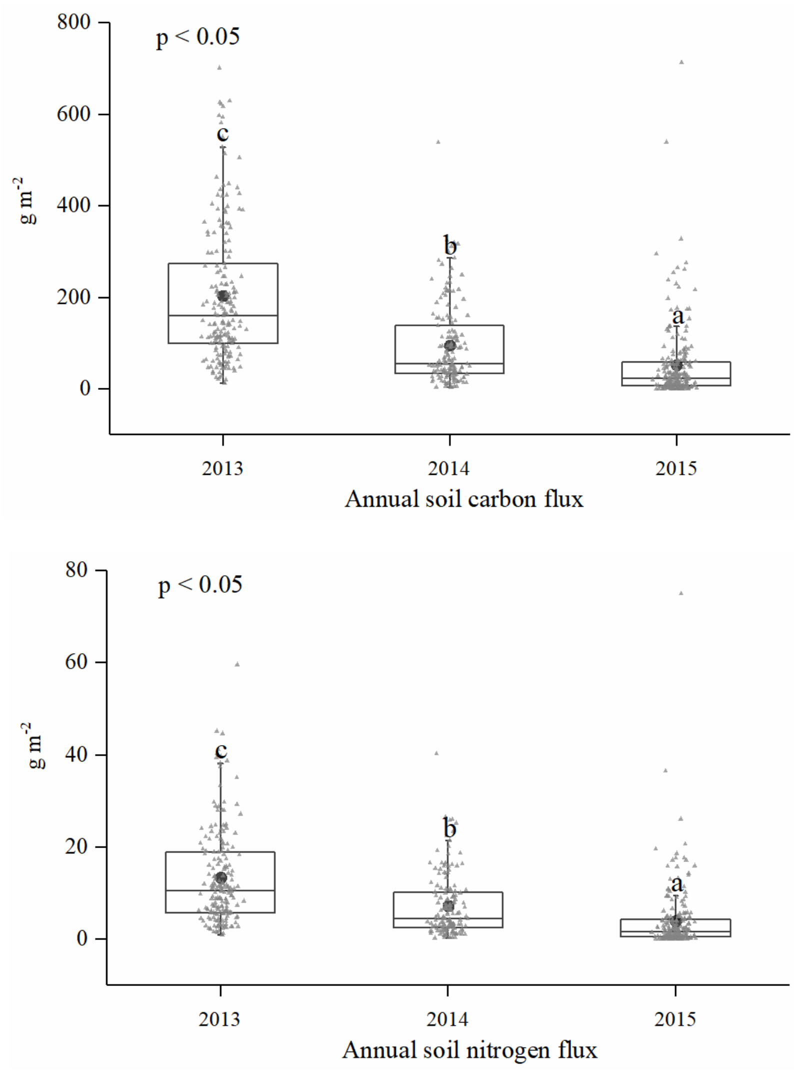

Fig. 2 Annual soil C and N fluxes in 2013, 2014 and 2015 at the BEF China experiment in Xingangshan, Jiangxi Province, PR China (n = 550). Triangles represent annual soil carbon and nitrogen fluxes from runoff plots (n = 182 in 2013, n = 158 in 2014 and n = 210 in 2015). Horizontal lines within boxplot represent medians and diamonds represent means. Different small letters mean significant differences at $p < 0.05$.

4. Discussion

4.1 Soil carbon and nitrogen fluxes

Sediment transported by water erosion are normally enriched in C and N compared to their source soils (Wang *et al.*, 2013b). Our study confirmed these findings and showed ER of C and N in sediment to soil ranging from 1.2 to 2.0 for afforested areas. This is comparable with other land use systems which show ER varied from 1.2 to 4.0. For example, in an agricultural catchment in the Belgian Loess Belt, ER was between 1.2 and 3.0 in simulated rainfall events captured by runoff plots (Wang *et al.*, 2010). The enrichment process of C and N in eroded sediments can be attributed to the preferential removal of fine particles higher in mineral-organic complexes than coarser particles like sandy grains and micro-aggregates (Palis *et al.*, 1997; Six *et al.*, 2002; Zinn *et al.*, 2007). Another aspect is the transport of unprotected young organic material from the free and/or light fraction of organic matter in soils. This fraction is easily detached by water-induced surface erosion processes (Jacinthe *et al.*, 2004; Stacy *et al.*, 2015). Before the BEF China experiment was established and since the study area was previously covered by secondary forest, the organic horizon and topsoil is likely to contain a certain amount of unprotected organic materials with higher C/N ratio than the mineral soils (Wang *et al.*, 2014b; Stacy *et al.*, 2015; McCorkle *et al.*, 2016). Therefore, eroded sediment is not only richer in C and N but also shows a higher C/N ratio in the first year. Then, with soil erosion processes ongoing, the organic horizon depleted and more mineral soil was exposed to rainfall. Thus, eroded mineral soil was the dominate part within sediments, which resulted in C/N ratio decreasing in the following years and getting closer to topsoil ratio (Fig. 3). Annual soil C and N fluxes in our study were 116.7 g m^{-2} a^{-1} and 8.3 g m^{-2} a^{-1} which is as much as in deforestation areas of the Canary Islands (Spain), with annual soil C flux caused by water erosion of 114 g m^{-2} (Rodrıguez *et al.*, 2004), but far higher than in forested areas. In the southern Sierra Nevada (USA), annual sediment C and N fluxes in forested catchments were between 0.0025 ~ 0.42 g C m^{-2} and 0.0001 ~ 0.004 g N m^{-2} during 2005-2011 (Stacy *et al.*, 2015). Severe soil C and N fluxes within BEF China in 2013 (203 g m^{-2}) mainly resulted from high annual erosive rainfall amounts and less coverage of the soil surface (LAI and BSC) (Table 2) which caused considerable sediment delivery (Song *et al.*, 2019). Although the annual erosive rainfall amount increased in 2014 and 2015, LAI and BSC as the two main factors on soil erosion increased over time and contributed to decreasing sediment delivery and associated soil C and N fluxes in this early forest stage. Besides, soil C and N fluxes caused by water erosion accounted for approximate 24% of the 0-5 cm topsoil C (1403 g m^{-2}) and N (102 g m^{-2}) (Li *et al.*, 2019), which occupied a considerable part of soil organic carbon stock. Therefore, the study suggests that deforestation and afforestation both should be implemented with caution as high nutrient losses and important differences between afforested areas and the undisturbed forest might occur in the earlier years, although temporal forest recovery can reduce soil C and N fluxes by controlling water erosion.

Table 1. Multiple linear regression of factors on sediment carbon and nitrogen concentrations and annual soil C and N fluxes. SCC: sediment carbon concentration; SNC: sediment nitrogen concentration; SC: soil carbon; SN: soil nitrogen; ASD: annual sediment delivery; BSCs: biological soil crusts; LAI: leaf area index. DBH: diameter at breast height; n.s. means no significance at $p < 0.05$; * means significance at $p < 0.05$; ** means significance at $p < 0.01$; *** means significance at $p < 0.001$. / means the variable not fitted into linear regression models.

	Sediment CN concentrations		Annual soil CN fluxes	
	C	N	C	N
Adj.R^2	0.397	0.401	0.925	0.934
Factor	Beta (Sig.)			
Slope	n.s.	n.s.	n.s.	n.s.
Altitude	n.s.	n.s.	n.s.	n.s.
SCC	/	/	0.26***	/
SNC	/	/	/	0.24***
ASD	-0.05***	-0.05***	1.02***	1.02***
SC	0.07*	/	n.s.	/
SN	/	0.08*	/	n.s.
BSCs	0.18***	0.22***	-0.04***	-0.04***
Stone cover	-0.03***	-0.03***	-0.01***	-0.01***
Surface cover	0.18***	0.23***	-0.04***	-0.04***
Tree species richness	n.s.	n.s.	n.s.	n.s.
LAI	0.07***	0.05***	-0.03***	-0.03**
DBH	0.18***	0.20***	n.s.	n.s.
Crown cover	0.03***	0.03***	n.s.	n.s.
Tree height	0.10***	0.09***	n.s.	n.s.
Crown width	0.10*	0.10*	n.s.	n.s.

4.2 *Influences on soil carbon and nitrogen fluxes*

Topography (slope and altitude) did not play a significant role for sediment C and N concentrations and annual soil C and N fluxes. This is surprising since many studies have shown that gravity driven processes of particle movement along slopes are to a large extend a function of slope angle (Wischmeier, 1965; Martz and De Jong, 1987;

Jain *et al.*, 2001; Lal, 2001; Cerdan *et al.*, 2010a; Sun *et al.*, 2014; Hancock *et al.*, 2019). One explanation is the uniform inclination ranging from 20° to 40° for all plots (Supplementary 1). Further, the small size of our runoff plots does not allow rill formation and splash erosion is the main active process of particle detachment (Seitz, 2015). Thus, overland flow, the erosive power which is mainly controlled by slope (Wischmeier, 1965; Morgan, 2009), does transport the sediment to the collector but could not contribute significantly to erode topsoil during transport over such short transport distances of max. 0.4 m. In addition, sediment C and N concentrations was found to inversely correlate with sediment delivery, which is in accordance with other research (Lal, 1976; Owens *et al.*, 2002; Nadeu *et al.*, 2012; Wang *et al.*, 2014a; Stacy *et al.*, 2015). Given a certain slope length, more carbon-rich fine aggregates are depleted in the earlier stage of interill erosion (Lal, 1976; Polyakov and Lal, 2008; Jin *et al.*, 2009; Martínez-Mena *et al.*, 2012). Therefore, with the decrease of sediment delivery in BEF China every year, associated sediment C and N concentrations were increasing. Furthermore, BSC and plant traits were another two key factors on sediment C and N concentrations (Table 1). On the one hand, BSC not only improves the labile organic carbon as it is aggregating biotic components and soil particles in the topsoil but also reducing sediment delivery (Schulten, 1985a; Eldridge, 1993; Seitz *et al.*, 2017). With increasing BSC in the research plots every year from 2010 to 2015 (Seitz *et al.*, 2017), once water erosion occurred and BSC was destroyed and detached, sediment C and N concentrations would be enhanced. This also explained the increase of sediment C and N concentrations in bare plots. On the other hand, tree growth increases litter and root production in BEF China which can protect soil from splash erosion and reduce sediment delivery (Seitz *et al.*, 2015; Huang, 2017; Sun *et al.*, 2017). Hence, sediment C and N concentrations would be enhanced with sediment delivery decreasing.

Table 2 AER, crown cover, LAI and BSC in the observed three years. (AER: annual erosive rainfall amount; LAI: leaf area index; BSCs: biological soil crusts)

Year	AER (mm)	Crown cover (%)	LAI	BSCs (%)
2013	1319	47	1.04	24
2014	1885	50	1.15	36
2015	1920	62	1.45	45

Our measurements confirm that annual soil C and N fluxes in afforested areas are strongly affected by sediment delivery, which was also shown for undisturbed forest and agriculture and grassland ecosystems (Zöbisch *et al.*, 1995; Owens *et al.*, 2002;

Wang *et al.*, 2013b; Stacy *et al.*, 2015). This means that almost no dilution effects could be observed during the erosive events and particulate transport is the main mechanism of the C and N fluxes during erosion. Moreover, dissolved organic C and N (DOC and DON) contributing to less than 10% of the soil organic C and total N is regarded as an unignorable component of eroded C and N (McClain *et al.*, 1997; Cookson *et al.*, 2007; Doetterl *et al.*, 2016; Ma *et al.*, 2018). Many studies addressed that DOC fluxes under erosion ranging from 0.2 to 5.0 g m^{-2} a^{-1} in arable areas and forests (Kindler *et al.*, 2011; Graeber *et al.*, 2012; Doetterl *et al.*, 2016). For instance, Kindler *et al.* (2011) found leaching of DOC was 3.5±1.3 g m^{-2} a^{-1} in Europe forests. Considering the higher erosion rate in BEF China and the ratio of DOC/DON in upland water (McClain *et al.*, 1997; Song *et al.*, 2019), we assume that DOC and DON fluxes would be as high as 12 g m^{-2} a^{-1} and 1.2 g m^{-2} a^{-1}, respectively. From a soil conservation perspective, our results suggest that the first years after afforestation are most important to prevent high C and N fluxes due to erosion. One possible measure is to plant shrubs and to establish BSC on bare ground.

No significant effect of tree species richness but a tendency was detected reducing soil C and N fluxes (Table 1 and Fig. 3). This inspired the thinking of how the effect of tree species richness is defined. As many researchers declare, it is difficult to identify the impact of plant diversity as it interacts with other plant factors and soil properties (Bezemer *et al.*, 2006; Pohl *et al.*, 2009; Shrestha *et al.*, 2010). In this study, BSC and LAI as the two significantly negative factors on soil C and N fluxes ($P < 0.01$, Table 1) were detected to increase with tree species richness from 2014 (Song *et al.*, 2019), which masked the effect of tree species richness. Moreover, litter fall as a significant source of soil C and vital protection of soil surface from rainfall was reported to increase with tree species richness from 2015 (Seitz *et al.*, 2015; Huang, 2017). Considering these findings, we assume that tree species richness may reduce soil C and N fluxes in the future.

Finally, measurements of soil C and N fluxes caused by water erosion in our research area need to be adapted for a potential assessment on a regional scale due to the runoff treatment and measurement plot sizes. As we could not take the whole process of soil erosion (detachment, transport, deposition and export from the watershed) into consideration, further research is needed to accurately assess sediment export at afforested watershed scales. Furthermore, as remaining leaf litter and branches were removed from the ROP before the measurements, it has to be stated that the residuals protection on topsoil would improve erosion control by further decreasing sediment delivery and elements fluxes in ROP.

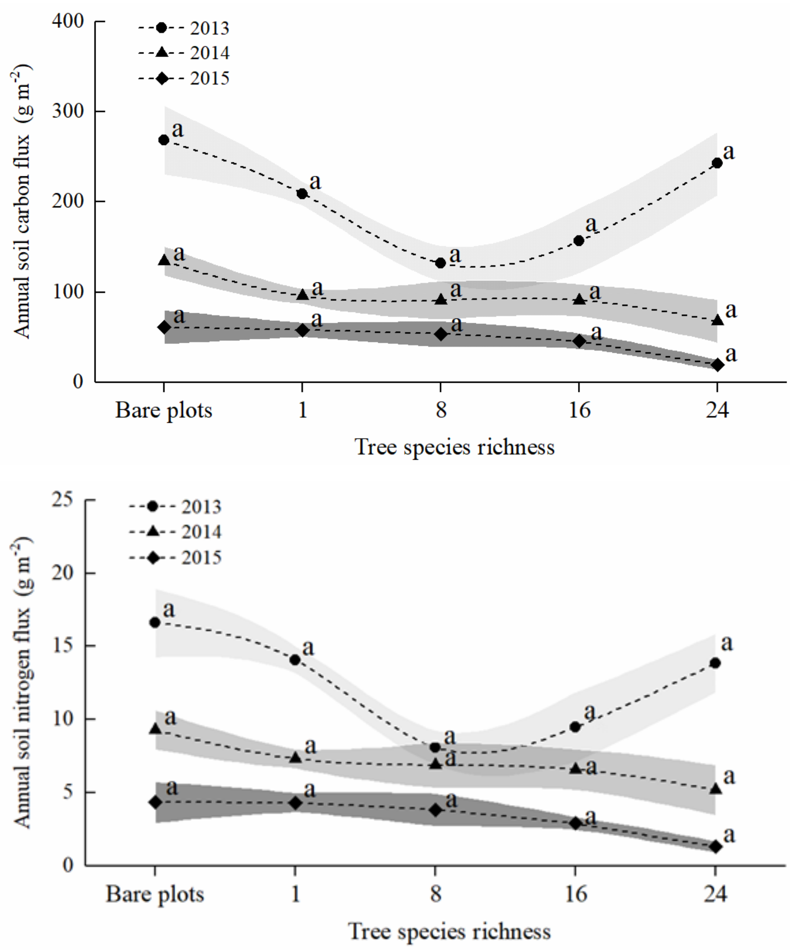

Fig. 3 Annual soil carbon and nitrogen fluxes in 2013, 2014 and 2015 under different tree species richness in the BEF China experiment in Xingangshan, Jiangxi Province, PR China. Black circles, triangles and diamonds represent mean and error bars represent standard error. Spline dashed lines connect mean values of each tree species richness for each year. Different small letters mean significant differences at $p < 0.05$.

5. Conclusions

At an early stage subtropical forest plantation in China, sediment C and N concentrations increased while annual soil C and N fluxes decreased with sediment delivery decreasing every year. Soil C and N fluxes in the study were as high as in deforestation areas even after 6 years tree growth. Therefore, afforestation areas should attract more attention as rich nutrient soil flux in the earlier years. To reduce sediment delivery especially by increasing soil surface cover such as BSCs recovery is recommended to conserve soil fertility and reduce C and N transported from soil to aquatic ecosystems and the atmosphere. In addition, soil C and N fluxes caused by water erosion as an important and dynamic component of terrestrial carbon stocks and should be taken into consideration in the C budget of afforested area.

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Supplementary 1 Information of topography, mean of soil properties (0-5 cm) and plant traits from 2013 to 2015 on 45 selected study plots at the BEF China experiment in Xingangshan, Jiangxi Province, PR China. (TSR: tree species richness; BD: soil bulk density; SOC: soil organic carbon; SN: soil nitrogen; BSCs: biological soil crusts; DBH: diameter at breast height; LAI: leaf area index. S.E: standard error)

	Topography			Soil properties (0-5cm)				BSCs
Plot	Slope (°)	Aspect	Altitude (m)	BD (g m^{-3})	pH	SOC (%)	SN (%)	(Mean ± S.E. %)
L20	24	W	229	0.86	3.68	3.96	0.23	8±3
Q23	23	N	153	0.78	3.39	3.47	0.21	39±11
D29	31	N	159	0.90	3.68	2.77	0.22	39±12
E31	22	S	144	0.95	3.86	2.48	0.19	41±8
E33	19	S	144	1.12	3.94	2.18	0.18	36±13
E34	21	S	125	1.06	4.09	2.84	0.25	25±7
G33	18	S	127	0.85	3.92	3.45	0.22	14±7
I22	28	S	119	1.07	3.58	2.28	0.21	39±12
I25	29	N	152	0.96	3.47	2.9	0.22	45±12
I28	26	S	163	0.90	3.81	3.29	0.21	49±9
J29	31	N	182	0.81	3.39	4.85	0.31	37±10
K19	24	N	199	0.80	3.70	4.18	0.3	54±8
L10	34	S	211	0.92	3.92	2.81	0.23	39±8
L11	28	S	201	1.04	3.87	2.95	0.24	34±6
L22	21	W	180	0.80	3.79	3.48	0.21	7±0
M07	31	S	129	0.89	3.55	2.52	0.18	16±4
M22	23	W	221	0.95	3.79	3.48	0.21	47±7
N02	41	S	129	0.89	3.61	3.01	0.23	36±9
N05	32	N	119	0.89	3.63	3.17	0.25	27±9
N09	33	S	218	0.86	3.58	3.6	0.23	34±8
N11	26	S	203	0.82	3.63	3.93	0.23	50±8
N13	31	S	182	0.78	3.78	3.62	0.27	34±7
N17	28	W	221	0.91	3.79	3.39	0.25	6±2
N28	19	E	167	0.97	3.56	2.21	0.19	38±10
O22	21	W	229	0.86	3.80	3.54	0.27	47±9

O27	21	W	185	1.07	4.12	2.41	0.17	22±4
Q13	30	W	215	0.90	3.86	3.84	0.28	2±1
Q17	22	N	131	0.99	3.52	2.91	0.21	25±6
Q27	35	S	160	0.97	3.66	2.78	0.19	34±9
Q29	33	E	144	0.9	3.74	2.42	0.2	17±5
R14	30	N	228	0.82	3.8	4.25	0.29	14±3
R18	36	W	215	0.91	3.82	3.50	0.27	5±4
R29	33	S	146	0.91	3.65	2.48	0.19	26±7
R30	27	S	136	0.95	3.67	2.46	0.2	41±11
S10	36	S	220	0.96	3.79	3.04	0.25	36±6
S22	33	W	145	1.00	3.61	3.26	0.24	48±9
T13	21	W	133	0.96	3.44	2.59	0.18	30±8
T15	30	N	244	0.87	3.67	3.42	0.22	27±8
U10	40	S	231	0.96	3.86	3.22	0.25	25±8
U16	20	W	147	0.94	3.44	2.65	0.17	38±8
V24	32	E	137	0.94	3.71	3.01	0.23	40±8
W10	27	E	147	0.92	3.45	2.35	0.17	63±8
W11	19	S	148	1.04	3.25	2.72	0.18	52±10
X21	24	N	132	0.91	3.65	2.54	0.19	13±5
Y09	32	E	126	1.10	3.71	1.93	0.17	19±8

Plot	TSR	Tree species	Tree height (Mean ± S.E. cm)	DBH (Mean ± S.E. cm)	Crown cover (Mean ± S.E.)	LAI (Mean ± S.E.)
L20	0	0	/	/	/	/
Q23	0	0	/	/	/	/
D29	1	M. flexuosa	N	N	N	N
E31	1	Q. fabri	160±10	0.9±0.1	0.31±0.06	0.43±0.11
E33	1	L. glaber	327±31	2.6±0.4	0.6±0.11	1.46±0.33
E34	1	C. henryi	737±36	4.6±0.2	0.92±0.01	2.66±0.16
G33	1	Q. serrata	N	N	N	N
I22	16	a	363±1	5.2±0.1	0.68±0.06	1.22±0.21

I25	1	M. yuyuanensis	204±11	2.0±0.1	0.16±0.04	0.25±0.09
I28	1	L. formosana	273±8	1.8±0.1	0.79±0.06	2.07±0.15
J29	8	b	215±19	3.4±0.3	0.66±0.07	1.44±0.29
K19	1	S. superba	486±27	3.9±0.2	0.73±0.09	2.96±0.29
L10	1	C. eyrie	174±10	1.6±0.1	N	N
L11	1	C. sclerophylla	171±6	1.5±0.1	0.76±0.06	1.89±0.23
L22	16	c	318±1	3.8±0.1	0.78±0.05	1.87±0.15
M07	1	B. luminifera	233±8	1.2±0.1	0.45±0.05	0.65±0.08
M22	16	c	351±16	3.0±0.2	0.87±0.02	2.31±0.22
N02	1	M. flexuosa	N	N	N	N
N05	1	A. altissima	124±3	1.3±0.1	0.03±0.01	0.03±0.01
N09	24	d	264±11	2.6±0.2	0.48±0.08	0.92±0.21
N11	1	S. saponaria	178±2	1.3±0.1	0.47±0.02	0.65±0.05
N13	1	S. sebiferum	364±13	3.3±0.1	0.15±0.03	0.18±0.04
N17	1	R. chinensis	198±3	1.8±0.1	0.4±0.04	0.54±0.07
N28	1	I. polycarpa	N	N	N	N
O22	1	C. myrsinaefolia	254±16	2.9±0.4	0.22±0.09	0.28±0.12
O27	1	Ch. axillaris	799±18	5.8±0.1	0.9±0.01	2.35±0.05
Q13	1	K. bipinnata	206±4	1.6±0.1	0.24±0.02	0.28±0.03
Q17	8	e	235±13	2.4±0.2	0.55±0.07	1.01±0.17
Q27	1	A. fortunei	441±21	3.5±0.2	0.72±0.04	1.38±0.11
Q29	1	M. leptophylla	71±2	1.4±0.2	0.02±0.01	0.02±0.01
R14	1	C. glauca	148±6	1.0±0.1	0.39±0.08	0.66±0.17
R18	24	d	413±1	4.3±0.1	0.85±0.04	2.13±0.18
R29	1	C. fargesii	184±10	1.3±0.1	0.14±0.03	0.16±0.04
R30	24	f	248±20	2.7±0.2	0.64±0.08	1.31±0.26
S10	8	g	492±21	4.0±0.1	0.95±0.01	3.29±0.20
S22	16	a	140±7	1.8±0.2	0.38±0.09	0.66±0.17
T13	1	M. thunbergii	N	N	N	N
T15	8	h	234±11	2.4±0.2	0.38±0.06	0.60±0.08
U10	16	c	386±10	4.1±0.1	0.8±0.07	2.16±0.46
U16	1	E. japonicus	259±9	2.4±0.1	0.35±0.07	0.61±0.14
V24	1	E. chinensis	323±22	3.0±0.1	0.62±0.07	1.32±0.11

W10	1	Ph. bournei	N	N	N	N
W11	1	E. glabripetalus	285±7	2.8±0.1	0.64±0.08	1.48±0.12
X21	1	M. grijsii	N	N	N	N
Y09	1	C. biondii	N	N	N	N

a: 16 tree species from monocultures N05, Y9, W11, U16, N28, X21, D29, W10, Q27, M07, R29, V24, Q29, T13, I25 and Q. phillyreoides.

b: 8 tree species from monocultures N05, Y9, W11, U16, N28, X21, D29 and W10.

c: 16 tree species from monocultures E34, L11, O27, I28, G33, N11, N13, E33, E31, N17, K19, L10, R14, O22, Q13 and N. sinensis.

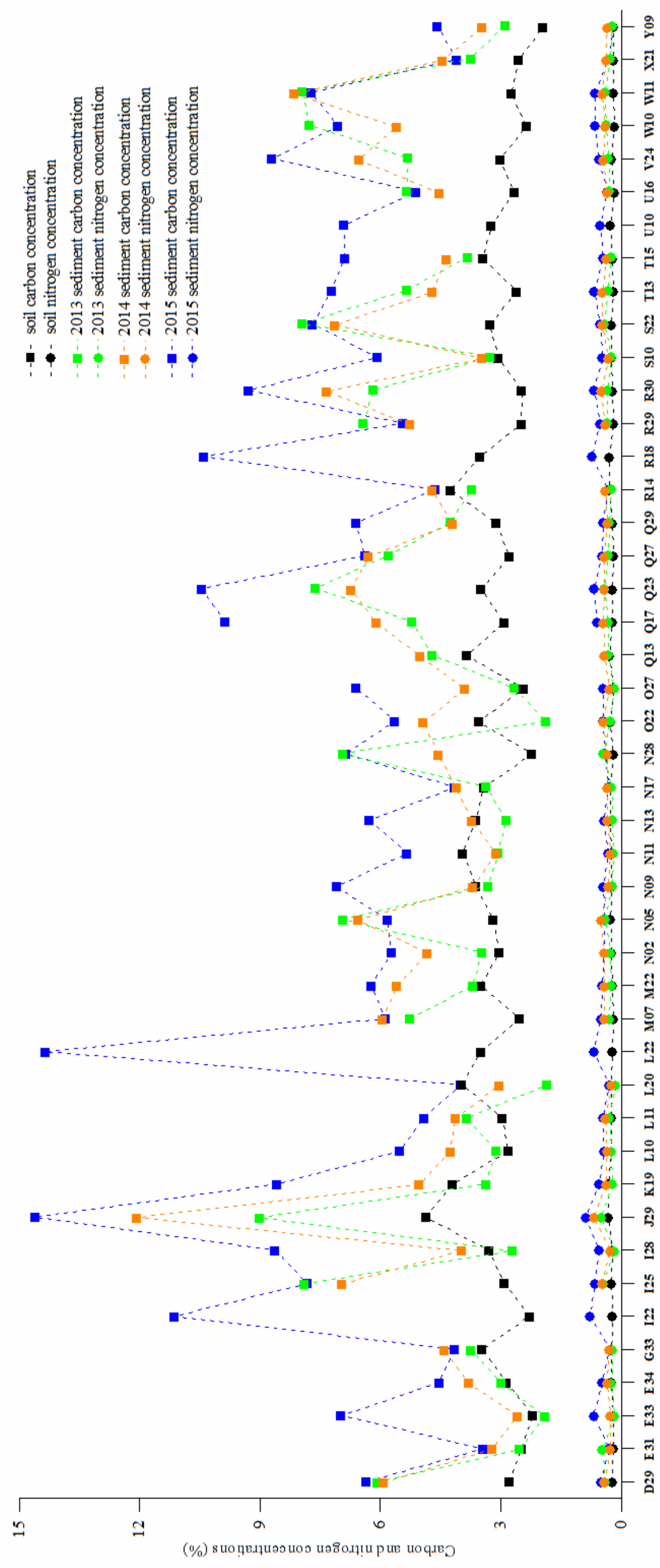
d: 24 tree species from c and *Cinnamomum camphora*, *Daphniphyllum oldhamii*, *Diospyros glaucifolia*, *Acer davidii*, *Castanopsis carlesii*, *Melia azedarach*, *Quercus acutissima* and *Sapium discolor*.

e: 8 tree species from monocultures Q27, M07, R29, V24, Q29, T13, I25 and *Q. phillyreoides*.

f: 24 tree species from a and monocultures E33, K19, R14, L10, L11, and *Cinnamomum camphora*, *Daphniphyllum oldhamii* and *Diospyros glaucifolia*.

g: 8 tree species from monocultures E34, L11, O27, I28, G33, N11, N13 and *N. sinensis*.

h: 8 tree species from monocultures E33, E31, N17, K19, L10, R14, O22 and Q13.



Supplementary 2 Means of carbon and nitrogen concentrations in soils and sediment sampled (0-5 cm) within 45 selected plots at the BEF China experiment in Xingangshan, Jiangxi Province, PR China.

Manuscript 4

(In Preparation)

Soil Organic Carbon Changes in the Earlier Stage of a Chinese Subtropical Tree Diversity Experiment

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Abstract

Forests in subtropical China were undergoing great changes in the last decades, mainly caused by extensive deforestation. Afforestation in turn can help to restore forest ecosystem and its related services such as soil organic carbon (SOC) storage and thus help to mitigate climate change. However, afforestation shows an inconsistent effect on SOC. In this respect, SOC changes in the earlier stage of afforestation and the driving factors on this process are still unclear. Therefore, based on a biodiversity and ecosystem functioning project in China (BEF China), soil profiles of 132 plots including 113 afforested plots, 14 failed afforested plots and 5 bare plots at five increments (0-5 cm, 5-10 cm, 10-20 cm, 20-30 cm, 30-50 cm) were surveyed in 2010 and 2014 to assess changes of SOC stocks and the driving factors. Results showed that SOC stock in 0-20 cm significantly decreased at a rate of 137 g m^{-2} a^{-1} in afforested areas while in deeper soils (20-50 cm) SOC stock suggested a slight difference between 2010 and 2014. These SOC stock changes along soil profiles were strongly negatively related with original SOC. However, geomorghy regarded as an important factor on SOC stock played a significant role only in the deeper soil. Furthermore, other factors such as tree species, tree species richness, aboveground biomass (AGB), belowground biomass (BGB), litter fall and soil erosion showed negligible effects on SOC stock changes. In addition, we found that approximately 274 Mg SOC reduction in total was caused in the earlier stage of afforestation in BEF China. The majority of this SOC reduction came from topsoils (0-20 cm). Hence, the study highlights afforestation in deforestation area contributes to atmospheric carbon accumulation in the first years and the original SOC could be an important parameter in modelling afforested ecosystem carbon balance in subtropical China.

Key words: afforestation, original SOC, tree species, biomass, soil erosion, topography

1. Introduction

Forests are undergoing great changes globally (Bonan, 2008; Smith *et al.*, 2016). According to the data of FAO (2015), the natural forest area declined about 240 Mha between 1990 and 2015 while planted forest increased by 110 Mha (Keenan *et al.*, 2015). Many countries make great endeavors to afforest for ecological restoration and forestry products (Paul *et al.*, 2002; Korkanç, 2014; Yosef *et al.*, 2018). For example, China is one of the largest cultivators of forest plantations in the world and its forested area was increasing by 1.5 Mha a^{-1} between 2010 and 2015 (FAO, 2015; Keenan *et al.*, 2015). Apparently, these areas attracted scientists' attentions due to the impacts on forest ecosystem services such as soil properties, C storage and climate change (Piao *et al.*, 2009; Assefa *et al.*, 2017; de Araújo Filho *et al.*, 2018; Hong *et al.*, 2018; Li *et al.*, 2018b).

Soil organic carbon (SOC) as the largest pool of terrestrial organic carbon accounts for approximate 40% of the whole C stock (to 1m soil depth) in forest (Dixon *et al.*, 1994; Jobbágy and Jackson, 2000; Pan *et al.*, 2011). It is sensitive to land use changes such as deforestation and afforestation (Jandl *et al.*, 2007; Laganiere *et al.*, 2010). Deforestation could reduce SOC stock due to the decreased organic matter inputs to soil and the increased decomposition rate of soil organic matter as well as the transportation of accelerated soil erosion caused by soil disturbance (Veldkamp, 1994; Murty *et al.*, 2002; Assefa *et al.*, 2017; de Araújo Filho *et al.*, 2018; Lal, 2019). However, no consistent effect of afforestation on SOC exists. Recently, many studies address that positive or negative effects of afforestation on SOC stocks largely depend on factors such as previous land use, tree species, stand age, and site management (Paul *et al.*, 2002; Laganiere *et al.*, 2010; Shi and Cui, 2010; Li *et al.*, 2012). For instance, a tendency of an initial loss in SOC is detected in the first few years of afforestation on former grassland where soils are rich in SOC (Paul *et al.*, 2002; Laganiere *et al.*, 2010; Shi and Cui, 2010). Therefore, influences of afforestation on SOC stocks in the earlier stage cannot be neglected due to the potential source of atmospheric CO_2 and its large areas around the world. However, research on this process do not attract enough attention.

Subtropical China with 70% coverage of mountains accounts for one quarter of the country land but has a half of the whole population (Wang *et al.*, 2007; Bruelheide *et al.*, 2014b). Due to economic benefits and ecology restoration measures, deforestation and afforestation are often occurring in this area. These intensive anthropogenic modifications can influence SOC stocks in forests. Therefore, our research was conducted to 1. quantify SOC stock changes and 2. detect driving factors on the process of the SOC stock changes in the earlier stage of afforestation after deforestation.

2. Materials and Methods

2.1 Study area

The study area was in the platform of biodiversity and ecosystem functioning project (BEF China), which is located in Xingangshan Town, Dexing City, Jiangxi Province, PR China (29.08°–29.11° N, 117.90°–117.93° E). The dominated climate is subtropical monsoon with mean annual temperature of 17.4 ^{O}C and mean annual precipitation of 1821 mm (Yang *et al.*, 2013; Goebes *et al.*, 2015b). The area is hilly with mean elevations of 189 m a.s.l. (Site A) and 137 m a.s.l. (Site B) (Scholten *et al.*, 2017). Main soil types are Cambisols, Acrisols and Ferralsols (Scholten *et al.*, 2017).

BEF China includes two parallel sites (Site A and Site B, which was planted in 2009 and 2010, respectively) with an area of 50 ha (Bruelheide *et al.*, 2014a). The sites were established by transplanting forty broad-leaved tree species after logoff of original forest (Bruelheide *et al.*, 2014a). The forty tree species were planted in monocultures and mixtures of 2, 4, 8, 16 and 24 species on 566 plots with each measuring 25.8 × 25.8 m (667 m^2) (Bruelheide *et al.*, 2014a). For each plot, 400 tree individuals were planted in 20 rows of 20 tree individuals with a planting distance of 1.29 m (Bruelheide *et al.*, 2014a).

2.2 Soil sampling

In this study, 132 of plots were selected for soil sampling (Table 1). Soil sampling was conducted in September and October in 2010 and 2014, respectively. Soil core with 6 cm in diameter was taken at a depth of 50 cm and then divided into five depth increments (0-5 cm, 5-10 cm, 10-20 cm, 20-30 cm and 30-50 cm) (Scholten *et al.*, 2017). For each plot, nine soil cores were collected (Figure 1) and mixed resulting in five soil samples. Soil samples were air-dried, sieved through a 2 mm mesh, handpicked to remove plant and animal residuals and then grounded for soil properties analysis. SOC and soil N was determined by a CN-analyzer (VARIO EL III, Elementar, Hanau, Germany) (Scholten *et al.*, 2017). Soil pH was determined in 1M KCl (Scholten *et al.*, 2017). In addition, five replicates of bulk density (BD) sample for each plot were obtained at the same depth increments as soil sampling in 2015 for soil BD determination. Soil organic carbon density (SOCD, kg m^{-2}) of five depth increments and SOC stock (0-50 cm) were calculated as given Eq. 1 and 2 (Don et al. 2009):

$$SOCD_i = T_i \times BD_i \times SOC_i \times (100 - C_i)\% \times 0.1 \qquad \text{Eq.1}$$

$$SOC\ stock_{0-50cm} = \sum_{0}^{n} SOCD_i \qquad \text{Eq.2}$$

SOCD represents soil organic carbon density (kg m^{-2}); i represents different five depth increments of 0-5 cm, 5-10 cm, 10-20 cm, 20-30 cm, 30-50 cm; T represents soil layer (cm); BD represents soil bulk density (g cm^{-3}); SOC represents soil organic content (%); C represents stone percentage (%).

2.3 Tree and litter measurement

Tree height and diameter at breast height (DBH) as two important parameters for biomass estimation were determined by the central 6 × 6 trees (36 trees) in the monocultures and 2 species plots and the central 12 × 12 trees (144 trees) in the 4, 8, 16 and 24 species mixtures (Li *et al.*, 2014a; Li *et al.*, 2017). Tree measurement of all plots started in September and October 2010 for Site A and in 2011 for Site B on a yearly base. Aboveground biomass (AGB) (kg dry mass) and belowground biomass (BGB) (kg dry mass) were calculated as given (Eq. 3 and 4) which were developed from 147 trees and 41 species in subtropical China (Xu *et al.*, 2015).

For litter measurement, 56 plots on Site A and 45 plots on Site B (Table 1) with 1, 2, 4, 8 and 16 species were selected and equipped with litter traps (Huang, 2017). Litter trap of 0.75 × 0.75 m was made of nylon nets (1 mm mesh) and fixed over a PVC frame at a height from 1 m to 1.5 m aboveground under tree canopy (Huang, 2017). For each plot, three litter traps were installed in the central area. The measurement started in March 2014 for Site A and March 2015 for Site B. Litter collection was done every month. Litter sample was put into oven and dried for 48 hours at 60 ^{o}C and weighed. The data was summed up for annual litter fall production. More details can be seen in Huang (2017). Then, biomass values (AGB, BGB and litter fall) were calculated to carbon stock using carbon conversion factor of 0.47 (IPCC) (Martin and Thomas, 2011).

$$AGB = \exp\left[-2.334 + 2.118 \times \ln(D) + 0.5436 \times \ln(H) + 0.5953 \times \ln(WD)\right] \quad \text{Eq.3}$$

$$BGB = \exp\left[-2.80346 + 2.004 \times \ln(D)\right] \quad \text{Eq.4}$$

D represents diameter at breast height (cm), H represents tree height (m), WD represents wood density (g cm^{-3}).

2.4 Soil erosion

Forty-five plots at five tree species richness levels (0sp, 1sp, 8sp, 16sp and 24sp) were selected for sediment discharge measurement on both Site A and Site B (Table 1). For each plot, five micro-scale runoff plots (ROP, 0.4 m × 0.4 m) with each of 20 L reservoir were randomly installed in 2013 (Seitz *et al.*, 2017). The measurement was operated during the rainy seasons from May to June in 2013 and 2014. Runoff volume was collected in situ and sediment discharge was calculated after sampling, drying at 40 ^{o}C and weighing (Seitz *et al.*, 2017). Then, annual erosive rainfall amount (AER, mm) and erosive rainfall during the runoff plot measurements (ERM, mm) in the rainy seasons were calculated based on precipitation curves from climate stations. Then, with sediment delivery acquired during the runoff plot measurements (SE, Mg ha^{-1}), an annual sediment delivery (ASD, Mg ha^{-1}) was calculated (Eq. 5).

$$ASD = \frac{AER}{ERM} \times SE \quad \text{Eq.5}$$

2.5 Topography

Altitude, slope, terrain ruggedness index (TRI), Monte-Carlo based flow accumulation (MCCA) were calculated (Scholten *et al.*, 2017). Moreover, 10 geomorphological units (geomorphons: flat, footslope, valley, peak, shoulder, ridge, spur, slope, pit and hollow) was computed in our study area according to the concept of openness and geomorphons (Yokoyama *et al.*, 2002; Jasiewicz and Stepinski, 2013; Scholten *et al.*, 2017). Depression, flat and valley summarized as valley while shoulder, peak and ridge summarized as ridge for further processing (Scholten *et al.*, 2017).

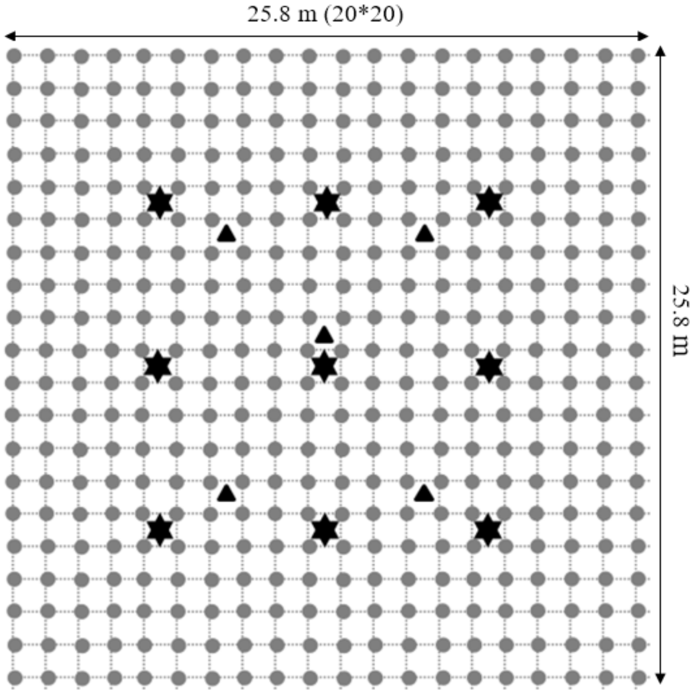

Figure 1 Positions of soil sampling for soil properties and bulk density on one plot. Grey dot means tree saplings. Black stars and triangles mean the positions of soil samples (n = 9, subsamples) and bulk density (n = 5, subsamples), respectively.

Table 1 Plots information of soil sampling, litter collection and soil erosion measurement

	Soil samples		Soil erosion		Litter collection	
Tree species richness	Site A	Site B	Site A	Site B	Site A	Site B
Bare plot	3	2	1	1	/	/
1	31	20	15	16	31	19
2	16	15	/	/	16	15
4	8	7	/	/	8	8
8	4	4	2	2	4	4
16	2	2	2	2	2	2
24	2	2	2	2	/	/
In all	63	50	22	23	61	48
	113		45		109	
Failed afforested	1	13	/	/	/	/

2.6 Statistical analysis

A one-sided, paired t-test was applied to determine the differences of SOCD, soil C/N ratio and soil pH between 2010 and 2014 at different soil depth increments. Before statistical analysis, normal distributions of variables were tested by Q-Q plot and SOCD2010 and SOCD2014 were log transformed. All the factors applied were scaled. Then, multiple regression was applied to detect the predictors of $SOCD_{2010}$, soil erosion, tree species richness, tree species, aboveground and belowground biomass, litter fall, aspect, elevation, TRI, MAAC and geomorphy on changes of $SOCD_{2014-2010}$. In the multiple regression, tree species of monocultures were set as dummy variables while geomorphy of summit ridge, spur, slope, hollow and valley in the study was set as 1, 2, 3, 4, 5 and 6, respectively. Variance inflation factors (VIFs) for each covariate in each model were calculated and lower than 3 lower (Chen *et al.*, 2017). All statistical analyses were performed with R 3.4.3 (R Foundation for Statistical Computing, Vienna, Austria) and SPSS 13.0 (SPSS Inc., Chicago, Illinois, USA). Graph and curve fittings were conducted in Origin 8.0 (OriginLab Corporation, Northampton, USA).

3. Results

3.1 Changes of SOCD after 5 years of afforestation

A significant decrease of SOCD at topsoil depth 0-20 cm was detected across the afforested plots from 2010 to 2014 (Figure 2). Means of SOCD at 0-5 cm, 5-10 cm and 10-20 cm in 2010 and 2014 were 1.69 kg m^{-2} and 1.48 kg m^{-2}, 1.25 kg m^{-2} and1.12 kg m^{-2}, 2.02 kg m^{-2} and 1.82 kg m^{-2}, respectively. The decreasing rates of SOCD were 13%, 11%, 10% at soil depth 0-5 cm, 5-10 cm, 10-20 cm. At deeper soil depth (20-50 cm), SOCD showed no significant difference between 2010 and 2014 (Figure 2).

3.2 Key factors driving SOCD changes

Multiple linear regression showed that $SOCD_{2010}$ could explain 75% to 14% of the variability of SOCD changes along soil profiles ($p < 0.001$). Geomorphy had no significant effect on SOCD changes at the topsoil but subsoils. At 10-50 cm soil depth, 7% to13% of the variabilities of SOCD could be explained by geomorphy ($R^2 = 0.07_{10\text{-}20\ cm}$, $0.09_{20\text{-}30\ cm}$ and $0.13_{30\text{-}50\ cm}$, $p < 0.001$, Table 2). Tree parameters (tree species, tree species richness, AGB, BGB and litter fall), soil erosion, aspect, elevation, TRI and MAAC did not play an important role in SOCD changes. Therefore, $SOCD_{2010}$ and geomorphy were the main factors on SOCD changes at the earlier stage of afforested area in the study.

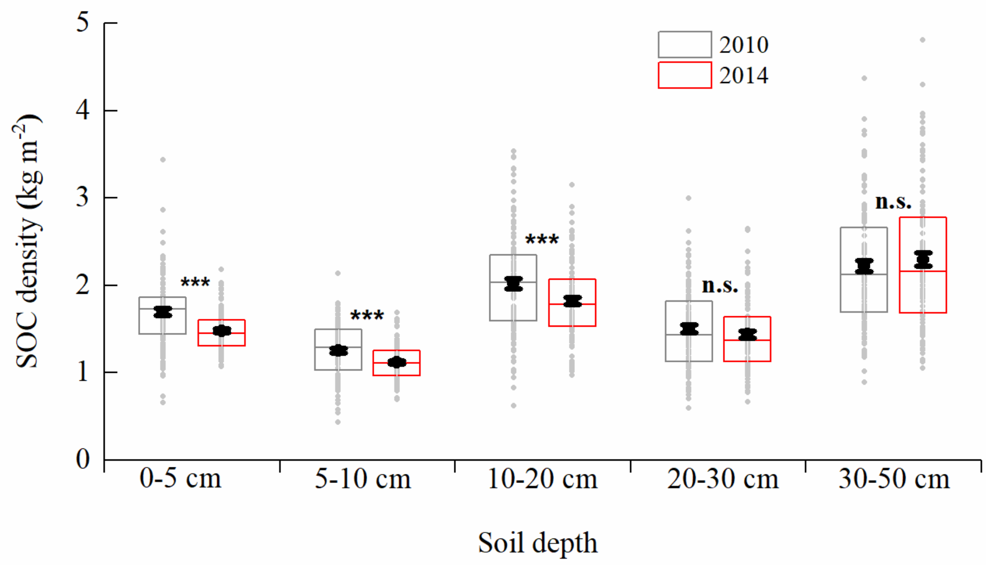

*Figure 2 SOC density at different soil depths in afforested plots in 2010 and 2014 in BEF China. Horizontal lines in boxplot represent medians and black dots represent means with standard error bars. Grey dots represent the SOC density of 113 plots and *** represent significant difference between 2010 and 2014 (paired t-tests $p < 0.001$). n.s. mean no significance at $p < 0.05$.*

Table 2 Multiple regression analysis of key factors on SOCD changes.

Factors	0-5 cm	5-10 cm	10-20 cm	20-30 cm	30-50 cm	0-50 cm
			Adj. R^2			
SOCD2010	-0.743***	-0.625***	-0.482***	-0.424***	-0.139***	-0.285***
Soil erosion	n.s.	/	/	/	/	n.s.
TSR	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.
TS	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.
AGB	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.
BGB	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.
Litter fall	n.s.	/	/	/	/	n.s.
Aspect	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.
Elevation	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.
TRI	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.
MAAC	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.
Geomorphy	n.s.	n.s.	0.065***	0.092***	0.126***	0.050***

n.s. means no significance at $p < 0.05$; *** significance at $p < 0.001$. SOCD_{2010}: soil organic carbon density of 2010. TSR: tree species richness; TS: tree species; AGB: aboveground biomass; BGB: belowground biomass; TRI: terrain ruggedness index; MCCA: Monte-Carlo based flow accumulation.

4. Discussions

4.1 SOCD changes in afforestation areas

The changes of soil C stocks depend on the balance of C inputs and outputs (Davis and Condron, 2002). Our study showed SOCD significantly decreased in the afforested areas where were covered by secondary forest in the first five years. The result was reasonable: on the one hand, soil C decomposition was still continuing and might be accelerated during site preparation and human disturbance in the first years. On the other hand, limited C inputs from litter and fine roots due to the tree saplings were flowing into soil from in the earlier stages of afforestation (Davis and Condron, 2002; Huang, 2017; Sun *et al.*, 2017). Therefore, these processes led to the decrease of soil C stock. Additionally, in the study the decreasing rate of SOC (87 g m^{-2} a^{-1} in the 0-10 cm and 137 g m^{-2} a^{-1} in the 0-20 cm) was accordingly higher than findings from other studies (Paul *et al.*, 2002; Shi and Cui, 2010; Deng *et al.*, 2016a; Moore *et al.*, 2018). For example, in afforestation areas less than < 5 years in China soil C reduced at an average rate of 20 g m^{-2} a^{-1} at 0-20 cm soil depth based on 55 observations (Shi and Cui, 2010). In another study, it could be shown that within the earlier ten years of a pasture converted to a tree plantation soil C at the 0-10 cm showed a decreasing rate of 60 g m^{-2} a^{-1} (Moore *et al.*, 2018). Meanwhile, at global scale, in the afforested agricultural areas less than < 5 years soil C in the < 10 cm layers generally decreased by 60.1 g m^{-2} a^{-1} based on 73 observations (Paul *et al.*, 2002). The higher decreasing rate of SOC in our research was mainly caused by denser tree plantations. In BEF China, the density of tree plantation (1.29 m × 1.29 m) implied more disturbance of soil and an accelerated decomposition of SOC during site preparation (Turner and Lambert, 2000; Guo and Gifford, 2002; Paul *et al.*, 2002; Turner *et al.*, 2005; Jandl *et al.*, 2007; Laganiere *et al.*, 2010; Tosi *et al.*, 2016). As shown in Figure 3A, bare plots without human disturbance almost kept the same amount of SOC stock as before afforestation along the whole soil profiles while failure afforested plots with soil disturbance had the decreased SOC. Moreover, tree growth could accelerate soil C mineralization for nutrients supply from soil. This point was enhanced by the finding that the afforested plots had a higher decreasing rate of SOC density than failure afforested plots (Figure 3). Besides, similar to tropical areas, subtropical areas has warm temperatures and moist soils as well as high soil microbial activity, which might facilitate high decomposition of organic matter, especially in soils without forest cover (Giongo *et al.*, 2011; Qiu *et al.*, 2015; de Araújo Filho *et al.*, 2018). Therefore, an appropriate density of tree plantation, lower soil disturbance and increased protection of soil surface should be considered during afforestation to reduce SOC depletion.

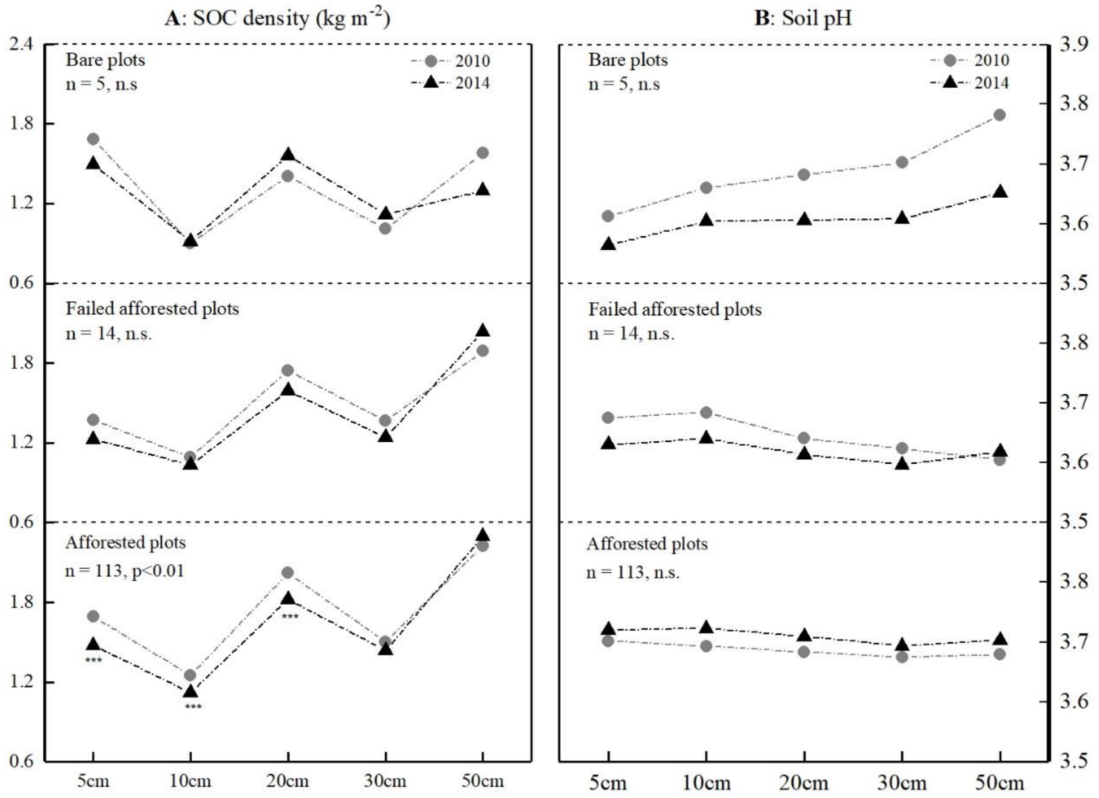

Figure 3 Means of SOC density and soil pH at different soil depths in bare, failed afforested and afforested plots in 2010 and 2014 BEF-China, respectively.

4.2 The driving factors on SOCD changes in afforestation areas

Recent studies have shown that SOC decreases in soils with high original SOC and increases in soils with lower original SOC (Garten Jr, 2002; Guo and Gifford, 2002; Paul *et al.*, 2002; Vesterdal *et al.*, 2002; Stevens and Van Wesemael, 2008; Shi and Cui, 2010; Chen *et al.*, 2017). Our results confirmed this point showing that SOC density changes in a strong negative relation with the original SOCD in 0-20 cm soil depth ($R^2 > 0.5$). One explanation might be afforestation can stimulate microbial activity and increase soil C decomposition by altering soil properties and microbial community composition (Deng *et al.*, 2016b; Pei *et al.*, 2016; Tosi *et al.*, 2016; Xu *et al.*, 2017; Hong *et al.*, 2018; Zhou *et al.*, 2018). For example, soil pH as an important index for microbial activity was found to increase after afforestation in BEF China (Figure 3). The increasing soil pH might improve total microbial biomass and the microbial activity (Pei *et al.*, 2016) and thus accelerate microbial respiration and soil C decomposition rate which led to soil C and C/N decrease. This process was illustrated in Figure 4 and Figure 5 by significant decreases of soil C/N from 2010 to 2014 and negative relationship between changes of soil C/N and $SOCD_{2010}$. In addition, the BEF China was previously covered by secondary forest and therefore in topsoil layer organic soils tended to have a high proportion of slowly decaying organic matter which mineralization rate is sensitive to temperature changes (Knorr *et al.*, 2005; Xu *et al.*, 2010; Wang *et al.*, 2013a; Li *et al.*, 2018a). Considering the elevated soil temperature after secondary forest clearance in the study area (Ma *et al.*, 2013), SOC mineralization might be accelerated and thus led to more soil C reduction. In general, topography as an important environmental factor indirectly affects SOC dynamics by soil temperature, soil moisture, soil fertility and vegetation (Raich *et al.*, 2006; Yimer *et al.*, 2006; Lybrand and Rasmussen, 2015; Tesfaye *et al.*, 2016; Tu *et al.*, 2018). In our study area, topographic heterogeneity leads to ecological gradients due to the significant relationships between geomorphological positions and soil fertility and trees survival and growth (Yang *et al.*, 2013; Scholten *et al.*, 2017). For instance, trees survival and growth increased with elevation decreasing. Therefore, hollow and valley areas with higher tree coverage might have less changes of soil temperature and moisture and then SOC was decomposed less slowly to accumulate. Moreover, our result showed topography played a stronger effect on SOC of the deeper soil layers (Table 2). This was mainly caused by: Deeper soil had less human disturbance and site preparation. And, compared with surface soils, deeper soils have a higher proportion of recalcitrant organic carbon which is sensitive to soil temperature and moisture changes caused by topography (Xu *et al.*, 2010; Wang *et al.*, 2013a).

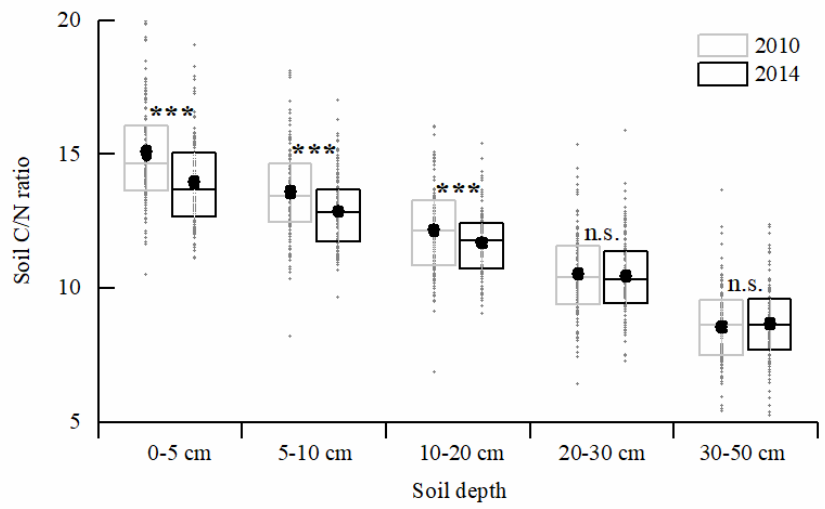

Figure 4 Soil C/N ratio at different soil depths in afforested plots of 2010 and 2014 in BEF China. Horizontal lines in boxplot represent medians and black dots represent means with standard error bars. Grey dots represent the Soil C/N ratio of 113 plots and *** represent significant difference between 2010 and 2014 (paired t-tests p < 0.001).

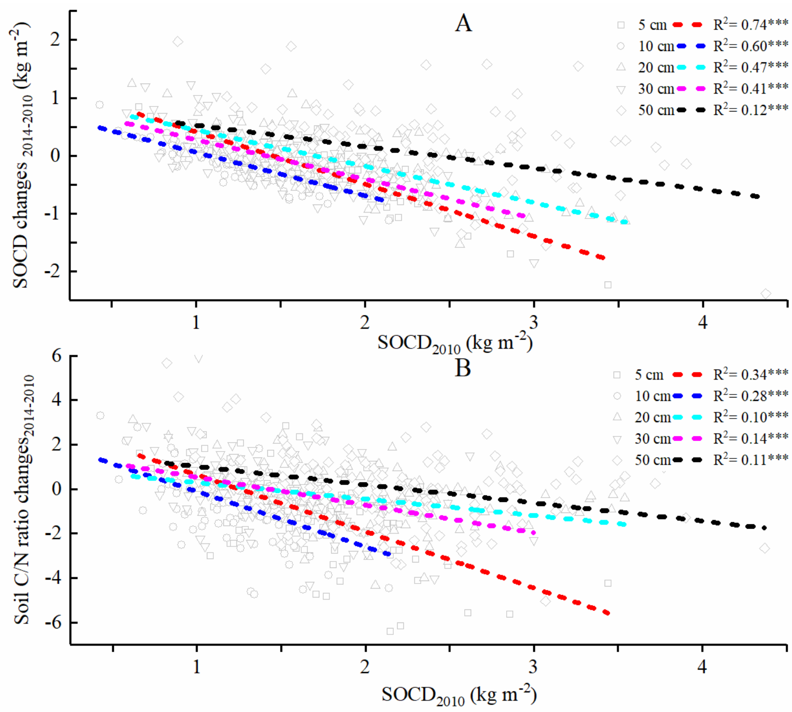

Figure 5 Relationships between SOC density changes 2014-2010 and soil C/N ratios changes 2014-2010 and 2010 SOC density in afforested plots of BEF China

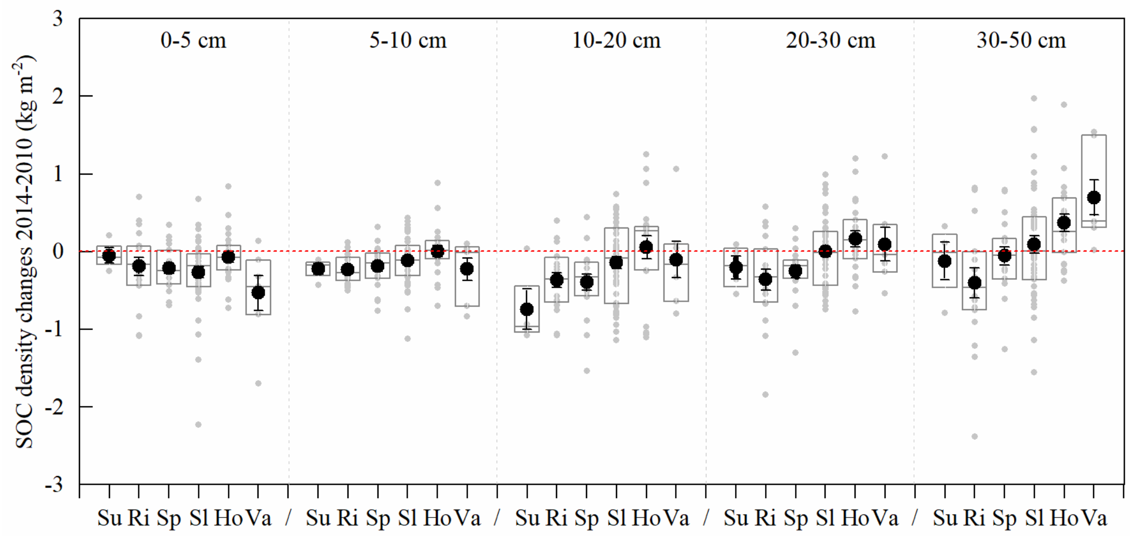

Figure 6 SOC density changes at different soil depths under six geomorphons in afforested plots of 2010 and 2014 in BEF China. Horizontal lines in boxplot represent medians and black dots represent means with standard error bars. Grey dots represent the SOC density changes of 113 plots. Su = summit (n = 4); Ri = ridge (n = 18); Sp = spur (n = 18); Sl = slope (n = 44); Ho = hollow (n = 21); Va = valley (n = 8).

Trees affect SOC mainly by C inputs from AGB and BGB such as litter and fine root (Kuzyakov and Domanski, 2000). For example, SOC in high forest productivity can be improved due to abundant C inputs from plant residence returning to soil (Dyckmans *et* *al.*, 2000; Kuzyakov and Domanski, 2000). Additionally, tree species and tree species richness can affect SOC by their impacts on the quality and quantity of litter production as well as on the transfer rate of litter to SOC (Vesterdal and Raulund-Rasmussen, 1998; Paul *et al.*, 2002; Huang *et al.*, 2017b). This point was proved by many studies showing that different tree species had different SOC changes in afforested area (Paul *et al.*, 2002; Laik *et al.*, 2009; Laganiere *et al.*, 2010; Shi and Cui, 2010). However, in our research no significant relationships existed between SOCD changes and tree species and tree species richness as well as forest biomass and litter fall (Table 2). It could be explained by that five years was too short for tree growth and C inputs from AGB and BGB transferring to SOC. As shown in Figure 7, limited C of AGB and BGB with a range of 0 to 2 kg C m^{-2} would flow into soil. For instance, litter fall as an important source of C input to soil only produced max. 0.3 kg C m^{-2} after five years of afforestation. Besides, tree not only allocates most of its biomass in the trunk but also has a slow turnover rate of its root biomass to soil (Cerri *et al.*, 1991; Kuzyakov and Domanski, 2000; Guo *et al.*, 2007; Laganiere *et al.*, 2010), which suggests the increased biomass C hardly contributes to SOC. Hence, in BEF China, the decrease of SOC could not be compensated by the increased C from forest biomass in the earlier stage of afforestation (Figure 7).

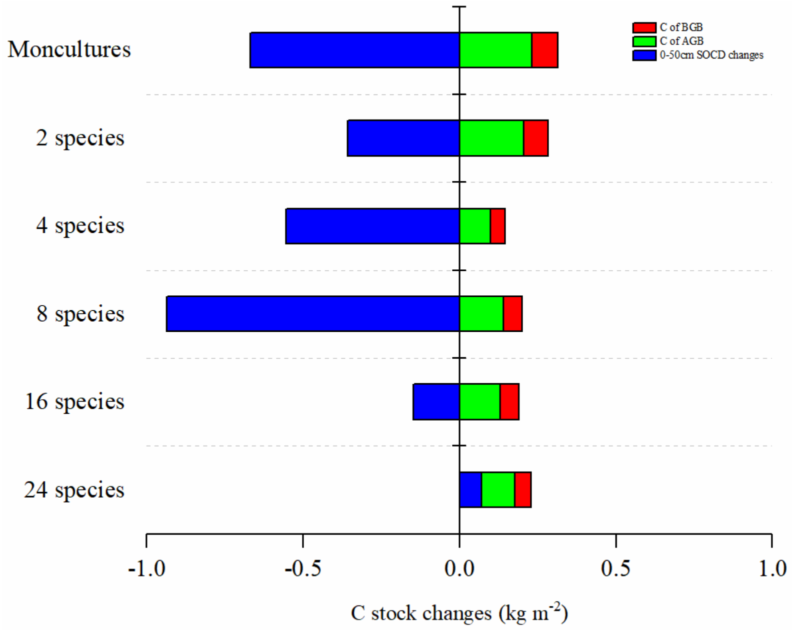

Figure 7 Carbon stocks of aboveground biomass (AGB), belowground biomass (BGB) (2015) and 0-50 cm SOCD changes 2015-2010 in BEF China.

Soil erosion did not influence SOCD changes in the study. We assume this was mainly caused by our experiment settings. On the one hand, we did not take the whole process of soil erosion into consideration (splash, transport, redistribution and deposition)

(Lal, 2003; Lal *et al.*, 2015; Lal, 2019). The operable measurement unit of ROPs in the experiment was 0.4 m ×0.4 m, that is small and limited for water erosion process. On the other hand, leaf litter and branches as a protective role against soil erosion (Seitz *et al.*, 2015), were removed from the ROPs, which is different from natural systems. Therefore, no certain relationship might be found when we linked sediment delivery at ROPs scale to SOC changes that were measured at plot scale (25 m × 25 m). In this respect, further research should be concentrated on water erosion influencing on SOC at the watershed scale.

5. Conclusions

We sampled soil profiles of 132 plots at five increments in BEF China from 2010 and 2014 to assess the changes of SOC stocks after afforestation on deforestation areas. Overall, afforestation in BEF China resulted in approximately 274 Mg SOC reduction in total in the earlier stage. Meanwhile, 90% of the total SOC reduction occurred in top-soils. In addition, afforested areas with higher original SOC stock had a higher decrease rate of SOC. Therefore, afforestation on where soils are rich in SOC should be taken seriously. Although C of forest biomass increase with tree growth, the amount of SOC stock reduction could not be compensated in the first years. Afforested areas in the earlier stage act as an atmospheric CO_2 source. Hence, further studies will be keen on how long the afforested area requires to play a role of carbon sink as well as SOC recovery at the pre-deforested level by different models.

Acknowledgments

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Manuscript 5

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On the combined effect of soil fertility and topography on tree growth in subtropical forest ecosystems—a study from SE China

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Abstract

Aims

The aim of our research was to understand small-scale effects of topography and soil fertility on tree growth in a forest biodiversity and ecosystem functioning (BEF) experiment in subtropical SE China.

Methods

Geomorphometric terrain analyses were carried out at a spatial resolution of 5×5 m. Soil samples of different depth increments and data on tree height were collected from a total of 566 plots (667 m^2 each). The soils were analyzed for carbon (soil organic carbon SOC), nitrogen, acidity, cation exchange capacity (CEC), exchangeable cations and base saturation as soil fertility attributes. All plots were classified into geomorphological units. Analyses of variance and linear regressions were applied to all terrain, soil fertility and tree growth attributes.

Important Findings

In general, young and shallow soils and relatively small differences in stable soil properties suggest that soil erosion has truncated the soils to a large extent over the whole area of the experiment. This explains the concurrently increasing CEC and SOC stocks downslope, in hollows and in valleys. However, colluvial, carbon-rich sediments are missing widely due to the convexity of the footslopes caused by uplift and removal of eroded sediments by adjacent waterways. The results showed that soil fertility is mainly influenced by topography. Monte–Carlo flow accumulation (MCCA), curvature, slope and aspect significantly affected soil fertility. Furthermore, soil fertility was affected by the different geomorphological positions on the experimental sites with ridge and spur positions showing lower exchangeable base cation contents, especially potassium (K), due to leaching. This geomorphological effect of soil fertility is most pronounced in the topsoil and decreases when considering the subsoil down to 50cm depth. Few soil fertility attributes affect tree height after 1-2 years of growth, among which C stocks proved to be most important while pH_{KCl} and CEC only played minor roles. Nevertheless, soil acidity and a high proportion of Al on the exchange complex affected tree height even after only 1-2 years growth. Hence, our study showed that forest nutrition is coupled to a recycling of litter nutrients, and does not only depend on subsequent supply of nutrients from the mineral soil. Besides soil fertility, topography affected tree height. We found that especially MCCA as indicator of water availability affected tree growth at small-scale, as well as aspect. Overall, our synthesis on the interrelation between fertility, topography and tree growth in a subtropical forest ecosystem in SE China showed that topographic heterogeneity lead to ecological gradients across geomor-

phological positions. In this respect, small-scale soil–plant interactions in a young forest can serve as a driver for the future development of vegetation and biodiversity control on soil fertility. In addition, it shows that terrain attributes should be accounted for in ecological research.

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On the combined effect of soil fertility and topography on tree growth in subtropical forest ecosystems—a study from SE China

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Abstract

Aims

The aim of our research was to understand small-scale effects of topography and soil fertility on tree growth in a forest biodiversity and ecosystem functioning (BEF) experiment in subtropical SE China.

Methods

Geomorphometric terrain analyses were carried out at a spatial resolution of 5 × 5 m. Soil samples of different depth increments and data on tree height were collected from a total of 566 plots (667 m^2 each). The soils were analyzed for carbon (soil organic carbon [SOC]), nitrogen, acidity, cation exchange capacity (CEC), exchangeable cations and base saturation as soil fertility attributes. All plots were classified into geomorphological units. Analyses of variance and linear regressions were applied to all terrain, soil fertility and tree growth attributes.

Important Findings

In general, young and shallow soils and relatively small differences in stable soil properties suggest that soil erosion has truncated the soils to a large extent over the whole area of the experiment.

This explains the concurrently increasing CEC and SOC stocks downslope, in hollows and in valleys. However, colluvial, carbon-rich sediments are missing widely due to the convexity of the footslopes caused by uplift and removal of eroded sediments by adjacent waterways. The results showed that soil fertility is mainly influenced by topography. Monte–Carlo flow accumulation (MCCA), curvature, slope and aspect significantly affected soil fertility. Furthermore, soil fertility was affected by the different geomorphological positions on the experimental sites with ridge and spur positions showing lower exchangeable base cation contents, especially potassium (K), due to leaching. This geomorphological effect of soil fertility is most pronounced in the topsoil and decreases when considering the subsoil down to 50 cm depth. Few soil fertility attributes affect tree height after 1–2 years of growth, among which C stocks proved to be most important while pH_{KCl} and CEC only played minor roles. Nevertheless, soil acidity and a high proportion of Al on the exchange complex affected tree height even after only 1–2 years growth. Hence, our study showed that forest nutrition is coupled to a recycling of litter nutrients, and does not only depend on subsequent supply of nutrients from the mineral soil. Besides soil fertility, topography affected tree height. We found that especially MCCA as indicator of water availability affected tree growth at small-scale, as well as aspect. Overall, our synthesis on the interrelation between fertility, topography and tree growth in a subtropical forest ecosystem in SE China showed that topographic heterogeneity lead to ecological gradients across geomorphological positions. In this respect, small-scale soil–plant interactions in a young forest can serve as a driver for the future development of vegetation and biodiversity control on soil fertility. In addition, it shows that terrain attributes should be accounted for in ecological research.

Keywords: soil fertility, topography, soil erosion, matter transport, biodiversity, DSM, carbon stocks, tree, forest, BEF-China, China

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INTRODUCTION

Most theories and concepts of soil formation (Glinka 1927; Hilgard 1914; Jenny 1941; McBratney *et al.* 2003) include the shape of the land surface as essential variable, which has been captured in the catena concept developed by Milne (1935). Topography as a primary terrain attribute is one of the most relevant soil-forming factors. Therefore, geomorphometric variables have been used successfully in numerous studies to predict soil attributes, soil classes and soil formation (e.g. Behrens *et al.* 2014; Hugget 1975; Pennock *et al.* 1987). With regard to soil chemical properties, e.g. Anderson and Furley (1975) found a negative effect of slope angle on soil organic carbon (SOC), nitrogen (N) and pH of topsoil horizons of Chalk soils in Berkshire and Wiltshire Downs in southern England. Wu *et al.* (2013) and Gao *et al.* (2015) found in a forest at Gutianshan National Nature Reserve that elevation of the study plots, SOC, soil moisture and total phosphorous content of the topsoil were important factors shaping the fungal community composition, and soil pH was correlated significantly to microbial biomass (Wu *et al.* 2012). The relationship between soil fertility and slope position has been described for upland soils under a tropical climate in northwest Vietnam (Clemens *et al.* 2010; Wezel *et al.* 2002) with fertile soils occurring on less eroded upper parts of hills. Concerning tree growth, the magnitude of phosphorous (P), N and K fluxes from leaf litter nutrient cycling in a tropical rain forest in Costa Rica varied significantly between Inceptisols with highest average leaf litter concentrations in valleys and Ultisols on slopes and plateaus, which showed lowest concentrations (Wood *et al.* 2006). In addition, terrain attributes were closely related to soil fertility and plant growth when Rossel *et al.* (2010) used visible near-infrared diffuse reflectance spectra of soils to develop a soil fertility index for sugarcane in Sao Paulo State, Brazil. Legendre *et al.* (2009) found in a close-by nature conservation area with comparable geomorphology that topography was a key factor explaining species richness and beta diversity.

The role of topography and soil fertility for tree growth has been described in many studies along large-scale climatic, altitudinal and topography transects (e.g. Griffiths *et al.* 2009; Hairston and Grigal 1991; Homeier *et al.* 2010). The same holds true for landscape-scale studies on the relation between terrain attributes, soil properties, soil classes and pedodiversity (Behrens *et al.* 2010a, b; Schmidt *et al.* 2008; Scholten *et al.* 1997). In general, landscapes with spatially heterogeneous abiotic site conditions provide a greater diversity of soil properties, and thus, offer more niches for different plant and animal species than homogeneous landscapes (Burnett *et al.* 1998; Schmidt *et al.* 2009). However, studies on small-scale heterogeneity of soil properties over distances of tens to hundreds of meters along slopes usually focus on crop land and precision agriculture (e.g. Qin *et al.* 2011; Blasch *et al.* 2015). Only a few studies investigated small-scale effects of elevation or slope position on decomposition (Enoki and Kawaguchi 2000; Gosz *et al.* 1973). Therefore, spatially-explicit analysis of topographic effects on soil fertility and nutrient cycling considering a large number of terrain and landform variables at different scales are rare.

It is clear that abiotic conditions, such as soil fertility, affect individual-tree growth (Baribault *et al.* 2012; van Breugel *et al.* 2011) and thus the productivity of forest stands, but also other ecosystem functions, such as nutrient cycling. More recently, the influence of biodiversity on ecosystem functions such as productivity has been studied intensively, mainly in grassland ecosystems (for recent reviews, see e.g. Cardinale *et al.* 2011; Tilman *et al.* 2014), but also in forests (Nadrowski *et al.* 2010; Scherer-Lorenzen 2014). Although several studies have documented a significant relationship between tree diversity and functions related to soil properties on a landscape scale, many studies also found strong effects of species identity (Goebes *et al.* 2015a; Li *et al.* 2017; Seitz *et al.* 2016). Such tree growth variations between tree species can be caused by differences in resource use efficiency and allocation patterns (Forrester *et al.* 2006; Riedel *et al.* 2013). However, we assume that local abiotic site conditions are very important for tree growth and may superimpose stand composition and structure (McNab 1989; McKenney and Pedlar 2003; Pretzsch and Dieler 2011; Forrester 2014). Under natural conditions, soil nutrient availability and water availability often showed a high small-scale variability (Boyden *et al.* 2012) and topography is considered to be a major controlling factor (Behrens *et al.* 2014). In this respect, terrain influences the spatial distribution of soil fertility given by SOC, soil pH, cation exchange capacity (CEC) and nutrients (e.g. Officer *et al.* 2004).

Soil fertility as such is not a technical term in soil sciences but describes a soil feature by an interchangeable set of soil properties and soil functions (Patzel *et al.* 2000). In our study, it integrates soil state variables, which characterize soil nutrient supply to plants and provides a framework to differentiate and valuate site conditions for tree growth. In our paper, we apply this framework to a biodiversity and ecosystem functioning forest experiment in subtropical China (BEF China, Bruelheide *et al.* 2011). Therefore, the main objective of this study was to investigate whether topography controls tree growth by small-scale differences of soil fertility expressed in soil texture, soil pH, SOC, N, CEC, base saturation (BS), exchangeable sodium (Na), K, Mg, Ca, Fe and Mn in a hilly forest area in subtropical China. We address three hypotheses about topographic effects on soil fertility and tree growth:

1. Topography affects soil fertility with increasing fertility from ridge to valleys, because of soil erosion processes and matter transport,
2. Individual soil fertility variables are explained by terrain attributes, and
3. Tree growth is positively influenced by soil fertility, and thus also by terrain attributes.

To test these hypotheses, soil fertility attributes and tree height were measured on two experimental sites A and B with 275 and 291 plots, respectively, in SE China, 18.4 and 20.0 ha in size, along a 200 m and 114 m elevation gradient at a spatial resolution of square plots of 667 m^2. Terrain attributes were calculated from a digital elevation model (DEM) with a spatial resolution of 5 × 5 m. BEF-China is the only biodiversity-ecosystem functioning experiment with such a large variation in topography. Thus, this is the first attempt to describe environmental heterogeneity in detail in the context of BEF research.

MATERIAL AND METHODS

Environmental settings

The research area of the BEF experiment established in a highly heterogeneous environment in subtropical China (Bruelheide *et al.* 2011) is located in SE China about 400 km west of Shanghai and situated close to the border between the two counties Dexing (Jiangxi Province) and Kaihua (Zhejiang Province). The two experimental sites A and B of the so-called Main Experiment are located close to Xingangshan Township at the eastern rim of Jiangxi Province (29°08-11 N, 117°90–93 E), China. Both sites belong to the colline altitudinal zone with mean elevations of 189 m a.s.l. (site A) and 137 m a.s.l. (site B) and a mean slope of 25° (site A) and 30° (site B).

Tectonically, both study sites are part of the Neo-Proterozoic Jiangnan belt located between the Yangtze craton in the northwest and the Cathaysia block in the SE, a Neo-Proterozoic orogenic belt (Shu and Charvet 1996) uplifted at about 1000 Ma ago. In the study area, the Middle and Upper Proterozoic sedimentary bedrocks are composed of a series of slightly metamorphosed (greenschist facies) gray-green sandstone, siltstone, and slate deposited between 1400 and 1050 Ma (Lengjiaxi group, Pt2ln) and gray–green and purplish red graywacke, siltstone, sandy slate, and slate (Banxi Group, Pt3bn) deposited between 1000 and 800 Ma ago (Gu *et al.* 2002). Due to rapid uplift of the area since the late Mesozoic (Xiao and He 2005), the structure of the fold-and-thrust belt are characterized by multifold duplexes and individual folds zoned from SE to NW with very steep to almost vertical angles of dip of the sedimentary rocks. Rock outcrops appear at shoulder positions and as spurs. The slopes are typically convex-shaped with inclinations of about <10° in the upper part and 20–35° at midslope positions with more pronounced convexity. The footslopes form the steepest part of the slope with a mean inclination of 30–40° and showed undercutting. Main drainage lines orientate along the striking lines fed by almost orthogonal tributaries that intersect the slopes.

Climatically, the Jiangxi and Zhejiang Provinces belong to the subtropics with moderately cold and dry winters and warm summers. Site A is located on a generally south facing part of a larger mountain chain. Site B is located within a smaller mountain range facing towards east and west. The mean annual temperature is 17.4°C and mean annual rainfall is 1635 mm (Yang *et al.* 2013). The climate of the study area is characterized by subtropical summer monsoon with a wet season from May to July and a dry winter (Goebes *et al.* 2015b, Seitz *et al.* 2015).

Experimental design

After the clear-felling of a *Cunninghamia lanceolata* plantation in 2008 (site A) and 2009 (site B), experimental forests were planted on a plot-level based approach (Bruelheide *et al.* 2014). In total, 40 broad-leaved tree species were planted on 566 plots on a net area of about 38 ha, each measuring 25.82 × 25.82 m (667 m^2), which corresponds to the traditional Chinese unit for area of 1 mu. Per plot, 400 tree individuals were planted in 20 rows of 20 tree individuals each, using a planting distance of 1.29 m. Species were planted in monocultures and mixtures of 2, 4, 8, 16 and 24 species. Species compositions of the different diversity levels were based on random and trait-informed (non-random) extinction scenarios. The random extinction scenarios were constructed by a broken stick design, starting from three different but overlapping sets of 16 species per site. The first set at each site was subjected to further subplot treatments by planting additional shrub species between tree positions (Very Intensively Studied Plots [VIPs]). For details on the design see Bruelheide *et al.* (2014).

Field methods

Soils were mapped and classified according to IUSS Working Group WRB (2014) and a geomorphological survey including landslides was carried out on both experimental sites (Fig. 1). Soil sampling was split into two parts: horizon-wise sampling for pedogenesis and soil classification using soil pits and schematic sampling conducted by drilling for soil physical and chemical analyses. In 2010 and 2011 on Site A and B, respectively, we sampled nine and seven key soil pits (pedons) and 275 and 291 plots. On each plot, nine soil cores (diameter of 3 cm), were taken to a depth of 50 cm and pooled. Soil cores were bulked to five depth increments (0–5, 5–10, 10–20, 20–30, 30–50 cm) resulting in five soil samples for each plot. Additionally, volumetric samples were taken on all VIP plots in 2014 and 2015 at equal depths for bulk density (BD). Tree height, which is an integral measure of growth performance and commonly used to indicate site quality for even-aged forest stands (e.g. Chen *et al.* 1998; McNab 1989), was determined for the central 6 × 6 trees in the monocultures and two-species mixtures (total = 36 trees) and the central 12 × 12 trees (total = 144 trees) in the 4-, 8-, 16- and 24-species mixtures (Li *et al.* 2014a, 2014b). Data were sampled for site A in September and October 2010 and for site B in 2011. Tree height was determined with a measuring pole as the length from stem base to the apical meristem at every plot (Li *et al.* 2014a).

Laboratory analysis

Soil sample preparation included hand sorting of coarse plant and animal residuals, sieving (<2 mm) and grinding of air-dried soil samples. Particle size analysis was done by combined pipette and sieving method (seven fractions, Koehn, DIN 19683-1) for all soil horizons sampled from

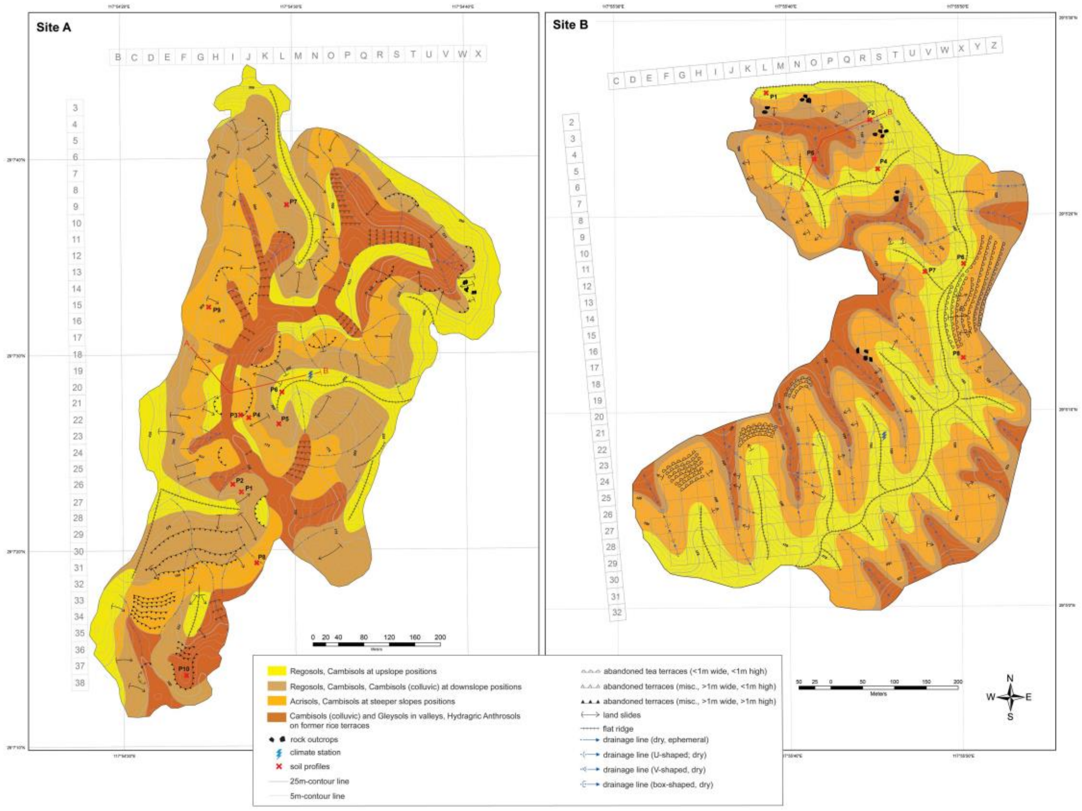

Figure 1: soil-geomorphological map of the experimental sites A (left) and B (right).

the pedons. Soil pH was measured in both 1M KCl and bidistilled H_2O potentiometrically and was determined for all plot samples. Total organic carbon (TOC) and total nitrogen (TN) were measured with a CN-analyzer (VARIO EL III, Elementar, Hanau, Germany) for all plot samples. Given the acidic soil conditions on both experimental sites, inorganic C does not occur and TOC represents the soil organic carbon content (SOC_{cont}). SOC stocks (t ha^{-1}) to a depth of 50 cm were calculated according to equation 1 (cf. Don *et al.* 2009):

$$SOC_{stocks} = \sum_{i=1}^{n} (Depth_i \times SOC_{cont} \times BD \times (1-(CM/100))) \qquad (1)$$

where $Depth_i$ is a specific depth increment (m), SOC_{cont} (g C kg^{-1}) represents the SOC content related to the increment, BD (kg m^{-3}) is the mean BD weighted by depth increment lengths, and CM (%) is the fraction of coarse material >2 mm in diameter, estimated following the German guidelines for soil description (Ad-hoc-AG Boden 2005). BD was determined gravimetrically on volumetric samples (five replicated per plot). As BD was sampled only on VIP plots, we used a Random Forest approach (Breiman 2001) to predict BD for all plots on both sites to obtain a consistent data set. CEC and concentrations of exchangeable Na, K, Mg, Ca, H and Al were measured with an ICP-OES (Perkin Elmer DV 5300 ICP OES) for sample from all VIP plots. The soil samples were percolated with an unbuffered 1 M NH_4Cl solution (effective CEC) to assess the potential fertility of the soil. BS percentage was calculated as proportion of the CEC accounted for by exchangeable bases Na, K, Mg and Ca, used as an indication for plant available base cations and soil acidification.

Terrain and landform analysis

A DEM with a cell size of 5 × 5 m was interpolated from elevation measurements with differential global positioning system (DGPS) using the ordinary kriging algorithm (Krige 1951). Based on the DEM we derived 30 terrain attributes to characterize the local, regional, climatic and complex features of the landscape. To avoid multicollinearity, we chose seven terrain attributes that (i) cover each feature of the landscape at least once, (ii) showed the highest correlation to all soil fertility indicators within each feature (averaging the absolute correlation coefficient over all soil fertility indicators and correlating this value to each terrain attribute) and (iii) are not correlated to each other with $r > 0.7$. The resulting attributes cover the local terrain attributes upstream steepest slope (USSSLP), downstream steepest slope (DSSSLP, both Tarboton 1997), and planform curvature (Zevenbergen and Thorne 1987). The heterogeneity of the terrain is described by the regional terrain attributes terrain ruggedness index (TRI, Riley *et al.* 1999) and relative richness (RR, Behrens 2003). Eastness and northness (Roberts 1986) were used to describe slope aspect indicating plant related climatic conditions. Monte–Carlo based flow accumulation (MCCA, Behrens *et al.* 2008) was used as complex terrain attribute to identify terrain driven water availability. Landform segmentation is based on the concept of openness and geomorphons (Jasiewicz and Stepinsky 2013; Yokoyama *et al.* 2002). These pattern recognition approaches compute elevation differences in the local neighborhood according to the line-of-sight principle and quantify local landform characteristics. The algorithm of Jasiewicz and Stepinsky (2013), which was used here, differentiates between 10 geomorphological units (geomorphons: flat, peak, ridge, shoulder, spur, slope, pit, valley, footslope and hollow). For further processing, we combined depression, flat and valley (resulting in valley) and shoulder, peak and ridge (resulting in ridge) since each of these single geomorphons cover a small number of 5 × 5 m cells on both experimental sites only. Geomorphons were allocated to plots using the spatial majority of one single unit within a specific plot.

Data analysis and statistical applications

Correlation analyses were done for all terrain and soil fertility attributes using the Spearman correlation coefficient. The influence of terrain attributes on soil fertility were investigated using the residuals of analysis of variance (ANOVA) models that were fitted for each soil fertility attribute used as dependent variable with tree species richness (factor) and tree composition as independent variables to account for treatment effects within the experiment. The adjusted residuals were further used to build linear models that consist of each soil fertility residual as dependent variable and all seven terrain attributes as independent variables. Model simplification was done using the stepwise backward selection method by deleting the least significant variable. CEC, K, Mn, Ca, Mg and BS were log-transformed to obtain normality. In total, we fitted 22 models for 11 soil fertility indicators on both experimental sites ($n_{\text{Site A}} = 135$, $n_{\text{Site B}} = 135$).

ANOVA models were used to test for effects of different geomorphological positions (Geomorphons, factor levels: Hollow, Spur, Ridge, Valley (only Site B) and Slope) on soil fertility attributes (dependent variable). We used the residuals of each soil fertility attribute that resulted from the models specified above which accounted for experimental treatments. In case of significant effects of geomorphons, Tukey Honest Significant Differences tests were used to distinguish between different landform segmentations factor levels. Within this approach, we fitted ANOVA models for the topsoil (0–5 cm), the deepest sampled soil depth increment (30–50 cm) and the entire soil (0–50 cm, averaged using depth increment weighted means) on both sites resulting in six models (n of each model = 135). Goodness of fit was measured as the adjusted R-squared.

To identify differences between site A and site B in soil fertility attributes, we fitted each soil fertility attribute against the two-level factor site with tree species richness as fixed and tree composition as random effects (see also Peng *et al.* 2017). To identify the influence of soil fertility and terrain attributes on tree growth, we fitted two linear mixed effect models using all soil fertility and all terrain attributes and tree species richness as fixed variables, respectively and tree species composition as random factors. In those models, tree height was log-transformed.

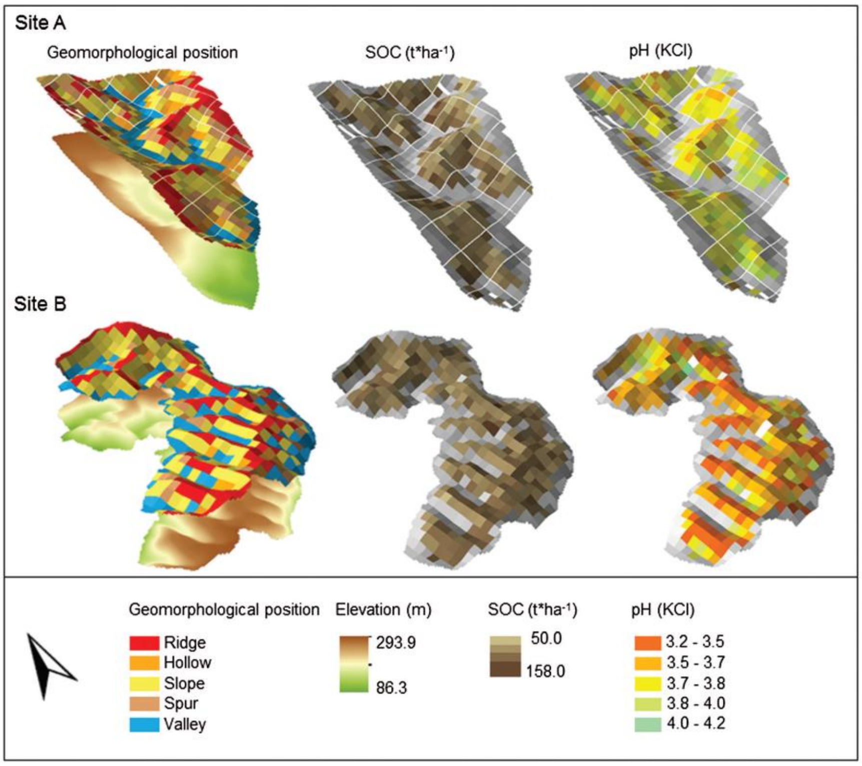

Figure 2: spatial distribution of geomorphological units, soil fertility attributes and C stocks for experimental sites A and B.

For each model, residuals met the requirements of normality and homogeneity of variances after outlier dismission due to cook's distance plots. All analyses were done using R 2.15.3 (R Development Core Team 2013) together with the 'Asreml' package to fit linear mixed effect models (Butler 2009) and the 'RandomForest' package to predict BD (Liaw and Wiener 2002).

RESULTS

Landform analysis

One of the most obvious differences between the two sites is that site A defines a valley while site B comprises a ridge (Fig. 2). The average elevation of site B is about 50 m a.s.l. lower compared to site A. In terms of standard deviation of the elevation values, site B gains only half of the relief energy. Site B showed a more structured relief and topographic heterogeneity (Fig. 2, online supplementary Fig. S1) as revealed by standard deviation (SD) alone and planform curvature cover a much larger range and, together with RR and TRI, showed a higher mean than at site A (Table 1). Furthermore, site B is more exposed to the west with mean values close to zero for northness and eastness as compared to site A with 0.18 for eastness and 0.32 for northness displaying a larger portion of NE facing slopes. MCCA is slightly higher for site B corresponding to slope length and catchment size. The plots at site A do not cover valley positions since the central valley is a swamp land and not part of the Main Experiment while site B has a number of plots in slightly inclined valley cuttings (Figs 1 and 2). The distribution of geomorphological positions across the total area given by geomorphons differs for hollows (site A: 14%, site B: 8%), ridge (site A: 13%, site B 37%), spur (site A: 24%, site B 5%) and valley (site A: 0%, site B 6%). On both sites, slope positions are dominant with 49% at site A and 44% at site B (Fig. 2).

The regular spatial distribution of rectangular experimental plots across a natural surface leads to mixing of members of different geomorphological units within one plot (Fig. 1). Site B contains more such intermediate plots that consist of more than one landform unit because the relief has a higher level of detail. With one well-defined valley situation, fewer small landslides and a larger spatial extent of homogenous surface areas, the delineation of geomorphological units per plot is more precise and unique at site A than at site B.

Table 1: terrain parameters of experimental sites A and B

	Minimum	Maximum	Mean	SD
Site A				
DSSSLP (radiants)	0.19	0.72	0.51	0.11
MCCA ($-\log_{10}$[Sum Px])	0.87	4.47	2.25	0.73
RR (%)	0	30.90	21.46	4.82
TRI (m)	0.72	3.82	2.53	0.66
Eastness (−)	−1.00	1.00	0.18	0.68
Northness (−)	−1.00	1.00	0.32	0.58
Planform curvature (rad m^{-1})	−3.04	2.83	0.25	0.99
Site B				
USSSLP (rad)	0.18	0.81	0.52	0.13
MCCA ($-\log_{10}$[Sum Px])	0.91	3.91	1.77	0.62
RR (%)	21.40	56.58	35.93	7.61
TRI (m)	1.20	4.44	2.80	0.65
Eastness (−)	−1.00	0.99	0.05	0.64
Northness (−)	−0.96	1.00	−0.04	0.57
Planform curvature (rad m^{-1})	−4.37	5.47	0.55	2.02

Abbreviations: MCCA = Monte–Carlo based flow accumulation expressed as the sum of pixels above each pixel of the DEM; TRI = topographic roughness index, planform curvature after Zevenbergen and Thorne.

Key soil profiles (pedons)

The soils cover the reference soil groups Regosols, Cambisols, Acrisols, Gleysols and Anthrosols (Fig. 1, IUSS Working Group WRB 2014), with Cambisols and Regosols on ridges, spurs and crests, Cambisols and Acrisols along slopes and colluvic Cambisols and Gleysols predominantly on footslopes and in valleys. Additionally, hydragric Anthrosols (paddy soils) are present in some valley cuttings and on lower footslopes (Fig. 1). Most soils are qualified as dystric, having a BS below 50%, and siltic with silt contents of 50.0% at site A and 43.5% at site B (online supplementary Table S1). At site A, brownish to yellowish Munsell soil colors dominated, whereas the soils at site B showed more reddish colors (online supplementary Table S1). Hydragric Anthrosols were located on abandoned rice terraces and terrace remnants indicative of past human activity. They have been modified profoundly through human activities, such as addition of organic materials or household wastes, and cultivation. Soil depth increases typically from several centimeters at steep upslope positions, on ridges and on spurs to more than 200 cm at downslope positions, in hollows and in valleys (online supplementary Table S1). The mean soil thickness, calculated as depth to the upper boundary of the C-horizon, was 66 cm at site A and more than double at site B with 143 cm.

In relation to the wide distribution of Jurassic sand and silt stones, the substrate composition, as well as the particle size distribution of all pedons on both experimental sites were quite similar having loam as the main texture class (online supplementary Table S1). Only at site A, a small NNW/SSE facing band represented by pedon 7 (online supplementary Table S1) showed distinctly lower clay and higher silt contents for all soil horizons. The main heterogeneity is related to the thickness of the soil cover (online supplementary Fig. S1) and downslope-increasing C contents, especially at site A (Fig. 2). Soil erosion led to a transport of topsoil material and soil components from ridge to valley positions. However, colluvial sediments were missing widely due to the convexity of the footslopes caused by uplift and removal of eroded sediments by adjacent waterways. Therefore, colluvial sediments occurred only in valleys and on concave footslopes connected to small valley incisions like pedons 2, 5, 8 at site B (Fig. 3). They showed higher C contents of about 1% below 50 cm depth (Table 1). Landslides are a common geomorphic feature at both experimental sites (Fig. 1) forming hollows and small spurs.

Soil fertility attributes

The soils are generally acidic at both experimental sites varying for pH_{KCl} values from 3.2 to 4.7 and pH_{H2O} from 3.9 to 6.0 (online supplementary Table S1). In general, soils at site B are more acidic than soils at site A with lower values of about 0.3 pH units (online supplementary Table S2). At site A, soil pH values showed a slight decrease with increasing elevation, whereas such a trend was not observed at site B (Fig. 2). Even though we measured nearly the same range of pH units for both experimental sites, the spatial extent of very acidic plots is much higher at site B with 37% of the area covered by ridges compared to site A with 13% only. Low pH values are typically accompanied by high exchangeable Al contents ($r = 0.7$).

Exchangeable bases were dominated by bivalent cations (online supplementary Tables S1 and S2) with maximum values of 53.2 (site A) and 52.8 $\mu mol_c\ g^{-1}$ (site B) for Ca and 32.5 (site A) and 11.0 $\mu mol_c\ g^{-1}$ (site B) for Mg. Potassium concentrations were slightly higher at site A compared to site B with a maximum of 3.2 $\mu mol_c\ g^{-1}$ for all VIP plots. Sodium was negligible with maximum values below 2 $\mu mol_c\ g^{-1}$ and a mean of 0.4 and 0.1 $\mu mol_c\ g^{-1}$ at site A and site B, respectively. Contrary to Ca and K, Mn reaches higher values at site B compared to site A while Fe does not differ between both sites. Al is the dominant cation of the exchange complex of the soils accounting for 71% of the CEC at site A and significantly more, 84%, at site B. Together with high H concentrations of 1.7 (site A) and 3.1 $\mu mol_c\ g^{-1}$ (site B), the low BS (18.8% and 8.4%, respectively) reflects strongly acidic soil conditions accompanied by a limited availability of Ca, Mg, Na and K. The CEC is almost equal at both experimental sites.

Generally, C and N contents are highest in the upper 5 cm of the soil and decrease continuously with depth at both sites (Fig. 3, online supplementary Tables S1 and S2). The pedons at site A showed slightly higher C and N contents (4.9% to 2.7% for C, 0.5% to 0.2% for N) in A horizons and within the upper 40 cm of the soil compared to site B. One outlier (site A P06, 10.7% C) may have been caused by incorporation of material from the humus layer during sampling. The mean

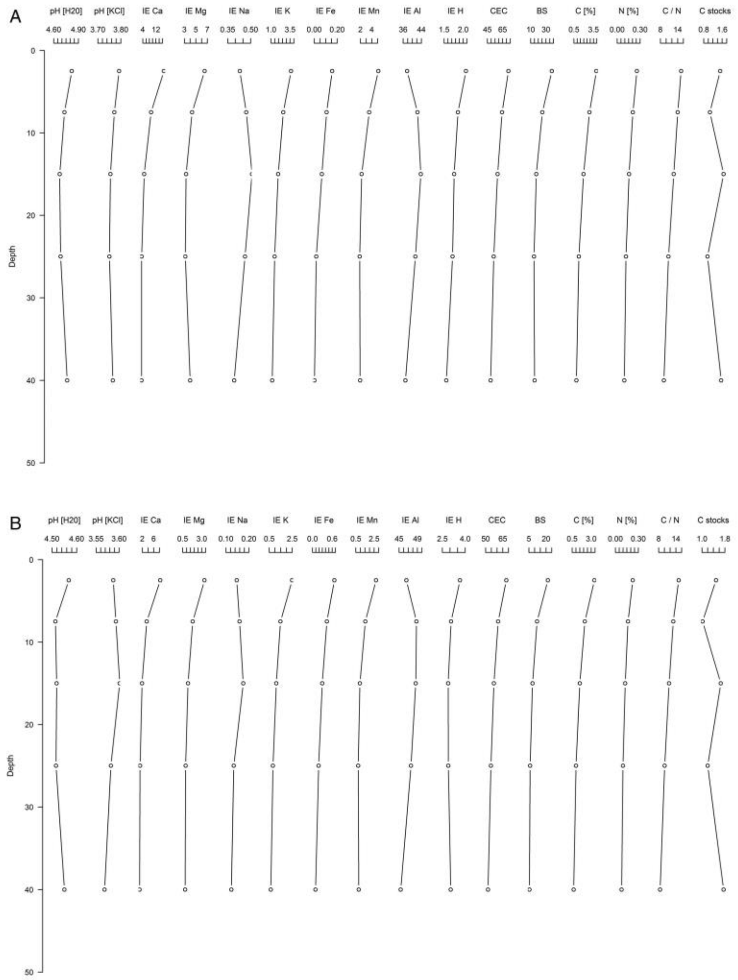

Figure 3: depth functions of soil fertility attributes on both experimental sites.

C contents of the upper 50 cm of all plots (1.7% at site A significantly higher than 1.3% P B) and a uniform BD of 1.3 g cm^{-3} results in mean C stocks of about 70.0 t ha^{-1} on both experimental sites. According to the small variability in BD, the spatial distribution of soil C content and C stocks coincide within the upper 50 cm (Fig. 2), with stocks showing an overall range from 50.0 to 150.8 t ha^{-1}. C stocks are lower on ridges and upper slopes than in hollows and valleys.

Terrain attributes and landform characteristics

Generally, all local terrain attributes showed significant relationships to soil fertility (Table 2). The majority of the 11 fertility attributes is closely related to planform curvature (eight at site A and six at site B). Slope significantly explains eight fertility attributes at site B, but only N at site A. MCCA as complex terrain attribute to identify terrain driven water availability and potential overland flow was of equal importance as planform curvature (six at site A and seven at site B). In contrast to all other fertility attribute, the spatial distribution of C/N ratio was not explained by any terrain attribute and CEC had only a weak relationship to relief at site A. Both regional terrain attributes RR (0 at site A and three at site B) and TRI (three on each site) had minor influence on the spatial distribution of soil fertility distribution. Due to the overall exposure of the experimental sites, northness played a more pronounced role at site A and eastness at site B.

Comparable to terrain attributes, typical landform segments clearly differentiate soil fertility on both experimental sites. Taking the residual of the ANOVAs (Fig. 4), ridge and spur positions were significantly different from

Table 2: results of multiple linear regressions (MLR) using soil fertility attributes as dependent and terrain attributes as independent variables for site A and site B

	pH	pH	IE Ca	IE K	IE Mn	CEC$_{eff}$	BS	C$_{org}$	N	C/N	C-stock
	H_2O	KCl	[µmol$_c$ g^{-1}]				[%]	[mass–%]			
Site A											
DSSSLP (radiants)	NS	NS	NS	NS	NS	NS	NS	NS	0.003***	NS	NS
MCCA (–log$_{10}$[Sum Px])	0.032***	0.011***	0.003**	NS	0.043***	NS	0.026***	NS	–0.003***	NS	NS
RR (%)	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
TRI (m)	NS	NS	0.026*	NS	NS	NS	0.033*	NS	0.016**	NS	NS
Eastness (–)	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
Northness (–)	NS	NS	NS	NS	NS	–0.013*	NS	–0.049*	–0.038*	NS	–0.239*
Planform curvature (radiants m^{-1})	NS	–0.015*	0.123**	–0.057***	–0.089**	NS	–0.082**	–0.068***	–0.074***	NS	–0.649***
Site B											
USSSLP (radiants)	0.005***	0.042***	NS	0.137**	0.277***	0.0009***	0.035***	0.055***	0.003***	NS	NS
MCCA (–log$_{10}$[Sum Px])	0.075***	0.051***	0.247***	0.068*	0.286***	–0.050***	0.223***	NS	0.0005**	NS	NS
RR (%)	NS	–0.023**	NS	NS	–0.126**	NS	–0.041*	NS	NS	NS	NS
TRI (m)	NS	NS	NS	–0.083***	NS	NS	–0.004***	NS	NS	0.137*	NS
Eastness (–)	0.018*	NS	0.073**	NS	0.099***	NS	0.059***	NS	NS	NS	NS
Northness (–)	NS	NS	NS	NS	NS	NS	–0.004*	NS	NS	NS	NS
Planform curvature (radiants m^{-1})	NS	NS	NS	–0.019***	0.065***	–0.009***	–0.017***	NS	–0.004*	NS	–0.212*

It was accounted for the experimental treatments tree species richness and tree species composition before fitting the MLR models for each soil fertility attribute.

*$P < 0.05$, **$P < 0.01$, ***$P < 0.001$.

Abbreviations: MCCA = Monte Carlo based flow accumulation expressed as the sum of pixels above each pixel of the DEM; NS = not significant.

all other segments, except for C/N ratios and C stocks, which were distributed evenly over all landforms (Table 3). Interestingly, slopes tended to show similar behavior for soil fertility attributes as hollows and valleys, except for K at site B.

This overall spatial pattern applies to the total upper 50 cm of the soil and was also valid for specific depth increments of CEC on both experimental sites (Table 3, CEC not affected by geomorphological position at site A, but affected at site B for all depth increments). However, all other soil fertility attributes showed a depth-specific effect, which can explain up to 48% of the spatial distribution. This relation to geomorphons is confirmed for soil pH and exchangeable K only for 0–5 cm at site A. Differences between the experimental sites were best explained by exchangeable base cations Na, Ca, Mg and BS, with higher values at site B than at site A.

Both soil fertility and terrain attributes affected tree height (Table 4). Tree height was significantly related to C stocks at both sites while pH$_{KCl}$, Mn and CEC affected tree height only at site A. MCCA and planform curvature as terrain attributes affected height growth at both sites while the geomorphological position showed an effect on tree height only at site A. At the time of this study, trees height was on average 120 cm (SD = 65.1 cm) and 74 cm (SD = 28.9 cm) at site A and site B, respectively.

DISCUSSION

Small-scale environmental gradients along land surfaces affects soil fertility

The topography of the study area reflects the geological and geomorphological history of folded sedimentary and slightly metamorphosed rocks. This might explain the much higher silt content in pedon 7 at site A representing a silty phase during sedimentation of the slates which is now exposed to the surface as narrow folding band. Joints are filled with siliceous-rich material, mainly quartz. Since slates weather easily under subtropical climate conditions and quartz is much more resistant to weathering (e.g. Scholten 1997), several cm-thick quartz veins cross the strongly isomorphic weathered slate (saprolite) shaping the large number of ridges and long spurs on both sites with 37% of the total area at site B and 42% at site B (Fig. 2). At site B, the higher values for RR and TRI compared to site A and the reddish Munsell colors of the soils (online supplementary Table S1) indicate that this site has been exposed to weathering processes for a longer time (Giaccio *et al.* 2002) supported by its lower height above sea level as a result of denudation and thus older land surface.

Typically situated at midslope positions, landslides affected substrate thickness and inclination with gently sloping flat surfaces and steep shoulders at the tear-off edge and

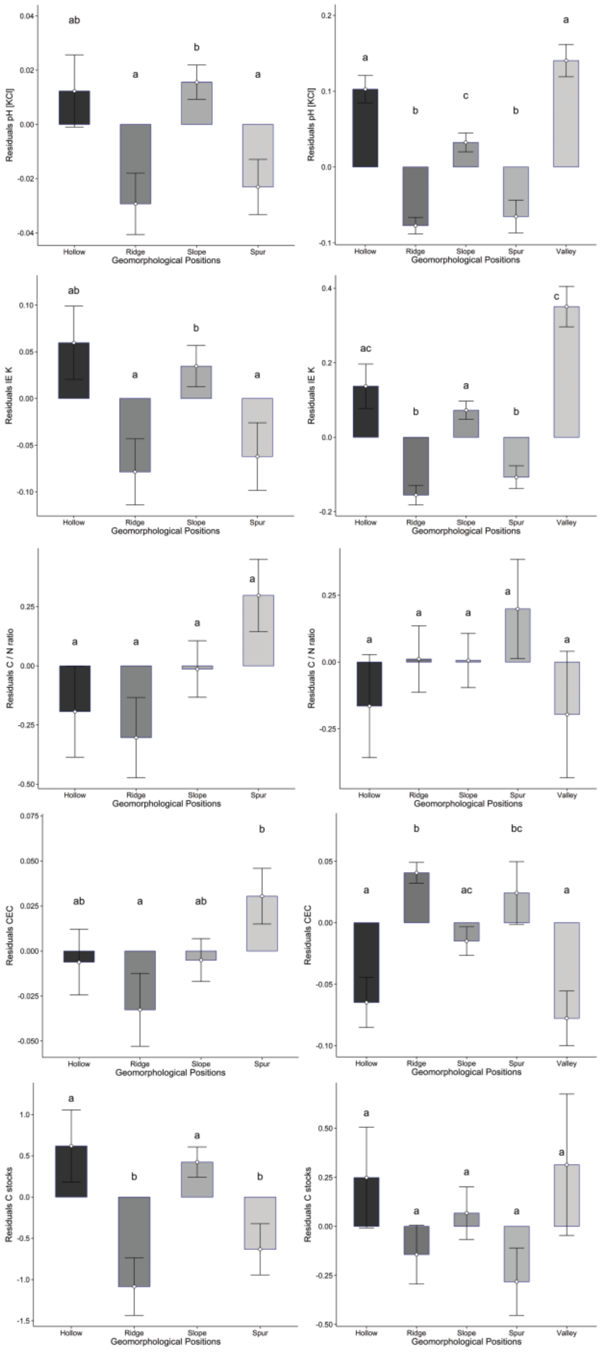

Figure 4: residuals of soil fertility attributes related to geomorphological position for both experimental sites (IE K: ion equivalent of potassium, C: carbon, N: nitrogen, CEC: cation exchange capacity).

Table 3: results of ANOVA using soil fertility attribute residuals as dependent and geomorphological units (factor with four and five different units for site A and site B, respectively) as independent variable for two depth increments and the complete profile

	Depth 0–5 cm		Depth 30–50 cm		Depth 0–50 cm	
	Geomorph. unit	Expl. SS [%]	Geomorph. unit	Expl. SS [%]	Geomorph. unit	Expl. SS [%]
Site A						
pH KCl	***	25	NS	—	**	12
pH H_20	***	19	NS	—	NS	—
IE Ca [$\mu mol_c\ g^{-1}$]	***	13	*	7	***	17
IE K [$\mu mol_c\ g^{-1}$]	***	13	NS	—	*	8
IE Mn [$\mu mol_c\ g^{-1}$]	NS	—	***	13	***	18
CEC_{eff}	NS	NS	*	7	*	5
BS [%]	***	26	NS	—	***	18
C_{org} [mass–%]	*	7	***	16	**	10
N [mass–%]	NS	—-	***	25	***	24
C/N	***	27	*	8	NS	—
C stocks	*	8	***	14	***	14
Site B						
pH KCl	***	47	***	41	***	46
pH H_2O	***	44	***	26	***	36
IE Ca [$\mu mol_c\ g^{-1}$]	***	20	***	35	***	39
IE K [$\mu mol_c\ g^{-1}$]	***	32	NS	NS	***	40
IE Mn [$\mu mol_c\ g^{-1}$]	***	36	***	43	***	48
CEC_{eff}	***	19	***	16	***	23
BS [%]	***	41	***	43	***	48
C [mass–%]	**	11	***	25	NS	—
N [mass–%]	NS	—	***	32	***	25
C/N	***	32	*	8	NS	—
C stocks	***	16	***	17	NS	—

Explained Sum of Squares (Expl. SS) were calculated as percentage of total Sum of Squares.
*$P < 0.05$, **$P < 0.01$, ***$P < 0.001$.
Abbreviations: NS = not significant; — = not calculated.

the lid of the landslide. At the scale of investigation, they interfere with slope formation caused by uplift and erosion over longer periods of time and can explain the high SD for most soil fertility attributes along slopes (online supplementary Table S2) and the irregular small-scale distribution of C at site B (Fig. 2). Also Zhang *et al.* (2012) found such a scattered spatial distribution for soil pH, C and N mainly affected by terrain convexity in a broad-leaved forest in Tiantong, Zhejiang Province, geologically belonging to the Neo-Proterozoic Jiangnan belt, as well as the experimental sites of BEF China.

Significant interrelationships between soil fertility and topography could be discovered by geomorphons (Table 3). In general, site A showed a more heterogeneous distribution of terrain attribute while site B was predominated by ridge positions (37% of the total area). Furthermore, many plots at site B belong to more than one specific geomorphon with high SD for terrain attributes. This is especially true for plots that cover both footslope and valley positions. Thus, site A showed a more precise image of how soil fertility attributes are related to terrain attributes since plot sizes of a regular grid without gaps were too large for the higher geomorphological heterogeneity at site B.

However, minor soil formation and relatively small differences in stable soil properties on both experimental sites suggest that soil erosion has truncated the soils largely over the whole area of the experiment. Soil horizonation processes such as advanced mineral weathering, clay translocation and ferralitization, which are typical for subtropical environments, are missing. Even if soil formation processes are generally proceeding at fast rates in this subtropical environment (IUSS Working Group WRB 2014), soil formation is still young and stable soil attributes like particle size distribution and BD vary only slightly (Fig. 3). Generally, the geomorphological units represent the recent function of relief for matter translocation processes rather than terrain attributes, which reflect small-scale redistribution of soil fertility attributes within such units (Table 2). The cumulated appearance of colluvic Cambisols on footslopes and weakly developed Regosols, as well as the scarce appearance

Table 4: results of the linear mixed effect model for tree height against soil fertility attributes, terrain attributes and geomorphological positions on site A and site B

	pH	IE Ca	IE Mg	IE K	IE Mn	CEC$_{eff}$	BS	N	C/N	C stocks
	KCl	[µmol$_c$ g^{-1}]					[%]	[mass–%]		
Site A	$F_{1,89} = 4.22$*	$F_{1,87} = 2.39$, NS	$F_{1,83} = 1.25$, NS	$F_{1,88} = 0.02$, NS	$F_{1,86} = 4.31$*	$F_{1,87} = 5.64$*	$F_{1,87} = 3.19$, NS	$F_{1,90} = 1.17$, NS	$F_{1,89} = 0.25$, NS	$F_{1,83} = 7.22$**
Site B	$F_{1,73} = 0.13$, NS	$F_{1,73} = 0.21$, NS	$F_{1,75} = 1.31$, NS	$F_{1,71} = 2.44$, NS	$F_{1,83} = 2.42$, NS	$F_{1,80} = 0.15$, NS	$F_{1,76} = 1.52$, NS	$F_{1,72} = 5.57$*	$F_{1,71} = 1.15$, NS	$F_{1,73} = 3.48$****
	DSSSLP/USSSLP	MCCA	RR	TRI	Eastness	Northness	Planform curvature			
Site A	$F_{1,189} = 0.50$, NS	$F_{1,178} = 5.70$*	$F_{1,184} = 1.00$, NS	$F_{1,178} = 1.10$, NS	$F_{1,186} = 0.20$, NS	$F_{1,187} = 0.80$, NS	$F_{1,173} = 11.70$***			
Site B	$F_{1,185} = 2.00$, NS	$F_{1,178} = 3.40$	$F_{1,200} = 0.80$, NS	$F_{1,199} = 0.01$, NS	$F_{1,190} = 0.90$, NS	$F_{1,195} = 6.80$**	$F_{1,182} = 8.20$**			
	Geomorphological position									
Site A	$F_{3,176} = 8.8$***									
Site B	$F_{4,190} = 1.3$, NS									

Tree species richness was used as fixed and plot composition was used as random factor.

*$P < 0.05$, **$P < 0.01$, ***$P < 0.001$, ****$P < 0.1$.

Abbreviations: MCCA = Monte Carlo based flow accumulation expressed as the sum of pixels above each pixel of the DEM; NS = not significant.

of further developed Acrisols underpin the actual influence of erosion processes. In valleys, the natural and man-made (former paddy soils from rice cultivation) influence of surface and groundwater forms gleyic properties.

A gently sloping relief with predominantly steep inclinations from 25° to 30° characterizes the study area. Many forest stands on such slopes have been cleared during the Great Leap Forward in the 1950s followed by severe soil erosion in particular in SE China and probably earlier periods of felling (e.g. Aldhous 1993; Huang 1987; Schönbrodt *et al*. 2013; Wang *et al.* 2005). The experimental sites might inherit such erosion pattern especially for SOC stocks at site A (Fig. 2). If we assume an erosion potential of 0.3–3.4 cm yr^{-1} after felling in humid subtropical regions (Jien *et al.* 2015), a mean topsoil SOC content of 2% (online supplementary Table S2), and 2 years' time between felling and soil sampling at both experimental sites, about 1.8–20.4 t ha^{-1} SOC could have been eroded since the establishment of the main experiment. In this respect, soil erosion explains the concurrently increasing CEC and SOC stocks along slopes, in hollows and in valleys, where deprotonating of carboxyl groups provides additional CEC. With an overall mean of 67.8 t ha^{-1} (site A) and 71.2 t ha^{-1} (site B) for the top 50 cm (25.9 and 25.1 t ha^{-1} for 0–10 cm and 52.7 and 53.7 for 0–30 cm, respectively), the recent SOC stocks are distinctly lower than for soils under forest in China in general. They showed 137.3 t ha^{-1} SOC for average soil depths of 75–88 cm, with 54.8 t ha^{-1} in surface soil horizons and 82.5 t ha^{-1} in subsurface soil horizons (Xie *et al.* 2007). Analyses of soils in subtropical forest plantations in China (e.g. monocultures of *Pinus massoniana*, *Castanopsis hystrix*, *Michelia macclurei* and *Mytilaria laosensis*) showed SOC stocks of 56–68 t ha^{-1} for the upper 30 cm (Wang *et al.* 2010). Subtropical hammock ecosystems at MacArthur Agro-ecological Research Centre, Florida, store about 34 t ha^{-1} (0–10 cm, Frank *et al.* 2012). Significant depth gradients and slope gradients both affected SOC stocks on slopes in *C. Lanceolata* stands in near-by Zhejiang Province, where the upper 40 cm of the soils accounted for 55% of the total C storage of 100 cm soil depth and significant differences in SOC stocks were measured for upper and lower slopes (Xue *et al.* 2012). The moderate SOC stocks at both experimental sites suggest that accumulation of SOC has taken place in a considerable amount since the last erosion events. The stable SOC turnover rate at MacArthur of 59 years suggest that the depletion of SOC in the soils of the experimental sites of BEF China by severe soil erosion cannot be explained by recent erosion events after the last felling and the establishment of the main experiment alone but should be inherited from former land use systems to a certain extent. It can be expected that the experimental forest will supply the soil with organic carbon over time at high rates through litter and fine roots (Sun *et al.* 2017; Bu *et al.* 2017). Near-by ecological service forests in Zhejiang Province stored 54 to 89 t C ha^{-1} in their biomass (Zhang *et al.* 2007).

Individual soil fertility attributes are specifically related to terrain attributes

The presence of hollows and spurs significantly affects hydrologic and sedimentary processes like hillslope discharge (O'Loughlin 1973). Since the total proportion of these landforms are equal for both sites (14% of the total area), the extent of eroded area is supposed to be similar and explain the only small differences (less than 0.5 times) between the experimental sites for C stocks, C/N ratio and CEC (Fig. 2, online supplementary Table S2). However, planform curvature or transverse curvature across slope direction covers a wider range and is much higher at site B. Discharge and erosion are more pronounced and explain the occurrence of deposition areas in valley positions as a typical geomorphological feature of site B. Higher rates of potential overland flow at site B given by MCCA support this finding. We assume that leaching and downslope interflow transported base cations downslope followed by accumulation on concave footslopes, in hollows and in valleys. This leads to distinctly lower K, Mg, Ca and Mn contents on ridge and spur positions (for K see Fig. 4). Especially at site B, exchangeable Mg and Ca is almost double in hollows and valley compared to ridge, spur and slope positions where these cations replace Al and H at the exchange complex. Further, desorption of these cations fixed on soil particles can result in significant loss of base cations from the catchment (Pacès 1985). Although a natural process in forest soils, depletion of base cations can be accelerated by harvest and leaching especially under acidic deposition (Huntington 2003), a process that was shown for the Hubbard Brook Experimental Forest (Bormann and Likens 1966) and many other forest ecosystems like the Solling Region in Germany (Matzner and Ulrich 1981) and the Strengbach catchment in NE France (Stille *et al.* 2009). However, some of the mobilized K, Ca, Mg and Mn might be absorbed by vegetation and partly returned to the soil through canopy leaching and litter decomposition as part of a closed plant-soil nutrient cycle (Likens *et al.* 1996; Perakis *et al.* 2006; Poszwa *et al.* 2000).

Among others, already Jenny (1941) stated that topography modifies the water relationships in soils to a considerable extent, and influences soil erosion and thus soil depth. Plots on ridges and spurs have a very low contributing area and limited depth and may suffer both from nutrient leaching and from water shortage during dry and hot periods of the year. Matter transport along slopes is likely for K on both experimental sites were K contents are significantly related to topography in the upper depth increment (0–5 cm), but decreased with increasing soil depth (Table 2). Such depth-dependend relationships between terrain attributes and soil nutrients were also observed for total N and P in soils of mixed forests of *Pinus tabuliformis* and *Quercus aliena* var. *accuteserrata* in Qinling Mountains (Wu 2015). The significant correlation of soil pH with MCCA at site A indicates that matter fluxes by interflow control spatial differences of soil acidity more than CEC (Table 2). Ridge and spur are higher in elevation compared to all other geomorphons of site A and site B. Leaching and downslope transport of base cations led to favorable soil conditions for tree growth in adjacent geomorphons at lower elevations. These processes explain why elevation was the only terrain attribute with a significant positive effect on seedling survival within the BEF China in June 2010 in contrast to aspect, slope and curvature as reported by Yang *et al.* (2013). TRI does not affect soil acidity since this regional terrain attribute is predominately related to structural features of the surface like the spatial distribution of quartz veins and faults, as well as the duration of weathering and soil formation rather than erosion and landslides, which are better reflected by RR. However, minimum, maximum and mean soil pH of the upper 50 cm tend to slightly lower values of about 0.2 pH units at site B compared to site A (Fig. 2, online supplementary Table S2), which might reflect the higher degree of weathering at site B as well. The low CEC (about 56 µmol$_c$ g^{-1} soil) and percent BS (<20%) at both experimental sites result in small exchangeable Ca pools, and are indicative of only slightly weathered, young mineral soil. These soils might be highly sensitivity to intensive forest harvesting practices, if most nutrient rich biomass is removed (Federer *et al.* 1989). The small depth gradients of most terrain attributes correspond to this finding and support the important role of soil erosion on both experimental sites.

Soil fertility on both experimental sites can be regarded as low when following criteria given for soil survey and agricultural land evaluation in the subtropics and tropics (Landon 1991). The soils are very acid, the CEC is low to very low with little difference over all geomorphic units, and base cations are deficient. BS is <50% on all plots emphasizing dystric properties throughout the whole experimental area. With bulk densities below 1.4 g cm^{-3} the soils are not compacted indicating that the area of the experiment has not been cultivated recently. The Ca/Al ratio of the exchange complex on both experimental sites is below 0.2 on most plots and in accordance with the very low pH values (Gruba *et al.* 2013). This can cause inhibition of Ca uptake by tree roots and the very high Al saturation of the exchange complex probably indicate Al stress to fine roots influencing tree growth (De Wit *et al.* 2010; Kinraide 2003; Marschner 1991). However, the Ca/Al rations in foliage are higher than 12.5 in most cases and BS is less than 15 only on four single plots, especially on site B, and do not indicate adverse impacts on tree growth or nutrition in general (Cronan and Grigal 1995).

Tree growth is affected by soil fertility at small-scale

The main experiment of BEF China represents a random spatial configuration of diversity treatments projected onto a heterogeneous and complex real-world landscape. Because topography and soil fertility attributes vary at the same spatial scale as the plot dimension, which could be shown by a large number of significant correlations between topography and soil fertility attributes, it is difficult to isolate the biotic signal from the environmental signal (Bruelheide *et al.* 2014; Healy *et al.* 2008). However, in the experimental framework of BEF China soil fertility and terrain affected tree height significantly (Table 4) after accounting for tree composition and tree species richness effects. Further interactions between biotic and abiotic control mechanisms were evident through altitudinal differences in survival rates of seedlings (Yang *et al.* 2013, 2017). Results from the mixed effects models for sapling growth responses at site A showed significant correlation to the local abiotic variables northness, N content and C/N ratio (Li *et al.* 2014a). However, Kröber *et al.* (2015) found only marginal effects of environmental variables on crown growth at site A, with slope being the best environmental predictor. According to our findings, soil C stocks were most closely related to tree height at this early stage of tree growth (1–2 years), whereas soil acidity, Mn and CEC were related to tree height only at site A. As trees at site A were planted 1 year earlier, we found more interactions between soil fertility and tree height growth at this experimental site. Soil acidification and high contents of exchangeable Al in the lower soil horizons could lead to restriction of nutrient uptake due to a poor replacement of base cations (Marschner 1991). Both nutrient deficiency and high Al contents can constrain fine root growth with soil depth, and cause a close dependence of tree growth on nutrient availability in topsoils, which showed higher C and N contents. Thus, plant nutrition most likely is coupled to a recycling of litter nutrients and root exudates rather than on supply of nutrients from the mineral soil and some trees might be able to bypass the common mineralization pathway by using a significant proportion of organic N as amino acids and proteins (Näsholm *et al.* 1998).

However, as shown before soil fertility was significantly affected by several terrain attributes and those additionally affect tree height. MCCA as indicator of water availability had a negative impact on tree growth, as well as the climatic terrain attribute northness showing the importance of irradiance for photosynthesis (see also Eichenberg *et al.* 2017). Planform curvature as a measure of soil erosion processes and matter transport showed that tree growth was reduced on very steep slopes. As tree heights varied more at site A, they also showed a relation to the geomorphological positions at which each tree grows. This again can be linked to erosion and accumulation processes along slopes in these two small catchments (Seitz *et al.* 2016). Generalized mixed-effects models showed that survival rates of tree seedlings were affected by species richness and negatively correlated to elevation (Yang *et al.* 2013). This can be explained by transport of base cations from ridge top and upper slopes downwards through interflow and erosion. Yang *et al.* (2017) found that tree richness did not affect shrub survival at this early stage of the experiment but single abiotic factors explained up to 5% of species survival, with a negative effect of slope inclination and a positive effect of the topsoil carbon to nitrogen ratio.

CONCLUSIONS

Our synthesis on the interrelation of soil fertility, topography and tree growth in a subtropical forest ecosystem in SE China showed that topographic heterogeneity led to ecological gradients across geomorphological positions. Although multilayered, the experimental design of BEF China with a high resolution of both terrain (5 m) and soil fertility attributes (approximately 25 m) allows to propose soil erosion and matter transport as key mechanisms for soil fertility and, thus, determine tree growth. Accordingly, we can confirm our first hypothesis. Our findings indicate low availability of exchangeable base cations and acid conditions in soils accompanied with high Al contents on both experimental sites, which could lead to limited tree growth due to insufficient soil nutrient supply. Especially plots on ridges and spurs may suffer both from nutrient leaching and from water shortage during dry and hot periods of the year. Such small-scale soil–plant interrelations in a young forest can serve as originator for the future development of vegetation and biodiversity control on soil properties in near-natural forest ecosystems. In addition, it showed that terrain attributes constitute an important predictor for the interpretation of soil fertility and tree growth in ecological research and it confirmed our second and third hypotheses that individual soil fertility variables are explained by terrain attributes and that tree growth is positively influenced by soil fertility, and thus also by terrain attributes. Nevertheless, in future years also the planted plot diversity may contribute to soil fertility besides topography.

SUPPLEMENTARY MATERIAL

Supplementary material is available at *Journal of Plant Ecology* online.

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Manuscript 6

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Bryophyte-dominated biological soil crusts mitigate soil erosion in an early successional Chinese subtropical forest

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Abstract

This study investigated the development of biological soil crust (biocrust) covers in an early successional subtropical forest ecosystem and their impact on soil erosion. Within a biodiversity and ecosystem functioning experiment in Southeast China (BEF China), sediment discharge and runoff measurements were conducted with micro-scale runoff plots under natural rainfall and biocrust covers were surveyed over a five-year period.

Results showed that biocrusts occurred widely in our experimental forest ecosystem and developed from initial light cyanobacteria- and algae-dominated crusts to later-stage bryophyte-dominated crusts in only three years. Biocrust covers were still increasing after six years of tree growth. Within later stage crusts, 25 bryophyte species were determined. The development of biocrusts was significantly influenced by the surrounding vegetation cover and terrain attributes. Besides high crown cover and leaf area index, the development of biocrusts was favoured by low slope gradients, slope orientations towards the incident sunlight and the altitude of the research plots. Our measurements showed, that bryophyte-dominated biocrusts were importantly decreasing soil erosion and more effective in erosion reduction than abiotic soil surface covers. Hence, their significant role to mitigate sediment discharge and runoff generation in mesic forest environments and their ability to quickly colonize gaps in higher vegetation layers are of particular interest for soil erosion control in early stage forest plantations. A detailed record of different biocrust species and their functional influence on soil erosion processes as well as a thorough monitoring of biocrust covers under closing tree canopy in subtropical forests is required in further studies.

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Abstract. This study investigated the development of biological soil crusts (biocrusts) in an early successional subtropical forest plantation and their impact on soil erosion. Within a biodiversity and ecosystem functioning experiment in southeast China (biodiversity and ecosystem functioning (BEF) China), the effect of these biocrusts on sediment delivery and runoff was assessed within micro-scale runoff plots under natural rainfall, and biocrust cover was surveyed over a 5-year period.

Results showed that biocrusts occurred widely in the experimental forest ecosystem and developed from initial light cyanobacteria- and algae-dominated crusts to later-stage bryophyte-dominated crusts within only 3 years. Biocrust cover was still increasing after 6 years of tree growth. Within later-stage crusts, 25 bryophyte species were determined. Surrounding vegetation cover and terrain attributes significantly influenced the development of biocrusts. Besides high crown cover and leaf area index, the development of biocrusts was favoured by low slope gradients, slope orientations towards the incident sunlight and the altitude of the research plots. Measurements showed that bryophyte-dominated biocrusts strongly decreased soil erosion, being more effective than abiotic soil surface cover. Hence, their significant role in mitigating sediment delivery and runoff generation in mesic forest environments and their ability to quickly colonise soil surfaces after disturbance are of particular interest for soil erosion control in early-stage forest plantations.

1 Introduction

Biological soil crusts (hereinafter referred to as biocrusts) are a living soil cover, which plays significant functional roles in many environments (Weber et al., 2016). In initial ecosystems, communities of cyanobacteria, algae, fungi, lichens, bryophytes and bacteria in varying combinations are the first to colonise the substrate (Evans and Johansen, 1999). Biocrusts are often dominated by one organism group, with cyanobacterial crusts being indicators of early-stage crusts and drier conditions (Malam Issa et al., 1999, 2007) and bryophyte-dominated crusts being indicators of later-stage crusts and moister conditions (Colesie et al., 2016; Seppelt et al., 2016). These highly specialised communities form a biological crust immediately on top of or within the first millimetres of the soil surface (Büdel, 2005). Biocrusts preferably occur under harsh conditions of temperature or light, where vascular vegetation tends to be rare (Allen, 2010). Therefore, biocrusts are generally widespread under dryland conditions (Berkeley et al., 2005; Belnap, 2006;

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Büdel et al., 2009), whereas under mesic conditions they mostly occur as a successional stage after disturbance or in environments under regularly disturbed regimes (Büdel et al., 2014).

In direct competition with phanerogamic plants, biocrusts are generally in an inferior position, and thus their development is limited under closed plant canopies or when leaf litter layers occur (Belnap et al., 2003a). This limitation is due to the competition for light (Malam Issa et al., 1999) and nutrients (Harper and Belnap, 2001). Disturbance of the phanerogamic vegetation layers, however, changes this competitive situation. Such disturbances can occur in forest ecosystems by natural tree fall or human-induced clear-cutting (Barnes and Spurr, 1998). Complete removal of a forest causes a harsh shift in vegetation development and creates a starting point for new vascular plant as well as biocrust communities (Bormann et al., 1968; Keenan and Kimmins, 1993; Beck et al., 2008). Biocrusts are able to quickly colonise natural clearances in tree layers (Belnap et al., 2003a) as well as gaps appearing after human disturbance (Dojani et al., 2011; Chiquoine et al., 2016). Generally, it can be stated that current knowledge on the relation between the development of biocrust cover and vascular plant cover leaves room for further research (Kleiner and Harper, 1977; Belnap et al., 2003b; Zhang et al., 2016). In particular, there are only few studies on the development of biocrusts in early successional forest ecosystems (Su et al., 2007; Zhang et al., 2016), but we assume that biocrusts are able to coexist in these mesic environments shortly after deforestation. Furthermore, descriptions of different biocrust types in mesic vegetation zones and investigations in southeast Asia are rare (Büdel, 2003; Bowker et al., 2016).

Functional roles of biocrusts have been investigated for decades, but less attention has been paid to their spatial distribution and characteristics (Allen, 2010). Biocrust cover varies across spatial scales (from centimetres to kilometres), and it could be shown that it depends not only on the surrounding vascular vegetation cover but also on soils, geomorphology, and (micro-)topography or terrain (Evans and Johansen, 1999; Ullmann and Büdel, 2003; Kidron et al., 2009; Bowker et al., 2016) in arid, semi-arid, temperate and boreal environments. Different biocrust distributions have been related to elevation and terrain-influenced microclimatic gradients (Kutiel et al., 1998), different geomorphic zones (Eldridge, 1999), varying aspects (George et al., 2000) and soil types (Bu et al., 2016). We assume that this is also true for mesic subtropical forest environments. To our knowledge, investigations of the influence of small-scale (centimetres to metres) topographic variations in biocrust development are rare, and further studies will help to understand the role of these small-scale factors (Garcia-Pichel and Belnap, 2003; Bu et al., 2016; Bowker et al., 2016). Furthermore, as the development of biocrusts is characterised by a high complexity and spatial heterogeneity with many microclimatic and micro-environmental factors, it is of great significance to conduct comparative studies on the spatial distribution of biocrusts (Bu et al., 2013).

Biocrusts were recognised as having a major influence on terrestrial ecosystems (Buscot and Varma, 2005; Belnap, 2006) as they protect soil surfaces against erosive forces by both wind and water (Bowker et al., 2008; Zhao et al., 2014). They can absorb the kinetic energy of rain drops (splash effect), decrease shear forces and stabilise soil particles with protonemal mats and fine rhizoids and thus decrease particle detachment and enhance soil stability (Malam Issa et al., 2001; Warren, 2003; Belnap and Lange, 2003). These effects differ with regard to soil texture, surface roughness, water repellency and finally different crust species and developmental stages (Warren, 2003; Belnap and Büdel, 2016). However, studies that directly relate different types of biocrust cover to rates of soil erosion are few (Allen, 2010). Furthermore, the influence of biocrusts on sediment delivery and runoff has mostly been investigated in arid and semi-arid climates and humid climates have been largely disregarded (Belnap and Lange, 2003; Weber et al., 2016).

This study aims to investigate the development of biocrust cover in an early successional subtropical forest ecosystem after human disturbance and the impact of those biocrusts on soil erosion. Therefore, interrill erosion was measured with runoff plots and the occurrence, distribution and development of biocrusts was recorded. The study was conducted in an experimental forest plantation, which aims to study biodiversity and ecosystem functioning relationships in southeast China (biodiversity and ecosystem functioning (BEF) China; for further information see Yang et al., 2013; Bruelheide et al., 2014; Trogisch et al., 2017). During the study, the following hypotheses were addressed:

1. Biocrusts are able to coexist in mesic early successional subtropical forest ecosystems, but crust cover decreases with ongoing canopy closure and decreasing light intensity.

2. The development of biocrusts in mesic subtropical forests is not only influenced by the surrounding vegetation cover but also by major soil attributes which influence biocrust growth and by terrain attributes which affect microclimatic conditions.

3. Biocrusts mitigate interrill soil erosion in early successional subtropical forest plantations.

2 Material and methods

2.1 Study site and experimental design

The study was carried out within the BEF China experiment (Bruelheide et al., 2014) in Xingangshan, Jiangxi Province, PR China (29°06.450′ N, 117°55.450′ E). The experimental area is located in a mountainous landscape at an elevation of 100 to 265 m a.s.l. with slopes from 15° to 41° (Scholten et al., 2017). The bedrock is non-calcareous sandstones, siltstones and slates weathered to saprolite, and the predominant soil types are Cambisols with Anthrosols in downslope positions and Gleysols in valleys (Scholten et al., 2017). The particle size distribution was quite homogenous throughout the experimental area, having loam as the main texture class (Scholten et al., 2017). The mean annual temperature is 17.4 °C, and the annual precipitation is 1635 mm with about 50 % falling during May to August (Goebes et al., 2015). The climate is typical for summer monsoon subtropical regions. The potential natural vegetation of this region is a subtropical broadleaved forest with dominating evergreen species. It was widely replaced by tree plantations of mostly *Cunninghamia lanceolata* for the purpose of commercial forestry in the 1980s (Bruelheide et al., 2014). The experimental area (approx. 38 ha) is structured in 566 research plots (25.8 m × 25.8 m each) at two sites (A and B) and was clear-cut and replanted with 400 tree saplings per plot in different tree species mixtures in 2009 and 2010 (Yang et al., 2013). A selection of 34 research plots was used for this study (cf. Seitz et al., 2016). Shrubs and coppices were weeded once a year from 2010 to 2012 to help the tree saplings grow, following common practice in forest plantations of this area.

2.2 Field methods

Biocrust cover was determined photogrammetrically in 70 selected micro-scale runoff plots (ROPs; 0.4 m × 0.4 m; Seitz et al., 2015; Trogisch et al., 2017) at five time steps (November 2011, May 2012, May 2013, May 2014 and May 2015). Biocrust species were first described in the field based on appearance and functional groups. Biocrust types were then determined based on the dominating autotrophic component (highest share of total biocrust cover per ROP). During the rainy season in summer 2013, an extended survey together with soil erosion measurements (see below) was conducted in five ROPs on 34 research plots each (170 ROPs in total; Table 1). At each ROP, perpendicular images were taken with a single-lens reflex camera system (Canon 350D, Tokio, Japan) and processed by the grid quadrat method in GIMP 2.8 using a digital grid overlay with 100 subdivisions (cf. Belnap et al., 2001). Stone cover and biocrust cover were separated by hue distinction. A continuous leaf litter cover, which may impede analyses, was not present during measurements. Biocrusts were collected in 2013, and samples were dried at 40 °C (Dörrex drying unit, Netstal, Switzerland). The identification of these sampled species was carried out by morphological characteristics using a stereomicroscope (Leitz TS, Wetzlar, Germany), a transmitted-light microscope (Leitz Laborlux S, Wetzlar, Germany) and ultraviolet light. Bryophytes (dominating taxa in 2013) were determined to the species level, wherever possible, and separated into mosses (Bischler-Causse, 1989; Moos flora of China: Gao et al., 1999, 2001; 2002, 2003; 2005, 2007; 2008, 2011) and liverworts (Zhu, 2006; Söderström et al., 2016; A. Schäfer-Verwimp, personal communication, 2016). Comparisons were conducted with specimens hosted in the herbarium of the State Museum of Natural History in Stuttgart, Germany (Herbarium STU).

Table 1. Erosion, soil, soil cover, vegetation and terrain attributes in 170 runoff plots (ROPs) and on 34 research plots (with five ROPs each) in Xingangshan, Jiangxi Province, PR China, in 2013.

	Min	Mean	Max	SD
Runoff plots (four measured rainfall events, $n = 334$)				
Sediment delivery ($\mathrm{g\,m^{-2}}$)	21.6	195.5	989.0	165.8
Surface runoff ($\mathrm{L\,m^{-2}}$)	3.1	40.3	111.8	21.7
Rainfall amount (mm)	25	94	178	28
Runoff plots (ROPs in use, $n = 170$)				
Slope (°)	5	29	60	6
Soil cover (%)	0	19	62	14
– Biological soil crust cover (%)	0	24	62	14
– Stone cover (%)	0	4	42	6
Crown cover (%)	0.00	0.32	1.00	0.34
Leaf area index (LAI)	0.00	0.73	5.35	1.04
Research plots ($n = 34$)				
Bulk soil density ($\mathrm{g\,cm^{-2}}$)	0.83	0.98	1.12	0.06
Soil organic matter (%)	4.2	6.5	9.7	1.7
pH (KCl)	3.24	3.66	4.00	0.18
Altitude (m)	119	167	244	37
MCCA	0.98	2.07	3.81	0.83
TRI	0.72	2.39	3.86	0.59
Eastness	−0.86	0.09	0.99	0.56
Northness	−0.87	0.23	0.99	0.62
Tree height (m)	1.0	2.2	7.4	1.7
Crown width (m)	0.4	1.2	3.0	0.8

Soil cover: proportion of soil surface area covered by biocrusts or stones; crown cover: proportion of soil surface area covered by crowns of live trees; leaf area index: one-sided green leaf area per unit soil surface area; MCCA: Monte Carlo based flow accumulation (Behrens et al., 2008); TRI: terrain ruggedness index (Riley et al., 1999); eastness and northness: state of being east or north (Roberts, 1986); tree height: distance from stem base to apical meristem; crown width: length of longest spread from edge to edge across the crown; min: minimum; max: maximum; SD: standard deviation.

Sediment delivery and surface runoff were measured within 170 ROPs in summer 2013 together with an extended biocrust survey (see above and Table 1), when tree saplings did not exceed 3 years of age and leaf litter fall was still marginal. After four time steps, 334 valid ROP measurements entered the analysis (for detailed information see Seitz et al., 2016). Sediment delivery was sampled, dried at 40 °C and weighed, whereas surface runoff and rainfall amount were measured in situ. At every ROP, crown cover and leaf area index (LAI) were measured with a fish-eye camera system (Nikon D100 with Nikon AF G DX 180°, Tokio, Japan) and calculated with HemiView V.8 (Delta-T devices, Cambridge, UK). Measurements of tree height and crown width were provided by Li et al. (2014) on a research plot scale ($n = 34$). Tree species richness and tree composition resulted from the experimental set-up of BEF China (Bruelheide et al., 2014).

Soil attributes (Table 1) were determined for every research plot ($n = 34$, sampling in 2013) using pooled samples from nine point measurements each (sampling depth 0–5 cm). Soil pH was measured in KCl (WTW pH meter with Sentix electrodes, Weilheim, Germany), bulk soil density was determined by the mass-per-volume method, and total organic carbon (TOC) was measured using heat combustion (Elementar Vario EL III, Hanau, Germany). Soil organic matter (SOM) was calculated by multiplying TOC by a factor of 2 (Pribyl, 2010).

2.3 Digital terrain analysis

Terrain attributes (Table 1) were derived from a digital elevation model (DEM; 5 m × 5 m; Scholten et al., 2017) on a research plot scale ($n = 34$). Attributes were the terrain ruggedness index (TRI; Riley et al., 1999) to describe the heterogeneity of the terrain, the Monte Carlo based flow accumulation (MCCA; Behrens et al., 2008) to diagnose terrain driven water availability, altitude above sea level to address elevation effects, and the eastness and the northness (Roberts, 1986) to describe plant-related climatic conditions. These terrain attributes cover major landscape features of the experimental area and were not correlated. Slope was additionally measured with an inclinometer at every ROP ($n = 170$; see Seitz et al., 2016).

2.4 Statistical methods

The temporal development of biocrust cover (hypothesis 1, above) from 2011 to 2015 was assessed at five time steps within 70 ROPs (see above) by an analysis of variance (ANOVA) and Tukey's honestly significant difference (HSD) test ($n = 350$).

The influences of vegetation, soil and topographic attributes on biocrust cover (hypothesis 2) in 170 ROPs (see above) were assessed by linear mixed effects (LME) models ($n = 334$). Crown cover, bulk soil density, SOM, pH, altitude, slope, MCCA, TRI, eastness, northness and tree species richness were fitted as fixed effects and biocrust cover as a response variable. The attributes were tested with Pearson's correlation coefficient before fitting. LAI was fitted individually in exchange with crown cover due to multicollinearity. The experimental site and research plot were fitted as random effects, and hypotheses were tested with an ANOVA type 1 with a Satterthwaite approximation for degrees of freedom.

The influences on soil erosion (hypothesis 3) were assessed by LME models with a restricted maximum likelihood ($n = 334$) and sediment delivery and surface runoff as response variables. Crown cover, slope, surface cover, SOM, rainfall amount and tree species richness were fitted as fixed effects. Surface cover was then split into surface cover by biocrusts and by stones, which entered the analysis as fixed conjoined factors. Precipitation events nested in plot, tree species composition, experimental site and ROP nested in plot were fitted as random effects. Attributes were not correlated. The hypothesis was tested with an ANOVA type 1 with a Satterthwaite approximation for degrees of freedom. Moreover, the Wilcoxon rank sum test was applied to test for differences between biocrust cover and stone cover on sediment delivery and surface runoff. Therefore, the dataset was split into data points where biocrust cover exceeded stone cover ($n = 281$) and data points where stone cover exceeded biocrust cover ($n = 53$).

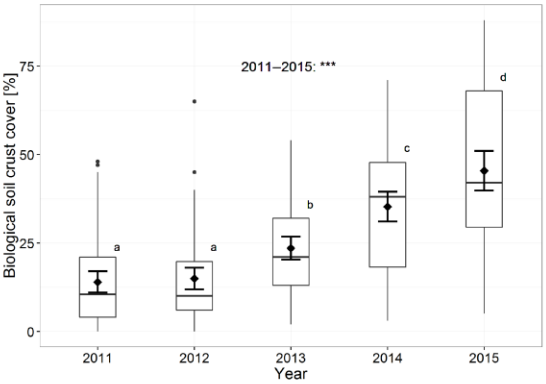

Figure 1. The development of biological soil crust cover in runoff plots of the BEF China experiment from 2011 to 2015 in Xingangshan, Jiangxi Province, PR China ($n = 350$). Horizontal lines within box plots represent medians, and diamonds represent means with standard error bars. Points signify outliers and small letters significant differences ($p < 0.001$).

All response variables were log-transformed before modelling. The dataset was tested for multicollinearity and met all prerequisites to carry out ANOVAs. All analyses were performed with R 3.1.2 (R Core Team, 2014). LME modelling was conducted with "lmerTest" (Kuznetsova et al., 2014) and rank sum tests with "exactRankTests" (Hothorn and Hornik, 2015). Figures were designed with "ggplot2" (Wickham, 2009).

3 Results

3.1 Temporal development of biocrust cover

Biocrusts occurred in 94 % of all ROPs, and their cover within ROPs ranged between 1 and 88 % over the course of 5 years. The mean biocrust cover of all ROPs more than tripled from their installation in 2011 to the last measurement in 2015 (Fig. 1). The increases were significant from 2011 to 2015 and from 2012 to 2013, 2013 to 2014 and 2014 to 2015 ($p < 0.001$).

Whereas a clear bryophyte dominance of biocrusts was evident at the time of sampling in 2013 (average ROP surface cover 24 %), different successional stages were identified in

Figure 2. Successional stages of biological soil crusts in two exemplary runoff plots (top row and bottom row, 0.4 m × 0.4 m each) in 2011, 2013 and 2015 (from left to right) at the BEF China experiment in Xingangshan, Jiangxi Province, PR China.

Table 2. Liverwort and moss species sampled in the BEF China experiment in Xingangshan, Jiangxi Province, PR China, in 2013.

Family	Species		Author
Liverworts			
Calypogeiaceae	*Calypogeia*	*fissa*	(L.) Raddi
Conocephalaceae	*Conocephallum*	*salebrosum*	Szweyk., Buczk. et Odrzyk.
Lophocoleaceae	*Heteroscyphus*	*zollingeri*	(Gottsche) Schiffn.
Marchantiaceae	*Marchantia*	*emarginata*	Reinw., Blume et Nees
Acrobolbaceae	*Notoscyphus*	*lutescens*	(Lehm. et Lindenb.) Mitt.
Mosses			
Polytrichaceae	*Atrichum*	*subserratum*	(Harv. et Hook. f.) Mitt.
Pottiaceae	*Barbula*	*unguiculata*	Hedw.
Bryaceae	*Bryum*	*argenteum*	Hedw.
Leucobryaceae	*Campylopus*	*atrovirens*	De Not.
Dicranellaceae	*Dicranella*	*heteromalla*	(Hedw.) Schimp.
Pottiaceae	*Didymodon*	*constrictus*	(Mitt.) K. Saito
Pottiaceae	*Didymodon*	*ditrichoides*	(Broth.) X. J. Li et S. He
Ditrichaceae	*Ditrichum*	*pallidum*	(Hedw.) Hampe
Entodontaceae	*Entodon*	spec.	sterile
Hypnaceae	*Hypnum*	*cupressiforme*	Hedw.
Hypnaceae	*Hypnum*	*macrogynum*	Besch.
Leucobryaceae	*Leucobryum*	*juniperoideum*	(Brid.) Müll. Hal.
Bartramiaceae	*Philonotis*	*marchica*	(Hedw.) Brid.
Bartramiaceae	*Philonotis*	*mollis*	(Dozy et Molk.) Mitt.
Bartramiaceae	*Philonotis*	*roylei*	(Hook. f.) Mitt.
Mniaceae	*Plagiomnium*	*acutum*	(Lindb.) T. J. Kop.
Polytrichaceae	*Pogonatum*	*inflexum*	(Lindb.) Sande Lac.
Thuidiaceae	*Thuidium*	*glaucinoides*	Broth.
Mniaceae	*Trachycystis*	*microphylla*	(Dozy et Molk.) Lindb.
Pottiaceae	*Trichostomum*	*crispulum*	Bruch

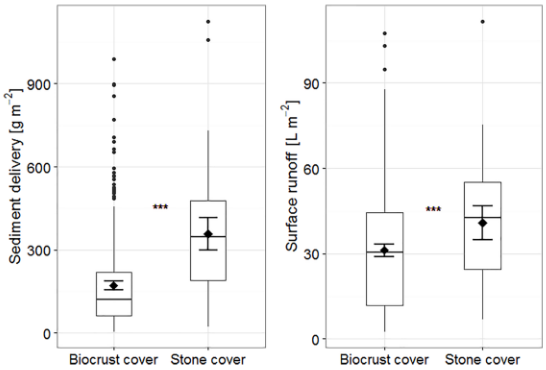

Figure 3. The influence of runoff plots dominated by biological soil crust cover ($n = 281$) and stone cover ($n = 53$) on sediment delivery and surface runoff in Xingangshan, Jiangxi Province, PR China, in 2013. Horizontal lines within box plots represent medians, and diamonds represent mean with standard error bars.

the field and on ROP photos from 2011 to 2015 (Fig. 2). In 2011, a smooth, light cyanobacteria- and algae-dominated crust with few lichens and bryophytes indicated an earlier stage of biocrust development (Colesie et al., 2016). In 2013, 25 moss and liverwort species were classified (Table 2) and formed a bryophyte-dominated crust, with some cyanobacteria, algae, lichens and micro-fungi still observed within ROPs. The same was true in 2015, but the first evidence of vascular plants (*Selaginella* and *Poaceae*) indicated a further change in the vegetation cover of the soil surface.

3.2 The influence of vegetation, soil and terrain on biocrust cover

The development of biocrust cover in 2013 was positively influenced by crown cover and LAI as attributes of the surrounding vegetation (Table 3). Furthermore, it was negatively affected by slope and northness and slightly positively affected by the altitude of the research plots as terrain attributes (Table 3). Further terrain attributes or any soil attributes did not affect the development of biocrust cover.

3.3 The impact of biocrust cover on soil erosion

Results reveal that biocrusts strongly affect soil erosion. ROPs with biocrust cover below 10 % showed a mean sediment delivery of 302 g m^{-2} and a mean runoff volume of 39 L m^{-2}, whereas ROPs with biocrust cover above 50 % showed a mean sediment delivery of 74 g m^{-2} and a mean runoff volume of 29 L m^{-2}. Both biocrust and stone cover, as well as total soil surface cover (comprising both biocrust and stone cover; $p < 0.001$) negatively affected sediment delivery (Table 4). In addition, soil surface cover negatively affected surface runoff ($p = 0.003$). However, only biocrust but not stone cover mediated the effect of runoff.

Table 3. Results of the final linear mixed effects (LME) model for vegetation, soil and terrain attributes on biological soil crust cover in Xingangshan, Jiangxi Province, PR China, in 2013.

	Biological soil crust cover			
	denDF	F	Pr	estim.
Fixed effects				
Crown cover	136	12.9	***	10.8
Bulk soil density	37	0.03	ns	3.65
SOM	39	1.11	ns	(−)0.95
pH (KCl)	38	2.47	ns	(−)16.7
Altitude	37	3.69	•	0.80
Slope	191	7.53	**	(−)2.72
MCCA	39	0.02	ns	0.33
TRI	38	0.04	ns	(−)0.40
Eastness	37	2.73	ns	(−)4.23
Northness	37	9.14	**	5.99
Tree species richness	38	1.22	ns	(−)0.27
Random effects		SD	Variance	
Site		< 0.01	< 0.01	
Plot		< 0.01	< 0.01	
Vegetation attribute fitted in exchange to crown cover				
Leaf area index	107	42.8	***	5.98

SOM: soil organic matter; MCCA: monte carlo based flow accumulation; TRI: terrain ruggedness index; denDF: denominator degrees of freedom; F: F value; Pr: significance; estim.: estimates; SD: standard deviation; *** $p < 0.001$; ** $p < 0.01$; * $p < 0.05$; • $p < 0.1$; ns: not significant, $n = 215$.

Furthermore, crown cover, SOM and rainfall amount affected sediment delivery, whereas runoff was affected by crown cover and rainfall amount. ROPs with increased stone cover showed higher sediment delivery and surface runoff compared to those with increased biocrust cover (Fig. 3).

4 Discussion

4.1 Temporal development of biocrust cover

Biocrusts were detected widely within the experiment and occupied a considerable area in the interspaces of the growing tree community. Thus, the first part of hypothesis 1, stating that biocrusts are able to coexist in mesic early successional subtropical forests, can be confirmed, as they successfully colonised the newly created habitats originating from the disturbance by forest clear-cutting and weeding (Bruelheide et al., 2014). Although biocrusts have been mainly defined to occur in dryland regions (Weber et al., 2016), they can also appear as a transient feature in mesic environments after major singular or repeated disturbance events (Büdel et al., 2014; Fischer et al., 2014). In the current study, deforestation provided a local arid micro-environment, which initiated early biocrust development. At this young stage of forest development, biocrusts were able to coexist with up-

Table 4. Results of the final linear mixed effects (LME) models for sediment delivery and surface runoff, with surface cover split into biological soil crust cover and stone cover in Xingangshan, Jiangxi Province, PR China, in 2013.

	Sediment delivery				Surface runoff			
	denDF	F	Pr	estim.	denDF	F	Pr	estim.
Fixed effects								
Crown cover	130	6.53	*	(−)0.15	173	9.11	**	(−)0.14
Slope	151	1.23	ns	(−)0.06	168	2.25	< ns	(−)0.06
Surface cover								
– Biocrust	151	50.2	***	(−)0.38	159	8.11	**	(−)0.12
– Stone	136	10.3	**	(−)0.19	188	1.66	< ns	(−)0.06
SOM	44	5.71	*	(−)0.08	72	2.43	< ns	0.12
Rainfall	95	5.46	*	(−)0.08	302	13.2	***	0.14
Tree species richness	22	0.46	ns	(−)0.05	68	0.11	< ns	(−)0.03
Random effects		SD		var.		SD	var.	
Precip. event: plot		0.199		0.040		0.537	0.288	
Tree composition		0.292		0.085		0.000	0.000	
Site		0.466		0.217		0.443	0.196	
Plot: ROP		0.441		0.195		0.269	0.073	

SOM: soil organic matter; denDF: denominator degrees of freedom; F: F value; Pr: significance; estim.: estimates; SD: standard deviation; var.: variance; *** $p < 0.001$; ** $p < 0.01$; * $p < 0.05$; • $p < 0.1$; ns: not significant, $n = 334$.

coming tree saplings and formed a pioneer vegetation on the soil surface (Langhans et al., 2009), which provides the basis for the growth of other plants by the input of carbon and nitrogen (West, 1990; Evans and Johansen, 1999). Biocrusts are known to facilitate the succession of vascular plants to more advanced stages (Bowker, 2007), but tree growth and thus crown cover can also lead to an advancement in biocrust development, e.g. due to the protection from direct sunlight (Zhao et al., 2010; Tinya and Ódor, 2016). The bryophyte dominance of biocrusts in 2013 documented this development into a later and somewhat moister successional stage. Later-stage bryophytes have received comparatively little attention in forest understorey (Gilliam, 2007) and biocrust studies (Weber et al., 2016), and in Asia only 23 different species have been reported within biocrusts up to now (Seppelt et al., 2016). Thus, this study with 25 recorded moss and liverwort species, most of them being new records within Asian biocrusts (Burkhard Büdel, personal communication, 2016), substantially increases the knowledge on biocrusts of this region.

The extent of biocrusts has been strongly increasing since 2012, i.e. 3 years after tree replantation, and was still gaining coverage in the sixth year after the experimental set-up. Thus, the second part of hypothesis 1, stating that crust cover decreases with ongoing canopy closure, has to be rejected. Even if single trees were already up to 7.4 m high (Li et al., 2014) and LAI was up to 5.35 in 2013, biocrusts still remained coexisting within the early-stage forest ecosystem. Furthermore, increasing crown cover and LAI seemed to foster the development of bryophyte-dominated biocrusts at this ecological stage. By the end of this study in summer 2016 (LAI up to 6.18), there were indications that biocrust cover may start to be pushed back, as the first vascular plants appeared in between. This is in line with the existing literature, demonstrating that continuing tree growth will cause biocrust communities to adapt with an altered composition of moss and liverwort species (Eldridge and Tozer, 1997; Fenton and Frego, 2005; Goffinet and Shaw, 2009). It has been shown that bryophytes switch from species favouring sunny habitats to more shade-tolerant species (Zhao et al., 2010; Müller et al., 2016). In addition, there might also be a reduction in bryophyte diversity due to shady conditions, where only a smaller number of species could prevail. In later stages, biocrust cover will be replaced by vascular vegetation (in light forests) or buried under persisting leaf litter (under darker conditions). In this context, the ecological roles of biocrusts in succession models and plant restoration are of interest (Hawkes, 2004; Bowker, 2007). In particular, biocrust succession in temperate climates has received limited scientific attention (Read et al., 2016). Furthermore, there are several projects underway to establish successful restoration techniques in arid and semi-arid environments (Rosentreter et al., 2003; Bowker, 2007; Chiquoine et al., 2016; Condon and Pyke, 2016), which could be adapted to mesic environments. Nevertheless, it has to be stated that biocrust restoration might be dispensable in some mesic systems, as natural reestablishment appeared to be very fast in this study.

4.2 The influence of vegetation, soil and terrain on biocrust cover

In the current study, the development of biocrusts was influenced by vegetation and terrain but not by the three soil attributes investigated in this study. Thus, hypothesis 2, stating that the biocrust development is not only influenced by surrounding vegetation but also by soil and terrain, can only partly be confirmed for this ecosystem. As demonstrated above, high crown cover and LAI positively affected the development of biocrust cover in 2013. This increase in biocrust cover is likely caused by successional alteration of biocrusts towards bryophyte dominance. Mosses and liverworts profit from humid conditions and a higher protection from light compared to cyanobacteria- or lichen-dominated crusts (Ponzetti and McCune, 2001; Marsh et al., 2006; Williams et al., 2013). The successional development of biocrusts within the BEF China experiment was faster than reported by Zhao et al. (2010) for Chinese grasslands (Loess Plateau), who claimed biocrusts from a 3-year-old site were early successional and dominated by cyanobacteria. The recovery rate was also faster than described by Eldridge (1998) and Read et al. (2011) for semi-arid Australia, two of the very few studies on biocrust recovery under woodland. In the study presented here, the rapid change in biocrust community composition is mainly linked to the growth rates of surrounding trees in this subtropical forest. As functions of biocrusts, such as erosion reduction, are species-dependent, the rapid change in species composition might also lead to considerable variations in functional responses. Further studies are required to investigate species changeover times in different environments and particularly in disturbed mesic ecosystems.

Furthermore, several terrain attributes affected biocrust cover. Slope was the most prominent of those factors, causing a considerable decline in biocrust cover with increasing slope. This finding was explained by their decreasing ability to fix themselves on the soil surface at high slope angles and thus their tendency to erode from the soil surface when large surface water flows occur during rainfall events (Chamizo et al., 2013; Bu et al., 2016). Thus, the surface-protecting effect of biocrusts decreases at steep plantation sites and during heavy monsoon rainfall events, which frequently occur in the broader research area in Jiangxi Province, PR China (Yang et al., 2013; Goebes et al., 2015). Moreover, microclimatic factors played a role in the development of biocrusts. Northness showed a positive impact on biocrust cover and indicated that slope orientations towards the incident sunlight directly influence the biocrust development. This was also observed in other studies in arid and semi-arid areas (Bowker et al., 2002; Zaady et al., 2007). Furthermore, biocrust development depended on the altitude, which probably also affects microclimatic conditions (Kutiel et al., 1998; Chamizo et al., 2016; Bu et al., 2016). Those microclimatic factors are additionally altered by the growing tree vegetation itself.

Interestingly, SOM and pH did not affect biocrust cover in this study, whereas generally, underlying substrates are a main factor for bryophyte development (Spitale, 2017) and soil attributes are known to strongly influence biocrust cover (Bowker et al., 2016). In the experimental area, increased organic matter contents and acidic conditions have been determined (Scholten et al., 2017) which favour the development of bryophyte-dominated biocrusts (Eldridge and Tozer, 1997; Seppelt et al., 2016). Nevertheless, the variation between the research plots was small and apparently not large enough to cause prominent differences in biocrust development. Comparisons between forest plantations on different substrates would help to clarify the influence of soil attributes on biocrust development in those environments and to assess their effect in a broader environmental context (Spitale, 2017). Furthermore, a broader range of soil parameters should be included in future studies.

4.3 The impact of biocrust cover on soil erosion

Biocrust cover clearly mitigated interrill soil erosion in this early-stage ecosystem, and thus hypothesis 3 was confirmed. Sediment delivery was strongly reduced with increasing biocrust cover. For arid environments, Cantón et al. (2011) and Maestre et al. (2011), for example, showed that sediment delivery from soil surfaces covered with biocrusts decreases compared to bare soil surfaces with physical crusting (from 20 to $< 1\,\mathrm{g\,m^{-2}}$ and 40 to $< 5\,\mathrm{g\,m^{-2}}$, respectively), both studies using micro-scale runoff plots ($0.25\,\mathrm{m^2}$). Bu et al. (2015) and Zhao and Xu (2013) found similar erosion-reducing patterns for the subarid temperate Chinese Loess Plateau. The study presented here shows that biocrusts fulfil this key ecosystem service also within a particular mesic habitat, even if their biomass and soil penetration depth is low compared to trees. This functional role is due to the fact that biocrusts attenuate the impact of raindrops on the soil surface and greatly improve its resistance against sediment detachment (Eldridge and Greene, 1994; Goebes et al., 2014; Zhao et al., 2014). Moreover, they have the ability to glue loose soil particles together with polysaccharides extruded by cyanobacteria and green algae (Buscot and Varma, 2005). In the current study, protonemata and rhizoids of mosses and liverworts were observed to be most effective by weaving and thus fixing the first millimetres of the topsoil, as also described by Bowker et al. (2008). *Pogonatum inflexum* and *Atrichum subserratum*, for example, have shown positive effects on erosion control due to their sustained protonema system (present authors' personal observation, 2014). Furthermore, bryophytes increase the formation of humus, which in turn assists with binding primary particles into aggregates (Scheffer et al., 2010; Zhang et al., 2016).

Whereas a partial stone cover did not decrease surface runoff in this study, bryophyte-dominated biocrusts positively influenced the hydrological processes in the topsoil layer regarding erosion control. Thus, they actively mitigated initial soil erosion compared to abiotic components such as stones and pebbles. Biocrusts have been frequently shown to influence hydrological processes such as surface runoff and infiltration rates (Cantón et al., 2011; Chamizo et al., 2012; Rodríguez-Caballero et al., 2013). Recently, Chamizo et al. (2016) showed that biocrusts decrease runoff generation on a larger scale ($> 2\,\mathrm{m}^2$), but the converse behaviour has also been found (Cantón et al., 2002; Maestre et al., 2011). Reducing effects on runoff are related to biocrusts species composition (Belnap and Lange, 2003), and later developmental biocrust stages with higher biomass levels provide more resistance to soil loss (Belnap and Büdel, 2016). Bryophyte-dominated crusts in particular have shown to enhance infiltration and reduce runoff due to their rhizome system, causing soil erosion rates to stay low (Warren, 2003; Yair et al., 2011). Other field studies also revealed that later-stage biocrusts, containing both lichens and bryophytes, offer more protection against soil erosion than cyanobacterial crusts (Belnap and Gillette, 1997) as they provide a higher infiltration potential (Kidron, 1995). On the other hand, Drahorad et al. (2013) found an increase in water repellency and a decrease in water sorptivity with ongoing biocrust succession in a temperate forest glade, which could also strongly affect runoff and sediment transport on subtropical forest soil surfaces. Moreover, biocrusts dominated by bryophytes increase surface roughness and thus slow down runoff (Kidron et al., 1999; Rodríguez-Caballero et al., 2012). Finally, biocrusts also absorb water and provide a comparably high water storage capacity (Warren, 2003; Belnap, 2006). For example, *Leucobryum juniperoideum*, which was widely found in the study area, showed a high water absorbing capacity (present authors' personal observation, 2014). Thus, the observed rapid change in biocrust composition from cyanobacteria to bryophyte dominance improved soil erosion control in this forest environment. This effect should be considered for the replantation of forests in regions endangered by soil erosion.

5 Conclusions

This study investigated the development and distribution of biocrusts in an early-stage subtropical forest plantation as well as their impact on interrill soil erosion after human disturbance. The following conclusions were drawn:

1. Biocrusts occurred widely in this mesic early successional forest ecosystem in subtropical China and were already dominated by bryophytes after 3 years of tree growth (25 bryophyte species classified). After 6 years of continuing canopy closure, biocrust cover was still increasing. Further monitoring under closing tree canopy is of importance to detect changes in biocrust cover and species composition. As this study discusses a very particular subtropical forest environment, where trees were replanted after clear-cutting, results have to be viewed with this particular set-up in mind. Further studies on biocrust development in different disturbed forest ecosystems appear to be of high interest.
2. The surrounding vegetation and underlying terrain affected biocrust development, whereas soil attributes did not have an effect on this small experimental scale. Besides high crown cover and LAI, the development of biocrusts was favoured by a low slope gradient and slope orientations towards the incident sunlight and altitude. Further research appears to be necessary to explain effects of terrain attributes such as aspect or elevation and effects of underlying soil and substrates.
3. Soil surface cover of biocrusts largely affected soil erosion control in this early stage of the forest plantation. Bryophyte-dominated crusts showed erosion-reducing characteristics with regard to both sediment delivery and surface runoff. Furthermore, they more effectively decreased soil losses than abiotic soil surface covers. The erosion-reducing influence of bryophyte-dominated biocrusts and their rapid development from cyanobacteria-dominated crusts should be considered in management practices in early-stage forest plantations. Further research is required on functional mechanisms of different biocrust and bryophyte species and their impact on soil erosion processes.

Data availability. Data are publicly accessible and archived at the BEF China data portal (http://china.befdata.biow.uni-leipzig.de/datasets/376, http://china.befdata.biow.uni-leipzig.de/datasets/598, Seitz, 2016a, b).

Author contributions. SS and TS designed the experiment, and SS, ZS, KK and CLW carried it out. MN and KK classified biocrust types and determined bryophyte species. SS, PG and KS performed the statistical models. SS, XS and BW prepared the manuscript with contributions from all co-authors.

Competing interests. The authors declare that they have no conflict of interest.

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Toward a methodical framework for comprehensively assessing forest multifunctionality

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Abstract

Biodiversity–ecosystem functioning (BEF) research has extended its scope from communities that are short- lived or reshape their structure annually to structurally complex forest ecosystems. The establishment of tree diversity experiments poses specific-methodological challenges for assessing the multiple functions provided by forest ecosystems. In particular, methodological inconsistencies and nonstandardized protocols impede the analysis of multifunctionality within, and comparability across the increasing number of tree diversity experiments. By providing an overview on key methods currently applied in one of the largest forest biodiversity experiments, we show how methods differing in scale and simplicity can be combined to retrieve consistent data allowing novel insights into forest ecosystem functioning. Furthermore, we discuss and develop recommendations for the integration and transferability of diverse methodical approaches to present and future forest biodiversity experiments. We identified four principles that should guide basic decisions concerning method selection for tree diversity experiments and forest BEF research: (1) method selection should be directed toward maximizing data density to increase the number of measured variables in each plot. (2) Methods should cover all relevant scales of the experiment to consider scale dependencies of biodiversity effects. (3) The same variable should be evaluated with the same method across space and time for adequate larger-scale and longer- time data analysis and to reduce errors due to changing measurement protocols. (4) Standardized, practical and rapid methods for assessing biodiversity and ecosystem functions should be promoted to increase comparability among forest BEF experiments. We demonstrate that currently available methods provide us with a sophisticated toolbox to improve a synergistic understanding of forest multifunctionality. However, these methods require further adjustment to the specific requirements of structurally complex and long- lived forest ecosystems. By applying methods connecting relevant scales, trophic levels, and above- and belowground ecosystem compartments, knowledge gain from large tree diversity experiments can be optimized.

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ORIGINAL RESEARCH

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Abstract

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KEYWORDS
BEF-China, forest biodiversity experiments, high-throughput methods, multitrophic interactions, standardized protocols

1 | INTRODUCTION

Biodiversity–ecosystem functioning (BEF) research requires comprehensive methodical approaches to study overall ecosystem functioning based on the simultaneous assessment of multiple functions and services. Integral approaches that include species interactions and trophic networks are especially important because ecosystem performance strongly depends on complex interactions among organisms with tight interconnections of above- and belowground systems (De Deyn & van der Putten, 2005; Kardol & Wardle, 2010; Soliveres et al., 2016). This is particularly true for forests, which represent long-lived and highly complex dynamic systems (Scherer-Lorenzen, Körner, & Schulze, 2005).

Forests support a wealth of ecosystem functions and services, such as biomass production, carbon storage, and prevention of soil erosion, and promote the diversity of coexisting taxa (Pan, Birdsey, Phillips, & Jackson, 2013). Tree diversity has been shown to affect this multifunctionality at local and larger spatial scales (Gamfeldt et al., 2013; van der Plas et al., 2016; Scherer-Lorenzen, 2014). However, experimental research on the relationships between biodiversity and multiple ecosystem functions in forests has begun only recently (Scherer-Lorenzen et al., 2005; Verheyen et al., 2016). Considering the complexity of forest ecosystems, it is clear that the role of tree species richness and associated diversity of microorganisms and animal taxa, including their interactions, for ecosystem functioning can only be studied adequately in a multifunctional framework (Gamfeldt, Hillebrand, & Jonsson, 2008; Hector & Bagchi, 2007).

Although observational studies along natural forest diversity gradients have offered new insights into BEF relationships, their information value is often limited by inseparable effects of species diversity and identity as well as confounding abiotic factors (Nadrowski, Wirth, & Scherer-Lorenzen, 2010; Vilà et al., 2005). Thus, well-designed biodiversity experiments are required to study causal tree diversity effects on ecosystem functioning and the underlying mechanisms (Hector et al., 2011; Nadrowski et al., 2010). Over the last 15 years, an increasing number of large-scale forest diversity experiments has been established in different parts of the world, forming a growing global collaborative experimental network (www.treedivnet.ugent.be) of currently 25 tree diversity experiments (Verheyen et al., 2016). Despite their relatively young age, these planted forests already allow the evaluation of a large range of ecosystem functions also encountered in mature forests. In addition, they represent a unique large-scale field network to study tree establishment as a function of forest diversity soon after planting and during canopy closure (Scherer-Lorenzen, Potvin, et al., 2005).

One of the most striking features of many forest BEF experiments, in which tree species richness and composition are manipulated deliberately, is their much larger spatial dimension than comparable grassland BEF experiments. Forest BEF experiments with up to several hundred thousands of tree individuals planted often extend to the landscape scale. In small-scale grassland BEF experiments with fast-growing herbaceous species, environmental factors can be controlled reasonably well through applying a randomized block design. In contrast, at the landscape scale and in long-lived tree communities, it is more difficult to ensure spatial and temporal homogeneity within the necessarily larger blocks (and plots within blocks), thus increasing the chances of accidental confounding of randomized planting with abiotic environmental variables. Thus, the separation of treatment (biodiversity) factors and environmental covariates in explaining the variation in measured ecosystem functions remains challenging in forest BEF experiments (Balvanera et al., 2006; Bruelheide et al., 2014; Caspersen & Pacala, 2001; Healy, Gotelli, & Potvin, 2008). Consequently, the methods applied to assess ecosystem functions must be applicable to capture the variation in environmental gradients and the effects of tree diversity at the different spatial scales between and within blocks (and plots). Therefore, practical, repeatable, and standardized high-throughput methods are required to quantify ecosystem functions or variables on a large set of plots and across the network of diversity experiments. However, many currently applied BEF methods strongly differ in terms of scope and scale, complicating efficient cross-site comparisons and synthesis approaches.

In principle, measurements of processes in forest BEF experiments typically focus on two or three spatial scales corresponding to tree community organizational levels: the individual tree, the local neighborhood of the individual tree, and the plot or community level. The level of the individual tree is used, for example, to measure species-specific tree growth (Li, Härdtle, et al., 2014), herbivory (Schuldt, Bruelheide, et al., 2015), or fungal infestation (Hantsch, Bien, et al., 2014). Moreover, the assessment of functional plant traits is based on the measurement of individual trees with a strong focus on species identity (Kröber, Li, et al., 2015). Even if measurements are carried out on single leaves or branches, they will also refer to a particular tree individual (Brezzi, Schmid, Niklaus, & Schuldt, 2017). The local neighborhood comprises all immediate neighbor trees of a focal tree individual (Fichtner et al., 2017). Defining neighborhood in this way makes it independent of tree size. How the local neighborhood influences individual tree performance is of particular importance because positive tree–tree interactions at the local scale may translate into positive biodiversity effects at community scale (Forrester & Bauhus, 2016; Potvin & Dutilleul, 2009). In contrast, plot-level measurements integrate ecosystem functions over the entire tree community. Such measurements are used, for example, to quantify the impact of tree species richness and composition on decomposition processes (Eichenberg et al., 2017; Seidelmann, Scherer-Lorenzen, & Niklaus, 2016). Plot-level measurements also apply to mobile organisms at higher trophic levels that are not confined to particular trees (Vehviläinen, Koricheva, & Ruohomäki, 2008) and to combined effects of soil fertility and topography on tree growth (Scholten et al., 2017).

Given that each method aims to contribute information at the respective scale, a well-balanced mixture of methods is required to maximize knowledge gain from cost- and labor-intensive (land rent, plot clearing, tree planting, and weeding) forest BEF experiments. Therefore, a wide spectrum of easy and sophisticated BEF measurements must be combined in a multifunctional framework to quantify ecosystem functioning on a large set of plots. Standardized methods for key ecosystem functions (Meyer, Koch, & Weisser, 2015) and rapid biodiversity assessments (Obrist & Duelli, 2010) need to be developed or adapted for forest ecosystems to promote synthesis studies across tree diversity experiments. Because these experiments are commonly used by many research teams from different disciplines and backgrounds, careful consideration of the applied methods is required to measure and analyze data jointly and effectively. Together with an integrated project data management ensuring data harmonization, data validation, and metadata quality, synthesis projects can be catalyzed in a multifunctional context (Nadrowski et al., 2013).

Only if we succeed in combining the results obtained by different methods, a coherent account of forest ecosystem functioning can be achieved.

Based on an illustrative example of a forest BEF experiment (BEF-China), we provide an overview on state-of-the-art methods currently applied in one of the largest forest biodiversity experiments worldwide. Given the increasing number of tree diversity experiments and cross-site synthesis approaches (Verheyen et al., 2016), the present work is a first attempt to develop standardized BEF methods to measure forest multifunctionality. Methods for the assessment of multiple ecosystem functions and variables are briefly described with focus on their practicability as well as their challenges that have been encountered. In a second step, we outline how methods differing in scope and complexity can be combined to retrieve consistent data allowing novel insights into forest ecosystem functioning. Finally, we discuss and develop recommendations for the integration and transferability of diverse methodological approaches across present and future forest diversity experiments.

2 | BEF-CHINA AS A CASE STUDY OF A LARGE TREE DIVERSITY EXPERIMENT

BEF-China is the first tree diversity experiment in the humid subtropics, established 2009/2010 in southeast China (Xingangshan, Jiangxi Province) with a total net area of 38.4 ha (Figure 1) distributed across two hilly landscapes (site A and B). The overall design and establishment success of the experiment are provided by Bruelheide et al. (2014) and Yang et al. (2013). A unique feature of the experiment is the large range of tree species richness levels and different nonoverlapping species combinations within different random and nonrandom (trait-driven) extinction scenarios. The size of the total species pool is 40 tree species, and richness is varied along a log-2 series from monocultures up to 16 species with an additional richness level of 24 species for the most diverse plots. The experiment contains more than 500 plots of 25.82 m × 25.82 m area (in horizontal projection), each planted with 400 trees in a regular grid of 20 rows × 20 columns. In two of the random extinction scenarios, tree diversity is factorially crossed with shrub diversity planted in between the trees at the same density as those. The experiment has been established on sloped terrain that allows assessing plant diversity effects on the reduction in soil erosion—an ecosystem service of high environmental importance in rain-laden southeast China.

To separate tree diversity effects from influences of abiotic environmental covariates, environmental heterogeneity was quantified by assessing local and regional topography, microclimate, and edaphic conditions at the beginning of the experiment; in relation to the term landscape, we refer to this environmental heterogeneity as “ecoscape” (Bruelheide et al., 2014; Scholten et al., 2017). A wide range of functional responses and processes is being studied, such as tree growth, soil erosion, plant functional traits, importance of plant genetic diversity, plant–insect interactions, and nutrient cycling, including trophic interactions with microbial and animal decomposers. Rather than presenting an exhaustive compilation of currently obtained measurements, we provide a concise overview on key aspects of forest ecosystem functioning to illustrate the broad range of methods applied (Figure 2, Table 1). It is clear that the presented methods cannot serve as a blueprint for other tree diversity experiments but should be rather regarded as stimulus to rethink methodical concepts and approaches for large cooperative projects and networks. We begin with methods for assessing plant growth and facets of tree diversity (leaf functional trait diversity and tree genetic diversity) and extend the scope to multitrophic interactions, nutrient cycling, and soil erosion.

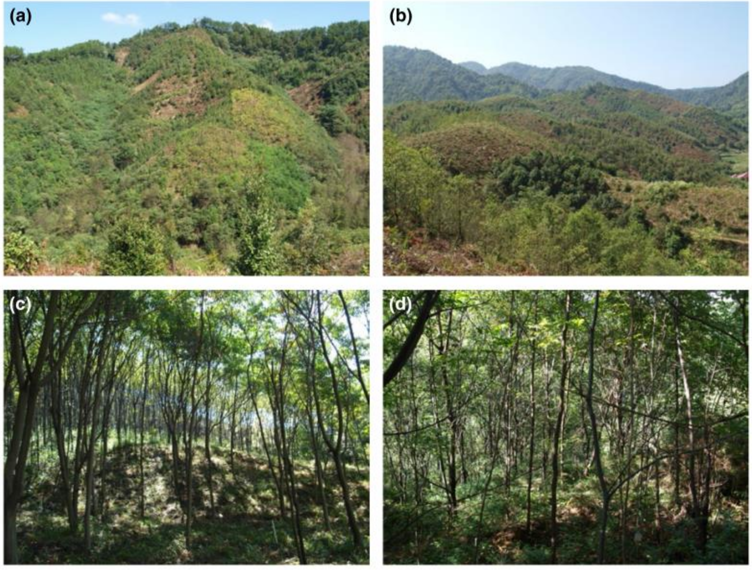

FIGURE 1 Example of a large tree diversity experiment: (a) partial view of site A and (b) site B of the BEF-China experiment seven and six years after planting, respectively. (c) Monoculture plot of *Triadica cochinchinensis* (site A) and (d) eight-species tree mixture of *Castanea henryi*, *Castanopsis sclerophylla*, *Choerospondias axillaris*, *Liquidambar formosana*, *Nyssa sinensis*, *Quercus serrata*, *Sapindus saponaria*, and *Triadica sebifera* (site A). To increase generality of BEF relationships, the experiment was established at two sites (about 5 km apart) with only small overlap of species pools. Photographs: S. Trogisch

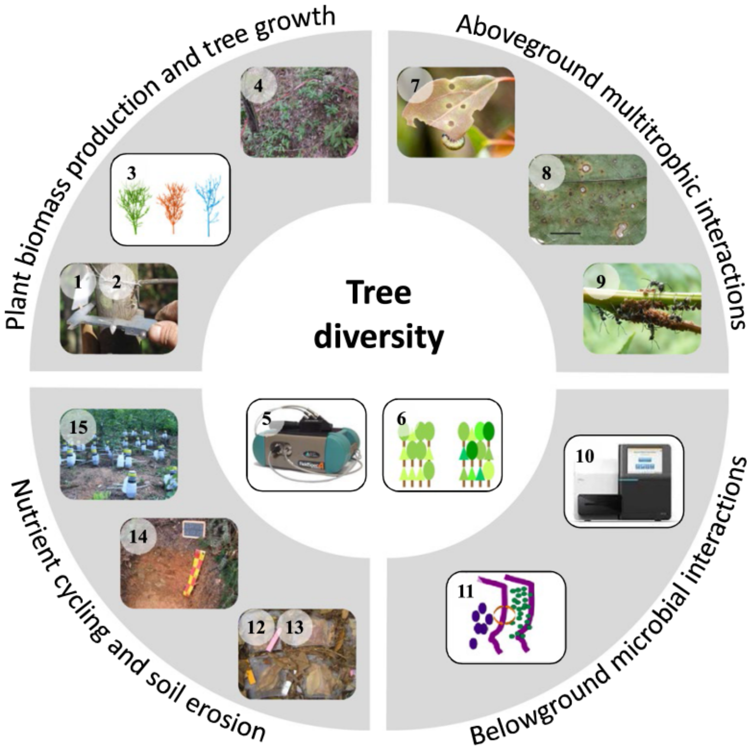

FIGURE 2 Range of methodical approaches applied in BEF-China to study effects of tree diversity including leaf functional trait diversity (5) and genetic diversity (6) on plant biomass production and tree growth (1 + 2 = aboveground and belowground tree biomass and productivity, 3 = tree growth and canopy architecture, 4 = herb-layer biomass and diversity), aboveground multitrophic interactions (7 = herbivory, 8 = plant-fungal pathogens interactions, 9 = trophobiosis), belowground microbial interactions (10 = microbial diversity, 11 = microbial biomass and activity), nutrient cycling and soil erosion (12 + 13 = leaf litter and deadwood decomposition, 14 = soil fertility and C storage, 15 = soil erosion). Numbers in this figure reflect numbering of ecosystem functions and variables in Table 1

2.1 | Plant biomass production and tree growth

2.1.1 | Aboveground tree biomass and productivity

The adequate assessment of tree biomass production in large BEF experiments is critical to investigate the influence of different facets of tree diversity (species richness, presence of particular species, species composition, functional diversity, and genetic diversity) on tree growth at the individual, neighborhood, and plot (= community) scale. Basically, tree biomass production is quantified by repeated measurements of tree size variables and subsequent calculation of tree biomass based on allometric equations, avoiding artefactual species identity effects which can be a result of using different functions for different species (e.g., Forrester, Benneter, Bouriaud, & Bauhus, 2017). However, comprehensive annual inventories with measurement of basal diameter, diameter at breast height (DBH, caliper, and measurement tape), and tree height (graduated pole for small trees and hypsometer) for all planted trees often exceed available project resources such as workforce and time. Therefore, in most cases, there is a trade-off between the number of sampled plots and the number of sampled trees. One solution is to carry out these measurements on a section within plots. In BEF-China, the central 16 of the 400 trees in every plot were defined as a core area and chosen for annual measurements.

In addition to quantifying woody biomass, leaf turnover has to be considered as a significant part of net primary production. Leaf production, herbivory, and mortality can be determined easily and cost-effectively by regular monitoring of marked leaf cohorts on selected tree individuals (Brezzi et al., 2017; X. Li, unpublished data). At the beginning of the observation period, branches are marked and leaves counted. Subsequent censuses can follow at for example half-yearly intervals, but interval length can be shorter during times of intensive growth because variable interval lengths can be accounted for using offsets in the data analysis (Egli & Schmid, 2001). Effects of tree species richness and time-dependent covariates on leaf demographic patterns can then be estimated (Castro-Izaguirre, 2016). Once trees have reached a certain height, community litter and seed production can be determined with litter traps (Huang et al., 2017).

The leaf area index (LAI), defined as the ratio of projected foliage area to ground area, is an important structural variable for key ecophysiological processes (e.g., energy interception and transpiration). Most commonly, LAI is indirectly measured as interception of photosynthetically active radiation (PAR) or by analysis of hemispherical photographs (Castro-Izaguirre et al., 2016; Peng, Schmid, Haase, & Niklaus, 2017). Both methods have their advantages and disadvantages, which are further discussed in Asner, Scurlock, and Hicke (2003) and Bréda (2003).

2.1.2 | Belowground tree biomass and productivity

Fine roots (diameter ≤ 2 mm) are the most active part of the root system (Asaye & Zewdie, 2013), interacting with soil microflora and

TABLE 1 Overview of methods for the assessment of key ecosystem functions and variables in tree diversity experiments. The spatial assessment level can be the individual tree (T), the local neighborhood (N) for studying tree–tree interactions, and the plot (P). References specific to the BEF-China tree diversity experiment are marked with an asterisk. Temporal scope and measurement intervals for respective methods have been adapted to the requirements of BEF-China and may depend on research focus and environmental setting

No.	Ecosystem function/ variable	Method	Details/considerations	Temporal scope	Spatial assessment level (T/N/P)	References
Plant biomass production and tree growth						
1	Aboveground biomass and productivity	Repeated measurement of DBH (caliper, measurement tape, and dendrometer) and height (graduated pole and hypsometer)	Often only applicable for a subset of inventoried tree individuals (e.g., central 4 × 4 individuals). Allometric equations required for biomass calculation.	Annual inventory.	T	Clark, Wynne, and Schmoldt (2000) Clark et al. (2001) Li, Härdtle, et al. (2014)*
		Repeated assessment of marked leaf cohorts	Species-specific leaf formation and longevity can be studied. Method restricted to young trees due to limited canopy access.	Half-yearly intervals.	T	Reich, Uhl, Walters, Prugh, and Ellsworth (2004)
		Litter traps	Determination of litter production and shed leaf area. Allows quantification of nutrient fluxes from canopy to soil. Litter collection from traps on regular basis time-consuming. Three litter traps per plot in core area (4 × 4 trees)	Biweekly litter collection over several years.	N/P	Bernier, Hanson, and Curtis (2008)
		Leaf area index (LAI)/ hemispheric photography	Repeated measurements in central plot area (6 × 6 trees) allow LAI quantification during stand development. Digital hemispherical photography using a fish-eye device less sensitive to uneven sky brightness.	Annual measurement.	N/P	Asner et al. (2003) Jonckheere et al. (2004) Peng et al. (2017)*
2	Belowground biomass and productivity	Soil cores	Destructive method for measuring root biomass, root distribution, and nutrient content. Image analyses of root scans can provide additional information on root diameter and length.	Annually or less frequently.	T/N/P	Sun et al. (2017)*
		Ingrowth cores	Destructive method for measuring root productivity.	Ingrowth core retrieval after 1 year.	T/N/P	Lei, Scherer-Lorenzen, and Bauhus (2012) Sun et al. (2017)*
		Minirhizotrons	Nondestructive assessment of fine-root dynamics in situ.	Pictures taken twice per year.	T/N/P	Taylor et al. (2014)
3	Tree growth and canopy architecture	Terrestrial laser scanning (TLS)	Three-dimensional (3D) structural elements of trees. Rapid, nondestructive, accurate, and extensive measurements of a large number of individual trees over time possible.	Annually or less frequently.	T/N	Li, Hess, et al. (2014)*
4	Herb-layer biomass and diversity	Herb-layer monitoring	Vegetation survey by transect-method (for inventory data). Additional composition analysis in subplot surveys. Biomass harvest in 0.5 m × 0.5 m quadrates.	Annually or less frequently.	N/P	Both et al. (2011)* Ampoorter et al. (2015) Germany et al. (2017)*

(Continues)

TABLE 1 (Continued)

No.	Ecosystem function/ variable	Method	Details/considerations	Temporal scope	Spatial assessment level (T/N/P)	References
Facets of tree diversity						
5	Leaf functional trait diversity	Near-infrared spectroscopy (NIRS)	Rapid and cost-effective assessment of important leaf traits to identify linkages between functional traits and ecosystem processes. Portable NIRS allows nondestructive and highly repeated measurements in situ. Trait-specific calibration required.	Intraday to annual measurements.	T	Serbin et al. (2014)
6	Genetic diversity	Maternal seed families, phytometer plants	Influence of seed family identity/genetic diversity on tree performance.	Annual measurements.	T	Avolio, Beaulieu, Lo, and Smith (2012) Zeng, Durka, & Fischer, (2017)* Zeng, Durka, Welk, et al. (2017)* Hahn et al. (2017)*
Aboveground multitrophic interactions						
7	Herbivory	Quantification of leaf damage (one-time measurement)	Allows quick assessment of herbivory on a large number of trees. Leaf age important, thus assessment of only young and fully expanded leaves. Visually estimated leaf damage verified by leaf scans. Assessment of 6 × 6 trees in monocultures to 12 × 12 trees in more species-rich plots.	Annually or less frequently.	T	Schuldt et al. (2012)* Schuldt, Bruelheide, et al. (2015)*
8	Plant—fungal pathogens interactions	Foliar fungal pathogens assessment	Quantification of pathogen infestation using a percentage class system of leaf damage with six damage classes. Susceptibility to pathogens as an additional species trait. Assessment of 6 × 6 trees in monocultures to 12 × 12 trees in more species-rich plots.	Annually or less frequently.	T	Hantsch, Bien, et al. (2014)*
9	Trophobiosis	Trophobiosis as model system	Systematic survey of aphids and tending ants on at least 20 young leaves per tree. Ideal model system to quantify multitrophic interactions. Assessment of 6 × 6 trees in monocultures to 12 × 12 trees in more species-rich plots.	Monthly survey during growing season.	T	Staab et al. (2015)*
Belowground microbial interactions						
10	Microbial diversity	Meta-barcoding of rhizosphere soils using next-generation sequencing platforms	Determine the structural and functional diversity and community composition of soil microbes (mainly fungi and bacteria). Central plot area (12 × 12 trees).	Annual measurements or less frequently.	T/N	Wu et al. (2013)* Lentendu et al. (2014)

(Continues)

TABLE 1 (Continued)

No.	Ecosystem function/ variable	Method	Details/considerations	Temporal scope	Spatial assessment level (T/N/P)	References
11	Microbial biomass and activity	Phospholipid fatty acid analysis (PLFA) combined with high-throughput method of lipid extraction; 15N dilution method, extracellular enzyme activity assays (EEA)	Determination of microbial community composition and total microbial biomass. Measurement of gross rates of N mineralization. Central plot area (12 × 12 trees).	Annual measurements or less frequently.	T/N	Oates et al. (2017) Pei et al. (2016)* Pei et al. (2017)*
Nutrient cycling						
12	Leaf litter decomposition	Litterbags with site-specific or standardized leaf litter	Inexpensive, highly repeatable and time-efficient. Standardized litter substrates (e.g., tea bags) facilitate global synthesis studies. Neglects effects of soil macrofauna.	Duration about 12 months with usually several retrieval dates.	N/P	Keuskamp et al. (2013) Trogisch et al. (2016)* Seidelmann et al. (2016)*
13	Deadwood decomposition	Litterbags with standard-sized wood pieces	Limited to smaller wood pieces. Size of wood samples important for decomposer fauna. Easy exclusion of certain decomposers (termites) by mesh size.	Wood pieces retrieval after one and 3 years.	N/P	Russell et al. (2015) Eichenberg et al. (2017)*
14	Soil fertility and C storage	Schematic soil sampling combined with near-infrared spectroscopy (NIRS)	Facilitate inexpensive analyses and rapid assessment of large number of samples in subsequent inventories.	Annual measurements or less frequently.	N/P	Scholten et al. (2017)* Ludwig et al. (2002)
Soil erosion control						
15	Throughfall kinetic energy	Splash cups	Allow indirect determination of rainfall kinetic energy at many measurement points in parallel during single rainfall events. Calibration by laser distrometer required. Eight splash cups in central plot area (6 × 6 trees).	Series of rain events.	T/N/P	Scholten et al. (2011)* Goebes, Bruelheide, et al. (2015)*
15	Soil erosion (interrill)	Microscale runoff plots	Determination of surface runoff and sediment discharge. Suitable to study vegetation effects on soil erosion processes. Five runoff plots per plot.	Series of rain events.	T/N/P	Seitz et al. (2015)* Seitz et al. (2016)*
15	Soil erosion (slope scale)	Erosion sticks	Simple and cost-effective method to quantify large-scale and long-term soil erosion. Nine erosion sticks per plot.	Reading of the height above ground once per year.	N/P	Shi et al. (2011)

fauna and being involved in nutrient and water uptake (Jackson et al., 1996). Thus, understanding fine-root dynamics is pivotal for understanding belowground interactions as well as tree growth and survival (McCormack et al., 2015). However, measuring belowground biomass and productivity is challenging as usually destructive sampling is required to separate the roots from the soil (Brassard et al., 2013). Furthermore, on sloped plots, such those in BEF-China, an important question regarding comparability with other experiments is whether layers of soil depth should be measured perpendicular to the soil surface or to its horizontal projection. Here, our recommendation is to use a direction perpendicular to the soil surface (Sun et al., 2017).

Standing fine-root biomass can be measured using the soil core method. Soil cores (10 cm in diameter, 30 cm in depth) are usually taken in the middle of two neighboring trees standing in the same horizontal row (Sun et al., 2017). Depending on soil type, fine roots should be sampled by soil depth increment to estimate the vertical variance of standing biomass. In BEF-China, we were able to assign washed roots to each of the 40 species using root morphology. This allowed us to estimate the contribution of different species to overyielding of total community-level fine-root biomass in mixtures (Bu et al., 2017; Sun et al., 2017). In addition, roots can be scanned for analysis of diameter and specific root length (Bu et al., 2017; Sun et al., 2017). For estimation of annual production of fine roots, we recommend the traditional method of ingrowth cores (Sun et al., 2017). Right after taking the soil core for standing biomass, the cavity is refilled with sieved soil from the same plot. Ingrowth cores are resampled after 1 year, and biomass of both live and dead fine roots is measured.

As a nondestructive method, minirhizotrons have been developed to monitor fine-root dynamics along time intervals (Guo et al., 2008; Majdi, 1996; Taylor, Beidler, Strand, & Pritchard, 2014). Minirhizotron tubes (typically length 90 cm and diameter 7 cm) are installed in the middle of two conspecific (in monoculture) or heterospecific (in mixtures) neighbored trees in an angle of 45° to the soil surface. Tubes are scanned at intervals, for example, twice per year in May and November, and pictures analyzed for fine-root length, area, amount, longevity, and turnover rate.

2.1.3 | Tree growth and crown architecture

Understanding the mechanisms of biodiversity effects in forests requires information about crown structure and space partitioning between trees within and between species (Jucker, Bouriaud, Coomes, & Baltzer, 2015; Niklaus, Baruffol, He, Ma, & Schmid, 2017; Pretzsch, 2014; Schmid & Niklaus, 2017; Williams, Paquette, Cavender-Bares, Messier, & Reich, 2017). However, conventional measurements are time-consuming and do not deliver much detail. In recent years, terrestrial laser scanning (TLS) has been established as a time-efficient and nondestructive approach for the measurement of the 3D structural elements of trees (Calders et al., 2015; Liang et al., 2016). Based on static laser range measurements, TLS delivers high-resolution 3D point clouds with accuracies in the range of millimeters. In order to obtain a complete 3D picture of all focal trees in each plot, several scans from different angles are required (Watt & Donoghue, 2005). Setting up the instrument and acquiring the 3D data are generally straightforward and fast. In the case of the BEF-China experiment, the central part of a plot with 6 × 6 trees can be captured in high detail (more than 100,000 points per tree) from nine scans within 45 min (Li, Hess, von Wehrden, Härdtle, & von Oheimb, 2014). Strong winds and occlusion by foliage may adversely affect the point cloud quality (Côté, Fournier, & Egli, 2011). Therefore, scans should to be performed under windless and—if possible—under leaf-off conditions. Thus, in stands with deciduous trees, the preferred time for applying TLS is winter. From the resulting point clouds, a number of conventional (i.e., height and DBH), but also more complex variables (i.e., branch demography, crown volume, and wood volume), can be obtained for every tree (Kunz et al., 2017; Raumonen et al., 2013). Meanwhile, the extraction of these variables has become highly automated. However, the separation of tree individuals from a large point cloud with many trees still is a challenge, and so far, is predominantly carried out manually. With repeated TLS measurements, it is possible to quantify spatial dynamics of individual crowns and canopy filling using cylinder-based (Raumonen et al., 2013) or voxel-based (Hess, Bienert, Härdtle, & von Oheimb, 2015) point cloud modeling approaches.

2.1.4 | Herb-layer biomass and diversity

It has been shown that trees exert strong controls on herb-layer biomass, composition, richness, and invasibility (e.g., by altering resource availability and variability) (Ampoorter et al., 2015; Barbier, Gosselin, & Balandier, 2008; Knight, Oleksyn, Jagodzinski, Reich, & Kasprowicz, 2008; Mölder, Bernhardt-Römermann, & Schmidt, 2008). Considering the special role of the herb layer in maintaining the structure and function of forests (Gilliam, 2007), improved understanding of how tree diversity affects herb-layer attributes and seedling establishment is critical. In this respect, tree diversity experiments allow for assessing the relationships between forest overstory and understory species richness, composition, and productivity, and how these relationships are influenced by spatial environmental heterogeneity and forest stand age (Both et al., 2012).

In large-scale forest experiments, full-vegetation relevés are laborious and time-consuming. At the plot level, W-transects (i.e., linear transects in the shape of a W) provide a time- and resource-efficient method for repetitively assessing herb-layer species inventory as species richness and composition with information on estimated proportions in cover on a large number of plots. In addition, herb-layer vegetation surveys performed on separate and integrated subplots (Germany, Bruelheide, & Erfmeier, 2017) can be used to explicitly test if the relationships between tree diversity and herb-layer attributes change under variable environmental conditions (Reich et al., 2001; Weigelt, Weisser, Buchmann, & Scherer-Lorenzen, 2009).

We recommend an integrated manipulation of resource supply and biotic impact (e.g., fertilization, annual weeding, no weeding, and functional group removal) at the subplot level. Biomass harvest by plant functional groups (forbs, grasses, climbers, and woody seedlings) on randomly located quadrates within each subplot can serve as a proxy for overall herb-layer productivity and its functional group components. In combination with a vegetation survey, this approach allows assessing relationships between tree species richness and the presence of particular tree species in a plot with the richness, composition, and productivity of the herb layer. Such a combined approach reveals the extent to which these relationships change at different environmental settings when taking spatial heterogeneity at the site level into account.

2.2 | Facets of tree diversity

2.2.1 | Leaf functional trait diversity

A primary goal of BEF research is to identify linkages between functional plant traits and ecosystem processes (de Bello et al., 2010; Díaz et al., 2007). In particular, the detection of key functional traits and their interrelationships and trade-offs is of great importance to derive a mechanical understanding of ecosystem functioning. For example, the link between key functional leaf traits (e.g., nitrogen concentration and specific leaf area) and photosynthetic capacity as well as carbon capture has been well established in across-site studies (Wright et al., 2004), but also occurs among subtropical forest tree species within sites. However, destructive sampling and time-consuming analyses often limit sample size. Thus, it is desirable to assess plant traits including intraspecific trait variation in high spatial and temporal resolution by nondestructive and cost-efficient high-throughput methods. Near-infrared reflectance spectroscopy (NIRS) has only recently been introduced to ecological research (e.g., Serbin, Singh, McNeil, Kingdon, & Townsend, 2014; Zuppinger-Dingley, Flynn, Brandl, & Schmid, 2015), although it is a well-established method for plant chemical analyses. As many leaf properties such as foliar C, N, phenolics, or leaf dry-matter content show specific NIR reflectance spectra, target leaf traits can be easily assessed at different scales, from ground leaf powder to fresh leaves, entire tree canopies or forest ecosystems, once compound-specific calibrations have been established (Couture et al., 2016; Foley et al., 1998). However, calibration requires a sufficiently high number of reference samples (approx. 200–300) with known trait information to yield reliable predictions for NIR spectrometry (Eichenberg et al., 2015).

Special attention has been given recently to field portable instruments which allow on-site and nondestructive measurements, thereby making sample preparation and transport unnecessary (Galuszka, Migaszewski, & Namiesnik, 2015; Serbin et al., 2014). Portable spectrometers accelerate data collection and make it possible to consider intraspecific trait variation, for example, within-canopy variation or temporal variation of leaf traits. An initial study conducted in the BEF-China experiment on 4,892 leaves from 2,759 trees showed that important leaf traits (e.g., leaf dry-matter content, specific leaf area, and C:N ratio) could be reliably predicted by portable field spectroscopy (Tobias Proß, unpublished data). It has been shown that quality of prediction differs among leaf traits because the high NIR absorption of water can reduce spectral information of other target components in fresh leaf samples. However, high measuring resolution outperforms potential drawbacks such as lower data quality and calibration efforts (Galuszka et al., 2015), which makes portable field spectroscopy an effective high-throughput method for assessing leaf traits in large tree diversity experiments.

2.2.2 | Genetic diversity

Genetic diversity, especially heritable genetic variation in plant traits and in trait plasticity, causes large variation in plant performance (Frankham, 1999; Zeng, Durka, Welk, & Fischer, 2017) and represents the raw material for future adaptive evolution. Genetic diversity should therefore be considered as an additional facet of diversity that can influence plant performance in biodiversity experiments (Booth & Grime, 2003; Hahn et al., 2017; Schmid, 1994; Zeng, Durka, & Fischer, 2017). Genetic variation is generally found in studies on variation between plants from different genetic entities, such as provenances, populations, or maternal seed families. Moreover, different genotypes often respond differently to environmental variation resulting in genotype–environment interactions (Stearns, 1992).

In experimental analyses of biodiversity–functioning relationships, two issues should be considered. First, controlling for variation among genetic entities of the planted material (e.g., seed families and provenances) very much increases the resolution and statistical power for finding variation at the species level. Thus, in tree diversity experiments, the genetic identity of planted trees should be considered, for example using maternal seed families. Seeds of a maternal plant representing a seed family need to be collected, and seedlings need to be raised with recorded seed family identity. Seed family identity then needs to be considered during the experimental set-up to control genetic variation, for example, by planting representatives of an equal number of seed families in all plots. Moreover, genetic variation can be manipulated using different numbers of seed families, for example, in order to assess the relative role of genetic variation at the inter- and intraspecific level (Hahn et al., 2017; Zeng, Durka, & Fischer, 2017).

Second, the omnipresence of genotype-by-environment interactions suggests that different genotypes may respond differently to experimental environments. Thus, members of seed families planted into experimental plots of different species diversity may serve as phytometer plants (Gibson, 2002; Mwangi et al., 2007) for diversity effects. Such phytometer plants offer the advantage that they can be planted into all experimental plots.

In the BEF-China experiment, trees of known seed family were used (1) as matrix species in the main experiment, where for 12 species (~58.000 planting positions), seed family identity was recorded; (2) in a factorial species diversity × genetic diversity experiment, where genetic variation was manipulated using different numbers of seed families (Hahn et al., 2017); (3) as an additional phytometer by planting *Machilus thunbergii* seed families into each plot of the experiment.

Using seed families as matrix species or as phytometers allows to assess the heritability, that is, the amount of heritable genetic variation, in plant performance or plant traits using quantitative genetic methods and assuming a certain sibship coefficient between maternal seed families (e.g., ¼ for the case of half-sib relations; Falconer, 1989; Lynch & Walsh, 1998; Zeng, Durka, Welk, et al., 2017). Moreover, genetic variation in phenotypic trait plasticity will become apparent, if seed families respond differently to differences between experimental treatments (Scheiner & Lyman, 1989). In conclusion, the use of multiple maternal seed families is a powerful experimental tool to increase the statistical power to detect variation at the species level, to quantify the heritability of plant traits and their plasticity, and to experimentally manipulate genetic variation.

2.3 | Aboveground multitrophic interactions

2.3.1 | Herbivory

Herbivory directly affects resource allocation, trait expression, and plant growth (Agrawal, 2007; Coley & Barone, 1996; Viola et al., 2010). These factors all influence plant community composition, primary production, and nutrient cycling (Schmitz, 2008). Large-scale herbivory assessments often trade-off time efficiency and sampling accuracy, and quantification of leaf damage has become a standard method. Leaf damage is measured by either recording herbivory rates (increase in damage between two time points) or by measuring standing damage levels (i.e., one-time measurements). For large-scale BEF experiments, we recommend the latter, less time-consuming method as a quick assessment tool (see also Johnson, Bertrand, & Turcotte, 2016). However, care needs to be taken as differences in leaf age can compromise comparisons among species (Poorter, van de Plassche, Willems, & Boot, 2004). We therefore recommend to use young (current season) leaves. If time of leaf flush differs substantially among tree species (which is not the case in BEF-China; Schuldt et al., 2012), these differences need to be considered in assessment timing and data interpretation. Most studies visually estimate leaf damage, by either comparing total and damaged leaf area (Poorter et al., 2004) or using predefined damage classes (Schuldt, Bruelheide, et al., 2015; Sobek, Scherber, Steffan-Dewenter, & Tscharntke, 2009; Unsicker et al., 2006; Vehviläinen, Koricheva, & Ruohomäki, 2007). For the BEF-China project, predefined damage classes (0%, ≤5%, ≤25%, ≤50%, ≤75%, and >75%) have proven useful. Estimation accuracy has been assessed with digital scans of randomly collected leaves (Schuldt et al., 2012). For each tree, seven young, fully expanded leaves are screened on each of three randomly selected branches. With increasing tree height, branches are selected to represent upper, mid, and lower crown conditions. The sampling design follows the assessment of tree growth, comprising 6 × 6 individuals in monocultures and two-species mixtures and up to 12 × 12 individuals in the more species-rich plots. As the number of trees of a certain species per plot decreases with increasing tree diversity (because of constant planting density), an increase in the number of sampled trees per plot is necessary to allow for species-level analysis at the tree level. Such analysis requires that all species are represented by a similar amount of tree individuals irrespective of the level of tree diversity (Bruelheide et al., 2014; Schmid, Baruffol, Wang, & Niklaus, 2017).

2.3.2 | Plant–fungal pathogens interactions

Parasitic interactions between plant hosts and fungal pathogens often cause a reduction in individual plant fitness by fungal consumption of photosynthetic products and negatively affect photosynthesis rates (Alves, Guimarães, Chaves, DaMatta, & Alfenas, 2011; Berger, Sinha, & Roitsch, 2007; Mitchell, 2003). The diversity and species composition of the plant host community affect fungal dispersal, infection, and infestation, mainly through negative density effects (Hantsch, Bien, et al., 2014; Hantsch, Braun, et al., 2014; Moore & Borer, 2012; Ostfeld & Keesing, 2012).

One advantage of a noninvasive rapid leaf damage assessment is the investigation of a high number of leaves and individuals of different plant species. For species comparability, we only use well-formed leaves from the current year which are macroscopically screened for leaf damage caused by fungal spot and lesion symptoms, mildews, rusts, and sooty molds, respectively, at the end of the vegetation period. Similar to the herbivory assessment, total fungal damage is evaluated by damage classes (i.e., 0%, ≤5%, ≤25%, ≤50%, ≤75%, and >75%) on seven leaves randomly chosen from three different branches (representing different crown conditions), which were randomly selected per tree individual. The fungal damage assessment included (like other tree-level measurements) an increasing number of tree individuals with increasing tree diversity to ensure a representative number of individuals per tree species per plot (i.e., 6 × 6 individuals in monocultures and two-species mixtures, 9 × 9 individuals in four-species mixtures, 12 × 12 individuals in eight-, 16-, and 24-species mixtures).

In contrast to the more common microscopic in-depth investigation of fungal pathogens (Hantsch, Braun, Scherer-Lorenzen, & Bruelheide, 2013; Hantsch, Bien, et al., 2014) or identification of foliar fungi with molecular high-throughput sequencing (Nguyen et al., 2017), fungal damage assessment needs not only less time allowing a higher sample size, but also works without specific expertise about fungal species.

2.3.3 | Trophobiosis

Tritrophic interactions between plants, sap-sucking Hemiptera (e.g., aphids), and tending ants, so-called trophobioses, are common in forests across climate zones (Ivens, von Beeren, Blüthgen, & Kronauer, 2016) and thus an ideal model system to quantify multitrophic interactions in forest BEF experiments.

We suggest and use in BEF-China the following simple protocol for trophobiotic interactions that allows time-efficient sampling of large numbers of trees (Staab, Blüthgen, & Klein, 2015). On each tree, at least 20 young leaves together with the attached branch sections are visually inspected for the occurrence of sucking Hemiptera and tending ants. If possible, surveys should be carried out monthly covering the main growing season. For Hemiptera and ant species that cannot be reliably identified in the field, voucher specimens are collected and stored in 70% ethanol for later identification. To ensure the sampling of a sufficiently large number of individuals of all tree species also in high-diversity plots, we suggest increasing the number of sampled tree individuals with the tree diversity level of a given plot (see *Herbivory*). The data can be analyzed for the effect of tree species identity and tree species diversity. The R-package "bipartite" offers all tools for ecological network analyses (Dormann, Fründ, Blüthgen, & Gruber, 2009). From our experience, network-level specialization H_2' (Blüthgen, Menzel, & Blüthgen, 2006) and weighted generality G_{qw} (Bersier, Banašek-Richter, & Cattin, 2002) are particularly useful to analyze the specificity and generality of plant–Hemiptera and Hemiptera–ant associations in response to tree diversity.

Besides simple and efficient sampling and data evaluation, a great advantage of trophobioses is that two fundamentally different forms of trophic interactions, consumption and mutualism (Thébault & Fontaine, 2010), can be studied simultaneously. If aphids are attacked by parasitoids, another trophic interaction can be added to the study system allowing an assessment of the ecosystem function parasitism (e.g., Gagic et al., 2011).

2.4 | Belowground microbial interactions

2.4.1 | Microbial diversity

Soil microbes are crucial components of terrestrial ecosystems. They deliver key ecosystem functions and influence important ecosystem processes, including nutrient cycling and nutrient acquisition (Bardgett & van der Putten, 2014). Recent advances in next-generation sequencing (NGS) techniques coupled with meta-barcoding approaches and the associated bioinformatics and statistical analysis tools enabled microbial ecologists to work in large-scale tree diversity experiments to shed light on the poorly understood role of microbial diversity on BEF relationships in forest ecosystems.

Although the advance in NGS and the possibility to analyze a large number of samples have led to large-scale and integrated biodiversity studies at the global scale (Shoemaker, Locey, & Lennon, 2017), standardized soil sampling, storage, and transportation across continents still are a challenge. Accordingly, we developed a soil sampling, freeze-drying, and preservation protocol that guarantees transportation of soil samples without nucleic acid degradation between laboratories across continents (Weißbecker, Buscot, & Wubet, 2017). The soil microbial nucleic acid extraction protocols have been optimized to a high-throughput protocol, and the classical PCR-based microbial diversity analysis protocols using microbial rDNA-based barcodes (e.g., 16S for bacteria and ITS for fungi) have been adapted to meta-barcoding protocols using NGS platforms (Lentendu et al., 2014; Wu et al., 2013).

Another crucial point is the sampling strategy. Soils are anything but a homogenous compartment, and even within each horizon, they are a complex patchwork of microhabitats with variable levels of resources and very specific communities. In BEF experiments, a crucial decision is whether to sample the roots and rhizosphere of each plant species used in the design or to sample the bulk soil. The rhizosphere has a selective filtering effect differing between plant species, while the bulk soil may better reflect the general effect of a plant biodiversity level on the whole microbial community. Even for mycorrhizal fungi directly linked to plant roots, it was shown in grassland studies that analyzing bulk soil better captures biodiversity than focusing on roots (Hempel, Renker, & Buscot, 2007). In addition, preliminary analyses in BEF-China found not only the highest soil microbial biomass and activities in the uppermost horizon under the plant litter, but also that this was the most reactive soil layer to variations in the biodiversity and age structure of the trees and understory (Wu et al., 2012). Based on our experience, we recommend that broad analyses of soil microbial communities in BEF experiments should be based on multiple samples from the upper soil layer at equal distance from neighbor plants. These samples can be pooled into a composite sample from which the DNA is extracted and analyzed (Wu et al., 2013).

Integrating the microbial species (operational taxonomic units—OTU) abundance matrices with other co-occurring organisms and environmental variables and using ecological statistical analysis tools enabled us to assess the significance of soil microbes on inter- and intrakingdom interaction networks, multitrophic interactions, forest ecosystem functions, and multifunctionality.

2.4.2 | Microbial biomass and activity

The effects of tree species diversity on soil microbial community structure and activity remain poorly understood, despite the important role of soil microorganisms for ecosystem functioning (Naeem et al., 2000; Zak, Holmes, White, Peacock, & Tilman, 2003).

Phospholipid fatty acid analysis (PLFA) has been validated as a valuable approach of investigating soil microbial community composition and viable microbial biomass (Bartelt-Ryser, Joshi, Schmid, Brandl, & Balser, 2005; Frostegård & Bååth, 1996; Frostegård, Tunlid, & Bååth, 2011; Pei et al., 2016; Vestal & White, 1989). Recently, a high-throughput method of lipid extraction and analysis has been developed, which allows for lipid profiling for large ecosystem studies (Gutknecht, Field, & Balser, 2012; Oates et al., 2017). In this method, the initial soil chloroform extraction is carried out in the standard procedure ("modified" Bligh and Dyer (1959) extraction) and then followed by the FAME procedure of saponification, acid methylation, and extraction (Schutter & Dick, 2000). This high-throughput method retains the sensitivity of traditional PLFA methods, but allows for much more rapid analysis of a large number of samples, for example enabling us to demonstrate how tree species identity and growth traits interact with soil characteristics across a large number of tree species to shape soil microbial growth (Pei et al., 2016). Another benefit of PLFA analysis is that the bacterial, fungal, or total microbial carbon pools can be calculated, for comparison with other measures of productivity and carbon cycling (Schmidt, Schulz, Michalzik, Buscot, & Gutknecht, 2015).

Besides, microbial species composition it is also important to understand how forest diversity alters microbial functional processes. To do this, we used a modification of the ^{15}N pool dilution approach (Stange, Spott, Apelt, & Russow, 2007) based on traditional methods (Booth, Stark, & Rastetter, 2005; Hart, Stark, Davidson, & Firestone, 1994). The ^{15}N isotope pool dilution approach can quantify gross rates of N mineralization, nitrification, and microbial immobilization. The limitation of this method is that it necessitates the usage of fresh soil and the usually laborious process of precipitating salt extractions for isotopic analysis (Hart et al., 1994). For analysis of extractions, we used a new spin mass system to analyze $^{15}NO_3$ and $^{15}NH_4$ directly from liquid samples (Stange et al., 2007), nearly halving the processing effort.

In addition to microbial nitrogen processing rates, soil microbial decomposition potential, measured through extracellular enzyme activities, is an important functional trait of microbial communities. For example, we are using this method to establish how forest and litter diversity alter decomposition through changes in soil microbial activities (Z. Pei, unpublished data). We examine extracellular enzyme activity according to the method described by Saiya-Cork, Sinsabaugh, and Zak (2002) and recently modified by DeForest (2009) and German et al. (2011). Due to the small-scale analysis in 96-well plates and the use of multiwell plate-reader technology, rapid processing of a large number of samples is feasible. With these methods, we are able to process several hundred soil samples per campaign in order to capture both individual-species and plot-level changes in microbial growth and activity (Pei et al., 2016).

2.5 | Nutrient cycling

2.5.1 | Leaf litter decomposition

Decomposition of organic matter is a highly integrative process in ecosystem biogeochemistry, which replenishes the pool of plant available nutrients, and releases photosynthetically fixed carbon back to the atmosphere (Berg & McClaugherty, 2008). Species diversity effects on litter mass loss and nutrient release have been reported at the level of plants and detritivores (Gessner et al., 2010; Hättenschwiler, Tiunov, & Scheu, 2005).

Litterbags filled with a standard litter substrate are commonly used to study diversity effects that act via changes in the microenvironment induced by tree diversity or species composition. If leaf litter of tree species planted in the experiment serves as standard substrate, the home-field advantage should be considered as potential bias because decomposition of plant litter might be faster on plots where the same species is planted (Ayres et al., 2009; Freschet, Aerts, & Cornelissen, 2012). To increase comparability across decomposition studies, common tea bags have recently been suggested as standardized litterbags and fast assessment tool (Keuskamp, Dingemans, Lehtinen, Sarneel, & Hefting, 2013). This low-cost and time-efficient approach allows a large sample size and can thus help to assess tree diversity effects on decomposition dynamics by combining data from experiments across the globe. However, the standard material used (green tea, rooibos tea) is absent from the studied ecosystem, hence it will be difficult to infer the multitude of mechanisms by which tree diversity may influence litter decomposition. Magnitude and direction of tree diversity effects can also differ among litter substrates. Thus, to account for possible species identity effects, plant litter with contrasting litter quality should be employed together as standard litter substrates (Seidelmann et al., 2016). As with any other standard material used (e.g., wheat straw, cotton strips, and standard litter of one species), only tree diversity effects that act via changes in the microenvironment can be assessed, but not any effects that act via the quality of litter present in the ecosystem (Scherer-Lorenzen, 2008). Thus, in addition, we suggest to measure community-specific litter decomposition in the corresponding plots to account for the combined effect of microenvironment and litter quality. Finally, to isolate the effects of litter quality, single-species litterbags can be incubated in a common plot providing a homogeneous environment (Trogisch, He, Hector, & Scherer-Lorenzen, 2016).

In large tree diversity experiments, a high number of litterbags are required to include as many plots as possible. For example, we used a total of 3,618 bags which were exposed on 402 subplots in the BEF-China experiment with bags retrieved after 2, 6, and 11 months (Seidelmann et al., 2016). Thus, preparation time of litterbags including collection of site-specific plant litter should not be underestimated.

The mesh should be UV-resistant in case bags are not buried but are exposed to high solar radiation. The chosen mesh size strongly controls the access for decomposer organisms, and a trade-off between small mesh size (excluding macrofauna, but minimizing the loss of litter fragments) and large mesh size (allowing access of most organisms, but increasing the risk of losing litter fragments) exists (Bradford, Tordoff, Eggers, Jones, & Newington, 2002; Prescott, 2005). To cope with this trade-off, litter bags with a micromesh (e.g., 50 μm) at the bottom part of the bag that has contact to the soil, and larger macromesh (e.g., 5 mm) at the top of the bag can be used (Harmon, Nadelhoffer, & Blair, 1999).

2.5.2 | Deadwood decomposition

Deadwood is a key driver of ecosystem functioning in forests (Cornwell et al., 2009; Harmon et al., 1986; Wirth, 2009) and one of the most important components of forest ecosystem biodiversity, carbon and nutrient cycling, energy flows, and soil-forming processes (Harmon et al., 1986; Laiho & Prescott, 1999; Lindahl, Taylor, & Finlay, 2002). On the one hand, care must be taken when choosing the size of wood samples with respect to the scope of individual studies. Smaller pieces allow a larger sample size with a feasible amount of labor and space requirements in the field. On the other hand, larger pieces can carry a higher diversity of decomposers due to the fact that especially larger decomposer species (e.g., cerambycid beetles) prefer larger wood pieces for development. We chose standard-sized stem wood of 25 ± 1 cm length and 8 ± 2 cm diameter (Eichenberg et al., 2017). The influence of certain deadwood decomposer organisms such as termites and other invertebrates is studied using different mesh sizes in a litterbag approach (Eichenberg et al., 2017). This allows a fast assessment of abiotic controls on wood decomposition in relation to invertebrate plus fungal- and microbial-mediated versus exclusively fungal- and microbial-mediated decay. Litterbags also ensure that no samples or fragments of samples are lost in steep terrain over the course of the experiment. In our case, replicated bags with wood pieces were retrieved one and 3 years after deposition. Similar to the tea bag index for leaf litter (Keuskamp et al., 2013), a common protocol defining standard wood substrates (i.e., ice cream sticks from birch wood and chopsticks) would greatly expand the comparability of wood decomposition rates for better global predictions.

2.5.3 | Soil fertility and C storage

Soil fertility is an important covariate in the analysis of effects of tree species richness on ecosystem functioning. Large forest BEF experiments, in particular those in geomorphologically heterogeneous landscapes, have inherently a considerable spatial variation in many attributes that also influence soil nutrient availability and fertility (e.g., Scholten et al., 2017).

Quantifying abiotic site conditions including soil nutrients is therefore critical for interpreting biodiversity effects on forest stand performance. Moreover, regular inventories of sensitive soil nutrient pools (e.g., content of available and N and P) in 5-year intervals may yield important insights into how tree species richness and composition modify soils during stand development. Tracking these plant-induced temporal changes in soil properties (see ecoscape approach above) permits the identification of forest compositions promoting nutrient cycling and nutrient use efficiency (Richards, Forrester, Bauhus, & Scherer-Lorenzen, 2010) and also the quantification of soil C accumulation—an important ecosystem service (Díaz, Hector, & Wardle, 2009).

In the BEF-China experiment, initial soil conditions have been thoroughly mapped before forest establishment (Scholten et al., 2017). Systematic soil sampling included taking nine soil cores in each plot to a depth of 50 cm which were pooled per plot and soil layer (0–5, 5–10, 10–20, 20–30, and 30–50 cm). Soil fertility has been characterized by measuring total soil carbon, nitrogen, soil pH, cation-exchange capacity, exchangeable cations, and base saturation. Many of these properties can also be determined with sufficient accuracy through near-infrared spectroscopy (NIRS) and mid-infrared spectroscopy (MIRS), once calibrated for the particular soil property, to facilitate inexpensive analyses and rapid assessment of large numbers of samples in subsequent inventories (e.g., Chen, Dong, Li, & Wang, 2017; Ludwig, Khanna, Bauhus, & Hopmans, 2002). Where information is to be gathered for entire soil profiles, the soils still need to be sampled conventionally (e.g., with corers) before soil samples can be analyzed with these indirect methods. For soils of the BEF-China experiment, NIRS models were developed to replace the onerous Hedley method employing a wet-chemical process of determining fractions of soil P corresponding with different plant availability through sequential extraction of samples (Niederberger et al., 2015). The potential of NIRS to save time and costs is particularly high for soil properties that cannot be determined through a single chemical analysis but require incubation approaches or repeated extractions, for example, nitrogen and carbon mineralization rates (e.g., Ludwig et al., 2002). In the context of BEF experiments, the approach may also be very interesting to trace the species origin of soil organic matter to disentangle the influence of tree diversity on soil carbon stocks (e.g., Dobarco, van Miegroet, Gruselle, & Bauhus, 2014).

2.6 | Soil erosion control

Large tree diversity experiments require a broad range of combined techniques to assess soil erosion processes. Measurements address the kinetic energy of raindrops (splash cups), runoff and sediment discharge (runoff plots), and long-term monitoring (erosion sticks).

Splash cups consist of a plastic flask attached to a carrier system, filled with a unit sand of 125–200 μm particle size (Scholten, Geißler, Goc, Kühn, & Wiegand, 2011). The sand loss calculated from the amount of sand remaining after exposition of the cup to rainfall is converted to kinetic energy using a linear calibration function derived from laser precipitation monitor measurements (Lanzinger, Theel, & Windolph, 2006). Splash cups are light, reliable and allow a high number of replications on different positions under a tree. Results permit detecting differences in kinetic energy between different tree species and diversity levels (Geißler et al., 2013; Goebes, Bruelheide, et al., 2015; Goebes, Seitz, et al., 2015).

Surface runoff and sediment discharge are observed using microscale runoff plots (ROPs) sized 0.16 m^2 (0.4 m × 0.4 m) and bordered by stainless steel panels in which soil surface cover (e.g., by stones or biological soil crusts) is recorded photogrammetrically (Seitz et al., 2016). ROPs can be equipped with pitfall traps to implement a soil fauna treatment (Seitz et al., 2015). Runoff is collected in 20-L containers connected to covered triangular gutters. Both sediment discharge and runoff are analyzed for C, N, and P contents. The small ROP size allows investigating interrill erosion precisely as other processes like rill erosion do not occur on such short flow distances (Agassi & Bradford, 1999) and those small ROPs are particularly appropriate to compare different diversity treatments (Wainwright, Parsons, & Abrahams, 2000). A further advantage is the possibility to use a high number of randomized replications at a time (220 ROPs in BEF-China), which is an important precaution in the design of ROP measurements (cf. Hudson, 1993).

Long-term monitoring of soil erosion characteristics on over 500 plots in the BEF-China experiment requires a reliable and cost-efficient technique (Shi, Wen, Zhang, & Yan, 2011). Erosion sticks, 1-m long UV-resistant PVC rods, are pushed into the soil at nine positions in each plot. Approximately 4,500 erosion sticks have been installed in the BEF-China experiment, and the length of the sticks above the soil surface is measured once per year.

3 | DISCUSSION

Based on methods currently applied in one of the world's largest tree diversity experiments, we highlighted how methods can be combined to simultaneously address multiple ecosystem functions and consequently maximize synergy in forest biodiversity research. By implementing harmonized methods, scientific knowledge gain can be optimized while simultaneously using the specific expertise of involved research teams efficiently. Only if consistent datasets for essential ecosystem functions can be amalgamated within and across tree diversity experiments, progress in BEF research can be achieved. For example, understanding how herbivory and leaf pathogens are influenced by tree diversity can provide deeper insights into the importance of multitrophic interactions for tree biomass (Schuldt, Bruelheide, et al., 2015) (Figure 3). Similarly, decomposition dynamics along tree diversity gradients can only be explained when we know how tree diversity affects microbial activity and the diversity and composition of decomposer communities. Ultimately, the combination of above- and belowground processes can help to identify direct and indirect drivers of vital ecosystem functions such as biomass production across ecosystem subsystems (Figure 3).

In order to fully explore the potentials of tree diversity studies that aim to quantify effects on multifunctionality, an "all-measurements-on-all-plots philosophy" should be adopted, despite the large number of plots (Baeten et al., 2013). This strategy might, however, restrict the choice of methods, as often such high-throughput methods can rather be seen as "proxies" for the ecosystem function of interest, because more sophisticated or detailed measurements are too time-consuming or expensive. Based on the knowledge we have gained from the BEF-China and other forest BEF experiments, we propose the following guidelines for present and future tree diversity experiments.

3.1 | Maximizing data density

Given the high number of tree individuals planted in tree diversity experiments, often only a subset of individuals can be measured in each plot. In particular, this is true for ecosystem functions like tree growth that require annual or even more frequent measurements of individual trees. Different methods therefore have a different range in terms of their spatial and temporal resolution. Whereas some methods are easily applicable to a relatively large subset of tree individuals per plot (e.g., portable spectrometers), others are restricted to only a few individuals due to high work intensity and time constraints (e.g., minirhizotrons). Thus, methods with a high sample size should always comprise those tree individuals or plot areas that are assessed by methods with a smaller range. The goal should be to maximize data density, that is, the number of measured variables, for a given subset of tree individuals in each plot. For example, in BEF-China, most measurement activities focus on the central 4 × 4 to 12 × 12 tree individuals in each plot. This means that for a certain subset of individuals, data on productivity, litter production, tree growth, microbial biomass, the plant microbiome, herbivory, or foliar fungal pathogen infestation are available and can be correlated at the tree level.

Furthermore, the combination of several rapid nondestructive methods allows measurements even on the same branches or leaves (e.g., leaf trait assessment using NIRS combined with herbivory survey). Ideally, aboveground and belowground methods should focus on the same tree individuals to increase data density across subsystems. In monocultures and low-diversity mixtures, the number of measured tree individuals can be reduced because of the high number of replicates (see above). The quantification of multifunctional responses at individual tree level to neighborhood- or plot-level implies that variables must be measured on the same tree individuals, which requires well-coordinated and time-adjusted measurement campaigns among involved research teams. Sampling effort can be considerably reduced if collected samples are shared among project partners. For example, subsamples of soil cores taken for nutrient analysis can be used for investigating soil microbial communities (Pei et al., 2016). Similar, different aspects such as nutrient cycling and microbial community composition can be effectively studied in joint decomposition experiments when taking a shared sampling strategy into account (Pei et al., 2017; Purahong et al., 2017).

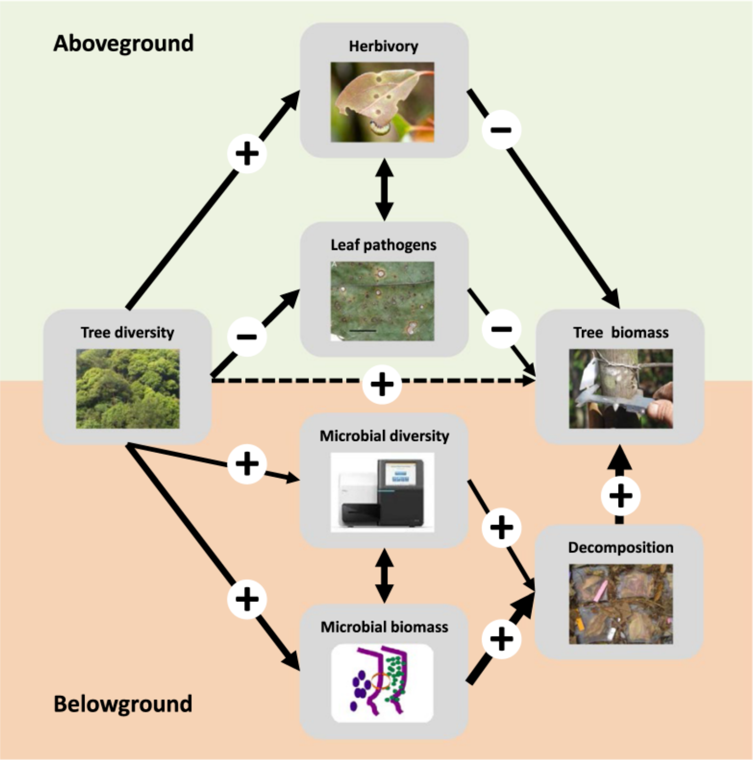

FIGURE 3 Identifying the links and underlying mechanisms between tree diversity and key ecosystem functions requires the coordinated assessment of forest multifunctionality across trophic levels and ecosystem subsystems. For example, consistent datasets of relevant ecosystem functions are needed to analyze the effect of tree diversity on tree biomass using structural equation modeling. Shown is a simplified conceptual structural equation model which links aboveground (herbivory, leaf pathogen infestation) and soil-related processes (soil microbial biomass and diversity, decomposition of leaves and roots and deadwood decomposition) affecting tree biomass. Solid and dashed arrows show hypothetical significant and nonsignificant positive or negative effects, respectively. Increasing arrow width specifies hypothetical strength of causal relationship between variables. Positive and negative relationships are indicated by "+" and "−" signs, respectively

3.2 | Applied methods should cover relevant scales

Tree diversity experiments with their large spatial extent are usually established with a long-term view on measurement activities and data acquisition. Thus, chosen methods should consider relevant spatial and temporal scales. The relationship between biodiversity and ecosystem functioning has been predominantly analyzed at the level of the community or plot, thereby neglecting the scale dependency of diversity effects (Chisholm et al., 2013; Schuldt, Wubet, et al., 2015). However, biotic interactions which determine the strength of biodiversity effects occur at the tree individual scale (Potvin & Dutilleul, 2009) and can be influenced by intraspecific (genotypic) trait variation (Johnson, Lajeunesse, & Agrawal, 2006) as well as the direct tree neighborhood (Barbosa et al., 2009). In BEF experiments, fully mapped and geo-referenced tree positions allow testing for neighborhood relationships at different scales. Thus, it is not necessary to decide beforehand which scale is appropriate, but instead it is best to apply a spectrum of methods that can capture local neighborhood interactions up to stand-level dynamics. For example, upscaling water use from individual trees to neighborhoods to plot (community) level needs data on xylem flow rates measured on individual trees and reliable estimates of sapwood area at plot level (Kunert, Schwendenmann, Potvin, & Hölscher, 2012).

It is clear that each method tends to focus either on individual trees (e.g., herbivory assessment) or on the plot (community) level (e.g., litterbags, erosion sticks), which might require a trade-off between generality and precision for the large number of trees to be measured. Thus, methods should be ideally combined in a way that they bridge precision and generality. This critical trade-off between precision and generality should be methodologically addressed in order to allow reliable upscaling of the BEF relationship to relevant scales for ecosystem management.

3.3 | Consistency in method selection in time and space

It is necessary to adapt methods to tree size and forest development stage. For some ecosystem functions, this sometimes requires an inevitable change in methods. For example, while tree canopy measurements are easily carried out in the first years after planting, this is usually not the case anymore after trees have reached a certain height. Leaf demographic assessments using marked leaf cohorts are not practical anymore after trees have reached a certain height and are replaced by collecting leaf and fine twig litter fall in litter traps. Similarly, sampling for herbivory or plant pathogen assessment needs to be adapted to increasing tree height by considering lower, mid, and upper canopy layers. However, newly introduced methods or adapted sampling designs should always be consistent, that is, calibrated and validated compared to previously used approaches. Consistency in applied methods should be promoted to ensure adequate data analysis of long-time series and to reduce ecological uncertainty (Schimel & Keller, 2015). This is especially important given that biodiversity effects may develop and become stronger over time. For example, microbial adaptation to certain tree species over time can alter above-ground–belowground interactions and could influence or reinforce biodiversity effects (Mangan et al., 2010). However, consistency of time series measurements may be compromised by fluctuation in the composition of research teams, available funding, or adjustment of research questions during the lifetime of the experimental platform. To ensure that knowledge on respective methods is not lost with time, collected datasets should be linked to respective technical protocols in the platform's database. Publishing methods in novel formats such as scientific video journals could further promote reproducibility and consistency of measurements (Kröber, Plath, Heklau, & Bruelheide, 2015). On a wider level, reducing ecological uncertainty by application of consistent and standardized methods across globally distributed experimental forest sites would improve the evaluation of general tree diversity effects (Fraser et al., 2012). In the long run, we think that a central web platform that compiles innovative methods and provides detailed protocols would largely promote data harmonization in cross-site experimental studies on forest multifunctionality.

Moreover, large BEF experiments offer an ideal test platform for introducing new emerging methods in forest diversity research. For instance, drone-based remote sensing is currently a rapidly developing technology (Tang & Shao, 2015). Drone remote sensing has been successfully tested for example in forest inventories and to estimate tree canopy height and canopy closure (Getzin, Wiegand, & Schöning, 2012; Torresan et al., 2017). As tree positions in BEF experiments are fully mapped, remote sensing data can be easily related to ground-based measurements such as of DBH or LAI. In this way, the overlap with already well-established approaches not only ensures better calibration and consistency but also promotes the establishment of new technologies.

3.4 | Promoting rapid assessment of biodiversity and ecosystem functions

The scale of sampling in large tree diversity experiments necessitates rapid, standardized, and cost-effective assessment of biodiversity. These have been successfully developed for taxa such as arthropods (Obrist & Duelli, 2010; Oliver & Beattie, 1996; Yu et al., 2012), and meta-genomic methods are used for rapid multitaxa assessment of microbial and fungal diversity (Cannon, 1997; Gao et al., 2015). The bottleneck of the "taxonomic imperative" can be addressed with DNA-based methods, particularly those based on NGS of pooled communities (Yu et al., 2012). These use quantified criteria for delineation of species diversity (Pons et al., 2006) and assignment of taxonomic names (Hebert, Ratnasingham, & deWaard, 2003), allow a greatly increased throughput (Ji et al., 2013), and are amenable to digital storage and meta-analysis in a web-based framework (Ratnasingham & Hebert, 2013). DNA barcoding can be adapted to take advantage of greater information content of multigene and PCR-free sequence data (Chesters, Zheng, Zhu, & Yu, 2015). Additionally, wiki-based descriptions allow for integration with morphological taxonomy without imposing excessive time constraints (Riedel, Sagata, Suhardjono, Tanzler, & Balke, 2013).

With respect to plant functional diversity, morphological, and biochemical leaf traits that are known to be important for driving ecosystem functions can be quickly assessed by portable NIRS in the field, once calibration is established (see above). With its high sample throughput, NIRS makes it possible to study, for example, seasonal dynamics of leaf nutrients, which can offer new insights into trait variation at much finer temporal and spatial scales. NIRS can also help to resolve species composition in fine-root mixtures (Lei & Bauhus, 2010) and to determine some soil properties such as available P, which are otherwise only quantifiable with onerous laboratory methods (Niederberger et al., 2015). In this way, high spatial and temporal resolution of trait measurements can be achieved which will improve trait-based predictions of ecosystem functioning. The identification of easily measurable plant trait syndromes which reflect ecophysiological key functions could further strengthen this approach.

Besides rapid assessment of biodiversity there is a clear need to develop easy-to-use and quick methods for the measurement of key ecosystem functions. A standardized rapid ecosystem function assessment (REFA) has been recently suggested and conceptualized by Meyer et al. (2015). Low-tech, easy-to-use, repeatable, and cost-efficient measurements allow the harmonized assessment of ecosystem functions (e.g., biogeochemical cycles, tree productivity, or consumer–plant interactions) across a large number of plots and experimental sites. This approach is especially beneficial in a multifunctional context as the number of ecosystem functions considered in an experiment can be increased. Furthermore, in contrast to more traditional approaches, functions can be studied at the same spatial resolution, preferably on all plots or levels of tree diversity, due to reduced measurement effort. In this way, inherent interrelationships in multitrophic networks (Staab et al., 2015) or across below- and aboveground subsystems could be more adequately considered in BEF research. However, the measurement of ecosystem functions in structurally complex forest systems imposes special requirements in terms of spatial and temporal scale. This means that REFA methods and sampling designs need to be specifically adapted or developed for assessing forest multifunctionality. In this respect, our compilation of methods could serve as a first contribution for the development of a REFA framework for forests.

4 | OUTLOOK

The majority of previous studies in forest BEF research have focused on single ecosystem functions, thereby neglecting inherent feedback mechanisms, essential connections between above- and belowground subsystems, and important trophic relationships. However, knowledge of these interdependencies among multiple functions is crucial to understand and predict the responses of forest ecosystems to species loss. Considerable progress in forest BEF experiments can be promoted by applying harmonized methodical approaches to comprehensively assess forest multifunctionality. Method selection should therefore be guided by major principles such as consistent application of methods across spatial and temporal scales, maximizing data density and rapid assessment strategies to increase the number of replicates. Another important issue is to ensure data comparability across tree diversity experiments for the growing number of synthesis initiatives. Ideally, this requires space- and time-aligned measurement campaigns and common agreement on standardized protocols. Current methods need to be adapted to account for the specific requirements of structurally complex and long-lived forest ecosystems. New innovative approaches such as the identification of easy-to-measure indicators for ecosystem functioning or other rapid assessment strategies have to be developed. With these challenges ahead, we hope that our outline of key methods currently applied in one of the largest tree diversity experiments will help to promote synergy and comprehensive assessment of multifunctionality in forest biodiversity research.

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AUTHOR CONTRIBUTIONS

HB, BS, and KPM conceived the initial ideas; ST, AS, and HB designed the paper and led the writing of the manuscript. All authors (ST, AS, JB, JAB, SB, FB, NC, DC, WD, DE, AE, MF, CG, MG, PG, JG, CH, SH, WH, JSH, AH, LH, YH, AMK, PK, MK, KL, YL, YJL, PN, ZP, KAP, RP, TP, MSL, KS, TS, SS, ZS, MS, GvO, ChW, EW, CW, TW, BY, XY, CDZ, BS, KPM, and HB) wrote individual chapters, contributed critically to the drafts, and gave final approval for publication.

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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

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Tree species 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 run-off plots was conducted to investigate the influence of tree species and tree species richness as well as functional traits on interrill erosion in a young forest ecosystem. An interrill erosion rate of 47.5 $\mathrm{Mg\,ha^{-1}\,a^{-1}}$ was calculated. This study provided evidence that different tree species affect interrill erosion differently, while tree species richness did not affect interrill erosion in young forest stands. Thus, different tree morphologies have to be considered, when assessing soil erosion under forest. High crown cover and leaf area index reduced interrill erosion in initial forest ecosystems, whereas rising tree height increased it. Even if a leaf litter cover was not present, 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 interrill erosion. Long-term monitoring of soil erosion under closing tree canopies is necessary, and a wide range of functional tree traits should be considered in future research.

1 Introduction

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

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 biodiversity 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 (the diversity of tree species) as well as functional diversity (the diversity of functional traits as morphophysiophenological attributes of a given species; cf. Violle et al., 2007) of tree communities can play a critical role in improving ecosystem services such as water filtration or climate regulation (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 in soil protection (Romero-Diaz et al., 2010; Jiao et al., 2012). This also applies to the south-eastern part of China, which is known to be a hotspot of biodiversity and especially tree species richness (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. Considering that studies on soil erosion under forest have mostly focused on deforestation (Blanco-Canqui and Lal, 2008) and that counteracting measures such as afforestation often result in monoculture stands (Puettmann et al., 2009), it appears that the role of tree species richness for soil erosion has been largely disregarded. 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 a 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 most 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 cover with a high number of species includes a high number of plant functional groups which complement one another. Thus, they are more effective in controlling erosion processes than vegetative cover with few species (Pohl et al., 2012). For example, high tree species richness can result in an increased stratification of canopy layers (Lang et al., 2010) and a higher total canopy cover (Lang et al., 2012). In addition, a highly diverse structure within the leaf litter layer on the forest floor seems to improve its protective effect (Martin et al., 2010). Further research on the influence of tree species richness on erosion control appears to be necessary, but the complex system of interacting functional groups within the vegetation cover is also of great interest.

Vegetation cover is generally considered a key factor for the occurrence and extent 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 at the soil surface (Kim et al., 2014). Moreover, forests can provide a multistorey 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 by a factor of up to 2 to 3 compared to open fields (Nanko et al., 2008, 2015). This leads to considerable soil loss if the forest floor is unprotected, which may be the case if protective layers diminish, e.g. under shady conditions (Onda et al., 2010) or fast decomposition (Razafindrabe et al., 2010). While the effects of soil surface cover on soil erosion are 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, full-grown forests (e.g. Zhou et al., 2002; Nanko et al., 2008; Geißler et al., 2012a), whereas forests at an early successional stage are rarely mentioned. In these young forests, tree heights are lower than at later stages, but structural and spatial complexity is high and species-specific growth rates differ considerably (Swanson et al., 2011). It is assumed that these species-specific differences in structure and growth will influence soil erosion rates.

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

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

2 Methodology

2.1 Study site and experimental design

The study was conducted in Xingangshan, Jiangxi Province, PR China (29°06.450′ N, 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 of 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 interspersed by siliceous-rich joints. Prevailing soil types are Cambisols with Anthrosols in downslope positions and Gleysols in valleys (cf. IUSS, 2006) covering saprolites. Soil bulk density is low (0.98 g cm^{-3}) and soil reaction acidic (mean pH in KCl 3.68). Soil texture ranges from silt loam to silty clay loam. The climate in Xingangshan is humid and subtropical and ranked as Cwa after the Köppen–Geiger classification. It is characterized by an annual average temperature of 17.4 °C and a mean annual rainfall of 1635 mm (Goebes et al., 2015b).

The experimental area has been used as a commercial forest plantation (*Cunninghamia lanceolata* 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 4 to 5 years at the time of measurements. Trees were planted randomly in different species richness levels with a planting distance of 1.29 m, following a broken-stick design. This study focused on the very intensively studied plots (VIPs; cf. Bruelheide et al., 2014) of which 34 were used (Table 1). 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 26 tree species, 6 of which only appeared in mixtures (Table 2). Monocultures with tree heights lower than 1 m or crown covers of less than 10 % were excluded before the analysis.

2.2 Erosion measurements

To determine sediment delivery (as initial interrill erosion) and surface run-off volume, micro-scale run-off plots (ROPs, 0.4 m × 0.4 m) were used (cf. Seitz et al., 2015; without fauna treatment). Each ROP was connected to a 20 L reservoir and a rainfall gauge was placed next to it (Fig. 1). All 34 VIPs were equipped with five ROPs each, resulting in a total number of 170 ROPs. Within each VIP, areas of 220 m^2 were sectioned for ROP measurements to avoid interferences with other BEF China experiments. The selected areas were representative of the range of surface properties in the plot, and the ROPs were placed randomly therein. All leaf litter was removed from the ROPs prior to measurements. The ROPs were operated in May and June 2013 during the rainy season. Run-off volume and rainfall amount were determined in situ and sediment was assessed after sampling by drying at 40 °C and weighing. The capacity of the reservoirs was not exceeded in any rainfall event.

At each ROP, tree crown cover, leaf area index (LAI), soil surface cover, slope and rainfall amount were measured. Crown cover and LAI were determined 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) adjusted to the canopy area vertically above the ROP. Soil surface cover was measured photogrammetrically (grid quadrat method with GIMP 2.8) and separated into organic and inorganic cover by colour distinction. Slope was measured with an inclinometer. The rainfall amount at each ROP was determined by rainfall gauges (see above). At each VIP, total tree height, stem diameter at 5 cm above ground (hereafter, stem diameter) and crown width were measured and calculated as the mean of 36 tree individuals per VIP (Li et al., 2014). Additionally, soil organic matter (SOM) was identified for each VIP (5 cm depth, nine replicates) by measuring total organic carbon with a Vario EL III elemental analyser (Elementar, Hanau, Germany) and multiplying it by the conversion factor 2 (Pribyl, 2010). Tree species richness was known from the VIP set-up.

Figure 1. Measurement set-up showing a run-off plot (ROP, 0.4 m × 0.4 m) with reservoir and rainfall gauge at the experimental site in Xingangshan, Jiangxi Province, PR China.

2.3 Rainfall patterns

Weather conditions were recorded by an on-site climate station (ecoTech data logger with Vaisala weather transmitter and ecoTech tipping bucket balance) at 5 min intervals. In 2013, the total precipitation in the study area was 1205 mm and lower than the mean of the preceding 3 years (1635 mm). In May and June, 10 rainfall events were captured with ROP

Table 1. Mean characteristics of the 34 selected very intensively studied plots (VIPs) in 2013 in the BEF China experiment, Xingangshan, Jiangxi Province, PR China.

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

measurements in the study area. Events were determined by breaks in rainfall of at least 6 h. Four of these events (E1–E4) were strong enough to trigger soil erosion (out of 33 events over the entire year of 2013) following Wischmeier and Smith (1978), who used an event threshold of 12.7 mm. 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 3), reflecting a growing monsoon influence from the beginning to the middle of summer.

2.4 Statistical analysis

Linear mixed effects models with restricted maximum likelihood were used with R 3.0.2 (R Core Team, 2013) and “lmerTest” (Kuznetsova et al., 2014) to investigate the influences on sediment delivery. 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 (E1–E4) nested in plot, tree composition (species pool), site (A or B) and ROP nested in plot were used. Nesting was introduced to avoid pseudoreplication considering the degrees of freedom in our hypotheses tests. Tree and crown characteristics were fitted one after the other because they were highly correlated. Contrasts

Table 2. Twenty-six selected tree species used in the experiment according to the Flora of China web page (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.	*Machilus grijsii* Hance*
Castanopsis fargesii Franch.	*Machilus leptophylla* Hand.-Mazz.*
Castanopsis sclerophylla (Lindl.) Schott.	*Magnolia yuyuanensis* Hu
Celtis biondii Nakai*	*Nyssa sinensis* Oliver*
Choerospondias axillaris (Roxb.) Burtt et Hill.	*Rhus chinensis* Mill.
Cyclobalanopsis glauca (Thunb.) Oerst.	*Sapindus saponaria* Gaertn.
Elaeocarpus chinensis Gardn. et Chanp.	*Schima superba* Gardn. et Champ.
Elaeocarpus glabripetalus Merr.	*Triadica sebifera* (L.) Roxb.
Elaeocarpus japonicus Sieb. et Zucc.	*Quercus fabri* Hance
Idesia polycarpa Maxim.*	*Quercus phillyreoides* A. Gray*

Table 3. Characteristics of rainfall events considered erosive (threshold 12.7 mm) in Xingangshan, Jiangxi Province, PR China in May and June 2013.

Event	Mean intensity $(\mathrm{mm\,h^{-1}})$	Peak intensity $(\mathrm{mm\,h^{-1}})$	Total rainfall amount (mm)
E1	1.38	11.4	20.29
E2	2.34	23.04	25.74
E3	3.19	45.24	54.42
E4	14.60	83.04	34.01

between diversity levels (div0 to div1–div24, div1 to div8–div24) were introduced to quantify the effects of bare plots vs. tree plots and tree monocultures vs. mixtures, respectively. The effect of individual tree species (div1) was tested separately against the mean sediment delivery using crown cover, slope, surface cover, SOM and amount of precipitation as fixed factors and site and ROP nested in plot as random factors ($n = 200$). The maximum-likelihood approach was used to obtain model simplification by stepwise 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 analysis of variance (ANOVA) type 3 with a Satterthwaite approximation for degrees of freedom, and p values were obtained by likelihood ratio tests.

3 Results

The results were based on 334 ROP measurements out of a total of 378 measurements. Invalid measurements were caused by technical constraints such as plugged tubes or toppled rainfall gauges. Sediment delivery over all VIPs and rainfall events ranged from 14 to 920 $\mathrm{g\,m^{-2}}$ per ROP. Event-based mean sediment delivery increased with peak intensity from precipitation event 1 to event 4 with 42 $\mathrm{g\,m^{-2}}$ (E1), 85 $\mathrm{g\,m^{-2}}$ (E2), 120 $\mathrm{g\,m^{-2}}$ (E3) and 283 $\mathrm{g\,m^{-2}}$ (E4). The interrill soil erosion rate determined by micro-scale ROPs and extrapolated for all erosive precipitation events (> 12.7 mm rainfall amount) in 2013 was estimated to be 47.5 $\mathrm{Mg\,ha^{-1}}$.

3.1 Species richness effects on interrill erosion processes

Tree species richness did not affect sediment delivery or run-off volume (Table 4 and Fig. 2). Sediment delivery and run-off volume did not differ between bare plots (div0) and plots with trees (div1–div24) nor between monocultures (div1) and species mixtures (div8, div16, div24). The standard deviations of sediment delivery $(\mathrm{g\,m^{-2}})$ and run-off volume $(\mathrm{L\,m^{-2}})$ in relation to diversity levels were high (Fig. 2 and Table 5). 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 this experiment tree height in mixed stands was not lower than 1.07 m and crown cover achieved at least 29 %.

3.2 Species effects on interrill erosion processes

Individual tree species in monocultures showed significant differences in sediment delivery (Fig. 3) ranging from 90 $\mathrm{g\,m^{-2}}$ (*L. formosana*) to 560 $\mathrm{g\,m^{-2}}$ (*Ch. axillaris*) per rainfall event.

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

Table 4. Results of the basic linear mixed effect model for sediment delivery (^{a}: $p < 0.001$; ^{b}: $p < 0.01$; ^{c}: $p < 0.05$; ^{d}: $p < 0.1$; 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": denominated degrees of freed; "F": F value; "Pr": probability).

		denDF	F	Pr	Estimates
Fixed effects	Run-off volume	204	49.0	$< 0.001^{a}$	0.33
	Crown cover	120	7.25	0.008^{b}	(–) 0.18
	Slope	141	1.33	0.250 n.s.	0.05
	Surface cover	140	56.1	$< 0.001^{a}$	(–) 0.46
	Soil organic matter	42	5.61	0.022^{c}	(–) 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 for crown cover				
	Leaf area index	95	5.16	0.026^{c}	(–) 0.17
	Tree height	31	3.58	0.069^{d}	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 5. Mean sediment delivery in $\mathrm{g\,m^{-2}}$ and surface run-off volume in $\mathrm{L\,m^{-2}}$ (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 delivery	199 (106)	361 (187)	188 (90)	202 (105)	103 (57)	135 (123)	204 (107)
Run-off 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)

tree species differed considerably between the species (Table 6).

3.3 Effects of species-specific 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 delivery. Tree height marginally positively affected sediment delivery ($p < 0.1$), whereas stem diameter and crown width had no influence (Fig. 4, Table 4). The soil surface cover consisted of stones and biological soil crusts and covered on average one fifth of the ROP surfaces in May and June 2013. It affected sediment delivery negatively ($p < 0.001$). Sediment delivery decreased with increasing SOM content ($p < 0.05$). 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 delivery (Fig. 4, Table 4).

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 7).

4 Discussion

The soil loss rate determined by micro-scale ROPs ($47.5\,\mathrm{Mg\,ha^{-1}\,a^{-1}}$) for 2013 was considerably higher than the average rate Guo et al. (2015) recently calculated for southern China (approx. $20\,\mathrm{Mg\,ha^{-1}\,a^{-1}}$) in a study based on small-scale and field ROPs. Pimentel (1993) reported an average rate of $36\,\mathrm{Mg\,ha^{-1}\,a^{-1}}$ for the same area. Zheng et al. (2007) stated an average soil loss rate of $31\,\mathrm{Mg\,ha^{-1}\,a^{-1}}$ determined with $^{137}\mathrm{Cs}\,/\,^{210}\mathrm{Pb}$ tracing techniques in Sichuan Province, PR China. These different rates are due to different

Table 6. Sediment delivery and growth characteristics (means) of tree species with significant differences in delivery at the experimental site in Xingangshan, Jiangxi Province, PR China.

	Sediment delivery ($g\,m^{-2}$)	Crown cover (%)	Leaf area index	Tree height (m)	Stem diameter (m)	Crown width (m)
Mean	199	32	0.75	1.84	0.03	0.94
Monocultures	202	22	0.63	1.63	0.02	0.78
Tree mixtures	135	44	1.18	2.30	0.04	1.26
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

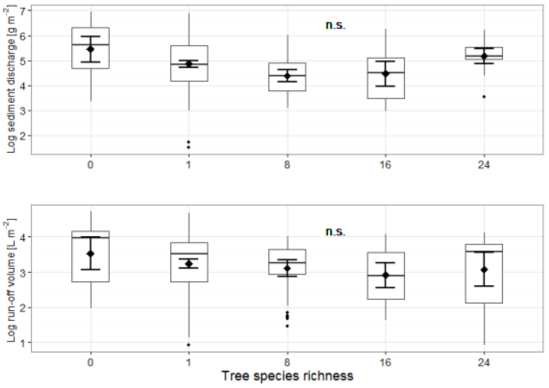

Figure 2. Sediment delivery and run-off volume at five diversity levels based on four rainfall events in May and June 2013 in Xingangshan, Jiangxi Province, PR China (n.s.: not significant; $n = 334$). Horizontal line within box plot represents median and diamond represents mean.

Table 7. Growth characteristics of the 20 tree species in monocultures 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	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
Soil organic matter (%)	6.4	1.4	9.4	4.3
Slope (°)	27	5	35	19

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

land use types and measurement techniques but also to the scale-dependent nature of soil erosion and run-off generation (cf. Boix-Fayos et al., 2006; Cantón et al., 2011). The micro-scale ROPs used in this study quantified interrill wash and sediment detachment by raindrop impact (Agassi and Bradford, 1999; cf. Cerdà, 1999; Parsons et al., 2003; García-Orenes et al., 2012). However, a significant amount of erosion occurs in the rilling system, and the influence of interrill processes on soil erosion varies greatly (Govers and Poesen, 1988). Nevertheless, Mutchler et al. (1994) stated that micro-scale ROPs are suitable to study basic aspects of soil erosion, and, furthermore, these measurements are particularly appropriate when defining impacts of vegetation through an inter-plot comparison (Wainwright et al., 2000).

4.1 Species richness effects on interrill erosion processes

Tree species richness did not affect sediment delivery or run-off volume, and thus the first hypothesis has to be rejected. Nevertheless, a trend of decreasing sediment delivery and run-off volume from diversity level 0 to 8 was visible.

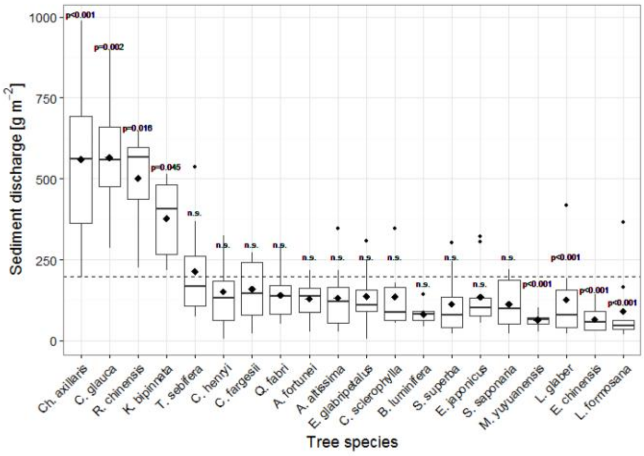

Figure 3. Sediment delivery under 20 tree species in monocultures based on four rainfall events in May and June 2013 in Xingangshan, Jiangxi Province, PR China. Dashed line indicates mean sediment delivery of all 20 species. Horizontal lines within box plot represent medians, and diamonds represent mean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

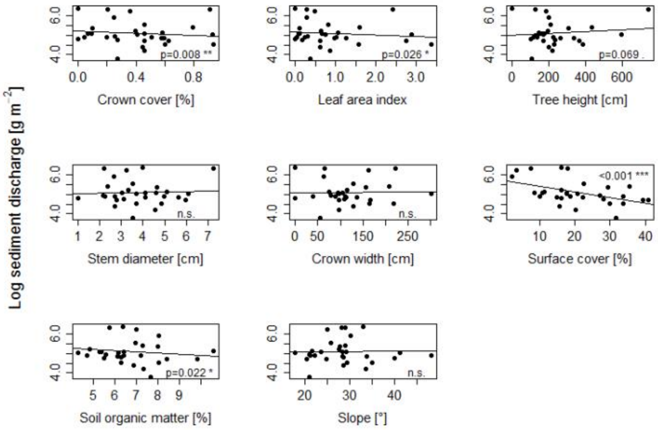

Figure 4. Effects of species-specific functional traits and site characteristics on sediment delivery. Analyses were based on four rainfall events in May and June 2013 in Xingangshan, Jiangxi Province, PR China. Black lines represent linear trends.

However, both parameters were nearly the same at diversity level 1 and 24 and standard deviations were high. In contrast to tree growth patterns in monocultures, which were highly variable, mixed stands indicated a more balanced development (cf. Kelty, 2006). All species mixtures in this experiment ensured a high level of tree height and ground coverage after 4 to 5 years of tree growth, whereas in monocultures the canopy cover was lower and highly tree-species-specific. Thus, several monoculture plots were excluded before measurements because some species could not provide enough ground coverage. At the same time, sediment delivery in 8- and 16-species mixtures 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 interrill erosion is likely attributable to the early successional stage of the forest experiment with low tree ages. Full canopy cover with high stratification and overlap has not yet been developed at the study site and the trees were far from reaching terminal height (Goebes et al., 2015b; Li et al., 2014). It is assumed 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 to one another 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, full-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). As the experiment progresses and tree height increases, increasing throughfall kinetic energy is expected, which in turn increases the general soil erosion potential if an understorey is missing.

4.2 Species effects on interrill erosion processes

Trees in monocultures differed in their impact on interrill 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 this study, four tree species (*Ch. axillaris, C. glauca, R. chinensis, K. bipinnata*) seemed to promote interrill erosion rates, whereas another four species (*M. yuyuanensis, L. glaber, E. chinensis, L. formosana*) showed a mitigating effect on interrill erosion at this initial stage of the forest ecosystem. Thus, a species-specific effect on sediment delivery for this subtropical experimental area can be confirmed. Species-specific effects can result from different throughfall kinetic energy, which was recently shown by Goebes et al. (2015a) at the same study site in China. The effect of throughfall kinetic energy was ascribed to different tree architectural characteristics and leaf traits. The authors found 3 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. At the 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 delivery in this study. Raindrops falling from leaves of this species nearly reached terminal velocity and hence throughfall kinetic energy was high (Morgan, 2005; Goebes et al., 2015a). This finding explained the high erosion rates below this fast-growing species. Further stands with significantly higher erosion rates and the four tree species with a mitigating effect on interrill erosion showed lower tree heights and thus lower throughfall kinetic energy. Their effect on sediment delivery has to be explained by further functional traits.

4.3 Effects of species-specific functional traits and site characteristics

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

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

It has been shown that tree height is an import factor for sediment detachment under forest (Geißler et al., 2013), mostly due to increasing drop falling heights (Gunn and Kinzer, 1949). As trees had not yet reached adult height (mean height $< 2\,\mathrm{m}$) in this study, the kinetic energy of raindrops formed at leaf tips was lower than in full-grown tree stands and drops did not reach terminal velocities (Morgan, 2005; Geißler et al., 2013; Goebes et al., 2015a). Therefore, tree height had a weak effect on sediment delivery ($p < 0.1$) in this study and delivery under trees did not exceed sediment delivery on bare ground. Nevertheless, high sediment delivery under *Ch. axillaris*, by far the fastest-growing tree in this 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 delivery 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 of these features to influence soil loss and highlighted the complexity of factors mitigating soil erosion in forest ecosystems.

Results showed that soil surface cover and SOM 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 delivery. This finding underlines the general importance of covered soil surfaces for erosion control (cf. Thornes, 1990; Morgan, 2005) and shows that the protective 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 forests reduces soil erosion considerably compared to bare land (Blanco-Canqui and Lal, 2008; Seitz et al., 2015). Furthermore, SOM reduced interrill erosion, which could be explained by its ability to bind primary particles into aggregates (Blanco-Canqui and Lal, 2008). If we assume that SOM increases with increasing species richness, as was recently demonstrated in a grassland study by Cong et al. (2014), an indirect effect of biodiversity on soil erosion could be supposed. Finally, slope angle did not affect interrill erosion due to the short plot length that limits run-off velocities (cf. Seitz et al., 2015).

5 Synthesis and conclusions

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

Tree species richness did not affect sediment delivery and run-off volume, although mixed stands showed a more balanced and homogenous vegetation development than monocultures. This finding was ascribed 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.

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 as a measure against soil erosion becomes important already at an early successional stage.

Species-specific functional traits and site characteristics affected interrill 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 were effectively counteracting soil loss in initial forest ecosystem. In further studies, a wider range of functional tree traits such as leaf habitus or belowground stratification should be taken into consideration. Moreover, investigations into 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.

Author contributions. 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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