FACULTY OF AGRICULTURAL SCIENCE
Institute of Agricultural Sciences in the Tropics (Hans-Ruthenberg-Institute)
University of Hohenheim
Field: Plant production in the Tropics and Subtropics
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Modelling weed management effects on soil erosion in rubber plantations in Southwest China

Dissertation
Submitted in fulfilment of the requirements for the degree
"Doktor der Agrarwissenschaften"
(Dr.sc.agr. in Agricultural Sciences)

To the
Faculty of Agricultural Sciences

presented by

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Born in Sichuan, China
2018

This thesis was accepted as doctoral thesis (Dissertation) in fulfilment of the regulations to acquire the doctoral degree “Doktor der Agrarwissenschaften” (Dr.sc.agr./PhD in Agricultural Sciences) by the Faculty of Agricultural Sciences at University of Hohenheim on 20.11.2018

Date of oral examination: 20.11.2018

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Acknowledgements

Doing a Ph.D. is not an easy job. Thanks to all wonderful people I have met, who have contributed to this work and softened this hard process for me. Firstly, I would like to thank my supervisor Prof. Dr. Georg Cadisch, who always supported and inspired me, gave me the freedom to explore as well as stood by for backing. He guided me into the scientific field, made me understand "what the research is" and enriched my life as a junior researcher with his scientific passions and rich experience. I would like to express my gratitude to Dr. Sergey Blagodatsky, who helped me with administrative and field organization, and always supported me in my research work. A big thank you also goes to Dr. Carsten Marohn, who helped a lot during the modeling phase. I am very appreciated for the discussion with him, which inspired and nourished my research work.

My sincere gratitude conveys to WPA (Water-People-Agriculture) Research Training Group funded by the Anton & Petra Ehrmann-Stifung, which not only financially supported my three-year research work but more supplied precious opportunities enabling me to extend my scientific activities and communications.

As this thesis was part of the "SURUMER" project, I would like to thank Prof. Dr. Jianchu Xu and Mr. Feng Liu, who gave me lots of supports during my field work. Without them, I could never ever successfully finish my data collection. To Dr. Gerhard Langenberger, Ms. Sabine Baumgartner and Mrs. Lijuan Duan, the project coordinators in Germany and China, who made the field life and communications to local farmers much easier. To other subprojects supports, especially the data and extra information support from Dr. Arisoa Rajaona (SP2), Mr. Manuel Krauss, Ms. Lydia Seitz. Special thanks also to all the farmers and villagers in my study site Nanhuicang Watershed and Mansong Watershed, China to support me with the field work, sampling and questionnaires.

Big hug to my friends and colleagues from the Institute of Agricultural Sciences in the Tropics. Dr. Marcus Giese, Christian Brandt, Xueqing Yang, Gaby Kircher, Prof. Dr. Forkald Asch, Dr. Melvin Lippe, Dr. Thomas Hilger, Dr. Khalid Hussain, Dr. Frank Rasche, Thi Thanh Ngyuen, Krittiya Tongkoom, Ruj Kasetsuwan, Hannes Karwat, Konrad Egenolf. Thanks for their supports in various ways.

Last but not the least, I would like to thank my family, who selfless supported me since my master by standing the fact that their only daughter was more than 7000 km away. Mengjia Liu and Yuqiong Gou, thanks for always being there for me. Your encouragement and confidence really meant a lot to me.

Hongxi Liu

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List of abbreviations

AMD	Assimilation Rates
ANSWER	Areal Nonpoint Source Watershed Response
CREAMS	Chemicals, Runoff, and Erosion from Agricultural Management Systems
DBH	Diameter at Breast Height
DEM	Digital Elevation Model
ERODEP	EROsion and sediment DEPosition
LAI	Leaf Area Index
LTR	Light Transmission Ratio
LUCIA	Land Use Change Impact Assessment
NRWNNR	Naban River Watershed National Nature Reserve
SURUMER	SUstainable RUbber cultivation in the MEkong Region
SWAT	Soil and Water Assessment Tool
SWC	Soil Water Content
TOC	Total Organic Carbon
USLE	Universal Soil Loss Equation
WEPP	Water Erosion Prediction Project
WaNuLCAS	Water, Nutrient and Light Capture in Agroforestry System
WOFOST	WOrld FOod STudies

Chapter 1 General introduction

The work presented in this thesis was carried out within the framework of the project "Sustainable Rubber Cultivation in the Mekong Region: Development of an integrative land-use concept in Yunnan Province, China (SURUMER)" funded by the German Ministry of Education and Research (BMBF), grant NO. FKZ 01LL0919. The objective of SURUMER project was to develop an integrative, applicable, and stakeholder-validated concept for sustainable rubber cultivation in Yunnan. The program combined 9 subprojects ranging from soil, plant and animal sciences to economics and sociology. The work of the presented cumulative PhD thesis was conducted within the subproject SP1, which aimed to assess the impact of intensified rubber cultivation on spatial and temporal carbon dynamics in different land use systems.

1.1 Water erosion and its effects on soil health and stream water quality

Soils provide many life supporting ecosystem services to mankind including food and energy supply. Redistribution of soils is mostly carried by erosion from its formation place to deposition place at a depressional site (Lal, 2003). This process is composed of three basic parts: i) detachment, ii) transport, and iii) deposition. Based on the source of energy during the process, erosion can be further divided into different types such as water, wind and tillage erosion (Van Oost et al., 2006). In this study, we only focus on water erosion. Detachment means breakdown of the soil aggregates and separation of individual soil particles from the soil matrix. This separation during water erosion can be driven by kinetic energy of rainfall drops, namely rainfall splash, or by stream power of runoff flow, namely runoff detachment. The detached particles or micro-aggregates are then transported by overland flow or interflow to downslope and further deposited at a certain distance from its original detached position, which may range from a few millimeters to thousands of kilometers (Lal, 2001). Deposition happens when the carrying capacity of the overland flow is reduced by decrease in velocity, increase in surface roughness, and/or presence of vegetation cover or any other obstruction. Loss of turbulence by slope gradient decrease may also lead to stream power reduction (Hairsine and Rose, 1992).

This slow geologic erosion-deposition is originally a constructive process creating fertile soils of alluvial flood plains and loess plateaus around the world and has supported ancient civilizations and thriving cultures for millennia (Lal, 2003). However, the accelerated soil erosion is a destructive process with the adverse impacts on multiple aspects such as soil degradation (UNEP, 1992), food security (Scherr, 1999) and water quality (Lal, 1998). The on-site effects of erosion are particularly important on agricultural land (Morgan, 2005) as the loss of soil from a field, the breakdown of soil structure causes long-term decline in soil quality (soil fertility, soil depth, soil organic matter) resulting in a reduction of soil productivity, namely soil degradation. Water erosion induced soil degradation is estimated as 1094 million ha (Mha) worldwide, of which 751 Mha is severely affected (Lal, 2003). Soil degradation may further lead to desertification and land abandonment threatening sustainable development and the food security. The primary off-site effects are well realized and are related to the sedimentation downstream and water quality deterioration. Sedimentation at depressional sites may reduce storage capacity of reservoirs and river capacity therefore enhances flooding risks (Allen, 1965). Sediment transferred into water bodies, namely the suspended solids (SS), can threaten the aquatic ecology by increasing stream turbidity and bringing pollutants such as nutrients, agrochemicals, heavy metals attached to sediment particles (Novotny, 1994). Therefore, erosion-deposition is a process affecting both terrestrial and aquatic systems, food and water security.

1.2 Soil erosion assessment

Erosion assessment can be categorized by erosion measurement and hazard assessment. The former is to collect actual soil erosion data under natural or simulated conditions in the field or in the laboratory at plot or watershed scale. The latter is to spatially identify areas with different erosion risk, which reflect conditions of soil, climate, topography and land cover (Morgan, 2005).

1.2.1 Erosion measurement

Plot level

Plot level measurement is designed to assess soil loss from a relatively small area. Erosion plots are the most popular method to assess interrill erosion, which starts soil detachment and widely exists in various land use type (Morgan, 2005). It can be classified into bounded and unbounded plots. Bounded plot is a piece of land isolated by impermeable material (e.g. metal sheets, cements) with known size, slope steepness and length. Normally the standard bounded plot is 22 m long and 1.8 m wide, although other sizes or shapes (e.g. circular) are commonly used in the field based on the research objective. At the downslope end is connected to the sample collection systems composed of troughs or gutter and tanks, in most cases additionally with a divisor, which aims to split the flow into equal parts and only conduct one part into the second tank to avoid overflow of the first tank. By collecting the sediments from troughs/gutters and tanks, total soil loss from this specific area can be calculated. Bounded plots are assumed to be the most reliable system to determine soil loss per unite area. Establishment of such isolated plots inevitably cause strong soil interruption. Generally, the data for the first three months to one year should be rejected to avoid effects of plot constructions. Therefore, bounded plots are mostly employed as permanent research stations for long-term erosion studies.

Unbounded plots, also referred as Gerlach troughs, are composed of the collection system of bounded plots, namely the area is not isolated by impermeable material. Advantages of Gerlach troughs are the little edge effects and little plot construction recovering time compared to bounded plots while the disadvantages is the difficulty to estimate soil loss contributing area when an areal assessment is required. In this case, the slope of study site should be straight in plan to meet the assumption that loss of runoff or sediment from the assumed contributing area is offset by inputs from adjacent areas. Unbounded plots supply an efficient tool to study erosion-deposition at the slope scale but should be cautiously applied depending on the topography.

Watershed level

Plot level erosion measurement focuses on on-site soil loss affecting soil health; while watershed level measurement calculates total soil/sediment transported from terrestrial to aquatic system and its impact on water quality. Sediment yield of a watershed can be measured by monitoring the quantity of sediment leaving the watershed along a river. The traditional recording station should continuously measure discharge and sediment concentrations at the exit point of the river to establish a sediment rating curve, which represents the relation between sediment concentration and discharge, similar as following:

$$C = aQ^b$$

where C is the sediment concentration, Q is the water discharge.

Discharge can be obtained by weirs and depth recorders. The accuracy of this method depends on the sampling frequency (Morgan, 2005). According to different sampling conditions, sediment concentration can be calculated through the linear model (Brasington & Richards, 2000; Navratil et al., 2011) and mix linear model (Slaets et al., 2014).

1.2.2 Erosion hazard assessment

Modelling tools can be employed to assess erosion potential at large scale and offer spatial information such as rainfall erosivity, erosion risk distributions, and implicate possible soil conservation through better management (Koomen & Stillwell, 2007).

Aksoy & Kavvas (2005) reviewed hillslope and watershed scale erosion and sediment transport models. Based on it, a model can be developed using a large amount of data or by using mass conservation equation models, which can be classified into empirical and physically based models.

Most commonly used empirical model is the Universal Soil Loss Equation (USLE, Wischmeier & Smith, 1987), which computes the average annual soil loss based on five factors, namely rainfall erosivity, soil erodibility, slope steepness and length, cropping management and supporting conservation practice. Other empirical models are developed based on USLE by improving certain factor calculations. For instance, Revised Universal Soil Loss Equation (RUSLE) mainly improves the cropping management factor by dividing it into several sub-factors thus includes more physical explanation into this factor. The SEdiment Delivery Distributed model (SEDD, Ferro & Porto, 2000) is as well based on USLE and incorporates sediment yield computation by a Monte Carlo technique. Soil erosion assessment by USLE in most cases is combined with mapping from aerial photographs. Five factors (rainfall, soil erodibility, topography, conservation and management) are derived from aerial photographs; and the potential soil loss is calculated by multiplication. Combination of geographical information systems (GIS) with USLE can estimate regional variation in erosion potential, especially under data-limited conditions with basic input requirements (soil map, land use map, digital elevation map). Therefore, this method is preferably employed at large scale (national or global scale) (Dabral et al., 2008; Fistikoglu & Harmancioglu, 2002; Pradhan et al., 2012; Hassan et al., 2017; Zhu, M.Y., 2015).

Physically based erosion and sediment transport models are mostly extended from hydrological models (Aksoy & Kavvas, 2005) and use outputs of hydrological models as input for erosion simulation. Similarly, an erosion and sediment transport model can, as well, be easily extended to a nutrient or pollutant transport model as nutrients or pollutants are mostly transported through runoff by dissolving or attaching to sediment particles. By applying GIS, physically based models can provide spatially distributed results. On the other side, input data requirements of physically based models dramatically increase with the model complexity, namely more process and spatial details need more input data. This has brought about the data collection challenge, especially when the model is applied at large scale. Therefore, most physically based erosion and sediment transport models are applied at watershed or plot (with no sediment transport) scale erosion assessment. Some physically based models, such as ANSWER (Areal Nonpoint Source Watershed Response Simulation, Beasley et al., 1980) and LISEM (Limburg Soil Erosion Model, De Roo et al., 1996), build purely on physically based hydrology, erosion and sediment transport sub-models. In such cases, an input data file preparation can be rather extensive since the database provision is rather limited. Some physically based models, such as SWAT (Soil and Water Assessment Tool, Arnold, J.G., et al., 2012), CREAMS (Chemicals, Runoff, and Erosion from Agricultural Management Systems, Foster, G.R., 1981) and LUCIA (Land Use Change Impact Assessment, Marohn et al., 2013a), combine either empirical or process-oriented parts into physically based simulation, therefore require less detailed input data. In most practical cases involving management effects assessment or recommendations for decision makers, the latter is preferable by more friendly data preparation.

1.3 On-site and off-site soil conservation measures

To curb soil erosion problems, a range of soil conservation techniques and strategies have been discussed and applied worldwide. Since erosion-deposition is a natural process, the aim of soil conservation is not to prevent erosion but to reduce it to a level at which “the maximum sustainable level of agricultural production, grazing or recreational activity can be obtained from an area of land without unacceptable environmental damage" (Morgan, 2005). The ideal reference of tolerated soil loss should be the rate equivalent to (or below) the natural soil formation rate. However, neither this balance between loss and formation is recognized nor the slow soil formation can be easily determined. Currently the soil loss tolerance is mostly referenced to the measured erosion rate at small watersheds dominated by forest and grassland, where an equilibrium condition is assumed to exist (Morgan, 2005). This reference value, however, varies spatially and temporally.

Soil conservation is mainly based on three principles: i) increasing soil cover to reduce mechanical energy of soil detachment by raindrop/runoff; ii) improving soil structure (e.g. infiltration rate, soil carbon content) to reduce runoff production and soil erodibility; iii) manipulating the topography such as building terraces/contours to increase deposition. Normally, on-site conservation measures control erosion from the first two mentioned aspects as they are less expensive, more easily fitted into the farming systems and mostly directly benefit farmers (Ervin C.A. & Ervin D.E., 1982). Such conservation measures are easier to be widely accepted by farmers but normally require understanding of erosion processes in specific crop types as well as on-site experiments. For instance, organic mulching can not only reduce soil loss by increasing surface cover but also potentially increase crop yield by keeping soil moisture, reducing weed growth and providing nutrient and organic matter. However, timing, type and installation should be taken with care based on the local ecological situation; otherwise organic mulching may have negative impacts. Another example is intercropping; it controls erosion by increasing soil cover as well as offers extra income. Nevertheless, if the crop type, planting density, fertilizer and pesticide are not properly managed, it leads to economic loss. Topography manipulation, such as terracing, is labor demanding and in most cases less effective as it does not prevent the soil from detachment. Besides, the mechanical work also changes soil structure and exposes subsoil thus may lead to less crop yield. Regular and proper maintenance of terraces are required. Some failed terraces may cause more damage due to release of water ponded by the hillside. For these reasons, topography manipulation is mostly adopted as a supplementary measure on steep slopes (Morgan, 2005).

Above-mentioned conservation measures are aimed at soil retention on site and reducing the total amount of transported soil. These management strategies retard both on-site erosion and off-site deposition effects therefore lead to soil conservation and improvement of water quality. Other management options focus more on off-site influence of erosion on streams and aiming at improving the water quality by reducing total sediment entering the water body (Mekonnen et al., 2015). Riparian buffer strips (RBS) is such a typical management option. RBS are man-made vegetated zones (relatively narrow strips) adjacent to a river which function as buffer between the stream and the impact (such as sediments, dissolved substances) stemming from the surrounding catchment. Overland flow through the RBS is reduced due to the physical barrier of the vegetation (increasing roughness) which mechanically traps the sediments (Liu et al., 2008). Not only sediments are prevented to flow into the stream but also nutrients such as nitrate and phosphorous as well as pesticides and herbicides can be retained by deposits filtration, absorption during infiltration and decomposition (Dillaha et al., 1988). Advantages of such water managements are their additional benefit to the aquatic ecosystem such as streambank stabilization and stream temperature decrease. However, constraints of this management option as compared to soil conservation measures lie in their limited on-site soil protection effects, namely soil loss reduction. As a result, water management options are more suitable to improve aquatic ecology while soil conservation measures are more often used to maintain a sustainable terrestrial system. For some catchments, multiple measures are applied for integrative management of the watershed, which normally needs centralized and well-planned management programs.

1.4 Increase of erosion impact in mountainous regions

One major driver of erosion acceleration is land use change, especially when it comes with improper land management (Valentin et al., 2008). Rapid changes in land use are taking place globally by human activities such as intensive agriculture, grazing, urbanization and mining. In tropical areas of Southeast Asia, high pressure from food demand by population increase and urbanization in lowland areas have forced continued expansion of crop cultivation to steep upland areas, which are not fully suitable for usage as arable lands. From 1990 to 2010, the total forest cover of Southeast Asia is estimated to have dropped from 268 Mha to 236 Mha mainly driven by cash crop plantation expansion (Stibig et al., 2014). As most upland has steep slopes, replacement of natural vegetation, with high soil conservation potential, by cash crops with relatively low surface cover can strongly raise runoff and soil loss. Land use change from forest to oil palm (with Oxisols soil) increased soil loss from 1 to 13 Mg ha^{-1} $year^{-1}$ (Hartemink A.E. 2006). Increased on-site soil loss leads to higher sediment yields and deteriorates water quality. In an Indonesian research catchment, land use change from Rambutan plantation to cassava led to an increase of total sediment yield from 2.9 to 13 Mg ha^{-1} $year^{-1}$ (Valentin et al., 2008). Bruijnzeel (2004) hypothesized that a small increase in exploitation of land resources in headwater catchment areas < 1 km^2 may result in water quantity and quality deterioration to downstream users. Therefore, land use change induced accelerated erosion not only brings concerns to upland farmers but also influences water resources in the whole catchment.

1.5 Challenges in improving soil conservation in mountainous regions

The acceleration of land use change induced erosion in mountainous areas has recently raised high attention due to its essential impact on terrestrial and aquatic ecosystem functions in the whole catchment. Only in 2017, around 2400 papers on soil erosion (Scopus, with keywords “soil erosion”, 2017) and 2700 papers on soil conservation

(Scopus, with keywords "soil conservation", 2017) in mountainous regions have been published. Given such a large body of literature on this subject, it might be concluded that we now know almost everything about erosion and deposition involving the various processes, affecting factors, effects and the corresponding controls and, therefore, little new knowledge can be added or is necessary to be explored. However, we actually should be very cautious making such a conclusion since both critical analysis of existing research and practical management gaps has to be completed for sustainable exploitation of proper land management and environment improvement in the mountainous regions as will be explained in detail further.

Need for improved understanding of soil erosion processes in new land use types

Driven by governmental policies and increasing market opportunities, expansion of agricultural crops in mountainous region is presented mainly by diverse cash crops such as tea, coffee, banana, rubber and orchard plantation at the expense of traditional food-based agriculture. Soil erosion is an integrated process affected by climatic conditions (e.g. rainfall), plant properties (e.g. canopy cover) and topography. On-site conservation utilizes an improved management, which is based on knowledge of erosion processes in certain land uses. Knowledge gaps concerning the erosion process in some newly appeared land use types hamper conservation recommendation for such land uses. One example is the changed dominant erosion process for varied land uses. Stemflow contributes little to erosion in most agriculture crops (e.g. sorghum, maize) (Bui & Box, 1992) while it makes a substantial contribution to soil loss in macadamia trees plantations (Keen et al., 2010). Additionally, newly expanded perennial crops especially plantations like fruit orchards, teak, and rubber have a long life span ranging from 15 to 40 years, so that soil erosion may vary during this time. This also raises challenges to directly transfer current erosion and conservation knowledge mostly from annual crops to such land use types.

Need for innovative techniques and strategies for soil loss and sediment yield control in mountainous region

Despite the multiple research studies and programs focusing on reduction of erosion rate, the relative efficiency and feasibility of erosion control technologies at different scales have been poorly documented. For instance, most on-site experiments only focused on short-term erosion (soil loss) control efficacy at plot scale while lacking further study on long-term effects at watershed scale (sediment control) (Zuazo & Pleguezuelo, 2008). Some large scale studies at catchment or watershed level focused on total sediment yield while critical evaluation on soil loss control at hotspots was absent in most cases. The assessment of the most massive restoration project “Grain to Green” in the Loess Plateau, China discovered an erosion rate reduction of 34% of tested regions, unchanged in 48% and slightly increased in 18% regions respectively from 2000 to 2008 (Zhang et al., 2010). The mean erosion rate in areas with slope > 18° was still three times larger than the tolerable erosion rate. Though it was concluded that soil erosion is still a major ecological problem in the Loess Plateau, no further recommendation was proposed to improve the situation. Actually, innovation in erosion control research has been pointed out as a major research gap (Poesen, 2015) with the evidence that we still use techniques (grassed waterways, check dams) developed ca. 80 years ago to control gully erosion. New challenges of erosion control techniques in mountainous watersheds arise from its high erosion potentials with more strict standards. Though deforestation is not considered as a major contributor to large-scale flooding and sediment accumulation in riverbed (Kiersch & Tognetti, 2002; Sidle et al., 2006), it has been proved that expansion of soil cultivation with poor conservation managements increased the sediment yield and deteriorated the aquatic ecology. The latter is more crucial as aquatic systems not only offer another income for local farmers (e.g. fishery) but more importantly surface water often serves as a supply for drinking water in mountainous regions (Gao et al., 2013; Kathe et al., 2015; Muhammad et al., 2011). Thus, water quality provides a more strict reference for erosion control treating agriculture as the major contributor to water pollution through surface runoff. Therefore, innovative erosion control techniques are required not only for sustainable soil use but also for prevention of surface water pollution (e.g. by nitrogen, herbicides, pesticides) and for water quality improvement. As for conservation strategies in mountainous watersheds, challenges result from the conflict between traditional decision-making styles (e.g. top-down, bottom-up, representative) with scattered community distribution. Watershed management must take into account the needs of all those who depend on mountain water, namely people who live in mountain areas, who are however often marginalized from the decision-making processes. Fragmentation of natural vegetation and mosaic land cover distribution also hamper feasibility of traditional centralized erosion and sediment yield control measures.

Possible solutions for erosion and conservation challenges in mountainous region by short-term field investigation with the help of modelling

In order to overcome above-mentioned challenges, the need of long-term field data (e.g. land cover, management, soil loss, sediment yield, surface water quality) rises to disclose causal links between land use intensification and off-site impacts and supply a profound base for better management proposals. However, long-term multi-aspect catchment studies are time and labor consuming, and therefore, scarce. On the other side, new empirical and physical-based modelling techniques were developed since 1970s (Devia et al., 2015). The solid foundation by rich experience and techniques should play a crucial role in overcoming challenges as the past is the key to the future. Therefore, an alternative to long-term field studies might be the combination of relatively short-term field experiments with existing large datasets and modelling techniques to understand erosion process and problems induced to propose suitable conservation measures at both at plot and watershed scales. Therefore it is crucial to design correctly short-term field experiments, as they should cover erosion hotspots minimizing thus the knowledge gaps, and reasonably offset data insufficiency stemming from short time observations.

1.6 Justification of study

Xishuangbanna in Southern Yunnan, China is one of typical regions experiencing land use change including expansion of traditional agricultural crops and appearance of new tropical cash crop, i.e. rubber plantations, as this area is one of the few regions of China with a suitable climate for rubber cultivation. Due to the high demand of rubber in China's growing economy, the area of rubber cultivation increased drastically within the last decade. From 1992 to 2010, the area covered by rubber plantations in Xishuangbanna increased by almost 340,000 ha (a gain of 400%) (Xu et al., 2014). The rapid expansion of rubber plantations disturbs the indigenous rain forests and land occupied by traditional swidden agriculture (Fox and Castella, 2013) thus strongly affecting the natural local water balance, water quantity and quality. Despite building of terraces, which has been widely adopted to conserve soil when planting rubber trees on the steep slopes, increase of on-site and off-site soil erosion has been still observed (Figure 1.1). Therefore, typical management of rubber plantations threatens the maintenance of both soil fertility and stream water quality and weakens terrestrial and aquatic ecosystem services. For environmental managers and local farmers, questions arise how to reduce soil loss by simple and acceptable management in rubber plantations. However, long-term field studies are laborious and expensive. The use of a process-based erosion model is therefore required to simulate soil loss at mono-crop plot scale as well as spatially-explicit sediment yield at multiple land use watershed scale. However, the challenge of applying such an erosion model lies in the critical knowledge gap in understanding of erosion processes in rubber plantations, especially considering rubber plantation as a perennial crop with a lifespan ranging from 20 – 40 years. Data shortage of long-term observation on land use change impact make the problem solution even more challenging.

Figure 1.1 Rubber tree roots exposure due to serious soil erosion on the plot (left, source: W. Liu et al., 2015); high stream turbidity in rainy season at the outlet of a rubber-dominated watershed (right, source: H. Liu)

To overcome these challenges, the presented study took a sub-watershed (Nanhuicang) with rubber expansion as the case study site. It is located in Nabanhe Watershed National Natural Reserve (NRWNNR), Xishuangbanna, Southwest China (Figure 1.2). The space-for-time substitution was adopted by field investigation design assuming that 1) erosion processes in the spatially distributed rubber plantations with different standing age can present temporal change of erosion due to rubber growth; 2) sediment yield of the neighboring watershed with forest dominated land use can be regarded as the reference being comparable to the target watershed before land use change. After one-year field observations, field data was combined using models (empirical and physical based) to understand long-term erosion processes in rubber plantations and suggest better management dealing with sediment yield increase by land use change. Therefore, this research aims not only at proposing multi-scale (plot and watershed) conservation measures for a sustainable rubber cultivation concept but more importantly addressing the challenges of soil and water conservation in mountainous watersheds through an integrated method incorporating short-term field investigation, literature information and modelling techniques.

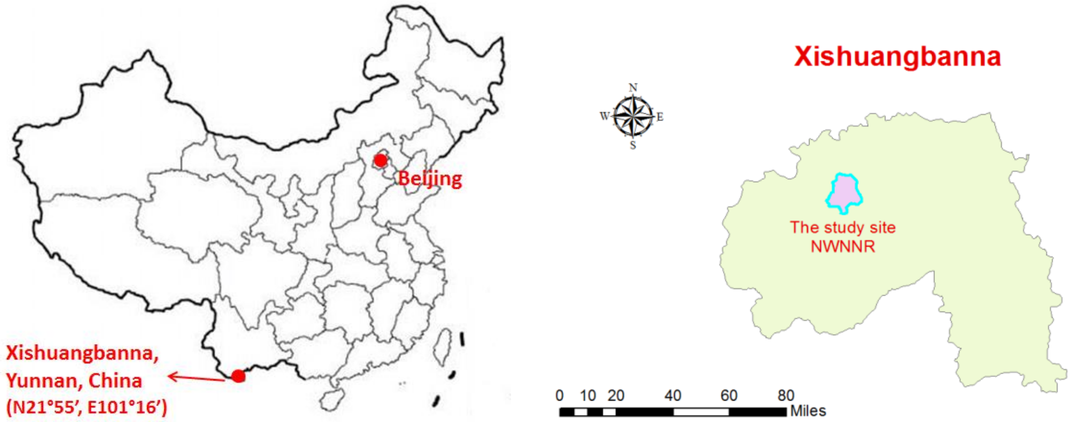

Figure 1.2 China with Xishuangbanna prefecture (left) and Yunnan with the study site Naban Water National Nature Reserve (NRWNNR, right)

1.6.1 Objectives and guiding hypothesis

The main objective of the thesis was to propose a management admissible by farmers in order to preserve soil from erosion in rubber plantations based on the results of trial experiments and modelling assessments on plot and watershed level (Figure 1.3).

The specific objectives of the presented study were to:

- Evaluate the change in erosive potential for one rubber rotation cycle and understand the dominant factors affecting erosion processes for rubber plantations of different age by field investigation.
- Test effects of different conservation measures (specifically different herbicide application frequencies) in a 12-year old rubber plantation, as a demo-site experiment, to propose a better management scheme for soil conservation.
- Simulate long-term (one rotation length) erosion processes and different soil conservation effects in rubber plantations with the help of a plant-soil model LUCIA by using one-year field data and literature data aiming at efficient long-term erosion control in rubber plantations.
- Assess different soil conservation strategies to mitigate sediment yield increase by rubber expansion through applying a spatially explicit model based on historic land use data, literature data and short-term (one-year) field data.

Corresponding to the specific objectives, the guiding hypotheses addressed in this thesis were:

- Erosion risk of rubber plantations change with developing rubber trees; and the major factor affecting the process should be the soil surface cover among selected indicators (canopy cover, canopy height, surface cover and fine root density with diameter $< 2mm$).
- Herbicide application is a common treatment to reduce understory vegetation after

plantation establishment. Different herbicide application strategies should be a key activity controlling erosion in rubber plantations by affecting understory plant cover.

- Short-term field experiments and literature reviews can supply a profound base for the simulation of erosion processes in new land use types (rubber plantation). With the help of models, long-term management effects on soil conservation can be simulated and evaluated.
- With a reasonable field monitoring site setting, spatially explicit models can be applied with short-term (one-year) data to simulate effects of cultivation expansion on sediment yield and offer a useful tool to test different conservation strategies in total sediment yield control.

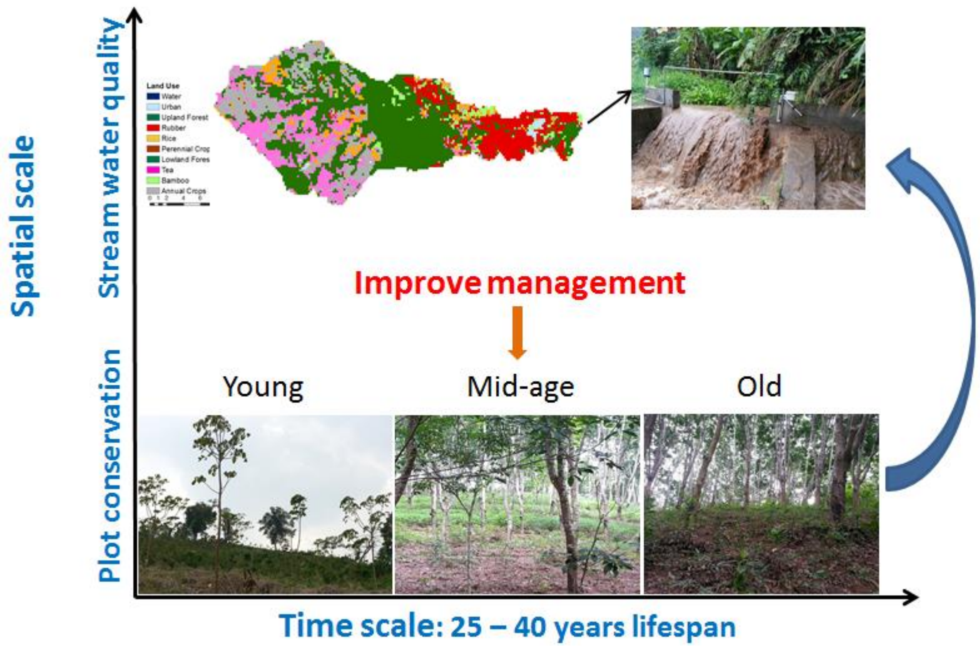

Figure 1.3 Work scheme of this thesis study

1.6.2 Outline of the thesis

This doctoral thesis is conceived as a cumulative thesis, where Chapter 2 - 4 are journal articles. The work started with the field investigation on erosion processes in rubber plantations of different ages (Chapter 2). Since rubber has a lifespan of 20 - 40 years, the space-for-time substitution combined with the empirical erosion model Revised Universal Soil Loss Erosion (RUSLE) were adopted to investigate erosion under rubber plantations of different age in a short-term field study, as well as to overcome spatial differences of soil properties and topography (slope). Simultaneously a 12-year old rubber plantation was selected as the demo-site implementing a herbicide application experiment to figure out a feasible and better management for soil conservation (Chapter 3). With the field data collected within one year, the plant-soil model Land Use Change Impact Assessment (LUCIA) was improved to better simulate long-term (one rotation length) erosion processes and conservation effects in rubber plantations (Chapter 4). Conservation measures were further simulated by LUCIA for a watershed with a mosaic land use that has experienced rubber expansion to assess how plot management controls total sediment exports to the stream (Chapter 5). To offset missed historic data reflecting land use change, a neighboring watershed with a forest dominated land use was selected referring the space-for-time substitution discipline, and set as the reference being simultaneously monitored and modelled. The final chapter (Chapter 6) discussed the management and conservation options in rubber plantations and elaborated the potential of integrating decentralized plot conservation into mountainous watershed management. Further aspects in Chapter 6 refer to possibility and risk of using short-term field monitoring data combining modelling techniques in data-limited environments such as mountainous regions for decision support in watershed and environment management.

Chapter 2 Impact of rubber plantation age on erosive potential studied with USLE model[a]

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[a] This chapter has been reprinted from:
Liu, H., Blagodatsky, S., Liu, F., Xu, J., Cadisch, G. (2018). Impact of rubber plantation age on erosive potential studied with USLE model. Journal of Applied Water Engineering and Research 6 (4): 252-261. (DOI: 10.1080/23249676.2018.1452647.) With permission from 2018 Taylor & Francis Ltd.
The original publication is available at:
https://doi.org/10.1080/23249676.2018.1452647

Abstract

Rubber (Hevea brasiliensis) plantations have a lifespan in the range of 25 to 40 years. We aimed at assessing soil losses in rubber plantations of different ages (4, 12, 18, 25 and 36 years old) and relating erosion risk to surface cover and fine root density by applying the Universal Soil Loss Equation (USLE) model. We measured in rubber plantations runoff and sediment yields during one year. Fine root density, surface cover and understory plant cover were measured monthly. Annual soil loss in different plantations varied from 0.33 to 2.80 Mg ha^{-1}. Combined annual cover, management and support practice factor CP of the USLE model varied with the growth phase of rubber in the range of 0.006 - 0.03. Monthly CP factor was related exponentially with surface cover. Inhibited undergrowth and high decomposition both contributed to decreasing surface cover in mid-age rubber resulting in highest erosion risk phase.

Keywords: erosion control, soil loss, RUSLE model, cover and management factor, field studies

2.1 Introduction

The last several decades have seen the rapid emergence and expansion of rubber plantations in Southeast Asia. Xishuangbanna, Southwest China, as the second largest non-traditional area, has experienced an increase of around 340,000 ha (a gain of 400%) of rubber plantations since 1992 up to 2010 (Xu et al., 2014). The large scale land use change can greatly accelerate water erosion processes and consequently land degradation (Dunjo et al., 2004). Potential increase of runoff by a factor of 3 and soil loss by a factor of 45 in rubber plantation was caused by increased soil compaction compared to natural forest (Wu et al., 2001). Therefore, it is important to investigate actual soil losses in rubber plantation to evaluate land use change impact on ecosystem functions.

The lifespan of rubber plantations ranges from 25 to 40 years. Soil erosion may change in time depending on tree growth and development. Having more bare soil due to an open canopy and less developed rooting system may make young rubber plantations more vulnerable to erosion. Once the plantation is established, the closed canopy and well developed roots should supply a much better protection from rainfall impact. Besides, soil is expected to be reconsolidated due to no tillage practice in rubber plantations thus becomes less erodible with increasing plantation age (Kabiri et al., 2015; Kasper et al., 2009). Previous studies on erosion in rubber plantation are rare and only reported soil loss in one specific plantation age (Liu et al., 2012). Insufficient field data on erosion in rubber plantations of different age makes it difficult to compare soil loss in rubber plantations with other land use types and raises uncertainty for an integrated soil conservation evaluation covering the whole rotation length.

Application of erosion models in rubber plantations helps to quantify contributions of different plant components like canopy and root to soil conservation, and therefore offer suggestions to improve soil protection by management. Besides, it allows upscaling of water erosion to the watershed level, relating soil loss at the plot level to water pollution in streams and provide an integrative assessment of soil conservation functions of different land use types (Guo et al., 2015; Shi et al., 2004; Zhang et al., 2006). The Universal Soil Loss Equation (USLE), developed by Soil Conservation Service of the US Department of Agriculture, introduced five factors known as rainfall (R), soil properties (K), topography (L and S), the cover and management (C) factor and support practice factor (P) to predict soil loss from agriculture (Kinnell, 2010). USLE is widely applied to agricultural lands in many countries. Attempts have been made to use USLE on forest lands, where C and P factors were in some cases combined together as the CP factor and were found changing with plantation ages (Kitahara et al., 2000; Özhan et al., 2005; Hattori et al., 1992). Dissmeyer and Foster (1997) linked CP factor of forest system to 9 sub-factors: surface cover, canopy cover, soil reconsolidation, high soil organic content, fine root, residual, onsite depression, step and contour tillage.

Quantifying relationship between *CP* factor and related environmental drivers (sub-factors) acting in rubber plantations can help to estimate the major reason for erosion risk change during the rubber rotation cycle, and to further direct better soil conservation management.

The specific objectives of this research were to: 1) evaluate and compare runoff and soil loss under rubber plantations of different age; 2) estimate the annual *CP* factor combined of cover and management (*C*) and support practice (*P*) in the USLE model for different rubber plantation age; 3) identify the impact of different factors (canopy cover, canopy height, soil cover, understory plant cover and root) on *CP* value and respectively on soil erosion. Our findings should help to improve soil conservation by better management and thereby also reducing off-site effects in surface streams.

2.2 Materials and Methods

2.2.1 Study site

The study was carried out in the Nanbanhe Watershed National Nature Reserve (NRWNNR), in Xishuangbanna, Yunnan Province South-West China. The study area is characterized by tropical monsoon climate with a rainy season from May to October and a dry season from November to April. The annual precipitation is 1100-1600 mm, among which 60%-90% of precipitation is distributed in the rainy season; and the mean annual temperature is 18-22°C. Soils in study area were classified as Acrisol.

Rubber has been introduced into NRWNNR since 1970s and expanded to 28.6 km^2 in 2013. Five ages of rubber plantation were selected within one rotation length: 4 year (4Y) as representative of young rubber with canopy height of 2 m and canopy coverage of around 40%; 12 year (12Y) and 18 year (18Y) as representative of mid-age rubber with canopy height of 7 m and coverage around 85%; 25 year (25Y) and 36 year (36Y) as representative of old rubber with canopy height of 8 m coverage of around 90%. All plantations were located in the similar elevation around 715 m. The slopes of selected plots were in the range of 20° - 37° (Table 2.1). Tree density in all plots was in the range of 450-600 trees per hectare, which is a common practice in the region (Yang, 2011).

Table 2.1 Slope, soil properties and calculated soil erosivity of five selected rubber plantations of different ages to measure soil erosion: 4-year, 12-year, 18-year, 25-year and 36-year old. Soil erosivity was calculated using the model of Wang (2013), which ba sed on the often used soil erodibility model, Revised Universal Soil Loss Equation (RUSLE) (Renard et al., 1997), with corresponding adjustments made for red soil erodibility in South China.

Ages	Slope	Soil						Soil erosivity (dimensionless)
		Sand (%)	Silt (%)	Clay (%)	OM (%)	Bulk density (g cm^{-3})	Infiltration rate (mm/h)	
4Y	26°	24.8±1.1	47.5±1.1	27.7±0.6	4.0±0.2	1.07±0.04	30.9±0.1	3.3
12Y	29°	7.2±0.6	56.8±1.6	36.0±2.1	5.2±0.7	1.11±0.04	19.6±0.4	2.8
18Y	20°	12.6±1.4	48.4±2.2	39.0±1.2	4.8±0.2	1.13±0.06	30.7±0.3	2.6
25Y	37°	17.6±1.7	46.6±2.3	35.8±1.2	4.9±0.3	0.92±0.03	36.1±0.2	2.1
36Y	35°	19.2±1.6	40.4±2.3	40.4±3.4	4.8±0.8	0.85±0.04	35.4±0.8	2.2

2.2.2 Experimental layout

In general, rubber trees are planted in rows on terraces with a tree spacing of 1 - 2 m and distance between two adjacent planting terraces of 4 - 6 m. However, terraces shape (e.g. terrace width, inclining angle) differ between plantations locations due to variation in building and maintenance. This can strongly affect soil loss, especially soil deposition, measured within big plots. We focused on the erosive potential changes caused by rubber plantation age, therefore chose the inter-terrace slope area as the study unit (Figure 2.1) to exclude impacts from terrace management. Liu et al. (2016) compared runoff and sediment collected by bounded and unbounded plots in rubber plantations, and concluded no significant difference of soil losses, runoff and sediment concentration by the two methods. Therefore we established in total fifteen unbounded plots with Gerlach troughs of 0.5 m width (Morgan, 2005) in the 4Y, 12Y, 18Y, 25Y and 36Y rubber plantations on the slope straight plane. All plots located in the mid-mountainside with three replicates of each age. At the top of the slope a non-permeable metal fence was inserted 10 cm into ground to isolate surface and subsurface run-in from above. The Gerlach troughs in mid (12Y and 18Y) and old (25Y and 36Y) plantations were placed at 5 m distance to the fence at the bottom of the slope while placed at 4 m distance to the fence in young plantation (4Y) due to shorter slope length between terraces. The contributing area was estimated to equal to the width of Gerlach trough (0.5 m) timed the distance between the trough and isolated metal fence (4 or 5 m), namely 2 m^2 for young rubber and 2.5 m^2 for mid and old plantations. Estimation of the plot area was based on the assumption that loss of runoff or sediment from the defined area can be balanced by inputs from adjacent areas if the slope was straight in plan (Morgan, 2005). Gerlach troughs were connected to collection vessels via a plastic pipe with a diameter of 5 cm, which delivered runoff and sediment to a series of two 200 L containers. The second container collected excess runoff of the first in case of storm events. Herbicide was applied twice per year to all plots in mid-rainy season and mid-dry season respectively, with the amount of 10 kg ha^{-1} of 10% glyphosate.

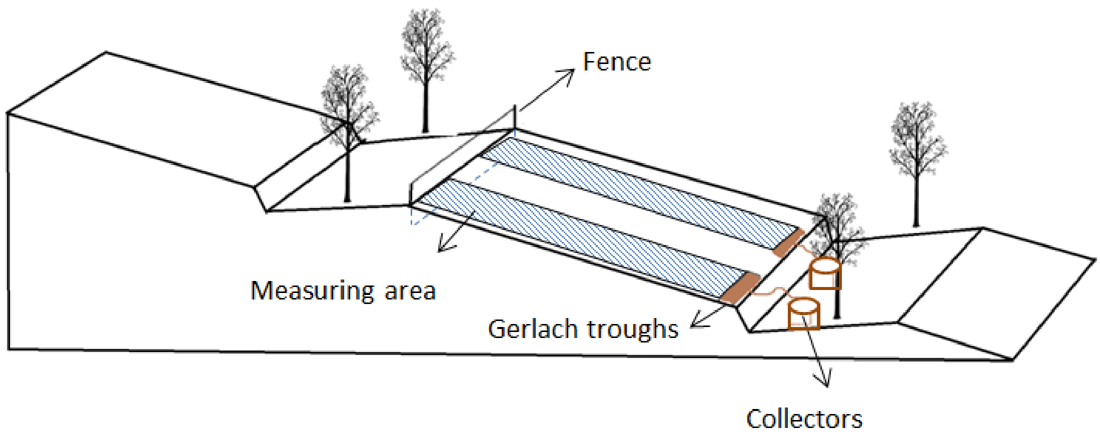

Figure 2.1 Sketch of plots overview. The measuring plots were located on the slope area between terraces. GT = Gerlach trough.

2.2.3 Data collection

Daily precipitation was recorded by a tipping bucket rain gauge (Campbell Scientific TB4), with frequency of every 15 minutes. Runoff and sediments were collected and measured after each event that produced erosion under natural rainfall in 2014. Runoff was determined by transferring the water level in collectors into volume using geometric equations. Sediments were firstly collected from depositions remaining in troughs and air-dried. A subsample was taken from air-dried sediments and stored for carbon determination. Remaining sediments were oven-dried at a temperature of 105℃ until constant weight was achieved. Additionally, solids containing in runoff collected in the containers were determined by taking a sub-sample of 500 mL from each tank after vigorous and homogeneous mixture. The sub-sample was then filtered through Whatman filter paper (d = 125 mm, 11Micron, 10.5 s/100 mL/sq inch flow rate) and oven-dried at 105℃. Then total weight of sediments was divided by the plot area (2.5 m^2) to obtain soil loss in gram per square meter. Top 5 cm soil samples under each plantation age were collected for bulk density, texture and organic matter analysis. Steady infiltration rates were measured with a portable rainfall simulator at a rainfall intensity of 40 mm h^{-1} for each plantation age. Canopy cover was estimated by following equation:

$$\text{Canopy Cover } (\%) = \frac{\pi \times R_1 \times R_2}{4} \times \text{Planting Density} \times 10^{-2} \quad (2.1)$$

where R_1 is maximum tree crown width measured along the slope (m); R_2 is maximum tree crown width measured across the slope (m). Surface cover, understory plant cover and fine root density (root diameter <1 mm and 1-2 mm) were monthly measured by photography and top 10 cm soil coring, respectively. Detailed description of sampling and analysis can be found in Liu et al. 2016.

2.2.4 Data analysis and statistics

Our study focused on investigating erosion in plantations of different ages. However, different soil properties and slope gradients do not allow a direct comparison of soil losses in plantations of different age. Therefore, we calculated the combined (*CP*) factor from cover and management (*C*) and the support practice (*P*) factor in the USLE model

(Wischmeier and Smith, 1978), which quantifies the vulnerability of the ecosystem to erosion depending on plant/crop cover. Thus, the impact of soil properties and steepness of the slope can be excluded by applying the USLE model.

CP (cover, management, and support practice) factor calculation: The USLE equation is defined as:

$$A = R * K * L * S * C * P \tag{2.2}$$

where A = soil loss (Mg ha^{-1}), R = rainfall erosivity (MJ mm ha^{-1} h^{-1}), K = soil erodibility (t h MJ^{-1} mm^{-1}) (i.e. the soil loss per unit of erosivity for a standard condition of bare soil, 5° slope of 22 m length), L = slope length factor (dimensionless), S = slope steepness factor (dimensionless), C = the cover and management factor (dimensionless), and P = the support practice factor (dimensionless).

In the plantation system, P was mostly treated either as a subfactor of C (Dissmeyer & Foster, 1980) or as a new combined *CP* factor (Özhan et al., 2005; Brooks et al., 1996; Hurni, 1982). In our study, in order to focus on the effects of rubber plantation age, deposition on the terrace platform was not included in the measurement. Therefore, instead of calculating the P factor separately, we followed the method of treating C and P together as *CP*.

The USLE model was initially designed to predict long-term average annual soil losses in a standard plot (5 m * 22.1 m) with a gentle slope area (9%). Then it was applied to short-term time-scales, such as monthly or event time-scales, with further development. Rubber plantations, as a perennial crop, should have different annual *CP* values depending on age of growing plantation. In addition, similar to annual crops, *CP* values for each age class should vary monthly. Therefore, we applied the USLE model at annual and monthly scale to identify erosion change within different plantation ages and to assess major drivers affecting *CP* factor dynamic in rubber plantations. The *CP* factor for rubber was calculated using measured annual or monthly sediment yields along with other estimated factors (Brooks et al., 1996; Wischmeier and Smith, 1978), namely $CP = A/(R * K * L * S)$. CP values calculated annually were related to canopy cover and canopy height, while monthly-calculated CP values were related to surface cover, understory plant cover and fine root density (root diameter < 2 mm). Correlation analyses were performed with the statistical package R version 3.1.3 (http://www.r-project.org/).

The R factor was calculated by summing up the annual or monthly rainfall kinetic energy (EI_{30}) (Brown and Foster, 1987) for two scales respectively;

$$EI_{30} = \left(\sum_{r=1}^{n} e_r v_r\right) I_{30} \quad (2.3)$$

where

$$e_r = 0.29[1 - 0.72\exp(-0.05 i_r)] \quad (2.4)$$

EI_{30} is the rainfall erosivity index of a single event (MJ mm ha^{-1} h^{-1}). I_{30} is the maximum rainfall intensity during a period of 30 minutes in the event (mm h^{-1}), e_r is the unit rainfall energy (MJ ha^{-1} mm^{-1}), v_r is the total rainfall volume (mm) of the single event, and i_r is the average rainfall intensity (mm h^{-1}) of the single event.

K is soil erodibility (Mg ha h ha^{-1} MJ^{-1} mm^{-1}), the soil loss per unit of erosivity for a standard condition of bare soil, 5 degree slope of 22 m length; K was calculated by equations (4) and (5) (Wang et al., 2012):

$$K = 0.0364 - 0.0013\left[\ln\left(\frac{OM}{D_g}\right) - 5.6706\right]^2 - 0.015 \times \exp\left[-28.9589\left(\lg(D_g) + 1.827\right)^2\right] \quad (2.5)$$

where

$$D_g = \exp\left(0.01 \times \sum f_i \ln(m_i)\right) \quad (2.6)$$

OM = soil organic matter content (%), $D_g =$ geometric mean diameter of the soil particles (mm), f_i = weight percentage of the silt, clay and sand fraction (%), m_i = arithmetic mean of the size limits for silt, clay and sand (mm).

L and S are factors reflecting slope length and steepness (uniteless). L was determined by equation (2.6) (Wischmeier and Smith, 1978); S was determined by equation (2.7), which was developed based on experiments in mountainous area with slope in the range of 15° - 39° (Chen et al., 2010).

$$L = \left(\frac{\lambda}{22.1}\right)^{0.5} \tag{2.7}$$

$$S = -8.43(sin\theta)^2 + 9.37sin\theta + 0.22 \tag{2.8}$$

Where λ and θ are horizontal slope length projection (m) and steepness (°) respectively.

2.3 Results

2.3.1 Rainfall, runoff and soil loss for rubber plantations of different age

Annual precipitation in 2014 was 1128 mm at the experimental site, similar to the previous 12 years (2002-2013) with an average rainfall of 1256 mm (data from Jinghong airport meteorological station). During 2014, 24 rainfall events generated overland flow ranging from 10 to 88 mm with respective estimated erosivities of 42 to 1075 MJ mm ha^{-1} h^{-1}, hence total cumulative rainfall erosivity was 3694 MJ mm ha^{-1} h^{-1} in 2014. Rainfall volume and erosivity had similar monthly distributions mostly concentrated in the middle of the rainy season in August (Figure 2.2). Over 90% of erosive events were recorded during July to September. Very high erosive events (EI_{30} > 300) were only observed twice in 2014 on 14th July and 17th August.

Runoff and soil losses were highly related to rainfall amount (Figure 2.3). Minimum rainfall event causing erosion was 16 mm for young rubber, 10 mm for mid-age rubber and 20 mm for old rubber. The highest runoff coefficients were recorded during a storm event on 17th August with a daily precipitation of 88 mm and resulting in 7.7, 21.2, 9.8, 4.5 and 4 mm runoff for plots with increasing plantation ages. Maximum sediment yields were produced by the same event with an estimated rainfall erosivity of 1075 MJ mm ha^{-1}, corresponding to 0.23, 0.94, 0.40, 0.10 and 0.09 Mg ha^{-1} for the five increasing plantation ages. Runoff and sediment load caused by storms (event with total rainfall volume > 50 mm) comprised over 45% and 50%, respectively for all rubber plantations.

Highest erosion was detected in mid-age plantations (12Y and 18Y) (Figure 2.4). Old rubber plantations (25Y and 36Y) had the lowest runoff coefficient (0.04 and 0.03, respectively) defined as ratio of total runoff to precipitation and lowest soil losses with no significant differences between them. We found the following statistically significant order for both runoff and soil loss in rubber plantations of different ages:

12Y > 18Y > 4Y > 25Y, 36Y.

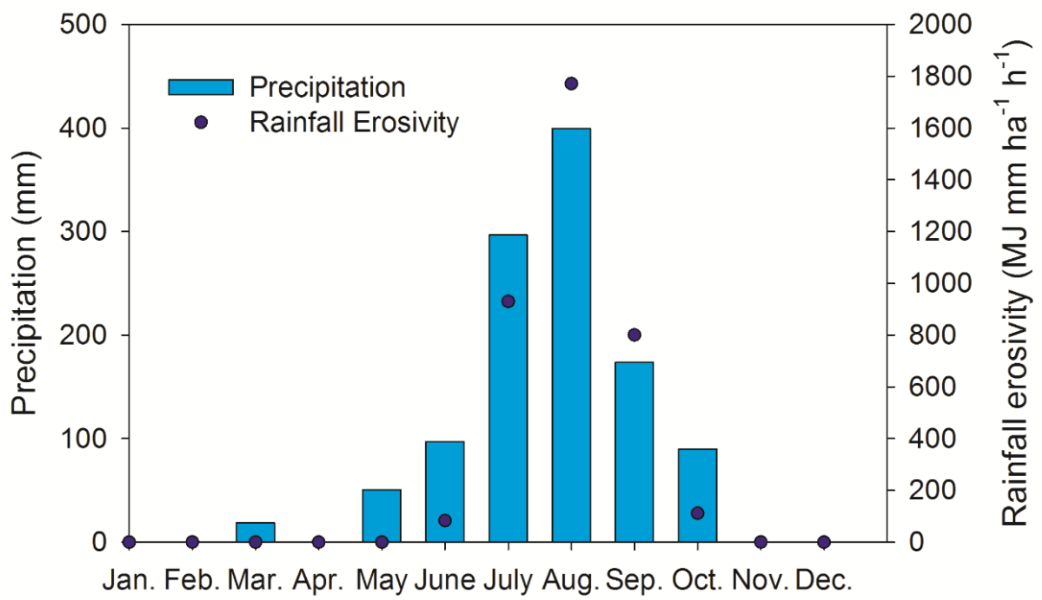

Figure 2.2 Monthly rainfall amount and cumulative erosivity distributions for the monitored period of 2014 at the study site in the Nanban River Watershed National Nature Reserve, Xishuangbanna, Yunnan Province, South-West China. Rainfall erosivity was calculated using the model of Browns and Foster (1987).

2.3.2 Soil properties and plant dynamics

Properties of top 5 cm soils for all plots were summarized in Table 2.1. A clay loam texture in 4Y plantation differed from a silt clay loam texture found in other plots. Soil erodibility of all but 4Y rubber plantations fell in a medium range (< 3), while the youngest rubber plantation had slightly higher erodibility with lower clay content (< 35%).

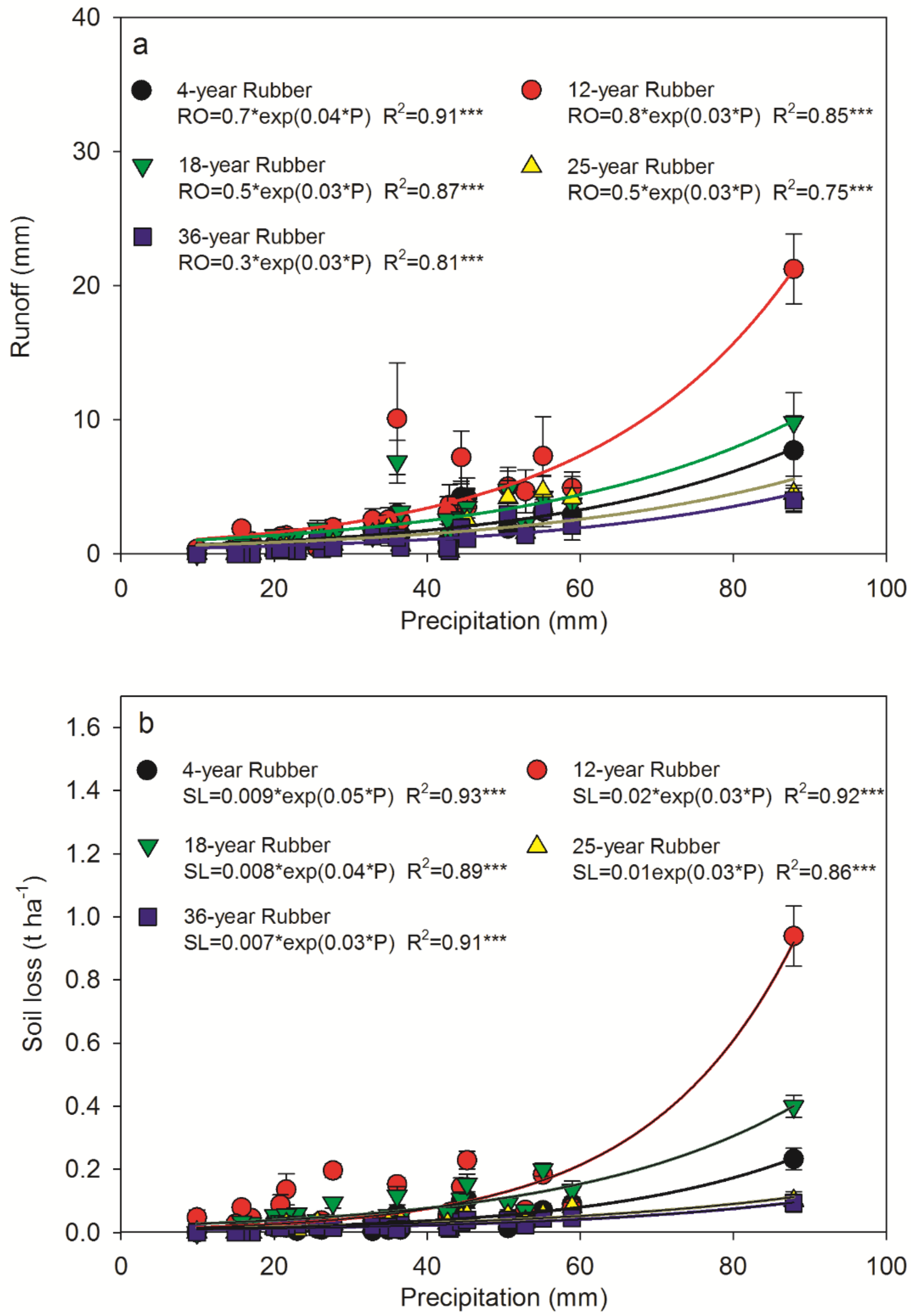

Figure 2.3 Relationship between precipitation and event-based a) runoff and b) soil loss under different rubber plantation ages. P = Precipitation, SL = Soil Loss, RO = Runoff

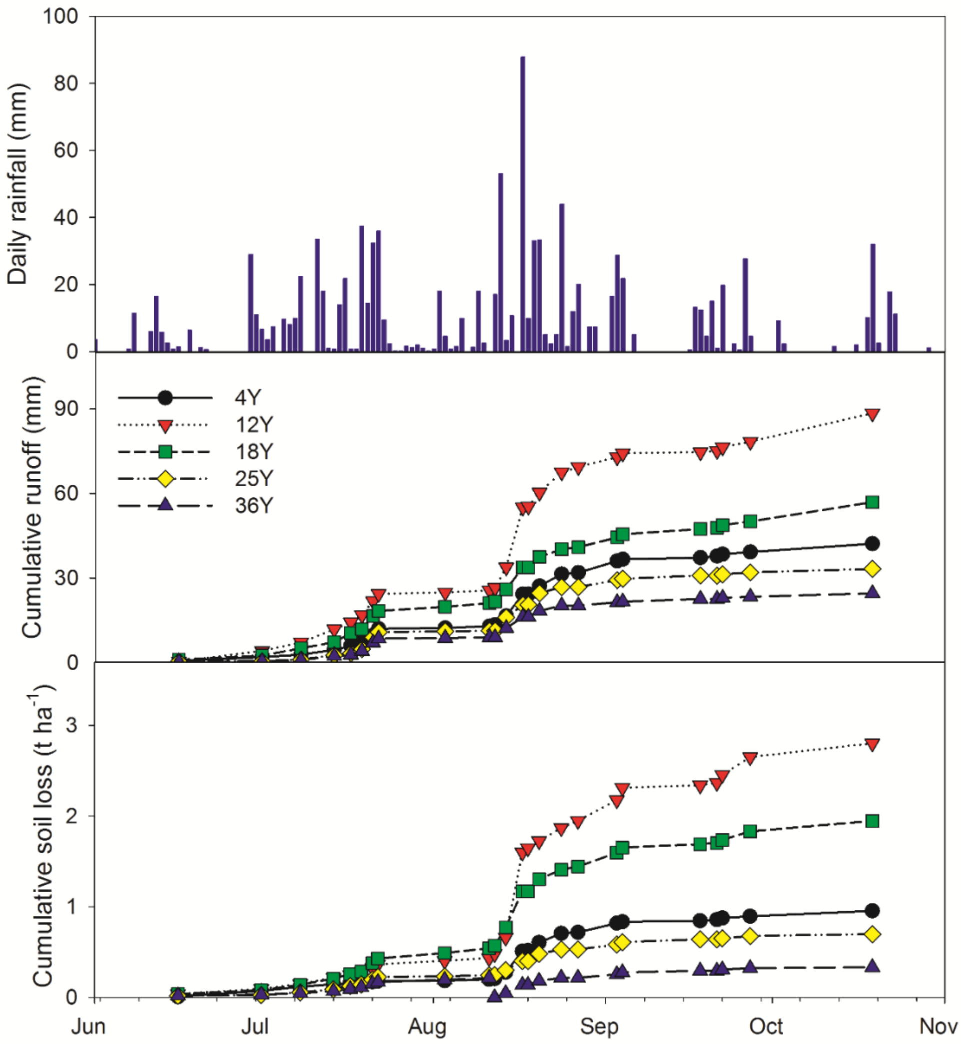

Figure 2.4 (a) Daily rainfall, (b) cumulative runoff and (c) cumulative soil loss in rubber plantations of different age during the rainy season (June to October) in 2014. 4Y: 4-year-old rubber plantation, representative as young rubber; 12Y: 12-year-old rubber plantation, representative as mid-age rubber; 18Y: 18-year-old rubber plantation, representative as mid-age rubber; 25Y: 25-year-old rubber plantation, representative as old rubber; 36Y: 36-year-old rubber plantation, representative as old rubber.

Table 2.2 Percentage of roots from undergrowth (d < 1 mm) to total fine roots (d < 2 mm) in different rubber plantation ages (4-, 12-, 18-, 25- and 36-year old)

	4Y (%)	12Y (%)	18Y (%)	25Y (%)	36Y (%)
June	67	23	26	10	11
July	65	18	20	13	8
August	60	19	27	11	10
September	76	16	27	11	11
October	60	13	20	8	7

Surface cover of young (4Y) and old (25Y, 36Y) plantations was kept stable and high (over 80%) during the rainy season (Figure 2.5). In contrast, surface cover of mid-age rubber (12Y, 18Y) declined continuously in rainy season and dropped below 40% in October. All plots showed increase in understory plant cover from June to July. Different trends in understory cover after July were caused by different responses to herbicide application. Understory plant cover in young (4Y) and mid-age (12Y and 18Y) rubber both largely decreased by over 50% after herbicide application while a later increase up to 60% (by October) was observed in young rubber. Old rubber (25Y and 36Y), however, showed a different pattern of more stable understory plant cover of around 40% during whole rainy season.

Fine root density (fRD) in old rubber (25Y and 36Y) was much higher than in young (4Y) and mid-age rubber plantations (12Y and 18Y) (Figure 2.5). Fine roots mainly derived from undergrowth in young plantation by contributing over 60% of roots with diam. < 1 mm, while under rubber trees in mid-age and old plantations undergrowth provided less than 25% of roots with diam. < 1 mm (Table 2.2). The change patterns for mid-age and old rubber were the same, namely fine root density kept stable in rainy season at around 0.5 kg m^{-3} for mid-age rubber (12Y and 18Y) and 1.4 - 1.7 kg m^{-3} for old rubber (25Y and 36Y) (Figure 2.5). Dynamics of fine roots in young rubber (4Y) were more variable: increasing from 0.5 to 0.8 kg m^{-3} from June to September and then decreasing to 0.7 kg m^{-3} in October.

2.3.3 Annual and monthly CP values of different rubber growth stages

Dynamics of annual *CP* factor during development of rubber plantation was calculated using equation (1) and is shown in Figure 2.6. Annual *CP* values of two selected mid-age rubber plantations, namely 12Y and 18Y, were similar; *CP* values of old rubber plantations, namely 25Y and 36Y, were also close. Erosive potential (represented by annual *CP* value) was low during young rubber phase but increased rapidly and stayed high once rubber entered the latex production phase (mid-age: 12Y and 18Y). After that, erosive potential decreased and stayed low in mature rubber plantation (25 and 36 years old).

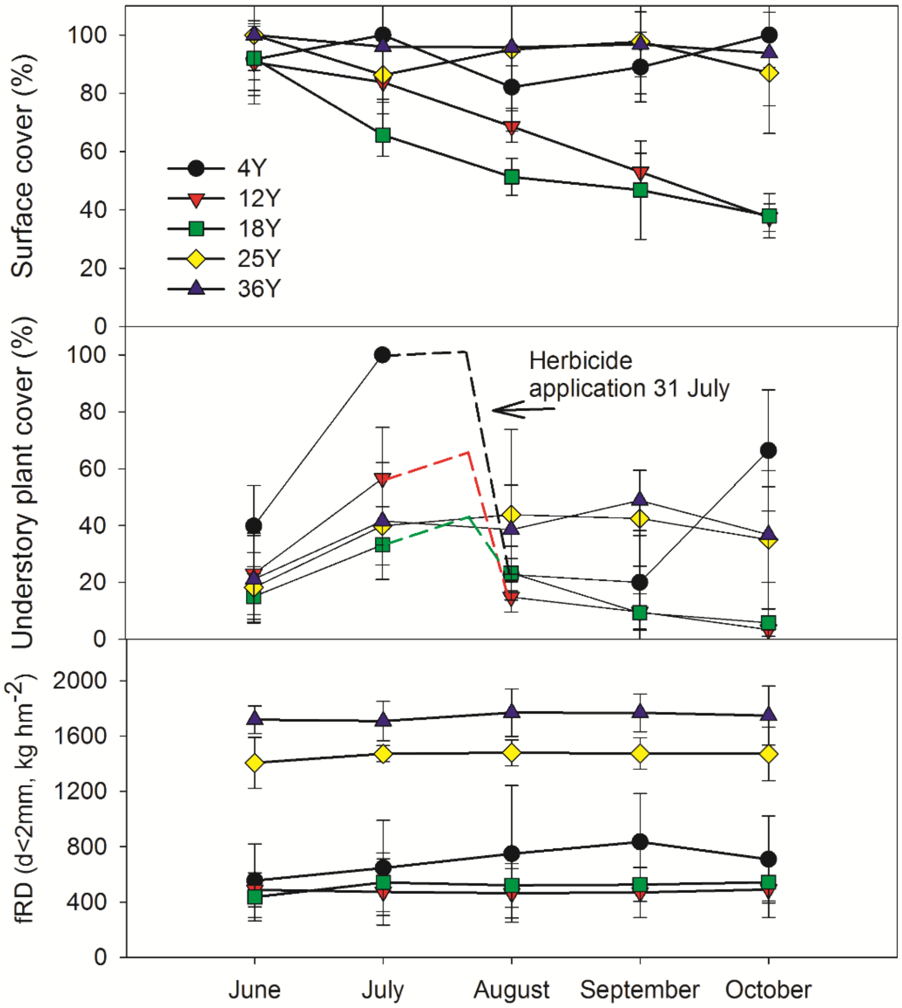

Figure 2.5 Temporal dynamics of (a) surface cover (understory plant and litter cover), (b) understory plant cover, (c) fine roots content (d < 2 mm) under different rubber ages from June to October in 2014.

Monthly variation of *CP* value in plantations of different age indicated that erosive potential of old rubber (25Y, 36Y) was more stable within one rainy season compared to young (4Y) and mid-age rubber (12Y, 18Y) (Figure 2.6). *CP* value of 4Y rubber plantation ranged from 0.004 to 0.02; while those of 12Y ranged from 0.013 to 0.05; and from 0.02 to 0.056 for 18Y. Highest monthly *CP* values of young and mid-age rubber were in October. Among tested environmental drivers, the surface cover showed highest correlation with *CP* factor (Table 2.3). Therefore, we chose surface cover as the indicator for CP factor. The relationship between them was best described by:

$$CP = 0.13e^{-0.029SC} \tag{2.9}$$

where SC = surface cover (%).

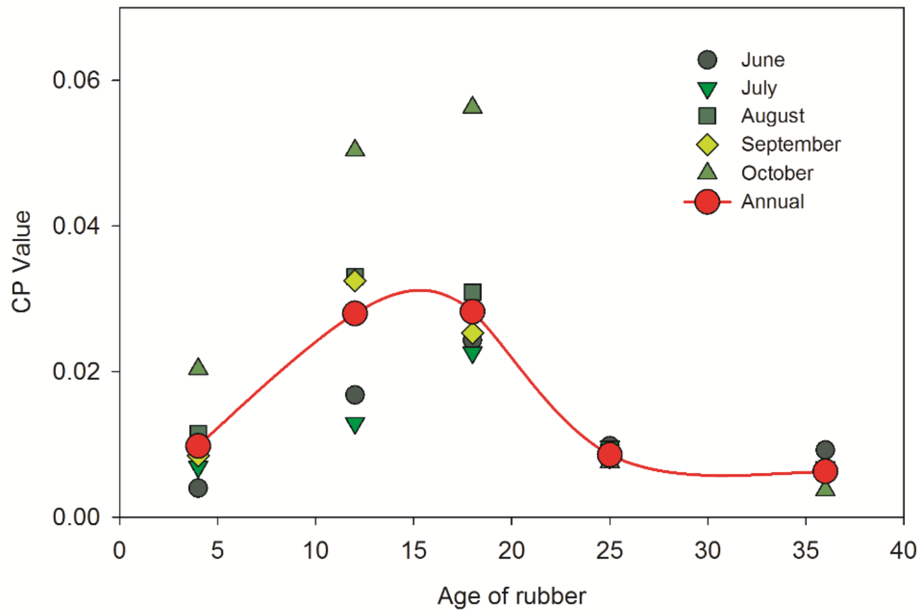

Figure 2.6 Annual and monthly *CP* facotr for different age rubber plantations under standard management. 4Y, 12Y, 18Y, 25Y and 36Y are 4-year-old rubber plantation, 12-year-old rubber plantation, 18-year-old rubber plantation, 25-year-old rubber plantation, and 36-year-old rubber plantation, respectively. *CP* factor (dimensionless) is the combination of cover and management factor (*C*) and support practice factor (*P*) in the Universal Soil Loss Equation (USLE) model.

2.4 Discussion

2.4.1 Erosion of growing rubber plantations

Soil loss in rubber plantations was generally higher than that reported under forest (0.05 Mg ha^{-1}) in Xishuangbanna (Li, 2001), measured under similar annual rainfall amounts and distributions. However, losses were much lower compared to annual agricultural systems in Southeast Asia, which showed values in the range of 8.90–174 Mg ha^{-1} (Valentin, 2008; Pansak et al., 2008; Tuan et al., 2014). Runoff production in rubber plantations of different ages was affected by soil steady infiltration rate (Figure 2.4, Table 2.1). Lowest infiltration rate of 12Y produced highest runoff, while similar infiltration rates for 4Y and 18Y, 25Y and 36Y produced comparable runoff amounts. Steady infiltration rate is mostly determined by soil texture and soil organic carbon (SOC) content (Franzluebbers, 2002). In our study, the land use history and soil properties of each selected plantation plot were not the same. Thus, we cannot attribute different soil infiltration rates to plantation age only. Several factors influence the soil infiltration rate under rubber; therefore, further study on yearly dynamics of soil properties under growing rubber plantation is necessary. Soil loss showed similar dynamics with runoff (Figure 2.4); and runoff, as a major driver for soil detachment, well explained different soil losses under plantations with different age (Table 2.3).

Table 2.3 Correlation between soil loss, runoff, *CP* factor in USLE equation, and precipitation, canopy, surface and understory plant cover and fine root density. *CP* factor (dimensionless) is the combination of cover and management factor (*C*) and support practice factor (*P*) in the Universal Soil Loss Equation (USLE) model.

	Precipitation (mm)[1]	Runoff (mm)[1]	Canopy Cover (%)[2]	Canopy Height (m)[2]	Surface cover (%)[3]	Understory plant Cover (%)[3]	fRD (<2mm)[3]
Soil loss (t ha^{-1})	0.55**	0.87**	0.14	0.07	-0.41*	-0.24	-0.38
Runoff (mm)	0.70**	-	0.05	-0.02	-0.14	0.16	-0.32
CP factor	-	-	0.21	0.13	-0.88**	-0.58**	-0.63**

[1] calculated based on event measurement in all plantation ages

[2] calculated based on annual value

[3] calculated based on monthly value

* $p < 0.05$, * $p < 0.01$

2.4.2 *CP* value of rubber plantation and its sensitivity to surface cover

Annual *CP* value for plantations of different ages was calculated to exclude impact of soil properties and topography. Annual *CP* value varied with plantation age similar to soil loss. However, unlike soil loss, the differences between *CP* values of the same growth phase, namely 12Y and 18Y, were less than measured differences between soil losses (cf. Figure 2.4 middle and Figure 2.5). This indicates that *CP* factor of perennial plantation is affected by tree age but also has a specific value for the same growing phase, which coincides to Dissmeyer & Foster (1997) conclusion based on natural forest investigation. The *CP* factor was a result of the crop/plantation condition (stage, cover) subjected to erosive rainfall. We derived the *CP* value in this study based on one year's monitoring. However, according to the reported similarity of annual rainfall erosivity in 2014 and long-term data (Liu et al., 2016), the annual *CP* factor in this study can be considered as a typical value for rubber plantation in this region.

Highest correlation between surface cover and *CP* value clearly indicate the main factor controlling erodibility change in rubber plantations. Although tree may contribute to soil protection, herbaceous plant is proved to be more efficient in soil conservation than woody plants under varied environments (Durán Zuazo & Rodríguez Pleguezuelo, 2008; Herweg & Ludi, 1998; Kort et al., 1998). Rubber trees in plantation may increase soil detachment instead of protecting the soil. Brandt et al. (1988) reported that canopy higher than 5 m tended to enhance soil detachment by accumulation of intercepted rain drops. Rubber canopy height can rapidly increase to 5 - 8 m in 7 years after planting (Yang, 2011), increasing the splash erosion compared to those in open area (Liu, 2012). Under such condition, surface cover becomes more important in soil protection from raindrop induced and runoff driven soil detachment. The coefficient of surface cover 0.029 observed in our study was close to Dunne's (1978) result (0.023) and Liu's (2010) result (0.021) for perennial plantations, which revealed the possibility to further develop a uniform equation of *CP* factor generally applicable for perennial plantations. Some studies on plantation *CP* or *C* factor related it with tree canopy height or canopy cover and established *CP* factor equation. In this study, we calculated correlation between *CP* factor and other environmental variables like canopy cover, canopy height, soil cover, understory plant cover and root density in order to estimate the major driver of *CP* factor. The establishment of general equation for *CP* factor would necessitate additional measurement of soil loss under the control plot (bare soil), consideration of further parameters like coarse roots (diameter > 2mm) in deep soil layers, and more data on soil erosion for plantations sampled with smaller time interval (e.g. biannually).

2.4.3 Sustainable management in rubber plantations

Different trends in surface cover change under growing rubber explained the erosion dynamic in rubber plantations. The weaker competition for light and water ensured dense undergrowth in the young plantation. With the start of rainy season, undergrowth increased plant surface cover up to 100% in July. Although herbicide (Glyphosate) application decreased understory plant cover to around 20%, it consequently raised litter cover, and as a result the surface cover still kept at high level (80%). Glyphosate was absorbed by understory foliage and influenced only actively growing plants but not roots or seeds. Therefore, litter cover together with fast undergrowth recovery since October provides a surface cover of around 100% (Figure 2.5). In contrary, strong competition for light and water restrained growth of understory vegetation in mid-age and old rubber, making litter as a major contributor to surface cover. High decomposition rate in rainy season (Jin et al., 2015) steadily reduced the litter cover. Sparse undergrowth supplied neither high understory plant cover nor enough litter to offset herbicide effects. As a result, mid-age rubber presented highest risk to erosion. Both lower decomposition rate due to higher C/N ratio in old rubber litter (Jin et al., 2015) and higher tolerance of undergrowth to the glyphosate contributed to surface cover stabilization (Figure 2.5) and, thus, reduced soil loss in the old rubber plantations.

Despite of moderate erosion levels, soil loss in mid-age rubber exceeded the limit of 2 Mg ha^{-1} hence revealing potential for water pollution (Jürgens and Fander, 1993). Based on our plot level study, we concluded that the fast decrease of surface cover in rainy season contributed to the high risk of erosion in the mid-age rubber. Therefore, increasing and stabilizing of surface cover by reducing herbicide application or intercropping could be an alternative to control erosion in steep areas or during extreme events (Liu et al., 2016). Intercropping can bring additional economic income, which is especially important during the no (young phase) or low (old phase) latex production period. Intercropping improves soil fertility and increases surface cover (Zhang et al., 2007). Soils under rubber-tea intercropping has been proven to sequester more atmospheric CO_2 in comparison with those under rubber monoculture (Zhang et al., 2007). At our study site, only maize intercropping was implemented during rubber immature stage. Intercropping in mid-age rubber is rare because the closed canopy limited annual crop growth. Therefore, more studies are still necessary to recommend potential species and proper management of intercropping for practical application (Langenberger et al., 2016). Weed control is another effective and labor saving measure to conserve soil better in rubber plantations. Liu et al (2016) measured soil erosion under different herbicide application scheme and found that presence of undergrowth reduced runoff as well as soil loss. Reduction of herbicide application frequency by once per year at the end of dry season was recommended as an optimal weed management. Considering the whole lifespan of a rubber plantation, reduced soil loss during older phases may offset high erosion in young and mid-age rubber. Therefore, expanding rotation length of rubber plantation could be another measure to improve soil conservation.

2.5 Conclusion

Soil loss and runoff production varied depending on rubber plantation age. Annual *CP* factor of the rubber plantation was in the range of 0.005 - 0.03. It also varied depending on plantation age, having, however, similar values within the same growth phase of rubber. The change of surface cover explained the monthly and long-term dynamics of the *CP* factor well in this study. Dense undergrowth during open canopy period can protect soil under young plantations while high and stable litter cover strongly reduced erosive potential during old rubber phase. Inhibition of undergrowth and rapidly decomposing litter made mid-age plantations the most vulnerable phase to erosion. Therefore, better management maintaining high soil cover (e.g. minimizing of weeding control or intercropping) should be taken to improve soil conservation function of rubber plantations.

Chapter 3 Impact of herbicide application on soil erosion in a rubber plantation of Southwest China[b]

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[b] This chapter has been reprinted from:
Liu, H., Blagodatsky, S., Giese, M., Liu, F., Xu, J., Cadisch, G. (2016). Impact of herbicide application on soil erosion in a rubber plantation of Southwest China. Catena 145: 180-192
(DOI: org/10.1016/j.catena.2016.06.007). With permission from 2016 Elsevier Ltd.
The original publication is available at:
https://www.sciencedirect.com/science/article/pii/S0341816216302120

Abstract

Rubber plantations are strongly increasing in Xishuangbanna, Southwest China. Herbicide applications controlling the undergrowth may increase erosion potential and carbon export by decreasing plant and litter cover. Quantitative evaluation of the erodibility of rubber systems and the impact of herbicides has not been studied. This study aimed at assessing the impact of herbicide application on soil loss and the induced carbon export in a rubber plantation. Runoff, sediment yield, and total organic carbon (TOC) content in sediments were measured under natural rainfall for one year in a 12-year old rubber plantation subjected to three different herbicide treatments: i) standard application twice per year practiced by the majority of farmers (Hs); (ii) no application to maintain a high understory plant cover (H-); and (iii) bimonthly application (adopted by some farmers) in order to largely avoid understory plant cover (H+). The infiltration rate under different treatments was measured with a rainfall simulator. Monthly measurements of fine root density using soil coring, surface cover, and understory plant cover making photography were carried out. The highest soil and TOC in sediments losses (425 g m^{-2}, 15 g C m^{-2} respectively) were observed under H+ treatment, while under H- treatment they were strongly reduced (50 g m^{-2} and 2 g C m^{-2} respectively). Compared to Hs, H+ increased soil and sediment TOC loss by 34 and 52%, while H- reduced soil and TOC loss, both by 82%. Notably, H- presented high conservation efficiency, reducing sediment yields by 86% for highly erosive rainfall events. The cover and management (*C*) factor and support practice factor (*P*) are essential components of the common Universal Soil Loss Equation (USLE) model. We combined the *C* and *P* factors into a single value (*CP*) and, for the first time, derived estimations of annual *CP* values for a rubber plantation (0.005–0.04) using our data. The dynamic change of the *CP* factor of plantations during the rainy season was quantified by relating relative soil loss to changes in understory plant cover (PC), which can be expressed as $CP = 0.04e^{-0.028PC}$ ($R^2 = 0.88$, $P < 0.0001$). Understory plant cover as affected by

herbicide application was thus a key factor controlling the soil loss of established rubber systems. This suggests options to improve the soil conservation and biodiversity through reduced herbicide management.

Keywords: Erosion, herbicide application, rubber plantation, cover and management factor, USLE

3.1 Introduction

High soil erosion threatens long-term soil fertility and consequentially sustainable land use (Battany and Grismer, 2000). Water erosion, as one of the most pervasive soil degradation processes, affects an area of approximately 1100 million hectares annually worldwide (Lal, 2008). Water erosion comprises splash effects caused by raindrops, runoff generation, and soil particle transportation and sedimentation (Mohammad and Adam, 2010). Vegetation may reduce the soil erosion rate significantly (Chen et al., 2004; Greene et al., 1994) by intercepting raindrops, with parts of the rain forming the stem flow while another portion eventually evaporates from the leaf canopy. Stem flow, which indirectly leads rain water onto the soil surface, reduces its kinetic energy for soil particle detachment and thereby the erosion potential (Greene et al., 1994; Nunes et al., 2011). Plant roots penetrate the soil and leave macro-pores that increase soil porosity and infiltration capacity, thereby decreasing surface runoff and erosion (Amezketa, 1999). Additionally, plant roots can improve soil aggregate stability by enmeshing fine particles into stable macro-aggregates using root secretions/exudates, leading to better resistance to the impact of raindrops.

Land use strongly affects runoff production and sediment yield (Mohammad and Adam, 2010). Natural vegetation especially was found to be most effective in reducing erosion (Chen et al., 2004), while land use change in tropical highlands in Southeast Asia greatly increased soil degradation (Valentin et al., 2008). Recently, rubber plantations have expanded by 1,000,000 ha to encompass areas that were not conventionally planted in countries such as China, Laos, and Thailand (Fox and Castella, 2013). The para rubber tree (*Hevea brasiliensis*) was introduced to Xishuangbanna, South China, during the last decades. From 1992 to 2010, the area covered by rubber plantations increased by almost 340,000 ha (a gain of 400%) (Xu et al., 2014). This large-scale land conversion from rainforests to rubber plantations resulted in biodiversity loss and environmental degradation (Li et al., 2010). In particular, most rubber in Xishuangbanna has been planted on steep slopes (more than 25°) with high erosive potential. Wu et al. (2001) reported that higher soil water repellency by increased soil compaction in rubber plantations could potentially increase runoff by a factor of three and thereby increase soil loss by a factor of 45.

Rubber plantation management (land terracing, removal of understory vegetation, etc.) may further influence the soil erosion potential of this monocrop system. Terraces can alleviate soil loss in rubber plantations (Cha et al., 2005), while poorly designed and managed terraces may be ineffective in controlling surface erosion and even contribute to land sliding (Sidles et al., 2006). Intercropping can greatly improve soil conservation and is normally adopted in young plantations (Rodrigo et al., 2005; Ulahannaan et al., 2014). Herbicide application is a common treatment to reduce understory vegetation after plantation establishment in order to facilitate access to rubber tree trunks for tapping. By eliminating protective understory plant cover and rooting systems, this management practice is most likely a key activity affecting sediment yield and runoff production in rubber plantations that has not yet been studied in this area.

The quantitative evaluation of vegetation type and management of soil erosion is an important factor considered in various soil loss prediction models. The most widely used empirical model, the Universal Soil Loss Equation (USLE) first proposed by Wischmeier and Smith (1978), introduced a cover and management (*C*) factor to calculate the impact of vegetation on soil loss compared to bare soils. This empirical model considers five major factors affecting sediment yield: rainfall (*R*), soil properties (*K*), topographic characteristics (*L, S*), cover and management (*C*), and support practice (*P*), where the *C* factor is the ratio of the soil loss from a vegetated area compared to the soil loss from an identical continuously tilled fallow area (Kinnell, 2010). This concept has been widely adopted in other erosion and hydrological models including empirical and physically based models like RUSLE (Revised Universal Soil Loss Equation), ANSWERS (Areal Non-point Source Watershed Environment Response Simulation), WEPP (Water Erosion Prediction Project), and SEMMED (Soil Erosion Model for Mediterranean regions) (Jong et al., 1999). Major efforts have been made to estimate the *C* value for annual crops (Grabriels et al., 2003; Schönbrodt et al., 2010; Wischmeier and Smith, 1978). However, there have been few studies on *C* value estimations for woodland systems or plantations (Özhan et al., 2005, Kitahara et al., 2000). When applied to a forest system, *C* and *P* factors were in some cases combined together as the *CP* factor. Nine subfactors of *CP* were identified by Dissmeyer and Foster (1980) to better predict erosion on forest land. Soil cover and fine roots are two subfactors that are likely to be affected by herbicide management and that further influence erosion in rubber plantations. Identifying the impact on the *CP* factor of the rubber system can help in understanding the cause and effect relationships between management practices and erosion.

Carbon loss through water erosion is another concern that can cause soil degradation and affect carbon dynamics and greenhouse gas emissions (Wang et al., 2014). Land use conversion from forests to rubber resulted in losses of soil carbon stocks of 37 Mg C ha^{-1}, which were attributed to soil disturbances during site preparation, terrace construction, and sparse vegetation cover (De Blecourt et al., 2013). Topsoil carbon stocks declined exponentially over the years since rubber plantations were converted from forest and then reached a steady state after around 20 years (De Blecourt et al., 2013). However, the lack of data on carbon loss through erosion makes it difficult to determine whether soil carbon losses in established rubber plantations are mainly caused by erosion or soil organic matter decomposition and how management influences the soil carbon stock. Häring et al. (2013) demonstrated that uncertainties in the soil erosion rate were considered to have the greatest impact on the misrepresentation of soil organic carbon (SOC) dynamics on steep slopes in tropical ecosystems.

The absence of existing estimations of a *CP* value for rubber systems makes it difficult to apply models like USLE or RUSLE for soil loss in rubber plantations and to further evaluate their impact on ecosystem functions. Studying the impact of understory vegetation control (herbicide application) on soil as well as carbon loss and estimating its contribution to anti-erosive effectiveness in rubber plantations can justify effective forest management options and improve soil loss model predictions in plantation (woodland) systems. Therefore, we conducted a natural runoff experiment with different herbicide treatments in a 12-year old rubber plantation in 2014. Thus, the main objectives of this study were 1) to evaluate soil erosion and carbon loss under different herbicide application frequencies; 2) to identify the major factors (understory plant cover, surface cover, and fine root density) that are affected by herbicide treatments and therefore influence soil erosion processes; and 3) to estimate the combined factor (*CP*) of cover and management (*C*) and support practice (*P*) in the USLE model for rubber plantations.

3.2 Materials and Methods

3.2.1 Study site

The study site (22°17' N, 100°65' W, 764 m asl) is located in the Nanbanhe Watershed National Nature Reserve (NRWNNR) in Xishuangbanna, Yunnan Province, Southwest China. The elevation decreases from northwest to southeast with the highest elevation of 2304 m and the lowest point at 539 m. The annual precipitation is 1100–1600 mm and the mean annual temperature is 18–22 °C. The region has a typical monsoon climate characterized by a distinct rainy season from May to October and a dry season from November to April. Sixty to ninety per cent of the precipitation is distributed during the rainy season. The soil in the study site is a silt clay loam, Acrisol (Table 3.1). High contents of silt and low permeability have made the soil more erodible, while a relatively high organic matter content can alleviate the erodibility.

Table 3.1 Physical and hydraulic properties of top 5 cm soil for all treatments with high intensity (H+), standard (Hs), and no herbicide (H-) application *.

	Soil properties before treatments	Soil properties after implementing treatments			
		H+	Hs	H-	ANOVA
Physical properties					
Sand (%)	7.2±0.7	7.2±0.2	7.1±0.4	7.1±0.3	n.s.**
Silt (%)	56.8±1.9	55.3±2.7	56.3±2.8	55.3±2.2	n.s.
Clay (%)	36.0±2.6	37.5±2.8	36.5±3.2	37.6±2.1	n.s.
OM (%)	5.2±0.9	5.4±0.5	5.4±0.4	5.5±0.4	n.s.
Bulk density (g/cm^3)	1.1±0.04	1.1±0.05	1.1±0.04	1.1±0.02	n.s.
Hydraulic properties					
Infiltration rate (mm/h)	-	17.5±0.1	19.6±0.5	24.8±0.4	P<0.001

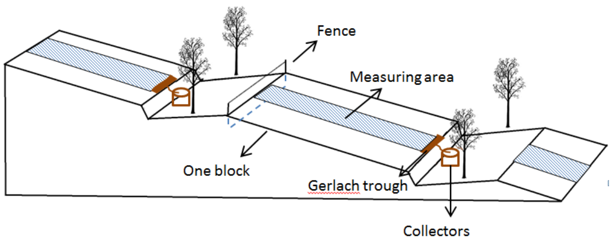
* n=9

**n.s.: no significant (p>0.05) difference among treatments

Rubber has been introduced to NRWNNR since the 1970s and expanded to cover 28.6 km^2 in 2013. Over 85% of established plantations already have a closed canopy and a rotation time that is generally about 25 years. Hence, a representative established rubber plantation of 12 years (12Y) with a canopy radius of 2.6 m was selected for the current study. The tree density in the selected plantation was 600 trees per hectare, which is within the range of the usual density of 450–600 trees per hectare in the area (Yang, 2011). Rubber trees are planted in rows on terraces located on a 29° slope with a tree space of 1–2 m and distance between two adjacent planting terraces of 5–5.5 m. The terraces were built 12 years ago before the tree plantation. The most common practice among local farmers is the application of herbicide twice a year in the mid-rainy season and mid-dry season, respectively, using 10 kg ha^{-1} of 10% glyphosate.

a

b



c

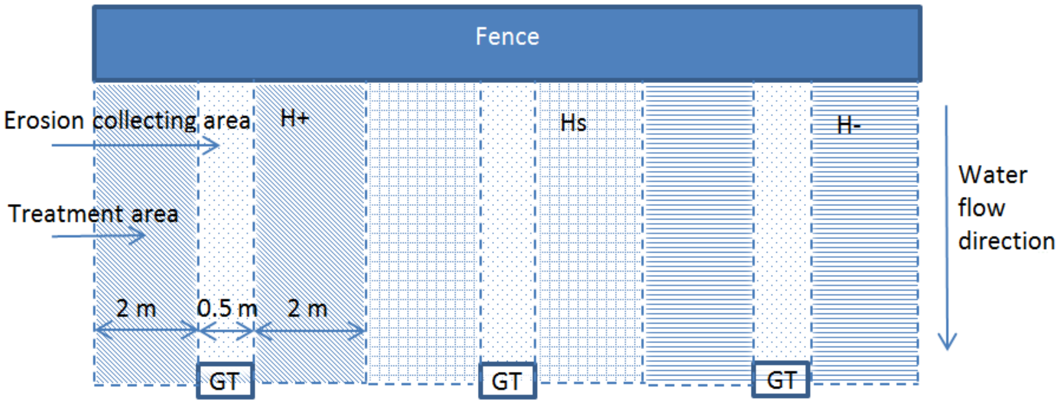


Figure 3.1 (a) Gerlach trough installed in rubber plantation. Slope between terraces was artificially arranged as straight in plane before planting the rubber trees. Depressions or rills were rarely observed on the slope. (b) Sketch of plots overview. The measuring plots were located on the slope area between terraces. (c) Sketch map of experiment layout in one block. Buffer area with 2 m * 5 m was left to both sides of erosion collection area to avoid inter disturbance from treatments in one block. GT = Gerlach trough; H+: bimonthly herbicide application; Hs: standard herbicide application – twice per year; H-: no herbicide application.

3.2.2 Experimental layout

In order to assess the impact of different management options on runoff and erosion, three herbicide treatments were implemented in 2014: (i) herbicide application practiced by the majority of farmers, namely two times per year in mid-February 2014 and late July 2014, respectively, using 10 kg ha^{-1} of 10% glyphosate (Hs); (ii) no herbicide application to maintain a high understory plant cover (H-); (iii) bimonthly herbicide application using the same amount as in Hs each time (as common practiced by farmers) in order to obtain a low or absent understory plant cover (H+). For treatments with herbicide application (Hs and H+), litter derived from understory vegetation was not removed but remained in the plots as is commonly practiced by farmers.

Herbicide was mostly applied on the slope area between terraces, where depressions or rills were rarely observed (Figure 3.1a). This study focused on the impact of herbicide management, and therefore we chose the inter-terrace slope area as the study unit (Figure 3.1b). The experiment was a complete randomized block design with three replicates. In total, nine open Gerlach troughs of 0.5 m width were installed in the 12Y rubber plantation on the same slope gradient. Local farmers had artificially adjusted the slope to create a straight plane during terrace construction. Therefore, we considered that the unbounded plots method for runoff and soil loss collection was applicable for our study. The variability in soil losses measured by different methods or using different plot sizes (Boix-Fayos et al., 2006; Sonneveld et al., 2005) should not be a problem in monoculture rubber plantations with a uniform distribution of soil properties and surface cover. At the top of the slope, an impermeable metal fence was inserted 10 cm into the ground to isolate surface and subsurface run-in. The Gerlach troughs were placed at 5 m distance from the fence at the bottom of the slope. The contributing area was estimated to be equal to the width of the Gerlach trough (0.5 m) multiplied by the distance between the trough and isolated metal fence (5 m), namely 2.5 m^2. Estimation of the plot area was based on the assumption that loss of runoff or sediment from the defined area can be balanced by inputs from adjacent areas considering the slope is straight and plane (Morgan, 2005). Gerlach troughs were connected to collection vessels via a plastic pipe with a diameter of 5 cm, which delivered runoff and sediment to a series of two 200 L containers. The second container collected excess runoff from the first in case of storm events. A buffer area of 2 m * 5 m was left on both sides of the erosion collection area to avoid inter-disturbance from treatments in one block (Figure 3.1c). In order to identify potential uncertainty arising from the application of the unbounded plots method, runoff and sediment yields under Hs were also collected by bounded plots with an area of 3 m * 5 m, which were built in 2013 at the same site. Soil losses (asymptotic p-value = 0.26 > 0.05), runoff (asymptotic p-value = 0.73 > 0.05), and sediment concentration (asymptotic p-value = 0.14 > 0.05) estimated by the two methods do not differ significantly, thus ensuring the applicability of the unbounded plots method at our study site (Figure 3.2).

Data collection:

Rainfall: A tipping bucket rain gauge (Campbell Scientific TB4) was installed in open ground next to the experimental site. The precipitation amount was recorded every two minutes and summarized every 15 minutes.

Runoff and sediment collection: Runoff and sediments were collected after each event that produced erosion under natural rainfall in 2014. Runoff was determined by measuring the water level in the collectors and translating it into volume with geometric equations. Sediments were firstly collected from deposits remaining in troughs and air-dried. After sieving with a mesh size of 2 mm, a subsample was stored for carbon determination. Remaining sediments were oven-dried at a temperature of 105 °C until constant weight was achieved. As was determined in a preliminary experiment in 2013, the greatest part of the eroded soil remained as sludge in the trough due to low runoff production. Therefore, after collecting deposits, the troughs were washed by collected runoff so that attached particles flowed into the container. Solids in runoff containers were easy to re-suspend by mixing due to the low sand fraction of soil in the study site. Sediment concentration in the runoff container was low (< 1.5 g L^{-1}). Therefore, the solids collected in runoff containers were determined by taking a sub-sample of 500 mL from each tank after vigorous and homogeneous mixing (Lang, 1992). The sub-sample was then filtered through Whatman filter paper (d = 125 mm, 11 µm, 10.5 s/100 mL/sq inch flow rate) and oven-dried at 105 °C.

Then the total weight of the sediments in the runoff container and those collected from the Gerlach troughs was summed and divided by the plot area (2.5 m^2) to obtain the soil loss in grams per square meter. Subsamples of sediments for each event were combined together every month and organic carbon content was determined with a C-N analyzer (Elementar Analysensysteme, Germany).

Soil properties: Samples of the top 5 cm of soil under each treatment were collected before and after measuring natural sediment runoff in 2014 to evaluate whether the understory vegetation can decrease soil erodibility. Soil samples were air-dried and passed through a sieve with a mesh size of 2 mm. Inorganic carbon was eliminated by concentrated HCl before sample analysis with a Vario TOC Cube (Elementar Analysen Systeme GmbH) analyzer by high-temperature catalytic oxidation up to 1200 °C and the emitted CO_2 was measured. Soil texture was determined by the pipette settling method (LY/T 1225-1999) and classified based on the United States Department of Agriculture (USDA) standard.

Steady infiltration rate: Two out of three plots under each treatment were randomly selected for repetition of the measurement of the steady infiltration rate with a portable rainfall simulator (Nanjing Nanning Electronic Company, NLJY-10) at a rainfall intensity of 40 mm h^{-1}. This measurement was conducted on 1 November, at the end of the herbicide treatment experiment, to avoid disturbing the soil within the plots. At measurement time, the soil was close to field capacity (moisture of around 40%) after a low intensity rainfall the previous day (10 mm). The simulator was supported by a metal frame and adjusted to 2 m above the ground. Runoff rates were measured from a plot of 2 m^2 (1 m * 2 m) for two hours. The infiltration rate was calculated by subtracting runoff from rainfall (Holden and Burt, 2002).

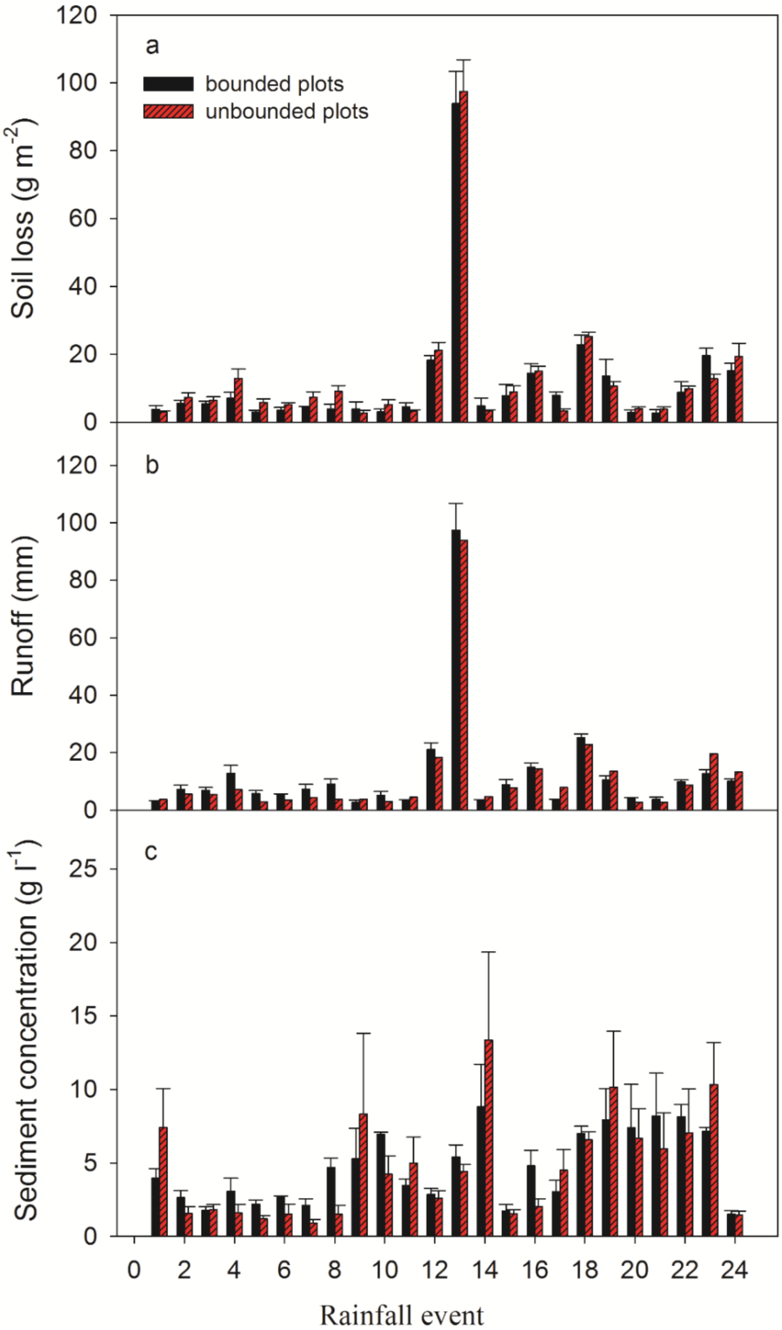


Figure 3.2 Comparison between (a) soil losses, (b) runoff and (c) sediment concentration collected from bounded and unbounded plots under standard management in 2014. Kruskal-Wallis test was used to test if there was significant difference between two methods. Asymptotic significance Kruskal-Wallis test for soil loss was asym. sig. = 0.26 > 0.05; for runoff was asym. sig. = 0.73 > 0.05; for sediment concentration was asym. sig. = 0.14 > 0.05, which means there is no significant difference between results from bounded and unbounded plots.

Surface cover and *understory plant cover* were measured monthly during the rainy season in 2014 by taking photographs from 2 m above the ground with a digital camera. Images were processed with 'SamplePoint' software (Booth et al., 2006) to identify the components of surface cover and to calculate the percentage of understory plant cover and remaining bare soil. The surface cover (percentage of area) was then calculated as 100 % minus the bare soil area (%).

Root sampling was performed every month with a soil auger (10 cm diameter) to 10 cm depth. Five root sampling sites were selected using the line transect method along the slope adjacent to the erosion plots. The soil samples were then washed in a 250-µm-aperture sieve and the roots were retained. Coarse roots with diameter > 2 mm were picked out and discarded, considering that 1) they mainly derived from rubber trees, which were not affected by herbicide, and 2) fine roots are generally more important for soil stabilization (Gyssels et al., 2005). Fine roots with diameter < 2 mm were further divided into diameter classes of < 1 mm and 1–2 mm. The roots of each group were categorized as live and dead roots based on color and elasticity (Sayerö & Haywood, 2006) and oven dried at 70 °C. Root density was used as an indicator for the occupation of soil by roots and was calculated using the weight of roots divided by the volume of soil auger.

3.2.3 Data analysis and statistics

This experiment was conducted under natural conditions, and therefore weather conditions, such as the variation in monthly rainfall amount and energy, were the major reasons explaining changes in sediment yields. In order to separate this effect from other factors (surface cover, understory plant cover, fine root density) mostly affected by herbicide treatments, we calculated the monthly *CP* values of the rubber system under different treatments and extrapolated the relation with the *CP* factor instead of soil loss.

Model selection for rainfall kinetic energy calculation: Yu and Rosewell (1996) recommended the model proposed by Brandt and Foster (1987) (Equation 3) to calculate rainfall kinetic energy (EI_{30}) for Australia's tropics, where the rainy season falls from November to April. As for China's tropics, where the rainy season lasts from May to October, Zhang et al. (2006) used the model by Yu and Rosewell (1996) to calculate monthly and annual rainfall kinetic energy, however, suitability was not tested. Therefore, we assessed the suitability of three widely used models to calculate EI_{30} for our local condition.

Model 1. Wischmeier and Smith (1958) first proposed an equation to characterize single rainfall events by rainfall intensity of the single event (i_r), which was widely accepted.

$$EI_{30} = 0.119 + 0.0873\log_{10} i_r \tag{3.1}$$

Model 2. Brandt (1990), following Wischmeier and Smith (1958), proposed a logarithmic equation to calculate storm kinetic energy from the maximum rainfall intensity within 30 minutes (I_{30}).

$$EI_{30} = 210 + 89\log_{10} I_{30} \tag{3.2}$$

Model 3. Brown and Foster (1987) proposed an approach to calculate rainfall erosivity that was adopted in the Revised Universal Soil Loss Equation (RUSLE) model and in the Rainfall Intensity Summarization Tool (RIST) by the United States Department of Agriculture (USDA).

$$EI_{30} = \left(\sum_{\mathrm{r}=1}^{\mathrm{n}} e_r v_r\right) I_{30} \tag{3.3}$$

where

$$e_{\mathrm{r}} = 0.29[1 - 0.72\exp(-0.05 i_r)] \tag{3.4}$$

In all models, EI_{30} is the rainfall erosivity index of a single event (MJ mm ha^{-1} h^{-1}). I_{30} is the maximum rainfall intensity during a period of 30 minutes in the event (mm h^{-1}), e_r is the unit rainfall energy (MJ ha^{-1} mm^{-1}), v_r is the total rainfall volume (mm) of the single event, and i_r is the average rainfall intensity (mm h^{-1}) of the single event.

EI_{30} is assumed to be linear with respect to soil loss (Renard et al., 1997), and therefore we subsequently estimated correlations between sediment yields of each rainfall event and the corresponding kinetic energy calculated by different models. The model with the best fit was used further and the obtained EI_{30} values were summed up as monthly rainfall erosivity for use in the USLE model.

CP (cover, management, and support practice) factor calculation: The USLE equation is defined as:

$$A = R * K * L * S * C * P \tag{3.5}$$

where A = soil loss (Mg ha^{-1}), R = rainfall erosivity (MJ mm ha^{-1} h^{-1}), K = soil erodibility (t h MJ^{-1} mm^{-1}) (i.e. the soil loss per unit of erosivity for a standard condition of bare soil, 5° slope of 22 m length), L = slope length factor (dimensionless), S = slope steepness factor (dimensionless), C = the cover and management factor (dimensionless), and P = the support practice factor (dimensionless).

In the plantation system, *P* was mostly treated either as a subfactor of *C* (Dissmeyer & Foster, 1980) or as a new combined *CP* factor (Özhan et al., 2005; Brooks et al., 1996; Hurni, 1982). In our study site, terrace management is the major conservation practice adopted in the rubber plantation. However, in order to focus on the effects of herbicide, deposition on the terrace platform was not included in the measurement. Therefore, instead of calculating the *P* factor separately, we followed the method of treating *C* and *P* together as *CP*.

The USLE model was initially designed to predict long-term average annual soil losses in a standard plot (5 m * 22.1 m) with a gentle slope area (9%). Then it was applied to short-term time-scales, such as monthly or event time-scales, with further development. In our study, we applied the USLE model at annual and monthly scale to identify the impact of herbicide on a relatively long period (one year) and to calculate the contribution of understory plant cover to the combined *CP* factor. So, the *CP* factor for rubber was calculated using measured annual or monthly sediment yields along with other estimated factors (Brooks et al., 1996; Wischmeier and Smith, 1978), namely $CP = A/(R * K * L * S)$.

The *R* factor was calculated by summing up the annual or monthly rainfall kinetic energy (EI_{30}) for two scales respectively. Analysis of rainfall kinetic energy showed that calculation according to Zhang et al. (2006) was appropriate for China's tropical area. Since the calculation of *R* in this study was based on one-year data, we compared it with mid-term and long-term calculated *R* factors (annually and monthly based) to verify the representativeness of the climate condition during our monitoring year (2014). Mid-term *R* was calculated based on daily rainfall from Jinghong airport (2001-2013) using the same equation as Zhang et al. (2006):

$$E_t = 0.697 \sum_{k=1}^{N} P_k^{1.5} \qquad P_k > 12.7 \text{ mm} \tag{3.6}$$

where E_t is the *t* month rainfall erosivity, and P_k is the daily rainfall amount.

The long-term *R* factor (annually and monthly based) was directly cited from Zhang et al. (2006), who calculated it directly from rainfall data of Jinghong Airport (1960–2000). Since the *CP* factor was a result of the crop/plantation condition (stage, cover) subjected to erosive rainfall, long-term *CP* values (annual and monthly) were calculated from mean *R* values (1960–2013) to evaluate the representativeness of short-term (2014) derived *CP* values. *K* was calculated by Equations (6) and (7), proposed by Wang (2013), which were improved based on the often used soil erodibility calculation model, the Revised Universal Soil Loss Equation (RUSLE) (Renard et al., 1997), to better estimate red soil erodibility in South China:

$$K\text{=}0.0364\text{-}0.0013\left[\ln\left(\frac{OM}{D_g}\right)\text{-}5.6706\right]^2\text{-}0.015\times\exp\left[\text{-}28.9589\left(\lg\left(D_g\right)\text{+}1.827\right)^2\right] \tag{3.7}$$

where

$$D_g = \exp(0.01\times \sum f_i \ln(m_i)) \tag{3.8}$$

OM = soil organic matter content (%), $D_g =$ geometric mean diameter of the soil particles (mm), f_i = weight percentage of the silt, clay, and sand fraction (%), and m_i = arithmetic mean of the size limits for silt, clay, and sand (mm).

The original expression for L in the USLE model (Wischmeier & Smith, 1978) is:

$$L=\left(\frac{\lambda}{22.1}\right)^{\mathrm{m}} \tag{3.9}$$

where λ is the slope length horizontal projection (m), and m ranges from 0.2 to 0.5 for different slope gradients. Previous studies recommended a slope length exponent, m, of 0.5 (Akeson & Singer, 1984; McCool et al., 1993; Liu et al., 2000) for steep slopes (45–60%), which is based on data from natural runoff plots with slope length varying from 2.4 to 40 m. Therefore, we took the exponent, m, as 0.5.

The slope steepness factor (*S*) in the USLE model was originally derived from gentle slope areas (3–18%). Different equations were then extrapolated for steep areas of up to 56% (Liu et al., 1994). Previous studies indicated the impact of soil properties, slope length, and the appearance of rill erosion on the steepness factor (*S*). Therefore, we took the equation developed by Singer and Blackard (1982), which was derived from soil loss data on steep (up to 50%) small inter-rill areas (0.6 m * 1.2 m) with silt-clay loam soil:

$$S = -8.43(sin\theta)^2 + 9.37sin\theta + 0.22 \tag{3.10}$$

where θ is steepness (°).

Quantification of the relationship between understory vegetation and soil loss: The significance of differences in sediment yields and runoff between the three treatments was tested by one-way ANOVA. Surface cover, understory plant cover, and fine root density could be highly correlated with each other, and therefore partial correlation analysis with SPSS Statistics version 22.0 was applied to evaluate the separate contributions of surface cover, plant cover, and fine root density to soil loss and the *CP* factor. Correlations and ANOVA analyses were performed with the statistical package R version 3.1.3 (http://www.r-project.org/).

3.3 Results

3.3.1 Rainfall, erosion, and carbon export

The correlations between monitored soil loss and calculated rainfall erosivity by the three tested models were 0.49 ($p < 0.05$, Model 1, Wischmeier 1958), 0.70 ($p < 0.01$, Model 2, Brandt 1990), and 0.98 ($p < 0.01$, Model 3, Brown and Foster 1987), respectively. Therefore, we applied the model of Brown and Foster (1987) for the calculation of rainfall erosivity in this study. During 2014, a total of 24 rainfall events generated overland flow and caused soil erosion, with amounts ranging from 10 to 88 mm with respective estimated erosivities of 42 to 1075 MJ mm $ha^{-1}h^{-1}$; hence the total cumulative rainfall erosivity was 3694 MJ mm ha^{-1} h^{-1} in 2014. Rainfall volume and erosivity had similar monthly distributions mostly concentrated in the middle of the rainy season in 2014 (Figure 3.3). Over 90% of erosive events were recorded during July to September. Very highly erosive events ($EI_{30} > 300$) were only observed twice in 2014, on 14 July and 17 August. Annual precipitation (1128 mm) and annual rainfall erosivity (3694 MJ mm ha^{-1} h^{-1}) in this study were both close to the values obtained over the middle-term (2001–2013): 1260 ± 185 mm and 3468 ± 769 MJ mm ha^{-1} h^{-1}, respectively, and long-term (1960–2000): 1100 mm and 3500 MJ mm ha^{-1} h^{-1}, respectively (Zhang et al, 2006). Differences were observed for monthly distribution. In 2014, rainfall erosivity was mostly concentrated in July to September and the highest monthly erosivity was recorded in August. For longer periods, rainfall erosivity was distributed more evenly from May to September (Figure 3.3).

The highest generated runoffs were recorded as 21, 26, and 14 mm under Hs, H+, and H-, respectively, accompanied by a storm event on 17 August with precipitation of 88 mm (Figure 3.4b). The annual runoff coefficients (ratio of total runoff to precipitation) during the rainy season were 0.11 under Hs, 0.18 under H+, and 0.05 under H- treatments. Compared to Hs, H- decreased total runoff by 48 and 42% for runoff produced by storms (event with total rainfall volume > 50 mm), while H+ increased total runoff by 73 and 54% for runoff produced by storms. The minimum event producing runoff increased from 10 to 15 mm when there was no herbicide application (H-).

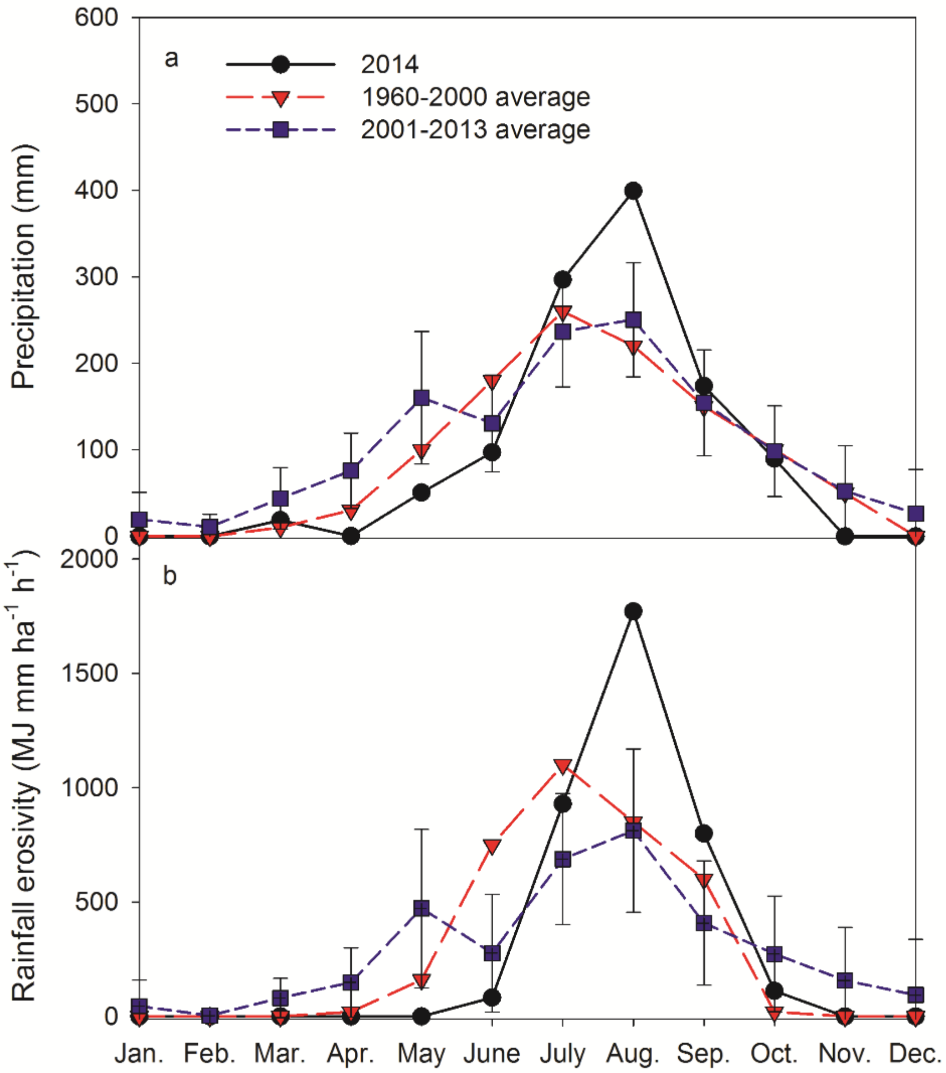


Figure 3.3 Comparison between a) monthly rainfall amount and b) monthly rainfall erosivity distributions of 2014, mid-term (2001-2013) and long-term (1960-2000) average value. Rainfall data in 2014 was monitored at the study site in the Nanbanhe River Watershed National Nature Reserve, Xishuangbanna, Yunnan Province, South-West China. Rainfall erosivity was calculated using the model of Browns and Foster (1987). Mid-term average value of rainfall amount and erosivity was calculated based on data from Jinghong airport (2001-2013). Long-term average value was from literature (Zhang et al., 2006) based on rainfall data from Jinghong airport 1960-2000. Mid-term and long-term rainfall erosivity were calculated using the model of Yu & Rosewell (1996).

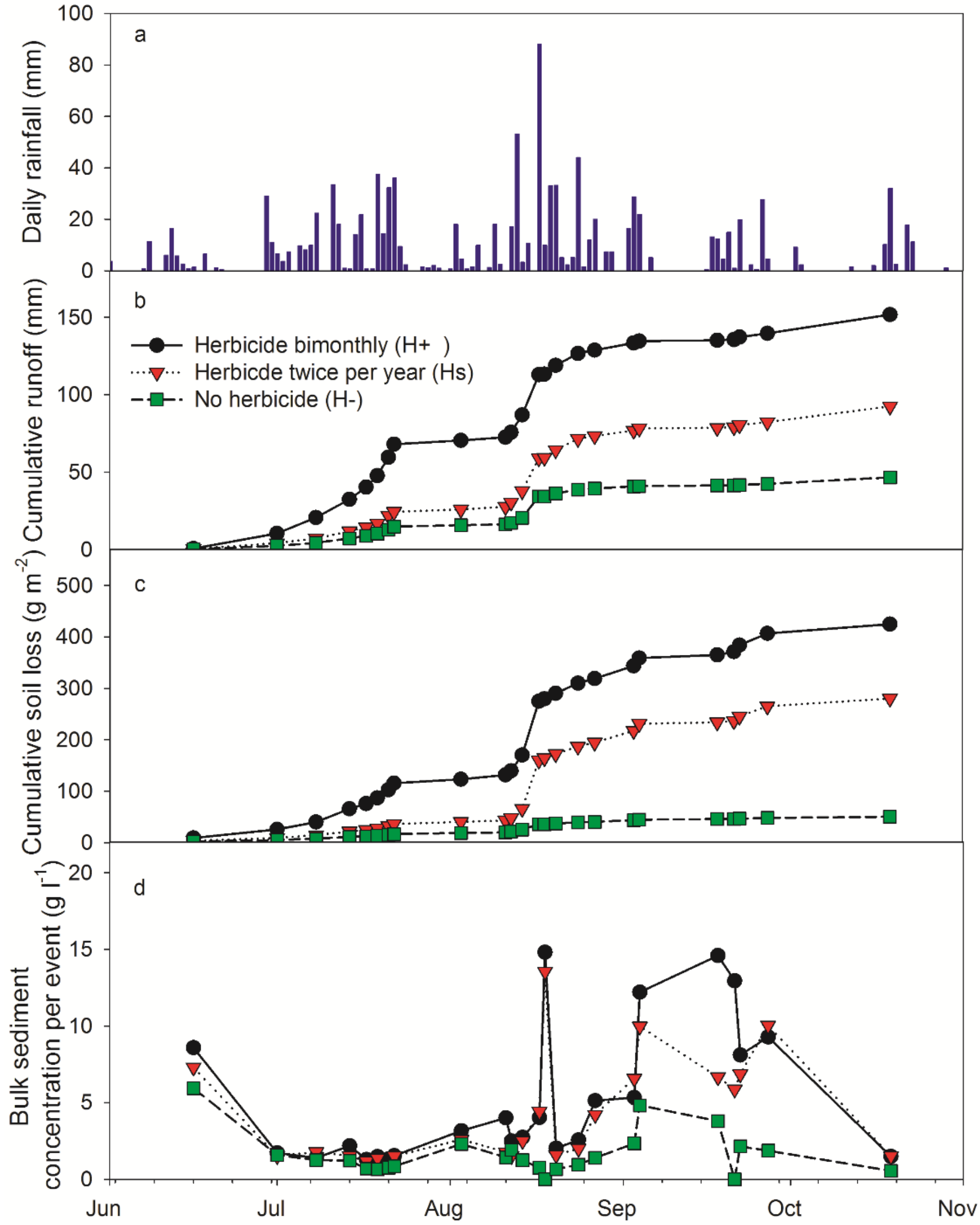


Figure 3.4 (a) Daily rainfall, (b) cumulative runoff, (c) cumulative soil loss with time and (d) bulk sediment concentration per event under different herbicide applications during the rainy season (June to October) in 2014.

The maximum sediment yield for a recorded single event was measured as 105, 94, and 10 g m^{-2} on 17 August with rainfall of 88 mm for the Hs, H+, and H- treatments, respectively (Figure 3.5). Cumulative soil losses in 2014 reached 280, 425, and 50 g m^{-2} for the Hs, H+, and H- treatments, respectively (Figure 3.4c). Compared to Hs, H+

increased soil loss by 52% and 29% for two highly erosive events (> 300 MJ mm ha^{-1} h^{-1}), while H- reduced soil loss by 82 and 86% for highly erosive events. Additionally, H- raised the minimum event inducing erosion from 42 to 63 MJ mm ha^{-1} h^{-1} compared to Hs. In general, sediment concentrations (defined as total soil loss / total runoff) were lower under H- treatment than under Hs and H+ (Figure 3.4d). Sediment concentrations under different treatments were similar in July but became remarkably different from August for treatments with (H+ and Hs) and without herbicide (H-). Sediment concentration strongly increased in the late rainy season (September) for plots with herbicide treatment (H+ and Hs) but remained at a low level during the whole rainy season with no herbicide (H-). The highest concentration was recorded as 15 g l^{-1} under H+ and Hs on 18 August with rainfall and erosivity of 10 mm and 42 MJ mm ha^{-1} h^{-1}, respectively. However, the same event caused no erosion under H- (Figure 3.4d). The total organic carbon export in eroded sediments was 15, 10, and 2 g m^{-2} under the H+, Hs, and H- treatments, respectively (Table 3.2). Compared to Hs, H+ increased carbon export by 52% and H- reduced it by 82%. Organic carbon concentrations in sediment were in the range of 33–37 mg g^{-1}, while the enrichment ratio was above 1 with a range of 1.1–1.2 over all cases. No significant differences in organic carbon concentration and enrichment ratios were observed under different herbicide treatments.

Table 3.2 Eroded soil organic carbon (OC) in different treatments with high intensity (H+), standard (Hs), and no herbicide (H-) application during 2014 rainy season.

	OC concentration in sediment (g C kg^{-1})			OC enrichment ratio *			Total OC eroded (g C m^{-2})		
	H+	Hs	H-	H+	Hs	H-	H+	Hs	H-
June	36.4[a]	35.3[a]	35.6[a]	1.2	1.2	1.2	0.3[a]	0.1[b]	0.1[a]
July	33.6[a]	33.2[a]	33.2[a]	1.1	1.1	1.1	3.6[a]	1.1[b]	0.5[c]
Aug.	34.2[a]	34.1[a]	33.1[a]	1.1	1.1	1.1	7.0[a]	5.4[b]	0.8[c]
Sept.	34.8[a]	34.7[a]	35.0[a]	1.1	1.1	1.2	3.1[a]	2.4[a]	0.3[b]
Oct.	35.2[a]	36.7[a]	36.6[a]	1.2	1.2	1.2	0.6[a]	0.6[a]	0.1[b]
Sum	-	-	-	-	-	-	14.6	9.6	1.8

* OC enrichment ratio = (OC concentration in sedimentation) / (OC concentration in soil)

Values with different letters within rows indicate significant ($p<0.05$) differences

3.3.2 Soil properties and plant dynamics

The soil in our experiment had high percentages of silt and clay and a low (< 10%) sand fraction (Table 3.1). There were no significant differences in particle size distribution for the 0–5 cm topsoil among different treatments. Plots with no herbicide application (H-) had a higher steady infiltration rate (24.8 mm h^{-1}) compared to Hs (19.6 mm h^{-1}) and H+ (17.5 mm h^{-1}).

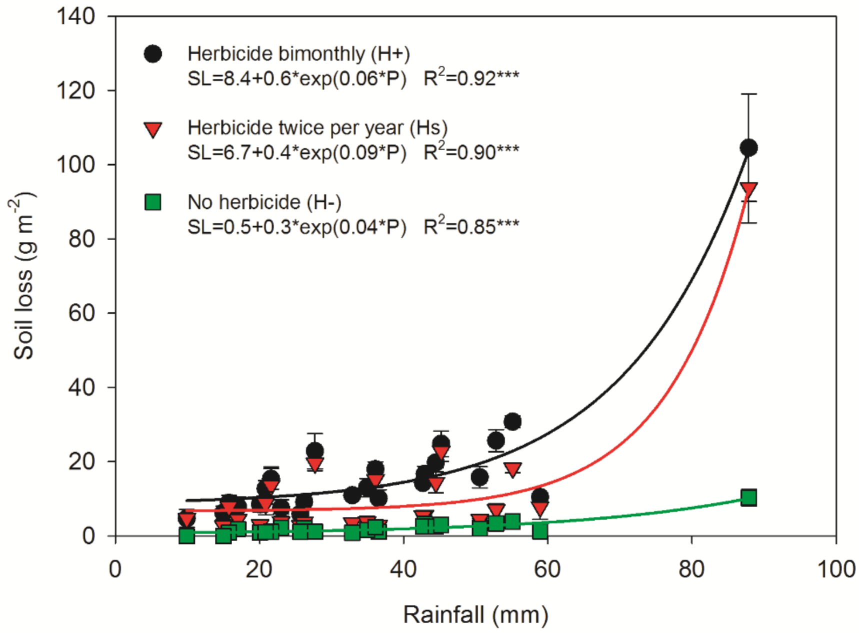


Figure 3.5 Relationship between precipitation and event-based soil loss under different herbicide treatments. P = Precipitation, SL = Soil Loss

No significant differences in surface cover were observed among the three treatments during the first three months. Surface cover, which in our study mainly included understory plant and litter cover, under H+ was maintained at over 80% from May to July, dropping to 27% in August, and remained below 30% until October (Figure 3.6). The change of surface cover under Hs declined slowly before August, at a rate of 8%. After herbicide application, the surface cover under Hs decreased constantly to below 50% at the end of the rainy season. Surface cover under H- remained high (over 75%) during the whole rainy season.

Understory plant cover was nearly completely suppressed under H+ during the whole rainy season (plant cover < 5%) (Figure 3.6). In contrast, under Hs, understory plant cover increased from May onwards and reached the highest cover (over 57%) in July. Following herbicide application under Hs at the end of July, plant cover dropped sharply to 15% in August and remained low until the end of the rainy season. The trend in plant cover development under H- was similar but more rapid compared to Hs at the beginning of the rainy season, increasing from May onwards, and subsequently remaining at a high level (over 70%) during the whole rainy season.

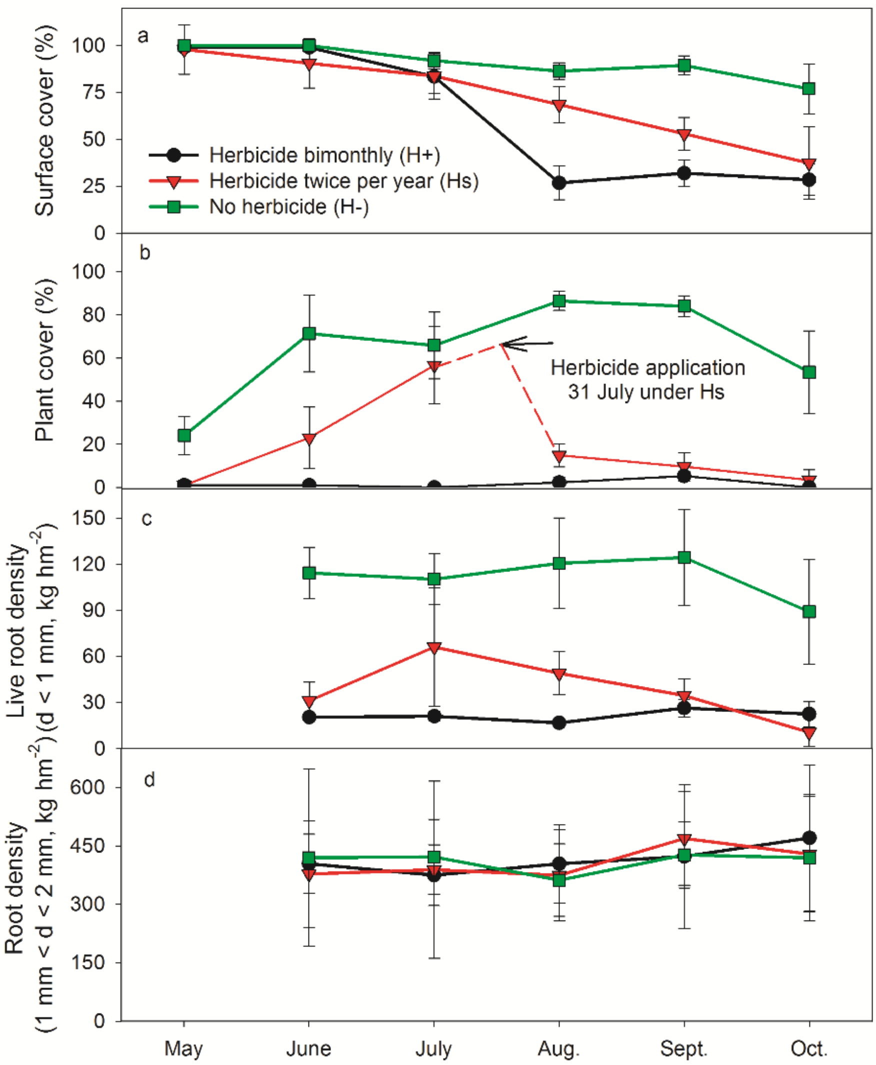


Figure 3.6 Temporal dynamics of (a) surface cover (understory plant and litter cover), (b) understory plant cover, (c) fine root density (d < 1mm) and (d) root density (1 mm < d < 2 mm) under different herbicide applications from June to October in 2014.

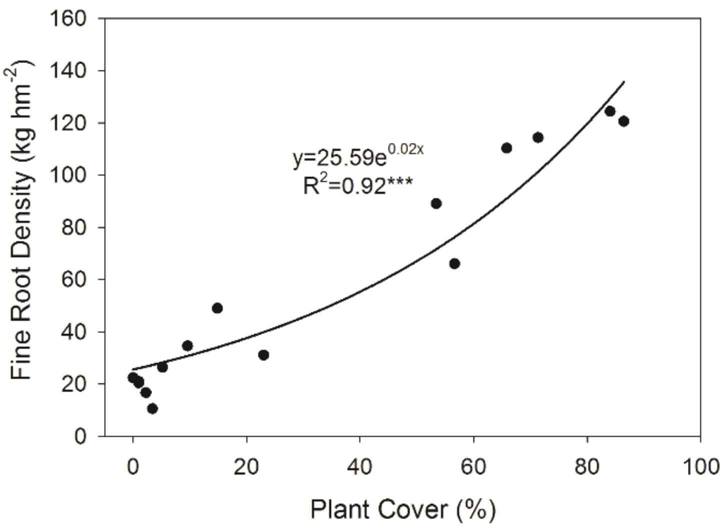


Figure 3.7 Relation between plant cover and fine root density (diameter < 1mm, 0-10 cm).

The total root density was in the range of 450–550 kg ha-1 but showed no significant differences between treatments. However, the density of fine living roots with diameter < 1 mm was considerably different among treatments. Fine living roots remained at low (17–26 kg ha^{-1}) density under H+ throughout the measurement period. A decline of live fine root density from 66 to 35 kg ha^{-1} was measured under Hs after the herbicide application at the end of July, whereas it remained high under H- during the whole rainy season (89–124 kg ha^{-1}). Plant cover and live root density (d < 1 mm) interdependencies were described well by an exponential relationship (Figure 3.7). Root density with diameters between 1 and 2 mm was not affected by herbicide application (Figure 3.6).

3.3.3 *CP* value depending on herbicide treatment, surface and plant cover, and root density

Annual *CP* values calculated using rainfall erosivity data in 2014 were 0.005 (under H-), 0.028 (Hs), and 0.04 (H+). They were close to those calculated using long-term data (1960–2013), which were 0.005 (under H-), 0.029 (Hs), and 0.045 (H+), respectively, while the monthly *CP* values for the two calculations differed widely (Figure 3.8). The annual *CP* value under Hs was lower than that under H+ and higher than that under H-.

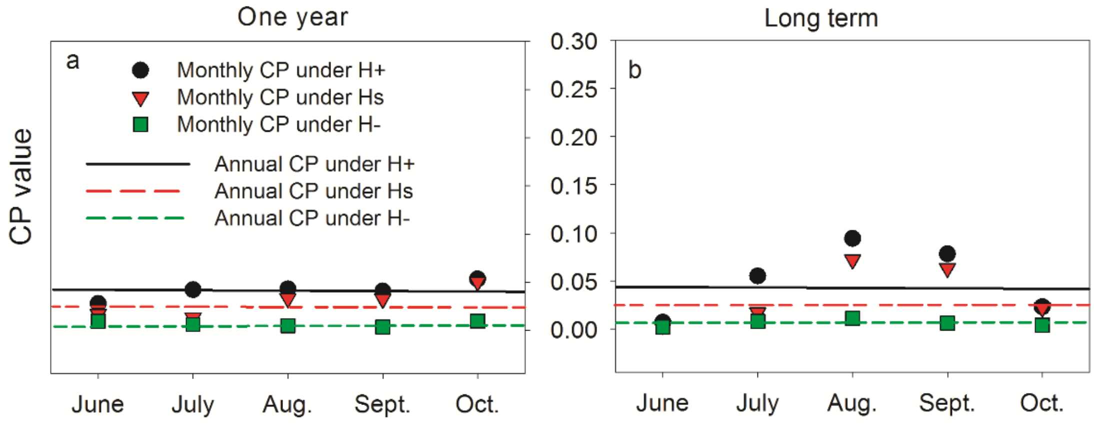


Figure 3.8 Value of annual and monthly *CP* factor calculated from a) one year (2014) rainfall data at study site and b) long term (1960-2013) rainfall data from Jinghong airport meteo-station located 22 km away from the study site. *CP* factor (dimensionless) is the combination of cover and management factor (*C*) and support practice factor (*P*) in the Universal Soil Loss Equation (USLE) model

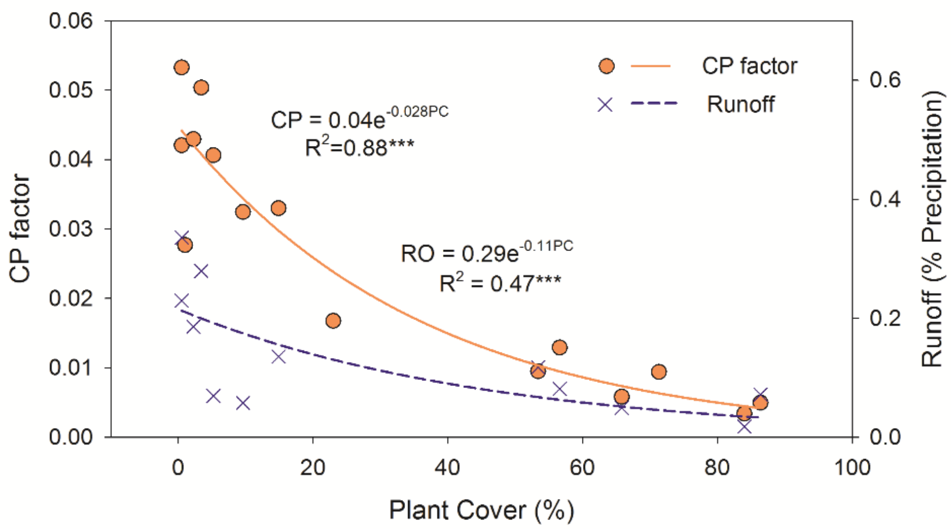


Figure 3.9 Relationship between average plant cover and *CP* value as well as runoff fraction of the total precipitation. *CP* factor (dimensionless) is the combination of cover and management factor (*C*) and support practice factor (*P*) in the Universal Soil Loss Equation (USLE) model

Soil loss showed significant positive correlations with rainfall erosivity and precipitation while a significant negative correlation was observed with root density < 2 mm (Table 3.3). Runoff was similarly regulated by precipitation and rainfall intensity.

The monthly CP value was highly correlated with plant cover and live fine root density (fRD) ($d < 1$ mm) (Table 3.3). The partial correlation between plant cover and C factor (–0.43) was higher than fRD (–0.19), and therefore plant cover was selected as an indicator to present the effects of understory vegetation on relative soil loss and runoff production. The relationship between plant cover and CP factor (Figure 3.9) was best described by:

$$CP = 0.04e^{-0.028PC} \tag{3.11}$$

where PC = plant cover (%).

3.4 Discussion

3.4.1 Soil loss under established rubber plantations

Soil loss measured under the rubber plantation (50–425 g m^{-2}) was similar to that reported by Wu et al. (2001) (269 g m^{-2}) but higher than that reported under forest (5 g m^{-2}) in Xishuangbanna (Li, 2001), measured under similar annual rainfall amounts and distributions. However, losses were much lower compared to annual agricultural systems in Southeast Asia, which showed values in the range of 890–17400 g m^{-2} (Valentin, 2008; Pansak et al., 2008; Tuan et al., 2014). Different soil losses among land uses could be partly explained by differences in soil organic matter content and variations in the spatial and temporal distribution of rainfall. The soil at the study site contained a high percentage of organic matter (around 5%) (Table 3.1) compared to intensively managed agricultural systems, which mostly have values below 4% (Loveland and Webb, 2003) but was lower than under tropical forest, which was in the rage of 5–12% (Blecourt et al., 2013; Guillaume et al., 2015). High soil organic matter content was likely to have increased aggregate stability and decreased soil erodibility (Duchicela et al., 2013). A significant soil carbon decrease (from 2.9 to 2.1%) after converting forest to rubber (De Blecourt et al., 2013) raises soil erodibility, while subsequent rapid reduction of the rate of decline in soil carbon (from 20% to 5%) after the establishment of the plantation guarantees lower soil erodibility compared to annual cropping systems. The lower carbon enrichment ratio (1.1–1.2) in sediments measured in our study compared to those under other land uses (1.9–5.5) (M-Mena et al., 2008) indicated that water erosion was not a major contributor to SOC loss after the establishment of rubber plantations.

The temporal distribution of rainfall contributes to high erosion in some areas. For example, Imeson (1990) reported that the main characteristics affecting the vulnerability of the Mediterranean area to erosion are intense rainfall after a very dry summer. However, in our study area, low precipitation after the dry season led to gradually increased soil moisture and, as a result, aggregate water stability could increase without soil losses (Ma et al., 2014a). Annual rainfall erosivity (3700 MJ mm ha^{-1} h^{-1}) in 2014 and for a long period (3500 MJ mm ha^{-1} h^{-1}) fell in the medium range based on Silva's (2004) classification. The highest kinetic energy of a single event (1075 MJ mm $ha^{-1}h^{-1}$) was lower than that in other areas in Southeast Asia, with events having values as high as 3000 MJ mm $ha^{-1}h^{-1}$ (Ma et al., 2014b; Tuan et al., 2014). The relatively low rainfall erosivity and low frequency of highly erosive rainfall (2 out of 24 events) partly explain the low erosion in our study.

3.4.2 *CP* value of established rubber plantation and its sensitivity to understory plant cover

Cover and management (*C*) and support practice (*P*) factors of the USLE were combined into a single factor (*CP*) and computed for a 12-year-old rubber plantation under different herbicide treatments. The results showed that under the same degree and length of slope and the same rainfall conditions, the soil loss varied depending on herbicide management due to the exponentially increased *CP* value with decreasing understory plant cover. The *CP* value derived in this study was based on one year's monitoring of a relatively small plot (0.5 m * 5 m) located on a steep slope. We chose equations from other studies under similar conditions (short slope length, the same soil properties, rills rarely observed) to calculate the slope length (*L*) and steepness (*S*) factors. Further tests on the suitability of *LS* equations could improve the accuracy of the USLE for this area. The similarity of annual rainfall erosivity calculated from one year and long-term data confirmed the validity of the annual *CP* factor derived for the rubber plantation.

Table 3.3 Correlation and partial correlations between soil loss, runoff, *CP* factor in USLE equation, and rainfall characteristics, plant and surface cover and root density. *CP* factor (dimensionless) is the combination of cover and management factor (*C*) and support practice factor (*P*) in the Universal Soil Loss Equation (USLE) model. EI_{30} is rainfall erosivity calculated by model of Browns and Foster (1987), and fRD is fine rood density

	Precipitation (mm)	EI_{30}[a] (MJ mm ha^{-1} h^{-1})	Plant Cover (%)	Surface Cover (%)	fRD (kg m^{-3}) (<1mm alive)	fRD (kg m^{-3}) (<1mm total)	fRD (kg m^{-3}) (1-2mm)	fRD (kg m^{-3}) (<2mm)
Soil loss (t ha^{-1})	0.58**	0.76**	-0.48	-0.49	-0.44	-0.4	-0.22	-0.57*
Runoff (mm)	0.76**	0.74**	-0.55*	0.02	-0.55*	-0.68**	0.23	-0.45
CP factor	-	-	-0.92**	0.71**	-0.90**	-0.83*	-0.33	-0.51

Partial correlation

Control	Variables	Plant Cover (%)	fRD ((kg m^{-3})) (<1mm alive)	Surface Cover (%)
Plant Cover (%)		-	-0.19	-0.43
Surface Cover (%)	*CP* factor	-0.86**	-	-
fRD (<1mm alive)		-0.43	-	-

a: In order to correspond to monthly measurement of ground cover, soil loss, runoff, precipitation and EI_{30} used in the analysis was monthly data

*p<0.05, **p<0.01

The estimated annual *CP* factors in the rubber plantation (0.005–0.045) were much lower than those under annual crops or grassland, which were reported as 0.25 (*C* factor) on average (Yang et al., 2003), but significantly higher than those under tropical forest

(0.001–0.004, *CP* factor) (Brooks et al., 1996; Bahadur 2009). Dissmeyer and Foster (1980) regarded the *P* factor as a subfactor of *C* and proposed nine sub-factors affecting the *C* (or *CP*) factor of woodland systems including surface cover, canopy, soil reconsolidation, soil organic matter content, fine root density, residual, onsite depression, step, and contour tillage. In our case, abundant litter from rubber trees provided high surface cover at the beginning of the rainy season as rubber sheds leaves in the middle of the dry season. Additionally, young leaves of rubber trees form shoots before the monsoon season and form a closed canopy that intercepts precipitation and reduces the impact of raindrops. No-tillage practice after establishment reconsolidates the soil structure and increases the soil organic content, while roots from rubber improve the soil physical properties by increasing the soil aggregate stability (Kabiri et al., 2015; Kasper et al., 2009). Therefore, established rubber plantations present a lower *CP* value compared to intensively managed annual agricultural systems, and hence lead to a lower vulnerability of the ecosystem to erosion.

Monthly rainfall distribution in 2014 was different from long-term observations, implying that the monthly *CP* value obtained here still needs further confirmation. The exponential relation between understory plant cover and the *CP* factor indicated that *CP* was particularly sensitive to the change of undergrowth in permanent crops or plantations like rubber. The exponential coefficient (0.028) in the equation linking understory plant cover and the *CP* factor in rubber systems was lower than that in agriculture and grassland (0.049) (Gyssels et al., 2005), reflecting the multi-layer structure in plantations supplied by trees and undergrowth. This quantified relationship enables further dynamic estimation of the *CP* factor in the rainy season, which is essential for a more accurate estimation of soil loss using the USLE equation. The impact of the quantified herbicide management on the *CP* factor was derived at small plot scale, neglecting the effect of terracing on erosion. However, terrace management in the rubber plantation should have a distinct influence on soil loss, especially at larger watershed scale (Cha et al., 2005). It is necessary to further study this impact under local conditions and to separate the *C* and *P* factors by considering plantation age and time since establishment of the terrace as well as the growth stage.

3.4.3 Impact of herbicide application on erosion and carbon export in the rubber plantation

Herbicide application significantly affected runoff production, sediment concentration, and sediment yields (Figure 3.4) by reducing understory vegetation. Reduced runoff under no or less herbicide was related to an increased infiltration rate (Table 3.1). Although no significant differences in soil bulk density were observed under different treatments, more fine roots (d < 1 mm) developed from undergrowth (Figure 3.6) can change the pore size distribution and thereby increase infiltration and decrease runoff (Greene et al., 1994; Zuazo and Pleguezuelo, 2008).

Soil covered both by plant litter and understory vegetation provides mechanical protection from rain, reducing the raindrop kinetic energy (Zuazo and Pleguezuelo, 2008) and decreasing sediment concentrations in runoff water (Garcia-Estringana et al., 2010). The low sediment concentration observed in July was due to the high surface cover comprised of rubber litter (H+) or rubber litter together with understory plant cover (Hs and H-). Herbicide application shifted the cover composition from understory plant cover to litter cover. Later during the rainy season, surface cover declined due to intensive litter decomposition (Ren et al., 1999), while enhanced soil cover was observed under reduced herbicide use. We found that the presence of bare soil was 35–75% higher with herbicide use (Hs and H+), and hence the loss of soil cover led to an increase in sediment concentration and a sustained effect on soil loss after herbicide application. On the other hand, a high and stable understory plant cover under H- treatment continuously maintained low sediment concentration and thereby ensured high soil conservation efficiency during the whole rainy season. Dense understory plant cover also protected the soil from short-term intensive rainfall events (peaking in mid-August, as shown in Figure 3.4d). In our study, we observed a decrease in sediment concentration with the appearance of vegetation, but other factors such as soil moisture and rainfall pattern may contribute to sediment concentration. Further studies are still necessary to quantify the relationship between sediment concentration and plant cover. The reduction in runoff and sediment concentration by vegetation verified the high soil conservation efficiency in extreme precipitation events under H- (Figure 3.5), revealing the possibility of coping with increasing climate variability and weather extremes through appropriate land management.

As for carbon export, Xu et al. (2015) proved that the soil water conservation measures adopted in Han River, China, efficiently increased SOC. For a rubber plantation, De Blecourt et al. (2013) partly attributed soil carbon loss to clearance of the understory vegetation after establishment of the rubber plantation. We found that herbicide application, as another option for soil conservation, affected the total carbon export amount by increasing sediment yield, but no difference in enrichment ratio was observed (Table 3.2). The transport of SOC by erosion is related to the detachment and transport of coarse or fine soil particles. The findings of Jin et al. (2009) and Zhang et al. (2011) both agreed that the cover percentage had no significant effect on the organic enrichment ratio, while Kisic et al. (2002) obtained different results. The similar enrichment ratios under different herbicide treatments found in this study can be explained by 1) the similar rainfall energy in the multilayer canopy system and 2) the dominating effect of slope steepness. Rainfall energy affected the ratio of release of micro- or macro-aggregates and therefore the sediment organic carbon enrichment ratio (Zhang et al., 2013). This effect might be stronger than the influence of soil cover on the enrichment ratio (Zhang et al., 2011). Herbicide application inhibited understory vegetation, while a major influence on rainfall energy may be attributed to the tree canopy instead of the undergrowth. The lower exponent in reducing sediment yield compared to annual crop systems, as explained above, reflected the less strong impact from surface cover in the multilayer system. The steep topography in our study (second explanation) mitigated the deposition of coarse particles during inter-rill erosion as well as the possible influence of soil cover on the selected transport of finer particles. Steep slopes also explain the observed enrichment ratio (1.1–1.2), which is lower than those found by other studies (M-Mena et al., 2008).

3.4.4 Potential improvement of the rubber system's anti-erosive effect by reducing herbicide

The most common type of management practiced by farmers is two herbicide applications per year (Hs), although some farmers prefer clean plots (H+) with more frequent herbicide use. However, both herbicide treatments have a negative impact on the environment by greatly increasing runoff and erosion and potentially increasing surface water pollution (Arias-Estevez et al., 2008) while decreasing the biodiversity of the system. In order to maintain the general highly anti-erosive effect of undisturbed established rubber systems while reducing environmental impacts, minimal use of herbicides is necessary. From the perspective of preserving a high level of ecosystem function and services, completely avoiding the clearance of understory vegetation (H-) would best fulfill this requirement by maintaining a low *CP* value and would reduce soil losses to under the limit of 100 g m^{-2} $year^{-1}$ that is considered necessary to support long-term soil sustainability (Jürgens and Fander, 1993). The main concerns of farmers who insist on the use of pesticides are 1) the risk of competition for soil water and nutrients from understory vegetation in the dry season, and 2) inconvenience for tapping. The results from our study revealed the importance of maintaining a good understory plant cover for soil conservation during months with highly erosive events (July and August). It was observed that understory plant cover was low (20%) even with no herbicide application in the dry season (H-). This indicated the natural control of understory vegetation due to water shortages in the dry season. However, undergrowth recovered fast once the rainy season started, with plant cover increasing and remaining at over 80%. Glyphosate is only active in foliage but not in roots or seeds, and therefore applying a single dose of herbicide at the beginning of the rainy season could inhibit the overgrowth of understory vegetation in June, when the high litter cover efficiently protected soil. Additionally, the rainy season supplies sufficient water for fast growth of understory plants. As a result, good understory plant cover is ensured, which offsets the decreasing litter cover in the mid-rainy season with highly erosive events. Rubber trees, as the dominant species in the established plantation, ensured strong competition for light, water, and nutrients through a closed tree canopy, high root density, and deep root system. Therefore, one herbicide application in the early rainy season can provide good soil conservation as well as preventing excessive understory growth.

3.5 Conclusions

Herbicide application is a major activity affecting soil loss in established rubber plantations by removing the protective understory plant cover. The large but dynamic litter cover could not fully compensate for the loss of understory vegetation. The change of understory plant cover explained the dynamic of the monthly *CP* factor well in this study, which indicated the sensitivity of the *CP* factor to undergrowth in the plantation system. We found that the annual *CP* factor of the rubber plantation varied from 0.005 to 0.04 under different herbicide treatments.

Reducing the use of herbicides is necessary to improve soil conservation in rubber plantations. Avoiding the application of herbicides can keep the understory plant cover above 70% and has the potential to reduce sediment yields and runoff by 83 and 48% respectively compared to the current application practice. This management, however, is unlikely to be adopted by farmers due to their concerns about understory competition with rubber trees in the dry season and inconvenience for tapping. We suggest reducing herbicide application to a single treatment per year in the early rainy season to improve soil conservation in rubber cultivation.

Chapter 4 Modelling weed management strategies to control erosion in rubber plantations[c]

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[c] This chapter has been reprinted from:
Liu, H., Yang, X., Blagodatsky, S., Carsten, M., Liu, F., Xu, J., Cadisch, G. (2019). Modelling weed management strategies to control erosion in rubber plantations. Catena 172：345-355.
(DOI: org/10.1016/j.catena.2018.08.041). With permission from 2019 Elsevier Ltd.
The original publicatioin is available at:
https://doi.org/10.1016/j.catena.2018.08.041

Abstract

The role of weeds in soil conservation in agroforestry systems has been largely ignored. We used the Land Use Change Impact Assessment (LUCIA) model to simulate the effects of weed management on erosion in rubber plantations (*Hevea brasiliensis* Muell. Arg). In order to quantify the impact of a dynamic, spatially explicit multi-layer plantation structure on erosion processes in agroforestry systems, we updated LUCIA's erosion module. Its new version simulates soil detachment due to rainfall and runoff, considering the separate effects of the tree canopy and surface cover on soil erosion. The updated LUCIA model was calibrated and validated based on an established rubber plantation experiment in Xishuangbanna, Southwest China, to evaluate the impact of different weeding strategies on soil loss. The model successfully represented the impact of the dynamic multi-layer structure on erosion and was able to predict well the effects of weed management on soil loss and runoff at the test site over 1 year, with a modelling efficiency (*EF*) of 0.5–0.96 and R^2 of 0.64–0.92. Subsequently, we validated the ability of the model to simulate surface cover changes under rubber plantations of different age (up to 40 years). Simulation outputs for 4-, 12- and 18-year-old rubber plantations revealed satisfying to good results. However, the predicted change in surface cover for old rubber plantations (25- and 36-year) failed to meet the field trends. The model predicted the greatest erosion in the year when the rubber canopy started to close. During this period, weed growth was limited by light, while litter input from rubber was insufficient to provide good soil cover. Four weeding strategies ("clean-weeding", "twice-weeding", "once-weeding" and "no-weeding") were designed for scenario simulations. Based on the results of 20-year runs, we concluded that "once-weeding" and "no-weeding" both efficiently minimized soil loss during one rotation length. A

high degree of surface and weed cover (over 95% and 60%) under “no-weeding” makes this management strategy with dense undergrowth hardly acceptable by local farmers due to reduced tree accessibility for tapping and increased potential danger through poisonous caterpillars. “Once-weeding”, on the other hand, controlled overgrowth of understory vegetation by keeping weed cover below 50%. We therefore suggest “once-weeding” as an improved herbicide management strategy in rubber plantations, to meet ecological system service maintenance and to facilitate adoption in practice.

4.1. Introduction

Soil erosion is exacerbated by rapid agricultural expansion in steep montane regions of Southeastern Asia, and threatens soil health and crop yields. The effects of erosion and conservation in traditional agricultural land uses, such as maize growing, have been well studied in this region (Pansak et al., 2010; Quang et al., 2014; Tuan et al., 2014). On the contrary, efficient conservation measures remain uncertain for more recently evolved land uses, especially in growth of perennial crops such as rubber plantations. Rubber plantations have rapidly expanded in Southwest China in the past decades. Although this land use type is mostly considered as forest cover by Chinese decision-makers (Zhai et al., 2018), its monoculture cultivation has resulted in biodiversity loss and environmental degradation (Li et al., 2010, Thellman et al., 2017). Compared to that in rainforests, total soil loss per year in rubber plantations has been estimated to increase by 45 times (Wu et al., 2001). In order to reduce potential soil losses, several conservation measures such as terracing and intercropping have been proposed and tested in short-term field experiments (Cha et al., 2005; Sidle et al., 2006). Particularly, minimization of weeding has been proved to be highly efficient in reducing soil loss in established rubber plantations (H. Liu et al., 2016) with little effects on latex yields (Abraham & Joseph, 2015).

However, the effects of longer-term weeding conservation remain uncertain due to a lack of long-term experimental data. Rubber is a perennial crop with a rotation length of 20–40 years, so that soil erosion as well as ground cover changes may vary during this time (Liu et al., 2018). Long-term tests of the potential impact and limitations of different weeding strategies are necessary but expensive and laborious. Crop and soil simulation models can provide an efficient tool and reduce associated cost (Matthews et al., 2001). Since the formulation of the Universal Soil Loss Equation (USLE) (Wischmeier and Smith, 1978), large efforts have been undertaken to develop advanced soil erosion assessment tools. Process-based models were developed to offset the conceptual limitations of simple empirical models such as USLE, with GUEST (Griffith University Soil Erosion Template; Misra and Rose, 1996), LISEM (Limburg Soil Erosion Model; De Roo and Wesseling, 1996) and WEPP (Water Erosion Prediction Project; Nearing et al., 1989) as prominent examples. These erosion models have proven their validity in plot-based studies with good hydrological (e.g. rainfall, runoff rate) and plant (e.g. ground cover, leaf area index) input (Barros et al., 2014; Cao et al., 2015; Fernandes et al., 2017; Poleto et al., 2014). However, simplification of dynamic plant growth and development routines hampers application of the above-mentioned erosion models for direct simulation of the impact of management on soil conservation. In particular, weed management simulation needs to present farmers' acceptance of weed growth, as well as the relationships between weed growth, tree development and erosion processes. The latter should represent both processes of plant competition for light and resources, and soil conservation. The Land Use Change Impact Assessment (LUCIA) model is a tool for both plot-level management and spatially explicit watershed-level simulations. Its plant growth module is based on the WOrld FOod STudies (WOFOST, Supit, 2003) approach and simulates tree-weed-soil interactions in plantation systems; while infiltration and runoff simulation is built on KINEROS 2 (Woolhiser, 1990). LUCIA has been successfully tested in tropical mountainous areas of Thailand and Vietnam (Lippe et al., 2014; Marohn et al., 2013a; Marohn et al., 2013b). LUCIA uses the Rose concept of erosion (Hairsine and Rose, 1992) and considers runoff entrainment-driven soil erosion dominant over rainfall-induced soil detachment (Lippe et al., 2014; Marohn et al., 2013a; Noordwijk et al., 2011). Splash erosion, hereafter called "rainfall detachment", has not yet been considered. In a plantation ecosystem such as a rubber plantation, the tree canopy intercepts raindrops and reduces rainfall amount and intensity, and therefore reduces the erosive power of rain events. On the other hand, accumulation of raindrops increases the kinetic energy of throughfall with rising canopy height. Therefore, the tree canopy should not be simply considered as a component of surface cover contributing only to soil protection. Field studies have proven that rainfall detachment is an important contributor to the total amount of soil detached in plantations (Ghahramani et al., 2011). The average potential splash erosion rate has been observed to be 2.1 times higher in rubber plantations than in open areas (Liu et al., 2015). Thus, it is important to include rainfall detachment in erosion process simulations.

This study aims to expand the runoff entrainment-driven (stream power) erosion approach with simulation of a multi-layer plantation structure by incorporating rainfall detachment into the erosion module of the LUCIA model. We then i) tested whether the updated LUCIA model could simulate erosion in a dynamic multi-layer system, specifically in rubber plantations, ii) tested how weed management, in particular the frequency of herbicide application, affects erosion during one rotation cycle (20–40 years) of rubber and iii) suggest an improved weeding strategy for rubber plantations, based on the model results, to efficiently control erosion.

4.2 Materials and methods

4.2.1 Model description

We simulated biophysical processes in rubber plantations at the plot scale using LUCIA model, which describes interactions between trees, weeds and soil in plant growth, water balance, erosion and soil organic matter modules. This study focused on including splash simulation into the erosion module. Where necessary, inputs provided by other modules are explained, while related equations are detailed in supplementary Table S1.

The erosion simulation in LUCIA follows the basic assumption that runoff-driven soil erosion, hereafter called 'runoff entrainment' (Hairsine and Rose, 1992), dominates over rainfall detachment. Runoff is simulated by water balance module as the remainder of daily rainfall minus interception and the water that infiltrates unsaturated soil (Supplementary, table S1). Runoff entrainment (c_{en} in kg m^{-3}) is calculated based on the maximum sediment concentration at transport capacity (c_{max} in kg m^{-3}), soil erodibility (β in the range of (0,1), dimensionless) to account for the resistance of flow detachment by the cohesive soil matrix, and cover efficiency (α, dimensionless) to exponentially reduce soil detachment with increasing surface cover (SF in the range of

(0,1), dimensionless):

$$c_{en} = {c_{\max}}^{\beta} \cdot \exp(-\alpha \cdot SF) \tag{4.1}$$

where $c_{\max}$ is the transport capacity, the theoretical maximum of sediment concentration (kg m^{-3}) limited by stream power, runoff flow depth and average sediment settling velocities (Misra & Rose, 1996). The coefficient β ($0<\beta\leqslant 1$) to account for the resistance of flow entrainment by the cohesive soil matrix (Misra and Rose, 1996). The influence of surface cover in reducing the force of sediment entrainment is accounted for the second part of Eq. (4.1) (Rose, 1993); more details can be found in the work of Lippe et al. (2014).

Surface cover (*SF*, dimensionless) is simulated as a function of dynamic leaf area index of rubber ($LAIRubber$, dimensionless), leaf area index of weed ($LAIWeed$, dimensionless), Lit_{eff} (ha Mg^{-1}) the effectiveness of plant litter covering the soil surface, and Lit_{surf} (Mg ha^{-1}) the surface litter amount (Marohn et al., 2013a)

$$SF = \left(1 - exp^{-\delta \cdot LAIRubber}\right) + [1 - \exp(-0.7 \cdot LAIWeed)] + \left(Lit_{eff} \cdot Lit_{surf}\right) \tag{4.2}$$

Three parts of Eq. (4.2) represent canopy cover (Gash et al., 1995), weed cover and litter cover (Marohn et al., 2013a), respectively. δ (dimensionless) is the coefficient of leaf distribution and light inclination, ranging from 0.6 to 0.8 for trees. *LAIRubber* and $LAIWeed$ are the simulated rubber leaf area index and weed leaf area index by plant module (Supplementary, Table S1). Lit_{eff} is an input from plant module, and Lit_{surf} is simulated in the soil organic matter module of LUCIA (Marohn et al., 2013b).

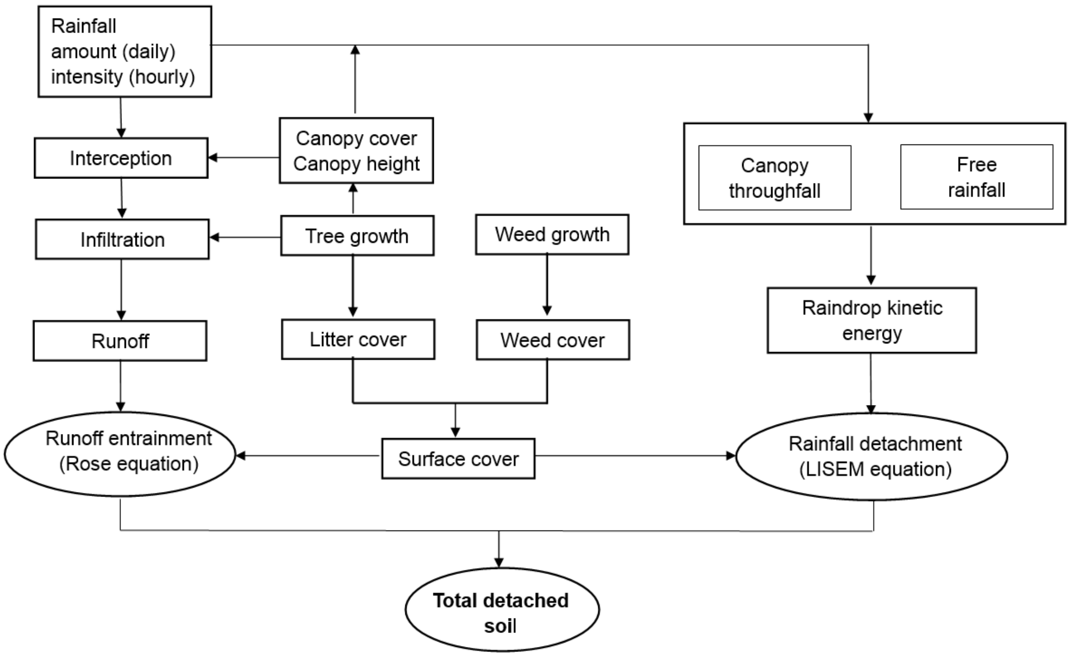


Figure 4.1 Structure of the updated LUCIA erosion module and involved parameters

4.2.2 LUCIA update: erosion simulation under multi-layer plant cover

In order to simulate the influence of multi-layer plant cover on erosion, we firstly redefined surface cover as litter and weed cover, excluding the tree canopy cover. Therefore, surface cover calculation changed from equation (4.2) to:

$$SF = [1 - \exp(-0.7 \cdot LAIWeed)] + (Lit_{eff} \cdot Lit_{surf}) \tag{4.3}$$

The influence of the tree canopy on soil erosion was simulated by calculating amount and intensity of free rainfall and canopy throughfall. Kinetic energy of rain drops driving rainfall detachment comes from two sources: free rainfall and canopy throughfall, considering the impact of canopy height with trunk height (H_T, m), and canopy thickness (H_C, m). Then the total detached soil is calculated as the sum of rainfall detachment and runoff entrainment. Fig. 4.1 provides an overview of the processes controlling erosion, which were included in the updated version of LUCIA, as described further below.

4.2.2.1 Canopy throughfall simulation

Free rainfall and canopy throughfall are differentiated based on the concept proposed by Rutter et al. (1971). Free rainfall amount (FR_{tot} in mm d^{-1}) and intensity (FR_{int} in mm s^{-1}) are calculated according to Lloyd et al. (1988) and Gash et al. (1995):

$$FR_{tot} = R_{tot} \cdot (1 - CanCover) \tag{4.4}$$

$$FR_{int} = R_{int} \cdot \frac{1 - CanCover}{3600} \tag{4.5}$$

where R_{tot} is daily rainfall amount (mm d^{-1}), R_{int} is hourly rainfall intensity (mm h^{-1}), *CanCover* (dimensionless, range 0 - 1) is the canopy cover of the rubber plantation by plant module, calculated as average cover percentage of per area (Supplementary, Table S1). 3600 is the conversion factor from mm h^{-1} into mm s^{-1}.

Based on the water balance concept of the Rutter model (Gash et al., 1995):

$$FR_{tot} + CD_{tot} + IE_{tot} = R_{tot} \tag{4.6}$$

where FR_{tot} is daily free rainfall (mm d^{-1}), CD_{tot} is daily canopy drainage (mm d^{-1}), and IE_{tot} is daily interception evapotranspiration (mm d^{-1}), calculated by water balance module (Supplementary, Table S1).

Canopy drainage (CD_{tot}, mm d^{-1}) is partitioned into stem flow (ST_{tot}, mm d^{-1}) and canopy throughfall (CR_{tot} in mm d^{-1}). CR_{tot} is therefore calculated by the drainage partitioning coefficient a (Gash et al., 1995), namely:

$$CR_{tot} = a \cdot CD_{tot} \tag{4.7}$$

where a was estimated as 0.9 based on datasets from rubber plantations in the work of Liu et al. (2018). The canopy throughfall intensity (CR_{int} in mm s^{-1}) is estimated according to the scaling technique in water balance module of LUCIA (Yu et al., 1997):

$$CR_{int} = \left(\frac{CR_{tot}}{R_{tot}}\right) \cdot R_{int} \cdot \frac{1}{3600} \tag{4.8}$$

where 3600 is the conversion factor from mm h^{-1} to mm s^{-1}, R_{int} (mm h^{-1}) is the maximum hourly rainfall intensity (model input). When data on rainfall intensity (R_{int}) are absent, as in the long-term simulation case of this study, an empirical function based on that developed by Lippe et al. (2014) using a dataset from mountainous Northern Vietnam (Ziegler et al., 2004) was applied (R^2 = 0.93, tested against one-year dataset in our case):

$$R_{int} = 0.9871 \cdot R_{tot}{}^{0.8851} \tag{4.9}$$

where R_{tot} (mm d^{-1}) is daily rainfall amount (model input).

4.2.2.2 Rainfall detachment in the erosion subroutine

Rainfall detachment (Ds in g m^{-2} s^{-1}) includes detachment caused by free rainfall and canopy throughfall. The general equation according to the LISEM model (De Roo and Wesseling, 1996) is:

$$Ds = (2.82/As \cdot Ke \cdot \exp(-1.48 \cdot D) + 2.96) \cdot P \cdot (1 - SF) \tag{4.10}$$

where As (dimensionless) is aggregate stability, calculated using a pedotransfer function proposed by Grønsten (2008) and adjusted by Kvaernø & Stolte (2012):

$$As = [91.6 + 3.5 \cdot SOM - 1.06 \cdot (fs + si)] \cdot 2 \tag{4.11}$$

where SOM (%) is topsoil organic matter content, fs (%) is sand content (0.02–2 mm) and si (%) is silt content (0.002–0.02 mm) in the topsoil.

Ke (J m^{-2} mm^{-1}) is the rainfall kinetic energy, which is the sum of free rainfall (Ke_{FT}, J m^{-2} mm^{-1}) and canopy throughfall (Ke_{CT}, J m^{-2} mm^{-1}). Equations are derived from the LISEM model (De Roo and Wesseling, 1996):

$$Ke_{FT} = 8.95 + 8.44 \cdot log_{10}(R_{int}) \tag{4.12}$$

$$Ke_{CT} = 15.8 \cdot sqrt(H_T + 0.5 \cdot H_C) - 5.87 \tag{4.13}$$

where R_{int} (mm h^{-1}) is the maximum hourly rainfall intensity (model input), H_T is trunk height (m) and H_C is the canopy thickness (m) of the rubber tree, taken from the plant module (Supplementary, Table S1).

D in Eq. (4.10), the depth of runoff flow (m), is calculated in a LUCIA water balance module proposed by Lippe et al. (2014), and *P* (mm s^{-1}) is the free rainfall intensity (FR_{int} calculated in Eq. (4.5)) or canopy throughfall intensity (CR_{int} calculated in Eq. (4.8)). The empirical factor 2.82 in Eq. (4.10) converts the dimensionless *As* into a factor with units g J^{-1}, while the factor 2.96 has as units g m^{-2} mm^{-1} (De Roo and Wesseling, 1996).

Sediment concentration (c_{rd} in kg m^{-3}), contributing to losses from rainfall detachment, is calculated as:

$$c_{rd} = \frac{(1-H)\cdot Ds/1000}{Q_{rate}/3600000} = \frac{3600\cdot(1-H)\cdot Ds}{Q_{rate}} \tag{4.14}$$

where H is the surface covered by the deposited layer and is assumed to be 0.9 (Heilig et al., 2001), Ds is the rainfall detachment rate (g m^{-2} s^{-1}, Eq. (4.10)), Q_{rate} is the hourly runoff rate (m h^{-1}) from water balance module (Supplementary, Table S1), and 3600 is the conversion factor to kg m^{-3}.

Soil detached by rainfall (S_{rd}, Mg ha^{-1} d^{-1}) is calculated as:

$$S_{rd} = c_{rd} \cdot Q_{tot}/100 \tag{4.15}$$

Total soil detachment per day (S_D in Mg ha^{-1} d^{-1}), in accordance to the daily time step of LUCIA model, then is:

$$S_D = (c_{rd} + c_{en}) \cdot Q_{tot}/100 \tag{4.16}$$

where c_{en} is the sediment concentration contributed by runoff entrainment (kg m^{-3}, Eq.(1)), Q_{tot}(mm d^{-1}) is daily runoff rate from the water balance module, and 100 is the conversion factor to Mg ha^{-1}.

Thus, LUCIA was updated by incorporating the above-derived canopy throughfall and associated rainfall detachment simulations (hereafter "updated LUCIA").

4.2.3 Test of the updated LUCIA model

4.2.3.1 Site description

The updated LUCIA model was applied to rubber plantations in Xishuangbanna, Yunnan Province, Southwest China. The annual precipitation for this region is 1100–1600 mm, and the mean annual temperature is 18–22 °C. The region has a typical monsoon climate characterized by a distinct rainy season from May to October and a dry season from November to April. Sixty to ninety percent of the precipitation falls during the rainy season. Rubber has been introduced to Xishuangbanna since the 1970s and covered 4787 km^2 (around 24% of the total area) in 2014 (Zhai et al., 2018). The lifespan of rubber plantations in this area is generally 20–25 years, at most 40 years. Rubber trees are planted at densities of 450–600 trees per hectare and in rows on terraces, with a tree space of 3–4 m and distance between two adjacent planting terraces of 5–7 m. The most common practice among local farmers is the application of herbicide twice a year in the mid-rainy season and mid-dry seasons, respectively, using 10 kg ha^{-1} of 10% glyphosate.

4.2.3.2 Available field data

Site-specific data for model simulations were taken from three field experiments conducted in 2013 and 2014 in Xishuangbanna. Available field data for model calibration and validation are summarized in Table 4.1.

Field data from a splash potential study (W. Liu et al., 2016) were used to evaluate the newly introduced rainfall detachment routines. Splash cups filled with quartz sand (0.125–0.2 mm) were positioned on the floor of rubber plantations to measure splash erosion potential. After each natural single rainfall event, the cups were emptied to calculate sand loss. Rainfall was monitored by a tipping-bucket data-logging rain gauge (3354WD; Spectrum Technologies Inc., USA) with 0.2 mm resolution. More detailed information about the experimental setup can be found in the work of W. Liu et al. (2016).

Field data from a herbicide application study (H. Liu et al., 2016) were used to calibrate and validate weed management simulations. The experiment was established in a 12-year-old monoculture rubber plantation with a slope of 55% in Nabanhe Watershed National Nature Reserve, Xishuangbanna in 2014. We used a complete randomized block design with herbicide application as the main factor, with three frequency levels. These included: (i) herbicide application twice per year in mid-February 2014 and late July 2014, respectively, using 10 kg ha^{-1} of 10% glyphosate (“twice-weeding”); (ii) no herbicide application, to maintain a high level of understory plant cover (“no-weeding”);

Table 4.1 Summary of available field data for model calibration and validation gathered in rubber plantations in Xishuangbanna, Southwest China

	Splash validation	Weeding management		Simulated long-term surface cover validation
		Calibration	Validation	
Study site	Xishuangbanna Tropical Botanical Garden (21°55'N, 101°15'E), Yunnan Province, SW China	Nanbanhe River Watershed National Nature Reserve (NRWNNR) in Xishuangbanna (22°17' N, 100°65' W), Yunnan Province, SW China		
Measurement period	June - Oct. 2013	2014		
Rubber plantation age and management	16-year rubber	"twice-weeding" in 12-year rubber	"clean-weeding" and "no-weeding" in 12-year rubber	"twice-weeding" in 4, 18, 25 and 36-year rubber
Field data used as model input	• Event-based rainfall amount (mm) and rainfall intensity (mm h^{-1}) • Soil texture	• Daily rainfall amount (mm) and hourly rainfall intensity (mm h^{-1}) • Soil properties		
Field data used for model calibration	-	• Monthly measured weed cover and surface cover (%)	-	-
Output parameters	Event-based splash potential (Mg ha^{-1})	• Event-based runoff (mm) and soil loss (Mg ha^{-1}) under "twice weeding"	• Monthly measured surface cover (%) under "clean-weeding" and "no-weeding" • Event-based runoff (mm) and soil loss (Mg ha^{-1}) under "clean-weeding" and "no weeding"	• Monthly measured surface cover (%) under "twice-weeding" in 4Y, 18Y, 25Y and 36Y rubber
Source	W. Liu et al., 2016	H. Liu et al., 2016		Liu et al., 2018

(iii) bimonthly herbicide application using the same amount as in “twice-weeding” each time, in order to obtain little or no understory plant cover (“clean-weeding”). Runoff and sediment were collected from the Gerlach troughs after each event that produced erosion under natural rainfall in 2014. Rainfall was monitored on the plot by a tipping-bucket data-logging rain gauge (Campbell Scientific TB4) with 0.2 mm resolution. Surface cover, including litter and weeds, and weed cover were measured monthly during the rainy season in 2014. Further details can be found in the work of H. Liu et al. (2016).

An additional set of data from a dynamic rubber erosion risk study (Liu et al., 2018) was used to compare observed and simulated surface cover in rubber plantations of different ages. This study was carried out to investigate erosion change along rubber plantation ages under “twice-herbicide” treatment. Monthly surface cover was monitored in 4-, 12-, 18-, 25- and 36-year-old rubber plantations.

4.2.3.3 Model calibration and validation

The model was set to run at plot scale with 5 m × 5 m area and 55% slope according to the conditions in the field experiment. Inputs for the plant module were taken from the default database provided by LUCIA, validated by the field investigation in the same study area by Yang et al. (2016).

Splash validation

The updated LUCIA introduced rainfall detachment calculations (Eqs. (4.10-4.13)) based on development of the LISEM model (De Roo and Wesseling, 1996). We firstly tested the suitability of the equations for our rubber system by comparing splash erosion potential (S_{rd}) calculated using Eq. (4.15) with the field measurements of W. Liu et al.

(2016). Daily rainfall amount and hourly intensity were measured in 2013. According to the field splash potential measurement setting, soil texture was set to be fs (sand content %) = 100. As was adjusted to the lowest value, 0.6 (with range of 0 to 200), based on Eq. (4.11). In total, nine events were simulated to evaluate model performance.

Weeding management calibration and validation

After the splash validation, the updated LUCIA was calibrated and validated to simulate the effects of weed management on soil conservation based on the 1-year field experiment (H. Liu et al., 2016). Rainfall (amount and intensity) and soil properties were entered according to the plot-scale measurements in 2014. Field data were split into a calibration set comprising variables measured under "twice-weeding", and a validation set measured under "clean-weeding" and "no-weeding". Calibration was carried out manually through trial and error until satisfactory results were obtained: 1) we determined upper and lower limits for target parameter based on prior information (e.g. literature, model manual) and run the model several times (normally 4 to 5) changing the target parameter by 20-25% of the estimated range; 2) model fit with established statistics (e.g. $R^2 > 0.5$, EF > 0.6) indicated the optimal parameter range; 3) the procedure was repeated for the narrow parameter range and was stopped, when the best fit between simulated and observed values was produced.

Weed cover simulation was best fitted by setting $LAIRGR_{(max)}$ to 0.04, which constrains the increase in $LAIWeed_t$ (Supplementary, Table S1). Then litter cover was best fitted by setting $Lit_{(eff)}$ to 0.5, the effectiveness of plant litter in covering the soil surface in Eq. (3). α, the coefficient reducing soil detachment in Eq. (1) due to cover efficiency, was set to 2.5 following the conclusion of Dune et al. (1978) from field measurements. Finally, simulation of soil loss was completed with best fitted by setting the value for soil erodibility in Eq. (1), β to 0.27.

A summary of calibrated parameters and the associated inputs from the plant module are shown in Table 4.2. After initial model calibration, two alternative weeding strategies ("clean-weeding" and "no-weeding") were set up using the weed management options in LUCIA. Model validation was performed against measured weed cover, surface cover, runoff and soil loss as reported by H. Liu et al. (2016).

Long-term surface cover validation

Good surface cover projection is a premise of reliable soil erosion simulation. Therefore, we ran the model for 40 years to further validate its ability to simulate erosion in plantations with a long rotation. Simulated surface cover in young (4-year), mid-age (12- and 18-year) and old (25- and 36-year) rubber plantations under "twice-weeding" was compared with monthly field measurements conducted in 2014 in these systems (Liu et al., 2018). Maize was taken as land cover for the second and third years according to the local common practice of intercropping maize with small rubber trees. Here we assumed that the effect of small rubber trees on soil erosion was negligible in the young rubber–maize intercropping system. All parameters in long-term runs were kept the same as those calibrated in the 1-year weeding management simulation. Weed management was set as "twice-weeding"; rainfall input was the same for all years to fit field conditions, as our aim was to study variation in soil cover effects and avoid confounding interactions with varying climate.

Table 4.2 LUCIA model parameters and their input values obtained during the calibration period

Parameters	Description	Value	Unite
Input of plant modules in LUCIA			
CR_{ini}	Initial canopy radius	0.5	m
CR_{max}	Maximum canopy radius	3	m
LAI_{max}	Maximum leaf area index of rubber trees	8	Dimensionless
$PlantDensity$	Plant density of rubber plantations	470	ha^{-1}
Calibration			
$LAIRGR_{(max)}$	Maximum relative growth rate of weeds LAI	0.04	ha ha^{-1} d^{-1}
$Lit_{(eff)}$	Effectiveness of plant litter covering the soil surface	0.5	Dimensionless
α	Coefficient of cover efficiency to reduce soil detachment	2.5	Dimensionless
β	Coefficient of soil erodibility in the absence of vegetation	0.27	Dimensionless

4.2.3.4 Model performance

Model performance was assessed by comparing predicted values against observed data for event-based runoff and soil loss. R^2, modelling efficiency, coefficient of determination and root mean square error were applied to evaluate model performance (model goodness of fit, GOF; Loague and Green, 1991).

Modelling efficiency (*EF*) was calculated as:

$$EF = \frac{\sum_{i=1}^{n}(O_i - \bar{O})^2 - \sum_{i=1}^{n}(P_i - O_i)^2}{\sum_{i=1}^{n}(O_i - \bar{O})^2} \tag{4.17}$$

and coefficient of determination (*CD*) as:

$$CD = \frac{\sum_{i=1}^{n}(O_i - \bar{O})^2}{\sum_{i=1}^{n}(P_i - \bar{O})^2} \tag{4.18}$$

and root mean square error (*RMSE*) as:

$$RMSE = \left(\frac{\sum_{i=1}^{n}(P_i - O_i)^2}{n}\right)^{0.5} \cdot \frac{100}{\bar{O}} \tag{4.19}$$

where O_i are the observed values, $\bar{O}$ is the mean of the observed data, P_i are the predicted values and n is the number of samples.

EF indicates how well the predicted values correspond to the observed values. A value of 1 means a perfect one-to-one fit. Following the studies of Pansak et al. (2010) and Lippe et al. (2014), an *EF* threshold of > 0.6 was used as the minimum performance criterion during model calibration procedures. *CD* is a measure of the proportion of the total variance of observed data explained by the predicted data; a value of 1 indicates a perfect prediction fit. We considered *CD* values between 0.5 and 2 during model calibration and estimation of validation success. *RMSE* describes the average error of predicted outcomes. The smaller the *RMSE*, the closer simulated values are to the observed ones; a value of zero indicates a perfect model fit (Bhuyan et al., 2002; Hussein et al., 2007).

4.2.3.5 Design of weed management scenario

Besides three weeding strategies described above, an additional “once-weeding” was included in management scenarios as recommended by H. Liu et al. (2016), namely four weeding strategies (“clean-weeding”, “twice-weeding”, “once-weeding” and “no-weeding") were evaluated in 1-year and long-term (40-year) runs. In the long-term run, weeding strategies were tested in the first seedling year, and the following rubber plantation years, namely the second and third maize-planting years, were excluded (setting as clean-weeding for all scenarios). As rainfall is the major driver of erosion, we used two sets of rainfall data in the long-term scenarios to estimate the potential change in erosion with plantation age. One was the real daily rainfall amount obtained from Jinghong airport (1975–2014), around 25 km away from the study site. Hourly rainfall intensity was calculated using Eq. (4.9). The other set of rainfall input data repeated the rainfall amount and intensity measured in 2014 for 40 years. Table 4.3 summarizes the resulting 12 simulation runs.

Table 4.3 Scenario runs to assess the impact of weed management strategies on erosion in rubber plantations in short and long-term simulations. "clean-weeding": no allowance of weed growth; "twice-weeding": herbicide application twice per year in mid-February and late July; "once-weeding": herbicide application once per year in mid-February; "no-weeding": no herbicide application.

Scenarios No.*	Running period	Weed treatment	Rainfall input
1	Short term (one year)	clean-weeding	Daily rainfall amount and hourly intensity measured in 2014
2	**	twice-weeding	
3		once-weeding	
4		no-weeding	
5	Long term (40 years)	clean-weeding	Looped daily rainfall and hourly intensity measured in 2014
6		twice-weeding	
7		once-weeding	
8		no-weeding	
9	Long term (40 years)	clean-weeding	Real rainfall amount measured at Jinghong airport (1975 - 2014) and calculated rainfall intensity
10		twice-weeding	
11		once-weeding	
12		no-weeding	

* Scenario 2 was used for model calibration; scenario 1 and 4 were for model validation on weeding strategies impact on soil loss in rubber plantations; scenario 6 was used for model validation on long-term surface cover simulation.

** In short term runs, rubber plantation was set as 12-year old.

4.3 Results

4.3.1 Performance of the updated LUCIA model

Rainfall detachment calculations from LISEM incorporated into LUCIA showed acceptable agreement (*EF* of 0.67) between simulated and measured splash erosion potential in rubber plantations (Figure 4.2). Furthermore, the updated LUCIA was capable of predicting well the long-term changes in surface cover in young (4-year) and mid-age (12-, 18-year) rubber plantations ($EF \leq 0.96$) in rainy season (June to October, when erosion occurred) but failed to produce comparable results with measured data for old (25-, 36-year) rubber plantations (Figure 4.3).

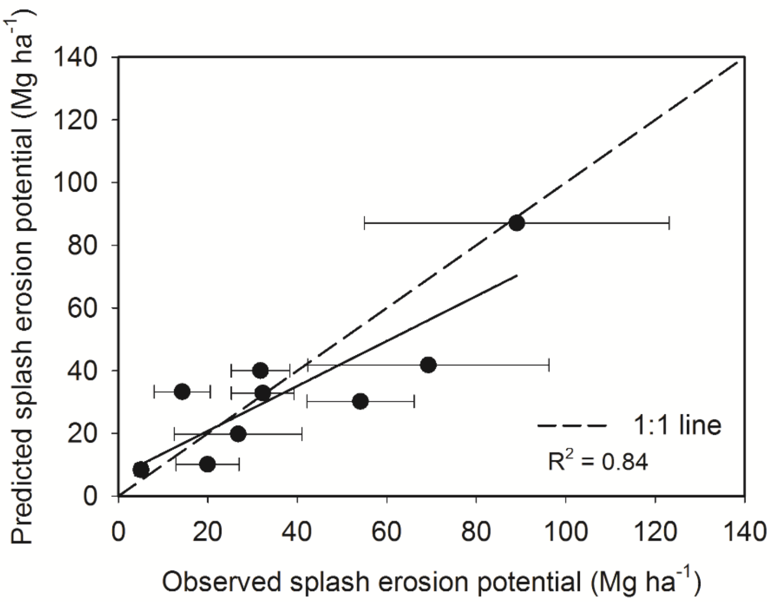


Figure 4.2 Predicted vs. observed event-based splash erosion potential under rubber in 2013 to validate the suitability of splash erosion calculations introduced from LISEM (Limburg Soil Erosion Model). The solid line refers to the regression curve and the dashed line is the one-to-one line.

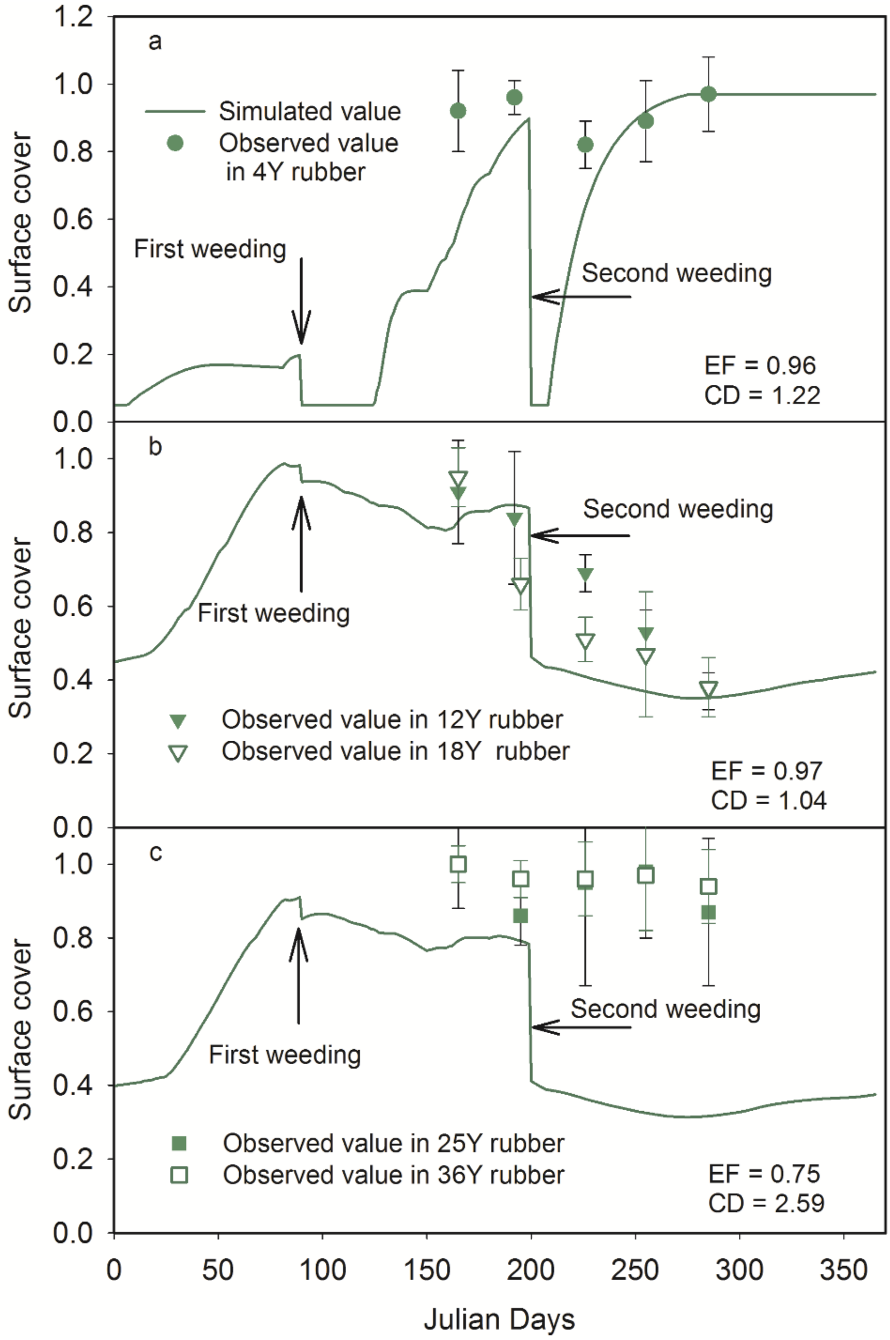


Figure 4.3 Predicted vs. observed surface cover change (including litter and weed biomass) under "twice-weeding" during long-term (40 years) runs for (a) young rubber (4-year old), (b) mid-age rubber (12-year, 18-year old), and (c) old rubber (25-year, 36-year old). First weeding was in mid-February, second weeding was in late July. EF: Modelling Efficiency. CD: Coefficient of Determination.

Considering hourly rainfall intensity resolution in the model, we ignored events with runoff of less than 2 mm, which accounted for 33 events contributing to less than 25% of total runoff and soil loss. The model detected a total of 39 events causing more than 75% of annual measured soil loss. Model performance of event-based runoff simulation was at an acceptable level, with *EF* of 0.75 for calibration and 0.5 for validation (Fig. 4.4). Event-based soil loss simulation demonstrated that LUCIA was able to imitate the effects of different weeding management strategies on erosion, with *EF* of 0.86 and 0.87 for calibration and validation, respectively (Figure 4.4). Considering that LUCIA was not able to reproduce correct surface cover changes in old rubber plantations, we limited further analysis to 20 years.

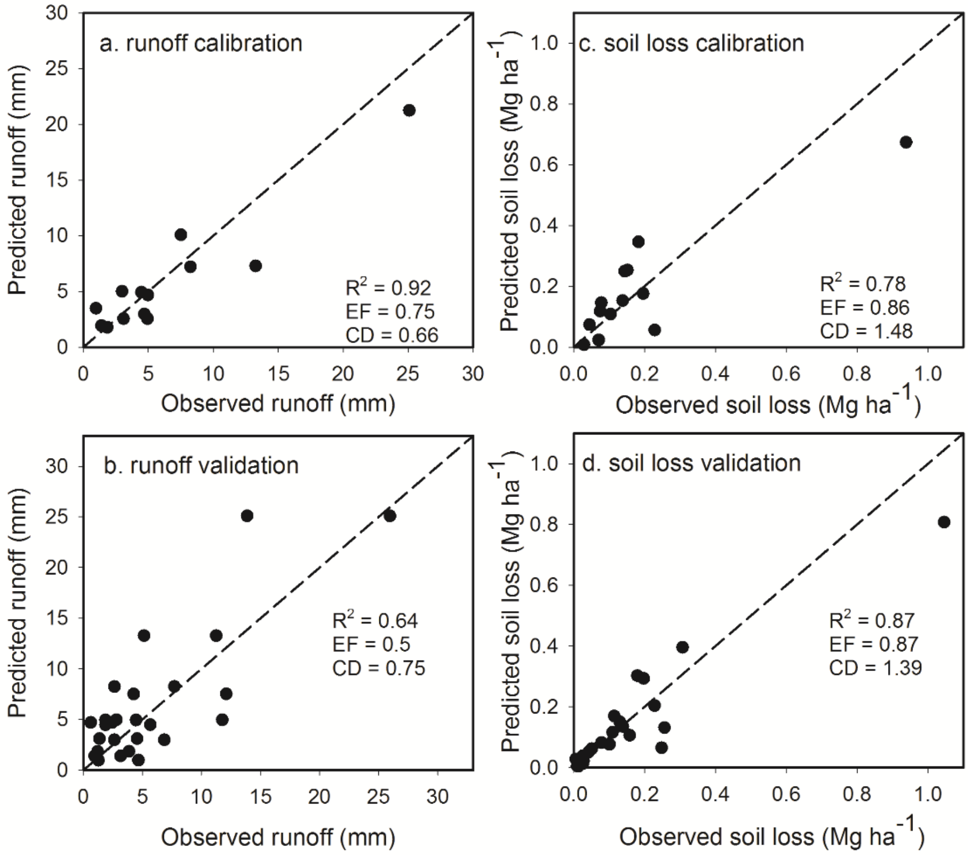


Figure 4.4 Model performance of different weeding management strategies in a 12-year old rubber plantation: (a) runoff (mm) under “twice-weeding” (Hs) for model calibration; (b) runoff (mm) under "clean-weeding" (H+) and "no-weeding" (H_0) for model validation; (c) soil loss (Mg ha^{-1}) under “twice-weeding” (Hs) for model calibration; (d) soil loss (Mg ha^{-1}) under "clean-weeding" (H+) and "no-weeding" (H_0) for model validation. EF: Modelling Efficiency. CD: Coefficient of Determination.

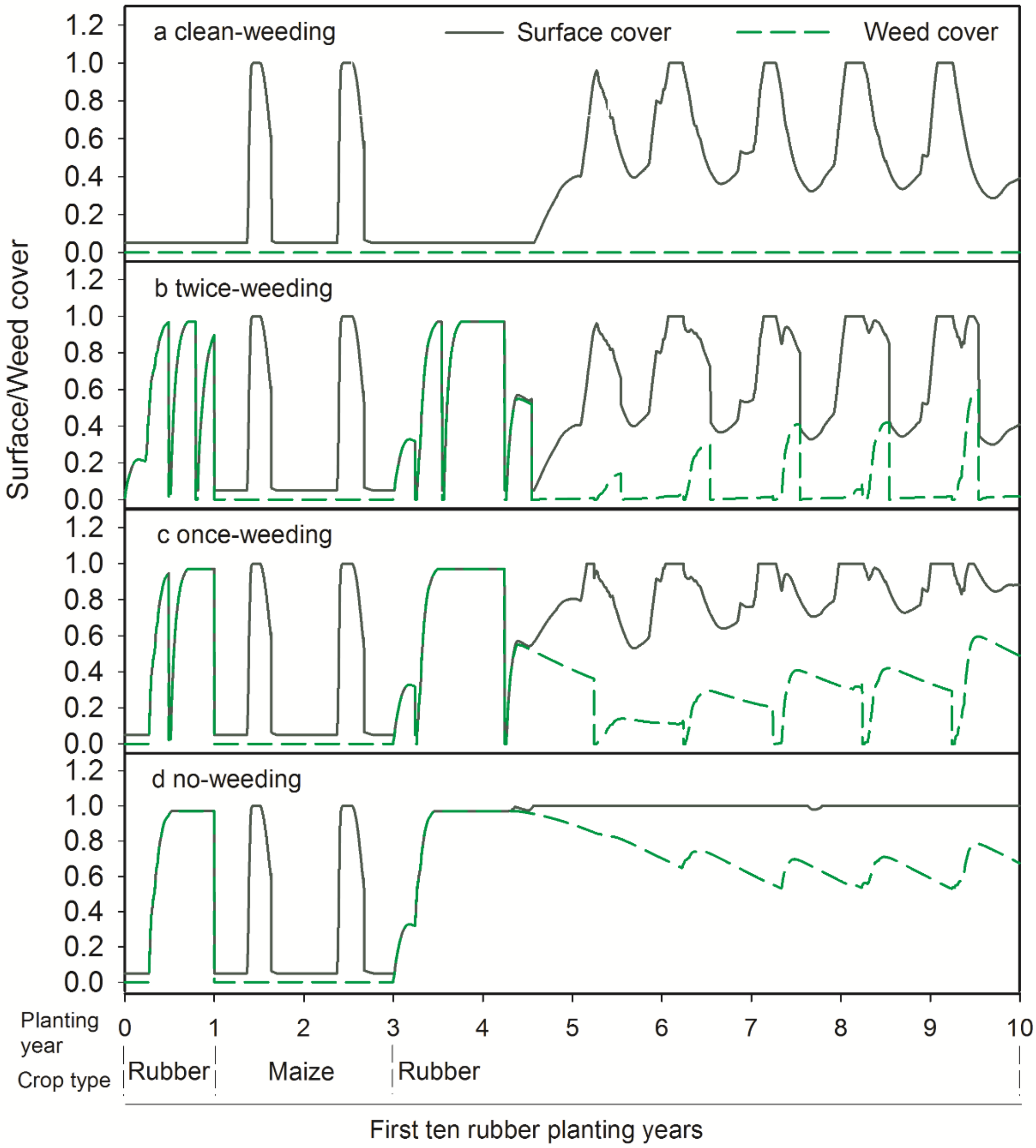


Figure 4.5 Simulated surface cover and weed cover change (including litter and weed biomass) under looped yearly (2014) rainfall pattern in rubber plantations for the first ten years under different weed management strategies. Maize was taken as land cover for the second and third year according to the local common practice of intercropping maize with small rubber trees. H+: "clean-weeding", no allowance of understory growth; Hs: "twice-weeding", herbicide application twice per year in mid-February and late July; H-: "once-weeding", herbicide application once per year in mid-February; H_0: "no-weeding", no herbicide application.

4.3.2 Effects of weed management on erosion control in rubber plantations

In the long-term runs, surface cover and soil losses differed between the various weed management options (Figures 4.5 and 4.6). Surface cover was simulated to be at a very low level (5%) under “clean-weeding” in the first and fourth year, but started increasing when rubber trees grew beyond the fifth year. It remained above 40% during the rubber mid-age period (5–20 years) but decreased sharply after reaching a maximum (98%) every year (Figure 4.5a). Surface cover under “twice-weeding” recovered quickly after weeding in the first year. The lowest level of surface cover (below 40%) was found during maize intercropping years (second and third years) and the latter half of the fifth year (Figure 5b). The level of surface cover simulated under “once-weeding” remained above 60% except for the two maize intercropping years (Figure 4.5c). The same trend was found under “no-weeding”, with the highest level of surface cover (almost 100%, Figure 4.5d).

Simulated annual soil loss with variable rainfall input presented analogous trends (Figure 4.6). “Clean-weeding” strongly increased soil loss in the first year, the maize intercropping years and the first two monoculture rubber years (fourth and fifth years) compared to the subsequent rubber phase. In contrast, soil loss was only strongly apparent in the maize intercropping years under reduced herbicide management (“once-weeding” and “no-weeding”). During the 20-year runs, using real rainfall input data, total soil loss was reduced by 70% under “no-weeding” and by 43% under “once-weeding”, while it increased by 33% under “clean-weeding”. Total simulated rainfall detachment (splash) within 20 years was reduced by 60% and 97% under “once-weeding” and “no-weeding”, respectively, compared to “twice-weeding”, while it increased by 35% under “clean-weeding” (Figure 4.7). Here we distinguished “splash detachment" and "splash to erosion". The former was the total amount of soil particles detached by raindrops; the latter combined raindrop-induced soil detachment and the amount transported out of the plot and therefore contributing to erosion. The key difference lies in whether the rainfall event caused runoff (by exceeding infiltration capacity) to transport detached soil particles down the slope.

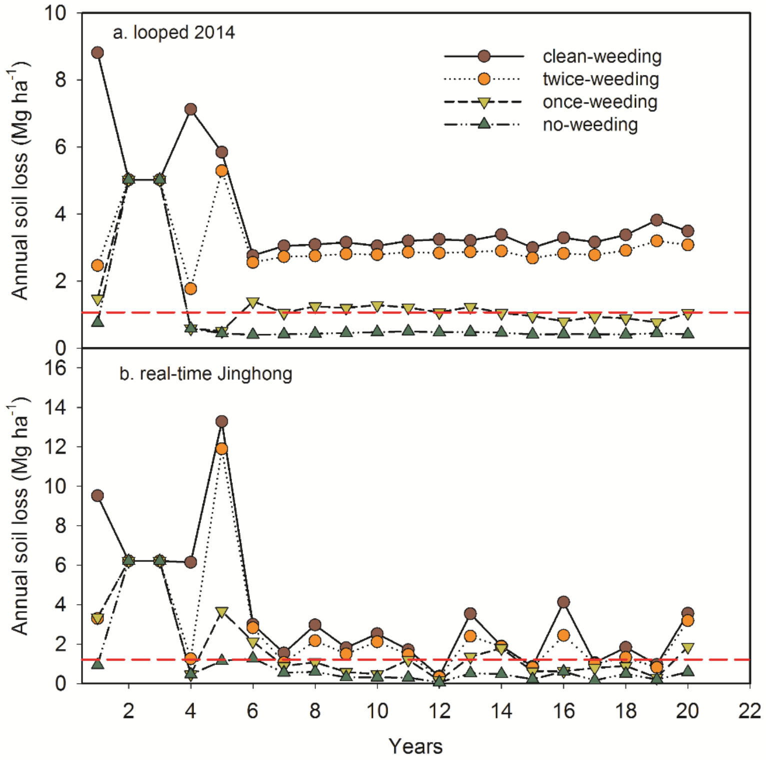


Figure 4.6 Simulated soil loss over 20 years in rubber plantations under different weed management strategies with (a) measured rainfall amount and intensity in 2014 repeated for 20 years, (b) daily real-time rainfall measured at Jinghong airport (1975 - 2014) and calculated rainfall intensity. Red line refers to the soil loss limit of 1 Mg ha^{-1} year^{-1} that is considered necessary to support long-term soil sustainability.

Simulated soil annual loss with variable rainfall input presented analogous trends (Figure 4.6). No allowance of weed growth (H+) strongly increased soil loss in the first year, the maize intercropping years, and the first two monoculture rubber years (4th and 5th year) compared to the subsequent rubber phase. Under twice-weeding management (Hs), the first year and monoculture rubber years (since the 4th year), excluding the 5th

year, presented less soil loss than the second and third maize intercropping years. In contrast, soil loss was only strongly apparent in the maize intercropping years under reduced herbicide management (H- and H_0). During the 20-year runs, using real rainfall input data, total soil loss was reduced by 70% under H_0, and by 43% under H-, while it increased by 33% under H+. Total simulated rainfall detachment (splash) within 20 years was reduced by 60% and 97% under H- and H_0, respectively, compared to Hs; while it increased by 35% under H+ (Figure 4.7). Here we distinguished "splash detachment" and "splash to erosion". The former was total detached soil particle amounts by raindrop; the latter combined raindrop induced soil detachment and transported amount out of plot therefore contributing to erosion. The key difference lies in whether the rainfall event caused runoff (by exceeding infiltration capacity) to transport detached soil particles down the slope. Runoff entrainment was reduced by 33% and 52% under H- and H_0, respectively, while it increased by 34% under H+ (Figure 4.7). Rainfall detachment in total contributed 17% and 18% to total detached soil under H+ and Hs respectively; and 14% under H-; while it only took up 1% to total detached soil under H_0 (Figure 4.7).

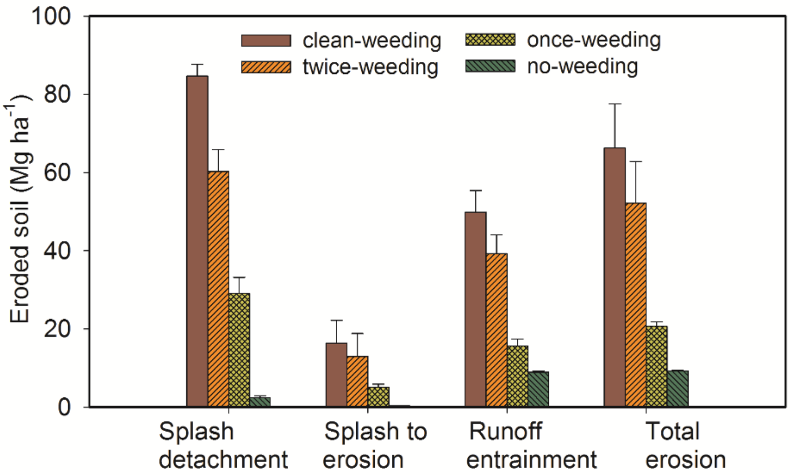


Figure 4.7 Simulated results of total detached soil particles by different sources (rainfall detachment and runoff entrainment) in rubber plantations under different weed management strategies during a 20-year simulation run.

4.4 Discussion

4.4.1 Updated LUCIA performance in predicting erosion in rubber plantations

Advantage of LUCIA is the ability to simulate dynamic vegetation growth and the interaction between management, vegetation growth and erosion (Marohn et al., 2013a). This is particularly important in assessing biological soil conservations such as strip cropping, cover crop and weeding management in our case. We updated the erosion module and applied it to evaluate weed management effects on soil conservation in rubber plantations. Results of surface cover simulations in young and mid-age rubber indicate that the model successfully represented weed growth by defining light competition with rubber trees through *LAI*-driven shading as well as water and nutrient constraints common to rubber and weeds. In particular, the model represented well the effects of different weeding strategies during rainy season on surface cover in rubber plantations of various age (Figure 4.3). Obviously, herbicide application caused a drop in surface cover in 4-year-old rubber; a decrease in older plantations was less evident, especially for the first weeding. This was caused by the contribution of litter cover to surface cover, which was not affected by herbicide. Along with litter decomposition, the effect of a second weeding in older plantations turned out to be more evident. The model failed to predict changes in surface cover during the late rubber phase (>20 years, Figure 4.3); therefore, additional factors affecting weed development and litter decomposition should be taken into consideration. Liu et al. (2018) reported that a change in weed species and resistance to glyphosate (the most common herbicide used in rubber plantations) are commonly observed in older rubber plantations in the study region. As a consequence, only minor reductions in weed cover are found after herbicide applications in old plantations. LUCIA, as an integrated process model of crop/tree growth and erosion, has a simplified weed simulation routine and implements full effectiveness of herbicide application. This simplification worked well to represent the situation during the first 20 years. Assuming that 20–25 years is a typical rotation length for rubber plantations, we concluded that the updated LUCIA still reached a satisfactory level in simulating the effects of long-term weed management in rubber plantations.

4.4.2 Significance of rainfall detachment processes in rubber plantations

The updated LUCIA results demonstrated that rainfall detachment contributing around 25% to total soil loss (Figure 4.7) is a non-negligible erosion source, as was also indicated by other field studies (Jiang et al., 2017; Liu et al., 2015). More importantly, the updated model estimated a cumulative amount of soil detached by the impact of rainfall splash in the range of 2–87 t ha^{-1} under different weeding strategies (long-term run), although only part of the detached material was effectively transported out of the field and thus contributed to soil erosion (total soil transported out of the field was 6–64 t ha^{-1}, Figure 4.7). Our current improvements to the model by including total splash effects could serve as a solid base for a more comprehensive assessment of detachment impacts on soil properties (e.g. soil texture or aggregate change by detachment).

The coefficient representing cover efficiency to reduce soil detachment (α) was calibrated as 2.5, slightly out of the range (5–15) proposed by Yu and Rose (1999). The suggested range of α (5–15) attributed protection only to runoff entrainment and neglected rainfall detachment, based on the basic assumption by Hairsine and Rose (1992). The updated LUCIA considers the effects of surface cover on both rainfall detachment and runoff entrainment; therefore, the proposed lower coefficient value (α)

is justified. The lower coefficient value of surface cover reflects the high degree of sensitivity of the dynamic multi-layer system to changes in surface cover. This has also been reported by different authors (Dunne et al., 1978; H. Liu et al., 2016; Rickson and Morgan, 1988) for field studies in various perennial cropping systems.

4.4.3 Erosion dynamics of rubber plantations under typical weed management strategies

“Twice-weeding” is the current major weed management strategy adopted by farmers in Xishuangbanna, China. Erosion in mid-age rubber under this strategy simulated by the updated LUCIA was 1.8 times higher than that simulated for young rubber (Figure 4.6a). These results were within the range (1.7–4) reported by Liu et al. (2018). Their field investigations showed that mid-age rubber has the longest period of erosion risk excluding establishing years. The simulation results of LUCIA, however, suggest that the transition period (the fifth year simulated by the model) from young to mid-age rubber has the greatest erosion risk. The model theoretically explained the low degree of surface cover by depression of weed growth due to light competition and insufficient litter cover from rubber trees that results in a high degree of erosion during the transition period (fifth year in the current study). It is noticeable that less soil loss was predicted under rubber in the first year than under maize (second and third year), attributed to our setting up a “twice-weeding” scenario. Practical management strategies of local farmers are mostly “clean-weeding” in the first year to protect rubber seedlings from weed competition. Though the model results implied efficient soil conservation under “twice-weeding”, good recovery of weeds from herbicide application may impede practical adoption of this management strategy by farmers. The first year is the seedling transplantation year. Therefore, other management strategies, such as intercropping with pineapple as suggested by Ulahannaan et al. (2014), should be implemented for soil conservation as well as seedling care in the first year.

The 5 Mg ha^{-1} year^{-1} soil loss simulated during maize cropping was similar to the value measured in maize intercropping with immature rubber (canopy radius < 50 cm) in Thailand (Khamkajorn et al., 2016). Relatively high erosion in maize cropping under twice-weeding suggested that additional management options, e.g. addition of straw mulch, should be considered to further reduce erosion in immature rubber intercropped with maize.

4.4.4 Role of weed management in soil conservation in rubber plantations

Weed management clearly alters changes in surface cover (Figure 4.5) and thus affects erosion dynamics in rubber plantations (Figure 4.6). Runoff entrainment normally decreases exponentially with increasing surface cover while rainfall detachment can be totally reduced by surface cover. “Clean-weeding” management is the most common management option adopted at the beginning of rubber planting, to protect the saplings from weed competition. From a soil conservation aspect, however, it completely removes potential protection from weeds and reduces surface cover strongly during the first five years until rubber trees shed sufficient litter which acts as effective soil cover. Annual soil loss under reduced herbicide application (“once-weeding” and “no-weeding", Figure 4.6b) was generally kept below the limit of 1 Mg ha^{-1} year^{-1} that is considered necessary to support long-term soil sustainability (Jürgens and Fander, 1993). Simulated high soil conservation efficiency under “no-weeding” was also proven by a 10-year field experiment by Abraham and Joseph (2015). Their study further proved improved soil health (soil OC, N, K, Mg) and no negative impact on latex yield under “no-weeding” practice. However, the surface cover (over 95%) and weed cover (60%) predicted by model results, coinciding with field observation of an increase in undergrowth biomass of 600% (Abraham and Joseph, 2015), revealed resulting dense undergrowth. This would be hardly acceptable by local farmers due to their concerns of potential danger from poisonous caterpillars, and reduced tree accessibility for tapping arising under such conditions. “Once-weeding” conserves soil well (Figure 4.6). Meanwhile, less weed cover (below 50%) under “once-weeding” (Figure 4.5c) implies overgrowth control of understory vegetation, therefore supplying a less favorable habitat for caterpillars. We recommend “once-weeding” as best-practice weed management, combining maintenance of certain ecological functions as well as acceptance by local farmers.

4.5 Conclusion

Assessment of the effects of management on erosion control is an essential component of sustainable land use strategies in Southwest China. We successfully simulated erosion using the updated LUCIA model, which incorporates a dynamic, multi-layer plant–weed–litter structure as well as rainfall detachment. The improved model was able to mimic soil loss under different weed management options in rubber plantations during one rotation length (20 years) reasonably well. The model simplified weed growth by considering water, nutrient (N, P, K) and light availability as *LAI* growth rate constraints and assumed high efficacy of herbicides. This simplification performed well for young and mid-age rubber plantations, but not for old rubber plantations due to increasing weed tolerance to the herbicide. Additional factors should, therefore, be taken into consideration if a detailed simulation of weed physiology in older plantations is required in future studies.

Scenario simulation results confirmed the important role of weed management in affecting erosion dynamics in rubber plantations. The model improved our understanding of erosion changes during a typical rubber rotation length (20–25 years) and highlighted some potential periods of a high degree of erosion under the current common "twice-weeding" weed management practice. During the early canopy closing period, depression of weed growth by the tree canopy and insufficient litter supply from rubber led to a low level of soil coverage, and therefore resulted in a high degree of erosion. "Once-weeding" and "no-weeding" both significantly reduced total soil loss by maintaining a high level of surface cover. The model results further implied that "no-weeding" largely protected soil from rainfall detachment by decreasing it to only 1% of total detached soil. However, this management option is unlikely to be adopted by farmers due to the long-term persistence of weeds if there is a high level of surface cover (over 95%). On the other hand, "once-weeding" is suggested as best practice to maintain a high level of surface cover (over 60%) while controlling overgrowth of understory vegetation by keeping weed cover below 50%. The updated LUCIA model can be regarded as a suitable tool for soil conservation planning and to support management decisions in rubber plantations.

Supplementary

Table S1 Input parameters for erosion module and their equations obtained from other modules of LUCIA. Further information could be found in Marohn et al., (2013a) and online documents (https://lucia.uni-hohenheim.de/en/85437) Parameters

	Module	Equations
$LAIRubber$ (Leaf area index of rubber, dimensionless)	Plant	$LAIRubber_t = LAIRubber_{t-1} + \Delta Wlv \cdot SLA \cdot 0.0001$
$LAIWeed$ (Leaf area index of weed, dimensionless)	Plant	$LAIWeed_t = LAIWeed_{t-1} + dLAIWeed_{open} \cdot (1 - CanCover) + dLAIWeed_{under} \cdot CanCover$
$CanCover$ (Canopy cover, %)	Plant	$CanCover = \frac{PlantDensity \cdot \pi \cdot CRadius^2}{10000}$
H_T (Trunk height of rubber tree, m)	Plant	$H_T = \frac{BiomassTrunk}{WoodDensity \cdot \pi \cdot \left[\left(\frac{DBH}{2}\right)^2 + \left(\frac{DBH}{2} \cdot TrunkShape\right)^2\right] \cdot 0.5}$
H_C (Canopy thickness, m)	Plant	$H_C = \frac{CRadius}{CanopyShape}$
Q_{tot} (Daily runoff, mm d^{-1})	Water balance	$Q_{tot} = R_{tot} - I_{tot} - IE_{tot}$
IE_{tot} (daily evapo-transpiration of intercepted water, mm d^{-1})	Water balance	$IE_{tot} = \frac{1}{3600} \cdot S \cdot \left(1 - \exp\left(-0.7 \cdot \frac{R_{tot}}{S}\right)\right)$
S (daily canopy storage mm d^{-1})	Water balance	$S = 0.8 \cdot LAI_{rubber} \cdot [1 - \exp(-0.4 \cdot LAI)] \cdot CanCover$
Q_{rate} (hourly runoff rate m h^{-1})	Water balance	$Q_{rate} = \left(\frac{Q_{tot}}{R_{tot}}\right) \cdot R_{int}/1000$

Table S2 Performance statistics for event-based runoff and soil loss simulations for short and long term runs corresponding to the field observation period from 2013 to 2014 in rubber plantations. "clean-weeding" (H+): no allowance of weed growth; "twice-weeding" (Hs): herbicide application twice per year on day 90 and day 200; "once-weeding" (H-): herbicide application once per year on day 90; "no-weeding" (H_0): no herbicide application.

Model performance	R^2 1[d]	EF[a] 1[d]	CD[b] 1[d]	RMSE[c] 0[d]
Splash potential simulation (event-based)	0.84	0.67	1.33	38.9[e]
Weed cover (monthly-based)				
Site calibration with "twice-weeding" (Hs) in 12Y rubber plantation in short term running	0.92	0.96	1.20	22.59[f]
Validation with "no-weeding" (H_0) in 12Y rubber plantation in short term running	0.45	0.97	1.30	17.12
Surface cover (monthly-based)				
Site calibration with "twice-weeding" (Hs) in 12Y rubber plantation in short term running	0.79	0.97	1.04	17.26[f]
Validation with "clean-weeding" (H+) and "no-weeding" (H_0) in 12Y rubber plantation in short term running	0.90	0.89	1.68	34.67
Validation with "twice-weeding" (Hs) in 4Y,12Y and 18Y rubber plantation in long term running	0.70	0.96	1.22	22.08
Validation with "twice-weeding" (Hs) in 25Y, 36Y rubber plantation in long term runs	0.03	0.75	2.59	49.92
Runoff (event-based)				
Site calibration with "twice-weeding" (Hs) in 12Y rubber plantation in short term running	0.92	0.75	0.66	42.18[g]
Validation with "clean-weeding" (H+) and "no-weeding" (H_0) in 12Y rubber plantation in short term running	0.64	0.50	0.75	69.66
Validation with "twice-weeding" (Hs) in 12Y rubber plantation in long term running	0.70	0.78	0.69	62.58
Soil loss (event-based)				
Site calibration with "twice-weeding" (Hs) in 12Y rubber plantation in short term running	0.78	0.86	1.48	48.38[e]
Validation with "clean-weeding" (H+) and "no-weeding" (H_0) in 12Y rubber plantation in short term running	0.87	0.87	1.39	54.84
Validation with "twice-weeding" (Hs) in 12Y rubber plantation in long term running	0.85	0.89	1.06	61.57

[a] Modelling efficiency [b] Coefficient of determination [c] Root mean square error [d] Value indicates perfect fit between observed and simulated data
[e] In $g\ m^{-2}$ [f] In % [g] In $mm\ d^{-1}$

Chapter 5

Soil conservation measures for the mitigation of land use change impact on sediment yield at watershed scale

– Case study of two small watersheds

Abstract

Agriculture conservation measures have gained their importance in reducing soil loss (on-site) and sediment export (off-site). The quantitative assessment on its effectiveness is of critical concern and assists greatly in cost-benefit analysis and decision-making in land management and landscape planning. In this research, we applied a paired watershed approach to monitor one-year sediment export of two watersheds with either a forest dominated (reference) or a mosaic (target) land use in Naban River National Watershed Natural Reserve (NRNWNR) in Xishuangbanna, South-West China. A distributed hydrological model (Land Use Change Impact Assessment, LUCIA) was calibrated and validated through field data from two watersheds, achieving satisfactory EF of 0.87, 0.72 for the runoff and 0.97, 0.96 for the sediment export, respectively. Agricultural management (business as-usual or conservation) was taken as the factor generating two scenario groups of the target watershed: mono-conservation and multi-conservation. Mono-conservation focused on soil conservation in newly appeared land use types, namely rubber plantations, and simulated different conservation measure effects (twice-weeding, once-weeding, no-weeding) on total sediment yield at the watershed scale. Multi-conservation applied conservation measures in major agricultural land uses, namely rubber (by different weeding strategies), maize (by adding residues) and tea (by adding residues). The model results simulated plot conservation efficiently reducing sediment yield by 18% - 49% at watershed scale. Multi-conservation strategy, namely once-weeding in rubber plantations, adding residues 4 t ha^{-1} in maize and tea managements, was able to reduce total sediment yield to the same level as reference value (0.43 Mg ha^{-1} and 0.42 Mg ha^{-1} from the target and reference watershed, respectively). We concluded that plot soil conservation provided an efficient tool to better manage mountain stream water quality by well controlling sediment yield, and multi-conservation in different agricultural types was required to fully compensate increased sediment export by agricultural expansion.

5.1. Introduction

Water erosion has both on-site and off-site effects. On-site impact is the reduction of soil fertility by the loss of nutrient-rich topsoil particles (Blanco and Lal, 2008; Fiener et al., 2008). Off-site effect is the transfer of sediments from upland and reduces storage capacity by silting-up deposition areas (Rompacy et al., 2002). Off-site effect can also deteriorate stream water quality by high loads of sediments, colloid and dissolved substances such as pesticides, herbicide (Ciglasch et al., 2005; Kahl et al., 2008). In most cases, erosion induced off-site problems are more serious as it threatens aquatic ecosystem functions immediately affecting people's life quality and security such as fishery, safe irrigation. This problem might be more severe in mountainous small watersheds as the local people tend to take the surface water as their drinking water resources. Therefore, they depend more on a healthy aquatic system with strict quality standards. However, large scale watershed management within integrated approaches may be difficult to apply in such a case. For example, typical watershed management methods as water protection zone and riparian buffer strip establishment, require scientific and detailed planning regarding zone/strip location, size and management regulations such as land use type (forest, bush or agriculture) and herbicide/fertilizer application. This centralized management highly depends on policy of decision-maker on land use management, who shall coordinate conflicts between upstream and downstream population as well as among landowners applying financial incentives. However, villagers in mountainous areas are normally marginalized from decision-making process. Therefore, challenges lie in agriculture expansion induced aquatic system deterioration and lacking appropriate watershed management. Appropriate agriculture soil and water conservation (SWC) techniques may play a more important role in such mountainous small watershed for the following reason: i) SWC at plot scale has been well studied for different crop types worldwide and offers simple but reliable techniques; ii) on-site SWC directly affects benefit for individual farmer's, therefore they may be easily accepted and implemented with less efforts from policy makers; iii) studies on integrated watershed management have proved that in most cases on-site SWC appears more effective than off-site measures in sediment control.

Assessment of the off-site effects by on-site SWC is laborious and expensive through conventional field methods. Therefore, such tools as erosion modelling should be able to quantify magnitude of SWC at both plot and watershed scale. Erosion modelling has shifted recent years from the plot scale to the catchment scale. Spatially-distributed and process-based erosion models subdivide catchments into smaller unites that either might be defined by users, such as in EUROSEM (Morgan et al., 1998), KINEROS2 (Smith et al., 1995), WEPP (Flanagan et al., 2001), WaTEM/SEDEM, or might consist of pixels in a grid, as for example in ANSWERS (Beasley et al., 1980), LISEM (De Roo et al., 1996). However, the above-mentioned erosion models simulate sediment control effects by on-site SWC based on artificially reduced soil loss at each pixel. Such an effect can be reached by e.g. increasing ground cover, leaf area index, without simulating direct management advises (e.g. increasing residues to a certain amount, intercropping with certain crops). This approach hampers application of multi-conservation measures corresponding to different land uses. We applied in our work the more advanced Land Use Change Impact Assessment (LUCIA) model, which is a dynamic and spatially explicit landscape-scale model. LUCIA integrates hydrological, geophysical, soil organic matter and vegetation growth routines in a single framework therefore is able to simulate different on-site conservation management effects (e.g. residue, weeding, intercropping) on sediment control.

Although SWC has been widely studied and assessed by field investigations or modeling, to our best knowledge, the evaluation loosely links SWC to aquatic ecosystem and includes its impact on local villagers. In this study, we have chosen two neighboring sub-watersheds with different land cover: one is forest dominated as reference; the other has a mosaic land cover as target). The major objective was to attempt an assessment of simple on-site SWC on the aquatic ecosystem by combining field investigation and model simulation. Specifically, we aimed at 1) evaluating surface water quality of two different land cover sub-watersheds by field investigation; 2) assessing different on-site SWC effects on total sediment control by LUCIA modelling.

5.2. Field investigation

5.2.1 Study site

Our study area has been Nanbanhe Watershed National Nature Reserve (NRWNNR, Figure 5.1) in Xishuangbanna, Yunnan province, southwest China. The region has a typical monsoon climate characterized by a distinct rainy season from May to October and a dry season from November to April. Sixty to ninety percent of the precipitation is distributed during the rainy season. The annual rainfall varies from 1200 to 1700 mm, and the annual mean temperature range is 18–22 °C. Observations in NRWNNR indicate a high dependence of local residents on aquatic services and functions supplied by surface runoff. Streams are the resource of irrigation, fishery and entertainment (e.g. swimming). Moreover, more than half of the villages in the study area use open springs or creeks as drinking water sources. Accompanied with agriculture (including rubber) expansion in recent years, local farmers have observed a reduction of drinking water quality, especially a high turbidity after rain events.

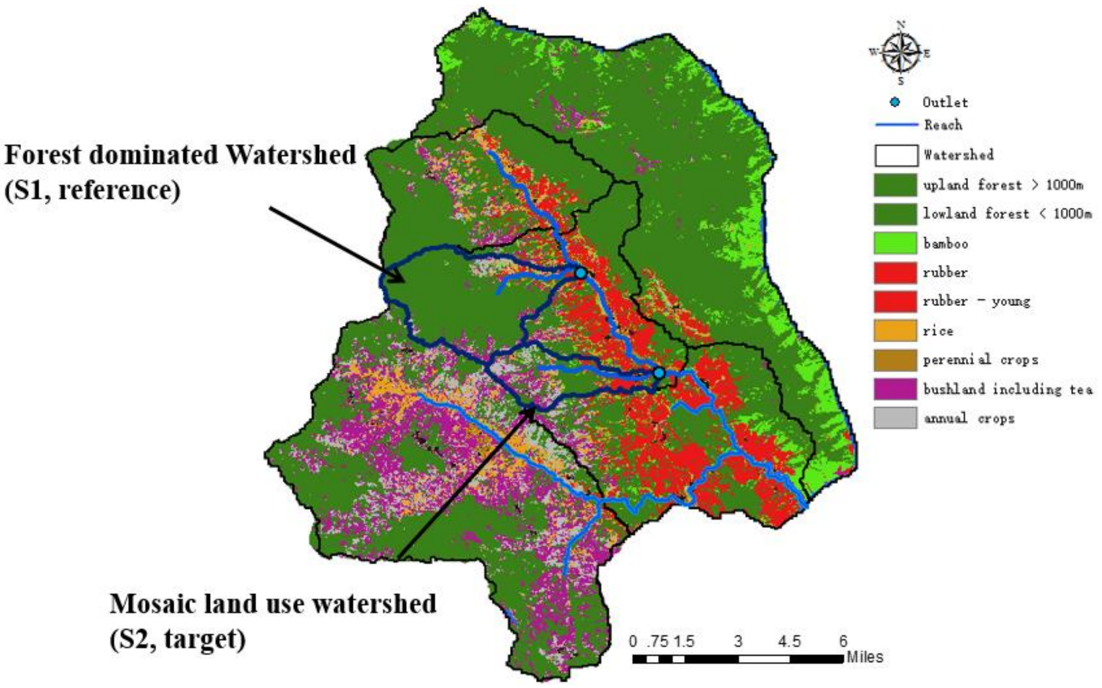


Figure 5.1 Overview of land cover of the two selected watersheds. S1 watershed with forest dominated (83% forest cover) land use was taken as reference; S2 watershed with mosaic land use (47% forest, 18% tea, 17% maize and 12% rubber) was our target watershed.

We selected two sub-watersheds (S1 and S2) which have similar soil types and average slope but a major difference on land covers (Figure 5.1). S1 (1608 ha) is forest dominated watershed with over 80% forest cover and was selected as the reference watershed; S2 (692 ha, 22°04′ - 22°17′ N, 100°32′ - 100°44′ E) is a typical Southeast Asia’s upland landscape covered with mosaic land use and selected as the target watershed. Major land uses in the target watershed were: (1) natural forest, 42%; (2) maize, 18%; (3) bush and tea, 16%; (4) Rubber plantations, 12%.

5.2.2 Watershed monitoring

We continuously monitored discharge and sediment yield at the outlets of both watersheds (Figure 5.1). The discharge of the streams was measured using the automatic recording station consisting of a sharp crested contracted weir with a V-notch weir and a stilling well with an automatic water-level recorder (Campbell Scientific CS451, pressure transducer SDI-12/RS-232) equipped with a data logger (Campbell Scientific CR200). The water level data were converted to discharge using a control rating curve (Walkowiak et al., 2013).

Turbidity is a basic and important indicator to quantify the aquatic ecosystem health. Sediments content is a main factor determining turbidity. The sediment particles include most contaminants and, thus, turbidity can be related to water quality. Therefore, reduction of sediment yields is one of the most important goals for watershed management. Together with the automatic water level recording stations, we installed automatic turbidity recorders (Campbell Scientific OBS-3+). Suspended sediments (SS) were collected by taking water samplers at time intervals from 2 min to 1 h depending on turbidity change during storm events. Therefore, a relationship between SS concentration (g m^{-3}) and turbidity (NTU) was established separately for both streams and used to transfer the continuous measured turbidity into SS concentration (see Supplementary Figure S1). Suspended load was calculated as the product of discharge and SS concentration. No obvious sedimentation was observed near our monitoring stations as the sediment texture was clay-dominated and suspended sediment concentration fell in the medium range. Therefore, total sediment yield in the watershed was calculated as the suspended load in the streams.

5.3. LUCIA model - watershed simulation concept

The Land Use Change Impact Assessment model (LUCIA, Chapter 4, https://lucia.uni-hohenheim.de/en) contains plant, hydrology and erosion subroutines with inner interactions. Plant subroutine is based on the concept derived from a PCRaster version of the WOrld FOod STudies (WOFOST) model and simulates process-based plant growth on a daily time step depending on photosynthesis, water and nutrient constraints

(Supit, 2003). Daily carbon assimilation in LUCIA depends on photosynthetically active radiation (*PAR*), leaf area index (*LAI*), crop- and development specific maximum assimilation rates (*AMD*) and day length. Net assimilation rates are extracted from daily assimilation by deducting respiration rates; and converted to biomass production by the parameter *Eff_C2Biomass*, which describes conversion efficiency of assimilated carbohydrates into biomass. Total biomass production is then partitioned to different plant organs (e.g. leaves, roots). Land uses in our two watersheds include annual crops (e.g. maize, rice) and perennial crops (e.g. rubber plantation). LUCIA separately considers these two types of crops and distinguishes annual single-layer structure and perennial multi-layer structure impact on erosion during simulation. This difference is simulated by defining surface cover (*SF*) for soil protection separately between annual and perennial crops. *SF* is the crop canopy, weed and surface litter cover for annual crops; while in perennial crops it includes only the weed and surface litter (details in Chapter 4, section 4.2.2). Canopy effects of perennial crops are considered in the hydrology subroutine (next section) to simulate canopy throughfall (amount and density) and further incorporated into the erosion subroutine to simulate changes in raindrop kinetic energy of canopy throughfall from free rainfall.

5.3.1 Hydrology subroutine

5.3.1.1 Plot scale

Daily rainfall amount (R_{tot} in mm d^{-1}) and hourly rainfall intensity (R_{int} in mm h^{-1}) are required hydrological inputs in LUCIA.

Rainfall is taken as fully free rainfall for annual crop land types while is partitioned into free rainfall and canopy throughfall for perennial crops. For perennial land use such as rubber plantations, free rainfall intensity (FR_{int} in mm s^{-1}) is calculated according to Lloyd et al. (1988) and Gash et al. (1995):

$$FR_{int} = R_{int} \cdot \frac{1-c}{3600} \tag{5.1}$$

with R_{int} hourly rainfall intensity (mm h^{-1}) and c the tree canopy cover, using the constant 3600 to convert from mm h^{-1} into mm s^{-1}.

Based on water balance concept by the sparse Rutter model (Gash et al., 1995):

$$FR_{tot} + CD_{tot} + IE_{tot} = R_{tot} \tag{5.2}$$

with FR_{tot} the daily free rainfall (mm d^{-1}), CD_{tot} the daily canopy drainage (mm d^{-1}), IE_{tot} the daily interception evapotranspiration (mm d^{-1}) estimated by the equation:

$$IE_{tot} = \frac{1}{3600} \cdot S \cdot \left(1 - \exp\left(-0.7 \cdot \frac{R_{tot}}{S}\right)\right) \tag{5.3}$$

with S the daily canopy storage (mm d^{-1}):

$$S = 0.8 \cdot LAI \cdot [1 - \exp(-0.4 \cdot LAI)] \cdot c \tag{5.4}$$

with LAI the leaf area index of tree (dimensionless), c the tree canopy cover. LAI of trees is simulated in the LUCIA plant subroutine determined by change of leaf biomass, extracted from total biomass change by a plant specific partitioning ratio, and specific leaf area SLA. Tree canopy cover (c) is calculated as average cover percentage, determined by the crown radius and the planting density (Marohn, 2009).

The canopy throughfall (CR_{tot} in mm d^{-1}) is estimated through partitioning of the canopy draining (CD_{tot}) by the drainage partitioning coefficient a (Gash et al., 1995). Namely:

$$CR_{tot} = a \cdot CD_{tot} \tag{5.5}$$

The canopy throughfall intensity (CR_{int} in mm s^{-1}) is estimated based on the scaling technique of Yu et al. (1997):

$$CR_{int} = \left(\frac{CR_{tot}}{R_{tot}}\right) \cdot R_{int} \cdot \frac{1}{3600} \tag{5.6}$$

Pixel runoff (Runoff_{pixel}, mm d^{-1}) is calculated as sum of hortonian flow and saturation overflow for both annual and perennial crops. Hortonian flow is the remainder of rainfall after interception, infiltration, deep infiltration and surface storage have been subtracted. Saturation overflow is added after the final soil water balance if the soil profile is filled with water (Marohn and Cadisch, 2011).

5.3.1.2 Watershed scale

LUCIA aims at simulating small catchments up to about 30 km^2 on a pixel-based grid. Therefore on the landscape level streams in some cases are too small to be detected from satellite data and classified in land use map. LUCIA creates a stream map by defining the effective slope length (LS_{eff}, m) reaching minimum travel distance 200 m:

Streams = ifthenelse (LS_{eff} >= 200, boolean (1), 0) (5.7)

Namely if the effective slope length (LS_{eff}, m) calculated for the pixel equaled or was larger than 200 m, then the stream was formed in this pixel by assigning 1 as stream value. If the effective slope length (LS_{eff}, m) of the pixel was smaller than 200 m, then the stream was not formed in this pixel by assigning 0 as stream value.

The effective slope length (LS_{eff}, m) was calculated by accumulative slope length (LS_{Ldd}, m) along the local drain direction (*Ldd*) and slope steepness (S, m m^{-1}):

$$LS_{eff} = LS_{Ldd} * S \tag{5.8}$$

with LS_{Ldd} (m) calculated by accumulating distance of neighboring pixel along the *Ldd*, with the operator *slopelength* by PCRaster, which is a dynamic modelling system for spatio-temporal environmental models (http://pcraster.geo.uu.nl) :

$$LS_{eff} = slopelength\ (Ldd,\ 1) \tag{5.9}$$

When a stream formed, LUCIA assumed part of pixel runoff going into the stream along the Ldd by the PCRaster function *accufractionstate*:

$$StreamIn = accufractionstate(Ldd,\ Q_{pixel},\ StreamRatio) \tag{5.10}$$

$$Stream = accufractionflux(Ldd,\ StreamIn,\ 1) \tag{5.11}$$

Following this functionality, certain ratio (*StreamRatio*) of pixel runoff (Q_{pixel}, mm d^{-1}) was assigned temporarily storing in the grid cell (*StreamIn*, mm d^{-1}) by the operator *accufractionstate*. Then this part of runoff (*StreamIn*) accumulated to form stream by the operator *accufractionflux*. The other part formed the landscape runoff and accumulated along the *Ldd*. It was calculated as temporary landscape runoff ($Q_{landscape}$, mm d^{-1}) by the operator *accufractionflux*:

$$Q_{landscape} = accufractionflux(LDD,\ Q_{pixel},\ StreamRatio) \tag{5.12}$$

Final landscape runoff (Q_{total}, mm d^{-1}) was calculated by subtracting potential infiltration on the landscape level ($I_{potential}$, mm d^{-1}):

$$Q_{total} = Q_{landscape} - I_{potential} \tag{5.13}$$

$I_{potential}$ (mm d^{-1}) was calculated based on updated water content in soil after infiltration on the pixel level.

In order to keep accordance with the erosion subroutine, a downscaling technique (Lippe et al., 2014) was applied to calculate a time-weighed discharge rate (Q_{dis} in m^3

m^{-1} s^{-1}) from daily runoff rate (Yu and Rose, 1999):

$$Q_{dis} = \frac{L \cdot Q_{rate}}{3600} \tag{5.14}$$

with L slope length (m) and Q_{rate} hourly runoff rate (m h^{-1}), 3600 is the constant to convert from m h^{-1} into m^3 s^{-1}. Hourly Q_{rate} is calculated from daily Q_{tot} (m d^{-1}) by the same scaling technique as canopy throughfall (Eq. (5.6)):

$$Q_{rate} = \left(\frac{Q_{tot}}{R_{tot}}\right) \cdot R_{int}/1000 \tag{5.15}$$

with Q_{tot} daily runoff rate (mm d^{-1}), R_{tot} daily rainfall (mm d^{-1}) and R_{int} hourly rainfall intensity (mm h^{-1}). 1000 is the constant to convert from mm into m.

5.3.2 Erosion subroutine

LUCIA follows the steady-state concept proposed by Misra & Rose (1996) and considers rainfall detachment, sediment entrainment, sediment re-entrainment and deposition of sediments in erosion simulation process.

5.3.2.1 Plot level detachment

At the plot level, the model assumes that in an erosion event, sediment concentration does not differ greatly from the equilibrium condition when the mass of the deposited layer remains constant with time. We adopted updated LUCIA erosion routine inheriting from Chapter 4, namely detached soil for each pixel cell coming from rainfall detachment and runoff entrainment. Detailed description on plot level erosion simulation is described in Chapter 4 section 4.2.1 – 4.2.2.

5.3.2.2 Watershed scale

LUCIA follows assumptions of ERODEP model (EROsion and sediment DEPosition, Lippe et al., 2014) and considers deposition and re-entrainment for watershed scale simulations. Deposition ($d_{(i,j,k)}$, dimensionless) is assumed always existing and limited by sediment settling velocity ($v_{(i,j,k)}$, m s^{-1}) and runoff velocity:

$$d_{(i,j,k)} = \frac{v_{(i,j,k)} \cdot f_{(i,j,k)}}{V} \tag{5.16}$$

with $d_{(i,j,k)}$ the deposition ratio of total sediment emerging in the cell per sediment size-class i, j and k. $v_{(i,j,k)}$ the settling velocity (m s^{-1}) per sediment size-class i, j and k. V the runoff velocity (m s^{-1}) calculated. The total deposition ratio d_t was calculated as the sum of each sediment size-class:

$$d_t = d_i + d_j + d_k \tag{5.17}$$

Instead of assuming a constant building up of sediment deposits, LUCIA uses the principle of ERODEP and takes r_t as re-entrainment ratio to move previously deposited sediments back into flow (Hairsine and Rose, 1992):

$$r_t = \left(\frac{H \cdot F \cdot \Omega}{g \cdot D}\right) \cdot \left(\frac{\sigma}{(\sigma - \rho)}\right) \cdot md_t \tag{5.18}$$

with H the fractional shielding of the original soil by the deposited layer assumed to be 0.9 (Heilig et al., 2001) and md_t the net deposition ratio with respect to sediment emerging in the cell after re-entrainment (Lippe et al., 2014):

$$md_t = d_t - r_t \tag{5.19}$$

LUCIA models the sediment cascade on the watershed scale based on detached soil in each pixel cell S_D (g m^{-2}) and the net deposition ratio md_t, with the PCRaster function *accufractionflux*:

$$S_{cum} = accufractionflux(Ldd, S_D/100, 1 - md_t) \tag{5.20}$$

Following this functionality, sediments flowing into a grid cell S_{cum} (Mg ha^{-1}) are the accumulation of detached soil particles (S_D, g m-2) in its upstream neighbors with transport of a certain fraction $(1 - md_t)$, 100 the constant to transfer g m^{-2} to Mg ha^{-1}.

The total sediment export of the whole watershed (S_{total}, t) is calculated as:

$$S_{total} = S_{cum} \cdot pixelsize/10000 \tag{5.21}$$

Pixelsize is determined by the input map resolution (e.g. land cover map, soil map)

Net deposition is the other part staying in the grid cell:

$$DP_{net} = accufractionstate(Ldd, S_D/100, 1 - md_t) \tag{5.22}$$

One of the advantages of LUCIA was the identification of soil erosion and sediment deposition hotspots in a spatially-explicit environment. The net sediment balance N_t (Mg ha^{-1}) at individual grid cell element for each single event was calculated as:

$$N_t = DP_{net} - S_D \tag{5.23}$$

with DP_{net} (Mg ha^{-1}) the net deposition with inflowing sediment from upstream, S_D

(Mg ha^{-1}) the detached soil at the pixel cell.

The cumulative net sediment balance (N_{cum}, Mg ha^{-1}) is calculated as the sum before:

$$N_{cum} = \sum_{t=0}^{n} N_t \quad (5.24)$$

If $N_{cum} > 0$, then the corresponding cell is a deposition hotspot; if $N_{cum} < 0$, the corresponding cell is an erosion hotspot.

5.4. Model application

5.4.1 Calibration of land use and climate parameters at one pixel (plot) level

A one pixel (plot) model area with the average watershed slope 55% was created to simulate soil losses in major land uses (maize, forest, young rubber, mid-age rubber, tea). The "pixel model" was roughly calibrated based on annual soil loss from literature (Table 5.1) by adjusting vegetation/crop parameters. Vegetation/crop parameters for forest and maize were taken from a study in Ban Tat, Northern Vietnam by Ayanu et al. (2011) considering similar climate conditions and planting patterns with our study site; parameters for young and mid-age rubber plantations were from Yang et al. (2017) which were calibrated and validated under the conditions in 2014 in the same study site; parameters for tea were taken from the default database of LUCIA. Climate information (temperature, radiation, evapotranspiration) was obtained based on data from Jinghong airport in 2014. Daily rainfall amount and hourly rainfall intensity were from the locally installed rain gauge (Liu et al., 2016).

Table 1.1 Comparison of pixel simulated soil loss with measurement in literature of different land use types

Land use type	Simulated soil loss (Mg ha^{-1} y^{-1})	Measured soil loss (Mg ha^{-1} y^{-1})	Reference
Forest	0.21	0.05-1.35	Kateb et al. (2013) Li et al. (2006)
Mid-age rubber	2.40	2.90	Liu et al. (2016)
Young rubber	1.80	0.94-5.32	Pansak et al. (2016)
Maize with no conservation	7.25	4-7.50	Kateb et al. (2013) Tuan et al. (2014)
Maize with conservation by adding residues 4 Mg ha^{-1}	2.5	0.83-3.5	Kateb et al. (2013)
Tea with no conservation	4.45	2.00-17.00	Kateb et al. (2013) Liski et al. (2003)
Tea with conservation by adding residues 4 Mg ha^{-1}	1.9	0.5-2.9	Liski et al. (2003)

5.4.2 Watershed simulation

5.4.2.1 Model input data for watershed scale simulations

LUCIA model builds on different spatial and climate information as common inputs. In our case, a land use map for 2014 and a digital elevation model (DEM) with 30 m resolution were available as geo-referenced datasets in ArcGIS format. A detailed locally measured field soil map was not available for the simulated watershed. Considering that Ferralsol is the main soil type in the watershed (Yang et al., 2016), we used for simulation a uniform soil map with the physical properties evaluated during field investigations in this region (Liu et al. 2016). Vegetation/crop parameters for major land uses (maize, forest, young rubber, mid-age rubber, tea) were obtained from pre-calibration pixel simulations; parameters for other land uses (rice) were taken from the LUCIA database. Crop management settings were based on interviews with local people in the watershed. Herbicide application in rubber plantations was set as twice per year, on 90^{th} (mid-February) and 210^{th} (late July) day, respectively. Climate input was the same as for one pixel simulation.

5.4.2.2 Model calibration and validation of watershed scale

The model was firstly calibrated for the S2 watershed by the modification of the parameter *StreamRatio* to best fit model outputs (discharge and total sediments export) and field measurements. Then the calibrated model was applied on S1 watershed for validation with corresponding changes in maps (land use, area, soil, DEM, LDD) and climate (rainfall) inputs. Other parameters (crop management setting, StreamRatio) were all kept the same as for S2 watershed simulations. Both model calibration and validation were based on an event-based resolution by comparing simulated with measured runoff and total sediment yield in the stream.

Model performance was assessed by calculating R^2, modelling efficiency (*EF*) and root mean square error (*RMSE*) as described in Chapter 4 section 4.2.3.4.

5.4.3 Design of plot soil conservation scenarios

We created two types of scenarios (in total 7 scenarios, Table 5.2) based on type i) mono-conservation management in rubber plantations. Despite various land uses in our study site, rubber plantations have attracted particularly high attention by their recent dramatic expansion. Thus, a number of improved rubber plantation management strategies have been proposed and implemented to ameliorate ecological functions. The first type of scenario was to test how the improvement of rubber management may affect watersheds' water quality; ii) Multi-conservation management was a second scenario type. Considering the specific mosaic land use character of the target watershed, we designed a multi-conservation management strategy by assuming conservation activities in major agriculture land types, namely, maize, tea and rubber. Such scenarios enabled us to test without traditional centralized watershed management (e.g. water protection zone, buffer zone), if implementation of plot soil conservation is sufficient to improve stream water quality. We simplified the water quality indicator as total sediment export to not complicate modelling process and applied different scenarios on S2 (mosaic land use) took S1 while taking measured sediment yield in S1 (forest dominated) watershed as the reference.

Table 5.2 Scenario runs to assess the impact of on-site different soil conservation measures on sediment yield control of the mosaic land cover watershed (S2). "twice-weeding": herbicide application twice per year on day 90 and day 200; "once-weeding": herbicide application once per year on day 90; "no-weeding": no herbicide application.

Scenarios	Scenario Number	Land use	Management
Baseline	1	Rubber	twice-weeding
		Tea	twice-weeding; no residue
		Maize	No burn; no residue
Mono-conservation*	2	Rubber	twice-weeding
	3	Rubber	once-weeding
	4	Rubber	no-weeding
Multi-conservation**	5	Tea	Add residue by 2 Mg ha^{-1}
		Maize	Same as baseline
	6	Tea	Same as baseline
		Maize	Add residue by 4 Mg ha^{-1}
	7	Tea	Add residue by 2 Mg ha^{-1}
		Maize	Add residue by 4 Mg ha^{-1}

* Scenario Mono-conservation only improves soil conservation in rubber plantation while managements in other land (maize, tea) remain the same as baseline.

** Scenario Multi-conservation contains soil conservation in rubber plantation as H- and stepwise applies soil conservations in tea and maize.

For type i) scenarios, we only changed weeding strategies in rubber plantations while keeping the same settings for other land uses. Weed management effects on soil conservation in rubber plantation have been well proved by our field experiment (Liu et al., 2016) and then well simulated at pixel scale (Liu et al., 2018). Same weeding strategies as plot level simulations were adopted here, namely: i) "no-weeding", no herbicide applied through whole year leading to 3 Mg ha^{-1} soil loss; ii) "once-weeding", herbicide applied once per year, on the 90^{th} day (mid-February) leading to 2.3 Mg ha^{-1}

soil loss; iii) “twice-weeding”, herbicide applied twice per year on the 90th (mid-February) and 210th (late July) day leading to 0.5 Mg ha^{-1} soil loss.

For type ii) scenarios, we adopted suggested weeding management options, namely “once-weeding” strategy, in rubber plantations and complemented soil conservation with a) in maize by adding residues 4 Mg ha^{-1}; b) in tea by adding residues 2 Mg ha^{-1}; c) in both maize and in tea by adding residues 4 and 2 Mg ha^{-1}, respectively. As we did not implement a field conservation experiment in these two land uses (maize and tea), soil conservation measures in maize and tea were referenced to literature, which has been proven efficient with no conflict to crop yields. Conservation in maize and tea firstly was tested in pixel simulation and compared to literature value; then applied at watershed scale.

5.5. Results and Discussion

5.5.1 Impact of land cover on total sediment yields in two watersheds

Good consistency between measured peak sediment and rainfall events in both watersheds indicates erosion as the major contributor to deteriorate aquatic environment. Some exceptions, on the other side, present non-uniform rainfall distribution across the whole watershed, which is the typical for uneven precipitation distribution in mountainous areas (Figure 5.2). For instance, rainfall (density and amount) recorded in S1 were similar on 24th August (16 mm h^{-1}, 52 mm) and 26th September (18 mm h^{-1}, 51 mm) while discharge was much higher on 24th August (2.3 m^3 s^{-1}) than 26th September (1.4 m^3 s^{-1}). Total sediment export in both watersheds (0.43 and 1.62 Mg ha^{-1} y^{-1}, respectively) is comparable with typical fallow/plantation dominated small watersheds in Southeast Asia (0.3 – 2.7 Mg ha^{-1} y^{-1}). S2 watershed export of total sediments was around three times (1.62 Mg ha^{-1} y^{-1}) higher than that of S1 (0.43 Mg ha^{-1} y^{-1}). Only three relative large events were detected in S1 which produced sediment slightly above 0.1 Mg ha^{-1}; while over 85% peak events (13 out of 15) in S2 produced sediments above 0.1 Mg ha^{-1}with a range of 0.1 – 2.8 Mg ha^{-1}. Noticeable, sediment yields by both watersheds were below the 3 Mg ha^{-1} y^{-1} considered as a tolerable soil loss under tropical conditions proposed by soil scientists (Valentin et al., 2008); as well as under 5 Mg ha^{-1} y^{-1} as tolerable soil loss according to “*Chinese standards for classification and gradation of soil erosion*”. Nevertheless, from the aquatic ecosystem aspect, high turbidity (over 800 NTU) caused by sediments appearing in peak events of S2 still strongly weakened aquatic ecosystem functions (e.g. fishing) and services as a resource of both domestic water and entertainment (e.g. swimming).

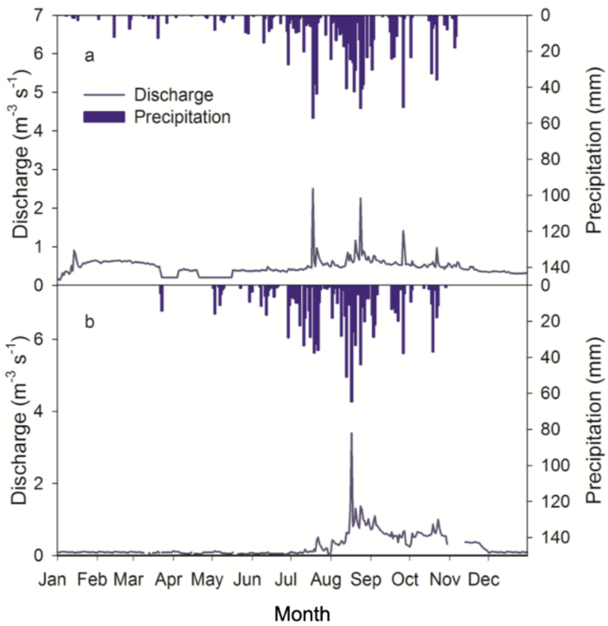


Figure 5.2 Field measured daily rainfall and discharge. a) refers to S1 (forest dominated) watershed; b) refers to S2 (mosaic land use) watershed.

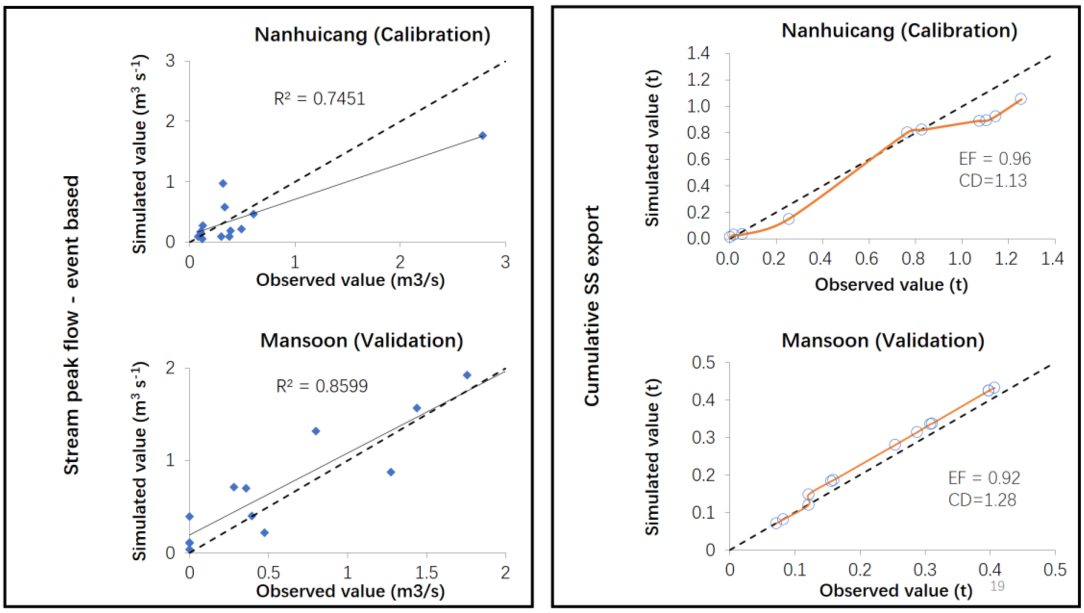


Figure 5.3 LUCIA simulation performances of runoff and sediment yield by comparing event-based field observation with model simulation results: (A) referring to runoff (mm) in S2 watershed for model calibration; (B) referring to sediment yield (Mg ha^{-1}) in S2 watershed for model calibration; (C) referring to runoff (mm) in S1 watershed for model validation; (D) referring to sediment yield (Mg ha^{-1}) in S1 watershed for model validation. EF: Modelling Efficiency. CD: Coefficient of Determination.

5.5.2 Baseline simulation

Table 5.1 presented the pixel simulated annual soil loss of different land uses compared to soil loss ranges found in literature. After calibration at pixel level, the model was further calibrated (S2) and validated (S1) at the watershed scale for runoff and total sediment yield on an event-based resolution (Figure 5.3). In total 12 out of 15, and 13 out of 18 events were detected by the model during calibration and validation, respectively. LUCIA aimed to simulate watershed smaller than 3000 ha, therefore, it assumed an even rainfall distribution within the watershed. In our study site, as discussed above, rainfall distribution is uneven due to mountainous topography for both watersheds. This explains the missing events by model simulation. The area of S1 is larger than S2 and, therefore, higher variability in rainfall distribution leads to more missing events, which were not detected by the model. The 13 detected events in S1 took up 85 % of total annual sediment yields and 12 detected events in S2 took up 80%. Moreover, both simulated runoff and total sediment yield match well with field observations as demonstrated by an EF coefficient of 0.70 and 0.71 for runoff and sediment yield during calibration, respectively; and 0.83 and 0.95 for runoff and sediment yield in validation phase, respectively. This is important, as different hydrological routines do not necessarily provide a perfect model fit (Clark et al., 2009). As an integrated model, LUCIA is able to capture major events within a given validation criteria. The model gave better performance in validation phase than calibration as the land use cover of S1 (for validation) is less diverse and dominated by forest compared to S2 (for calibration). Figure 5.4 presented the simulated spatially explicit annual (for 2014) soil erosion and sediment deposition patterns in the two watersheds. Net soil loss patterns corresponded well to land use types with a maximum soil loss rate of 13 Mg ha^{-1} in maize with 75% slope and a minimum soil loss rate of 0.2 Mg ha^{-1} in forest with 15% slope (Figure 5.4a). Simulations illustrated that net depositional areas were predominantly predicted in streambeds (Figure 5.4b). Our two watersheds are located both in the upstream of the national nature reserve area (Figure 5.1) and we observed no typical large depositional sites in the field. Rivers in both watersheds (Figure 5.1) were well recognized by simulated routines (Figure 5.4). Therefore, the simulations of deposition and stream pattern are in general reasonable, but more data on spatial distribution of eroded soils as well as long-term watershed total sediment yield is needed for further validation.

5.5.3 Effects of mono- and multi-conservation on watershed stream water quality

After the baseline calibration and validation, two types of conservation strategies were applied in the model. We tested different herbicide applications in rubber plantations as mono-conservation measure focusing on better management of rubber plantations. "Once-weeding" reduced on site soil loss by 78%, and “no-weeding" by 81% comparing to "twice-weeding". Mono-conservation strategy also revealed significant impacts on total sediment yield at S2 watershed outlet but none of them met the reference value by S1 forest dominated watershed (Figure 5.5). “Once-weeding” and “no-weeding” reduced the sediment yield by 14% and 16% than the baseline (“twice-weeding), respectively. This result coincided with other field studies and model simulations showing that soil conservation measures were effective on both plot and watershed scale erosion control (Hessel & Tenge, 2008). Based on field study (Liu et al., 2016) and long-term plot simulation (Liu et al., 2018), “once-weeding” was recommended as a better management considering both on-site erosion control and acceptance of local farmers. This study confirmed its efficiency in total sediments reduction.

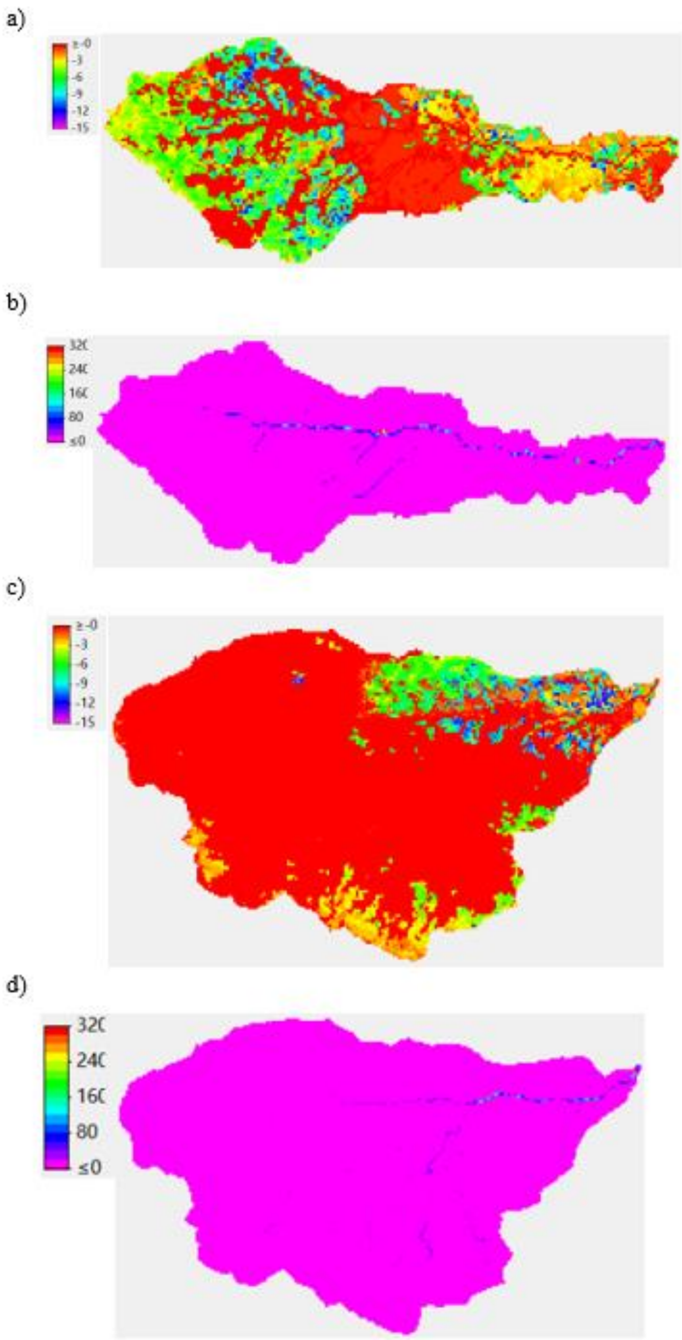


Figure 5.4 Spatial explicit of annual soil loss and deposition of two watersheds in 2014: a) referring to soil loss in S1 watershed; b) referring to deposition in S1 watershed; c) referring to soil loss in S2 watershed; d) referring to deposition in S2 watershed

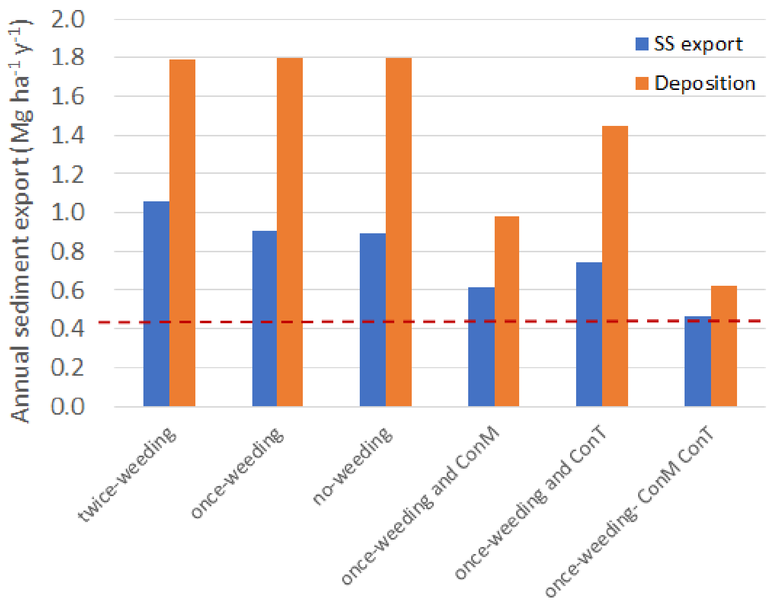


Figure 5.5 Simulated results of total sediment yield at S2 outlet under different conservation strategies; reference line is the sediment yield measured in S1 forest dominated watershed. “twice-weeding”: herbicide application twice per year on day 90 and day 200 in rubber plantations; “once-weeding”: herbicide application once per year on day 90 in rubber plantations; “no-weeding”: no herbicide application in rubber plantations; ConT: conservation in tea by adding residue 2 Mg ha^{-1}; ConM: conservation in maize by adding residue 4 Mg ha^{-1}.

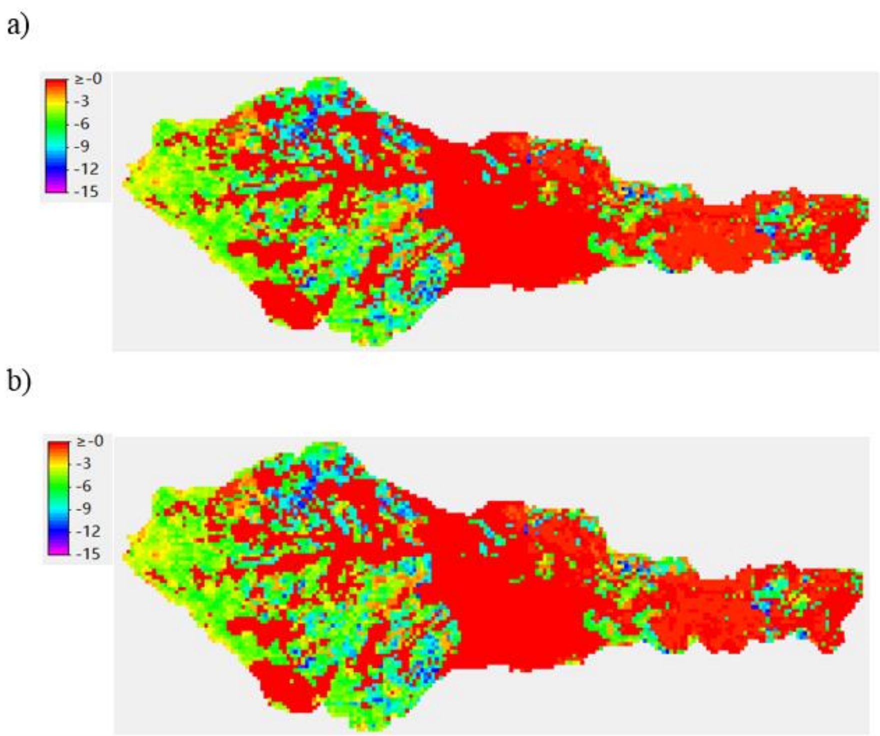


Figure 5.6 Spatial explicit model result of annual soil loss under a) once-weeding and b) no-weeding in rubber plantations of S2. “once-weeding”: herbicide application once per year on day 90; “no-weeding”: no herbicide application.

Figure 5.6 presented watershed spatial-explicit soil losses under different weeding strategies in rubber plantations. Obviously, when the weeding strategy shifted to "once-weeding" (Figure 5.6a), rubber plantations were no longer recognized as the major contributor to total sediment yields in the S2 watershed. At this stage, maize and tea devoted most to total sediment yield. Further reducing herbicide to "no-weeding" reduced on-site soil loss in rubber by 17% compared to "once-weeding", but no apparent effect was simulated at the watershed scale (Figure 5.6a). Hessel & Tenge (2008) suggested that most soil conservation measures were more effective at catchment scale for additional infiltration during transport through the catchment to the outlet. Our simulation, however, indicated potential effective limit of the single conservation measure at the watershed scale with a mosaic land cover. Therefore, it is important to consider dilution effects of the single conservation measure at the watershed scale under the condition of mosaic land cover.

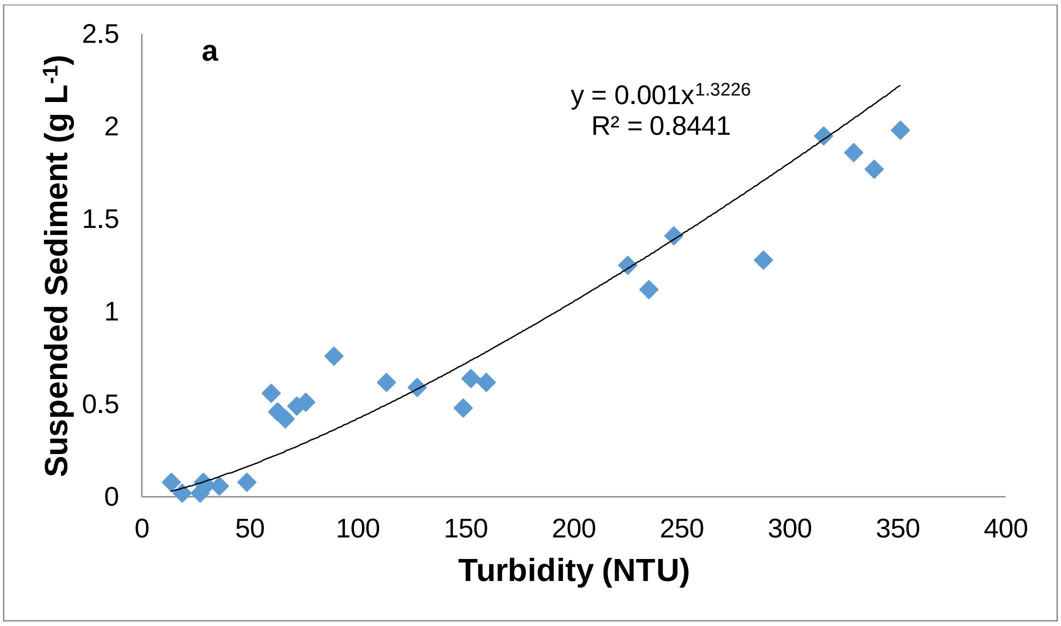
Multi-conservation strategy (2nd type scenarios) applied for maize and tea (adding the residue by 4 and 2 Mg ha^{-1}, respectively) reduced on-site soil loss from 7.3 to 2.5 Mg ha^{-1} (by 66%) and from 4.4 to 1.9 Mg ha^{-1} (57%), respectively. These results (i.e. for the plot/pixel level conservation measures) fell in the range of published data (Table 5.1). Conservation measures in rubber and tea reduced total sediments yield in the watershed by 18% compared to the baseline. Conservation measures both in rubber and maize reduced it by 32%, while conservation in rubber, tea and maize reduced total sediment yield to 0.46 Mg ha^{-1} y^{-1} (by 49%), which was very close to our reference line, 0.42 Mg ha^{-1} y^{-1} in forest dominated watershed S1 (Figure 5.5). The LUCIA simulation results support other watershed modelling cases showing that individual conservation measures in the field can highly decrease both soil loss and sediment yields. Furthermore, different from other sediment yield simulation models, LUCIA directly simulates the conservation measures instead of simple summation of changed soil loss per pixel as model input, namely to increase residue by 4 and 2 Mg ha^{-1} in maize and tea and simulates the measurement effects from plot to watershed scale. This physically based simulation includes impact of site-level effects such as temperature effect on litter decomposition and consequently on soil surface cover. Therefore, application of such models helps to give more specific conservation recommendations.

5.6. Conclusion

After calibration and validation, LUCIA generated good predictions of event-based runoff and total sediment yield in S1 and S2 watersheds, indicated by resultant EF of 0.83, 0.70 for the runoff and 0.95, 0.71 for the sediment yield, respectively. Although simplification of rainfall distribution in LUCIA (i.e. uniform distribution) hampered the model ability to catch all rain events, detected events by the model accounted for over 80% sediment yield within the watershed. Therefore, LUCIA captured major events and mimicked well sediment yield in the investigated watershed. Spatial pattern of sediment transportation and sediment deposition at watershed-scale was reasonable while further improved soil maps as well as long-term data is needed for further validation.

Both mono-conservation and multi-conservation measures reduced on-site soil loss as shown by the pixel-level simulations and total sediment yield as shown by the watershed-level simulations. Though lot of attentions have been paid on "environmental friendly rubber cultivation", the model simulation results indicated that single improvement of rubber management (mono-conservation) only reduced total sediment yield to 0.9 Mg ha^{-1} y^{-1} (by 17%). Sediment yields were still higher compared to reference forest dominated watershed (0.42 Mg ha^{-1} y^{-1}). If additional conservation measures (multi-conservation) were introduced to other land uses (maize and tea), sediment yields were reduced to 0.46 Mg ha^{-1} y^{-1}, which is comparable to a forest dominated watershed. Therefore, multi-conservation was able to offset increased sediment yield induced by land used change.

Supplementary



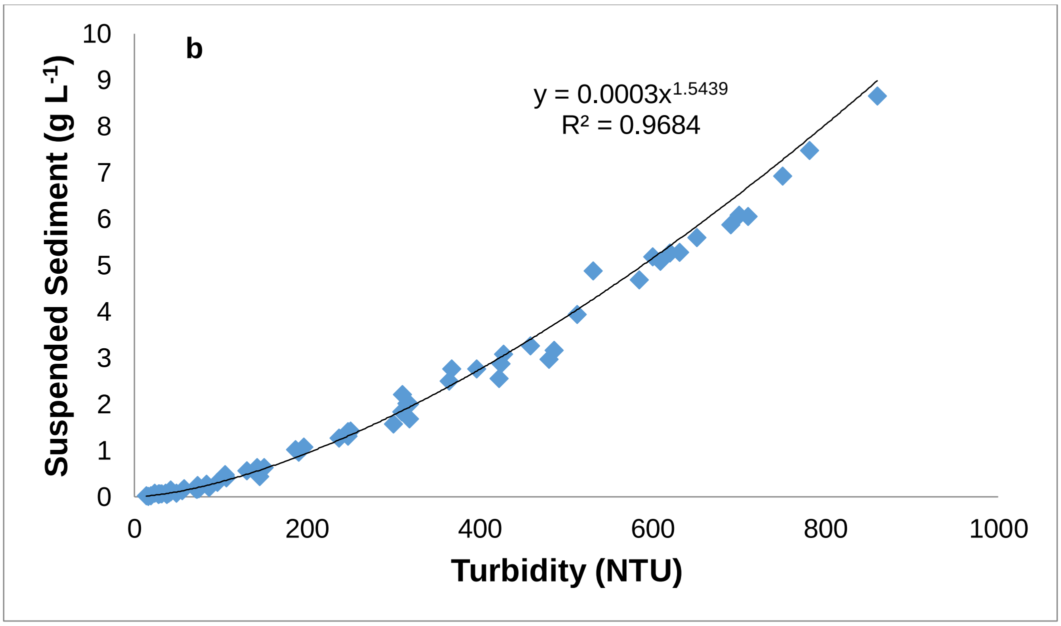


Figure S1 Established relation curve between suspended sediment and turbidity of two watersheds. a) was for S1 watershed; b) was for S2 watershed

Chapter 6 General discussion

6.1 Erosion measurement in subtropical mountainous regions

Though erosion measurement methods have been long-term established and worldwide applied, challenges still remain for mountainous regions due to the complex topography. Establishment of bounded plot is the most widely used erosion measurement approach but originally applied with 5° slope and 22 m standard plot length. The standard length is assumed to occur on a gentle direct slope plane where no deposition happens. Such a long uniform slope plane can be hardly found in mountainous regions. Therefore, the plot length has to be carefully adjusted depending on the topography. Additionally, steep slopes in mountainous regions may lead to stronger soil disturbance during the plot building thus highly increase time cost for soil and plant reestablishment.

Another method, Gerlach troughs, can well offset the methodical drawback of bounded plots due to a less time-consuming establishment period, while the big challenge lies in the estimation of the erosion contributing area, especially in mountainous region. Steep slopes and complex micro-topography lead to easily preferable runoff pathways and hereby high spatial heterogeneity of runoff production. As a result, estimated erosion (runoff and soil loss) highly depends on the position of Gerlach troughs, thus questioning the assumption of the underlying calculated erosion contributing area. Possible solutions could be 1) pre-investigate the topography of the study site and choose a suitable slope plane for erosion data collection; 2) increase the Gerlach troughs width and troughs numbers to counteract micro-topography effects; 3) establish a group of bounded plots as the reference to adjust the contributing area estimation. 1) and 3) were both applied in this study (Chapter 2 and 3). Gerlach troughs were applied based on previous investigations and analysis of topography in rubber plantations. Besides, bounded plots were built adjacent to Gerlach trough sets, which enabled us to compare soil loss from a known area with other estimated areas. It should be noticed that we focused on sheet erosion in rubber plantations in this study. Due to terrace maintenance the slope between terraces was straight and plane which enabled us to apply Gerlach troughs to measure soil loss. Still we missed investigations on soil losses at other spots in rubber plantations (e.g. connecting parts between terraces and slopes, total soil loss with contributing areas including terraces) resulting from choosing Gerlach troughs.

As for total sediment yield measurements of a watershed, the difference between traditional discharge recording station and sediment measuring station should be realized. The most important step for a discharge station is to choose a suitable type of weir or flume, which provides reliable data on water level and discharge. Walkowiak et al. (2013) describes various types of weirs and flumes in detail and indicates that river discharge has to be the most important indicator for station choice. Stations designed for sediment export estimations should take both discharge and sediment concentration as references. Similar to the traditional discharge station, selection of the right weir or flume type is the first step to establish a sediment measurement station. In our case, we chose a combination of V-notch and rectangular weir because of the large difference between seasonal river discharges. For instance, most mountainous watersheds, especially those with agricultural land use, may yield high sediment with relatively low discharge (Valentin et al., 2008). If low discharge measurement weir like V-notch weir is employed, suspended sediment will deposit and accumulate in front of the weir. Then the bedload sediment, namely the sediments trapped in the stilling weir basins, should be collected after each main rainfall event (Valentin et al., 2008). Another solution is to employ a flume instead of a weir for discharge recording, which can perfectly solve the sediment trapping problem. However, since flumes are more suitable for high discharge measurements, the discharge rating curve should be more carefully calibrated. A combination of V-notch and rectangular weirs can solve the problem caused by low discharge and high sediment yield. However, shape change of the weir from V to rectangular can cause turbulence near the weir and trap suspended sediments during storm events. Therefore, spatial heterogeneity of turbidity in river should be considered. In such cases, more sampling points should be set at the cross-section to generate the relation curve between SS and turbidity.

Overall, plot erosion measurement in mountainous region needs to be carefully treated and improved (e.g. topography investigation, increase sampling points) because the complex topography amplifies methodical drawbacks of traditional bounded and unbounded erosion plots. Watershed sediment measurement should pay more attention to the weir or flume selection by considering both discharge and sediment concentration.

6.2 Watershed/basin modelling for decision support

The research community has developed very diverse hydrological models. One of generally adopted approaches is the use of physically based models. These models can simulate the water cycle by solving equations that represent hydrological processes. Depending on research objectives, different models have been developed differing in representation of processes and spatial complexity. For instance, the ANSWERS (Areal Nonpoint Source Watershed Response Simulation, Beasley et al., 1980) model is built on a conceptual hydrological and a physically based erosion process. It has been used to simulate sediment concentration worldwide (Singh et al., 2006; Ahmadi et al., 2006). ParFlow is a numerical model that focuses on simulating surface and subsurface flow (Maxwell et al., 2008). This model can represent explicitly spatial controls on hydrological processes, but requires a supercomputer once applied at watershed scale due to the complicated numerical computation. These two models require extensive preparations of input files. For example, the soil input file of ANSWERS needs to parameterize antecedent moisture, infiltration, drainage response and potential erodibility, respectively. Despite detailed information provided by the manual, it is still difficult and very time consuming for the user, who is not an erosion/hydrology expert, to construct such a file. Therefore, such models are hardly employed by decision makers but more often used by academic researchers in detailed studies aiming at hydrology/erosion process simulation under various conditions.

On contrary, the physically based LUCIA model applied in this research (Chapter 5) is more user friendly, as the model itself estimates many physical parameters for the users (e.g. soil erodibility) via pedotransfer functions. Input of LUCIA only requires basic information of the watershed: land use, soil properties (texture, carbon content, depth), local drainage direction (LDD), digital elevation and climate files. The most complicated input file is the plant file while LUCIA provides default files for most crop types. In our case, we prepared only a detailed file for the rubber plantation as it was the new land use type in our study site. As for other traditional land cover (e.g. maize, forest, tea, rice), default files were sufficiently good for modelling (Chapter 5). Besides, LUCIA serves as not only erosion and sediment transport model, but also as an integrated watershed model with outputs describing plant status (e.g. crop yields, above/belowground biomass). Therefore, it can be a good tool for integrative evaluation of the ecosystem services and functions in the watershed. Another watershed model with similar functions is the public domain model SWAT. Such a model is more easily accepted by decision makers for management support while on the other side it presents less accurate information of a specific process (e.g. spatial heterogeneity of soil moisture effects on water yield) compared to erosion or hydrological models. In summary, different physically based models have specific advantage/disadvantage at the given scale; and it is critical to integrate them to obtain a general view on spatial, processional, hydrological and ecological complexity (Clark et al., 2017).

Another major approach applied in watershed modelling is the data-driven method. Taking hydrological model as an example, models based on data-driven methods intend to extract hydrologic variables from historical measured data by various algorithms such as statistic (Talei et al., 2013), machine learning (Mukerji et al., 2009) and data mining (Dawson & Wilby, 2001). Compared to physically based models, data-driven models can better reach the required accuracy with the limited watershed information. All above-mentioned physically based models require a minimum set of inputs covering climate (e.g. temperature, rainfall, and solar radiation), soil map, land use map, digital elevation map. In contrary, data-driven models enable satisfied runoff prediction with limited input. Artificial neural networks, genetic programming, evolutionary polynomial regression, support vector machines have all successfully applied in rainfall-runoff-sediment modelling (Hosseini et al., 2016; Panda et al., 2010). However, data-driven models simulate the whole system as black box, namely little information regarding soil infiltration, percolation, evapotranspiration can be obtained. Therefore, data-driven models are mostly applied in extreme events (e.g. flood) forecasting instead of land/water management.

In order to obtain better solutions in modeling, the hybrid combination of physically based and data-driven model has been proposed (Young & Liu, 2015; Hosseini & Mahjouri, 2016; Panda et al., 2010). Through the hybrid model, important hydrological/erosion processes presented by the physical-based model offset the black-box feature of a data-driven model while the powerful data-driven methodology alleviates the difficulty in accurate physical modeling. Such combinations should be further extended to spatial concepts. As the case in our study area, we successfully applied LUCIA model to assess plot conservation effects on sediment yields in the small watershed. However, if we further want to upscale the effects, namely to test the effects of sub-watershed on the big watershed, detailed information regarding the big watershed is required. However, this can be hardly met in most cases, as well as in our case. Therefore the hybrid model can serve as a solution by spatial combining the physical-based and data-driven models. Specifically, the target sub-watershed is simulated by the physical-based model, and rainfall, simulated discharge of sub-watershed is taken as input layer to simulate rainfall-runoff model at higher level by a data-driven approach. Thereby the observed sub-watershed impact at a higher-level scale can be evaluated without losing land management information (Figure 6.1).

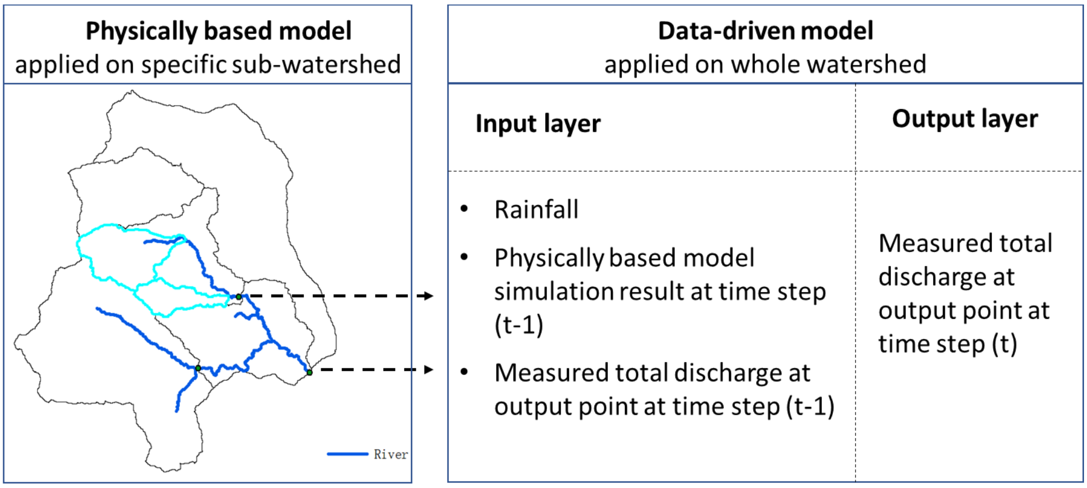


Figure 6.1 Conceptual scheme of spatial combination of physically based and data-driven models to study sub-watershed management effects on the whole watershed

6.3 Opportunities and risks in using short-term field data for watershed management

Long-term observation data is very valuable for studying ecology and hydrology in a study area. Ma et al. (2014) separated land use and climate change effects on water yield of Kejie watershed and proved an overwhelming impact from land use change based on 40 year data on climate, land surface and discharge. Valentin et al. (2008) summarized 5 years discharge and sediment yield data of 27 watersheds and correlated them to land use change and management. Wu et al. (2001) confirmed a more sustainable management in rubber plantations by a continuous 5-year experiment. Abraham & Joseph (2015) validated different weeding management effects on soil properties based on a 10-year experiment. On the other hand, lack of long-term data has become the most common challenge for land use change and management studies. In my research, we applied the space-for-time method and tried to derive appropriate conclusions and recommendations based on short-term (one-year) observation data.

Space-for-time substitution is a common method applied in ecology for long-term studies (STA Pickett 1989). It assumes an equivalent spatial and temporal variation. Uncertainty of this method mainly lies in taking time as a surrogate to explain the spatial heterogeneity. Space-for-time substitution can be successfully applied in general or qualitative trend studies. As for quantitative studies, the hypothetical equivalence of different plots need to be proved. For instance, De Blecourt et al. (2013) used space-for-time substitution to quantify soil carbon stocks change along conversion from secondary forests to rubber plantations. With a careful spatial sampling design (e.g. similar elevation, land change history, slope), they tested the soil texture of different plots. Based on the result that no significant differences were detected between different land use types, it was assumed that spatial changes of soil carbon can be attributed to changes of land use.

By our first assumption, namely "erosion process in the spatial distributed rubber plantations with different standing age can present temporal change of erosion with rubber growing (Chapter 1)", we chose different standing ages of rubber plantations to monitor their erosion process. We applied the uniform observation experiment (e.g. plot establishment, runoff and soil loss collection, ground cover and root density calculation) to each age rubber plantations. The challenge of this experimental approach lied in variation in topography (e.g. slope length, slope steepness) and soil properties, which were two important factors affecting soil losses. Our solution was to calculate the C factor of the USLE equation, which represented ecosystem erosive potential. Therefore, instead of analyzing soil losses, we focused on C factor affected by rubber plantation ages to possibly exclude the impact from topography and soil properties (Chapter 2). The second part of our research (Chapter 3 and 4) discussed the possibility to conserve soil by weeding management in rubber plantations. In order to assess long-term management effects, we firstly designed a one-year field experiment in a 12-year rubber plantation. With this set of data, we validated the model ability to simulate one-year weeding management effects on soil conservation. As surface cover was identified as the most important factor correlating with the erosive potential of the rubber system (Chapter 2, conclusion), we assumed that good representation of surface cover change within one rubber rotation length should guarantee trustable model predictions of soil loss in rubber plantations. Therefore, we validated long-term (40 years) surface cover change simulated by the LUCIA model against field data and tested long-term weeding management effects on soil conservation in rubber plantations (Chapter 4). The erosive potential change simulated by the physical-based model LUCIA fell in the range of C factor calculated by the empirical model USLE. One advantage of this field investigation was that it collected data of various indicators (e.g. surface cover, root density, soil properties, erosion, tree height, canopy coverage, splash potential) covering different aspects of plant, soil and water. Therefore, an integrated data collection helped us to understand the physical drivers of erosion process and management effects in rubber plantations. On the other hand, long-term monitoring in most cases focuses on one or two indicators (e.g. water level, soil loss), therefore can ideally present continuous change of the specific factor while miss an integrated process-based assessment.

As argued by most ecological studies, space-for-time is adopted due to necessity or convenience while it should not be treated as a replacement of long-term studies. Results of our study implied more a qualitative trend instead of accurate quantitative evaluation. Field investigation (Chapter 2) and physical-based model simulation (Chapter 4) both found higher erosive potential in mid-age rubber than in young rubber. The differences in erosive potential between mid-age rubber and young rubber plantations depended on the evaluation method. Application of USLE using field data resulted in the 1.5 – 3 times difference, while LUCIA simulated 1.7 times higher erosive potential. Well-validated LUCIA model at plot scales (Chapter 4) was able to capture physical processes in hydrology and erosion, as well as the effects of management. However, one-year observation data included only one big event corresponding to 88 mm precipitation while the long-term rainfall record of recent 10 years presented 3 extreme events higher than 100 mm (129, 202, 110 mm). Hence, missing long-term data led to information loss on ecosystem response to more diverse weather condition. Especially, understanding of physical processes under extreme weather (e.g. drought, storm) affected by different management strategies can help us to explore ecosystem adaption to climate change.

In summary, our study served as an example to evaluate long-term hydrological and erosion processes by short-term field observations. Combination of space-for-time substitution and modelling offered this possibility with the condition of an appropriate sampling and experimental design. This method resulted in an integrated qualitative trend but was insufficient for a full quantitative assessment due to lack of information on weather conditions, which were more diverse in long term.

6.4 Implications for management in rubber dominated mountainous watersheds

6.4.1 Rubber plantation management and soil conservation measures

Though Chinese decision maker considers rubber plantations as forest cover, monoculture rubber cultivation has been recognized as a threat for biodiversity and environmental degradation due to replacing natural rain forest (Zhai et al., 2017). Therefore, lots of efforts have been made to improve ecosystem services and functions of rubber plantations through improved management. Terracing is the typical conservation measure applied in most mountainous rubber cultivation areas. Based on local conditions (high precipitation in rainy season), bench terraces slopping inward should be the suitable type. This has been suggested since 1985 in "Rubber Tree Cultivation Technical Regulations" issued by Chinese Agriculture Ministry, where it states that "terrace width ranges from 1.8 m to 2.5 m with inward sloping within 12° - 15°". Standard terrace building has been well followed by state based rubber estates. In contrast, smallholdings mostly have built poorly constructed terraces due to absence of technological support. Since 1980s, small stakeholder's rubber plantations have continuously increased because of economic revolution. Poorly constructed and maintained terraces have led to more serious erosion problems instead of conserving water and soil. Though it states in "Rubber Tree Cultivation Technical Regulations" that "sloping area between terraces should keep a good surface cover to conserve water and soil", poorly maintained soil cover in this area is a common problem in both state and smallholding plantations. The reason of insufficient soil cover has been discussed in our work (Chapter 4). We found that weeds were almost cleared to retain a more convenient access to tapping as well as to keep a tidy plantation. Though both rubber litter and weed residues were left in the plantation, they only provided temporarily sufficient soil cover. The high surface cover by rubber litter was mostly supplied at beginning of rainy season and dramatically decreased by mid-rainy season due to fast decomposition rate. By distinguishing weed cover from litter cover, we found the important role of understory vegetation in erosion control; and recommended a simple and labor-saving way for soil conservation in rubber plantations, namely reducing weeding to once per year at beginning of rainy season.

Jungle rubber and intercropping are the two most widely discussed management strategies in rubber plantations. Compared to jungle rubber, weed conservation may supply worse ecosystem functions, specifically in increasing the plantation biodiversity. However, the low latex yield (650 kg ha^{-1} y^{-1}) in jungle rubber imped its wide implementation and acceptance by farmers. On the contrary, a 10 year field experiment has proved that latex yield was not affected by no weeding as compared to monoculture (Abraham & Joseph, 2016). Compared to intercropping, the weakness of no weeding on a soil conservation measure is the absence of extra economic income. Intercropping has attracted attention due to economic benefit of such plantations. Since 70s of the last century, rubber research institutes established a high amount of trials to explore promising intercropping species as well as integrated planting mode with livestock. However, on contrary to the booming intercropping research, actual adoption can hardly be found due to problems with availability of labor and crucial knowledge demanded for proper management (Langenberger et al., 2016). Widely applied intercropping has been only found in the first 2-3 years with maize, pineapple or banana (Baulkwill 1989). As concluded by Zhou (2000), adoption of intercropping is purely market driven and mainly impeded by extra labor costs. In recent years intercropping has become practically popular due to the decreasing latex price. However, its effect on improving ecological functions, especially on soil conservation, has been largely attenuated or even deteriorated in practical application by poor management. Field clearance by removing rubber litter and soil tillage, the common practices in intercropping, can highly disturb the soil and exacerbate soil erosion (Figure 6.2a). Despite of good soil cover provided during the mature rubber phase (Figure 6.2b), high soil disturbance during land preparation and harvest phase of the intercrop are most crucial for soil protection; and may possibly lead to even higher erosion than monoculture cultivation. Thus, more attention should be paid to the problem induced by intensive management of intercropping. From a soil conservation aspect, no weeding provides a simple effective way to reduce erosion with less labor requirements and management knowledge, and should be most easily adopted by local farmers.

Figure 6.2 (a) Understory growth and litter cleared during land preparation for intercropping in a rubber plantation; (b) good surface cover after three months of intercropping

6.4.2 Decentralized plot conservation as a tool for mountainous watershed management

Serious erosion directly leads to land degradation and negatively affects crop yields. Therefore, erosion is of direct concern for farmer and an ecological problem that most easily attracts considerable attention of small stakeholders. Based on the response to questionnaires by local farmers, erosion was the third ranking problem concerned by farmers (Wang & Aenis, 2015). Need of soil conservation is well recognized by farmers but hardly implemented due to lack of proper knowledge.

Eroded soil is transported further into the river systems by surface runoff. Fine sediments deposit on the river bed and in some cases clog the pore space therefore reduce the transport of dissolved oxygen into the hyporheic interstitial (Greig et al., 2007; Heywood and Walling, 2007). Aquatic biodiversity in Naban River has been found to decrease along increasing rubber intensity (Zhao et al., 2014). Land use change induced water quantity and quality deterioration are also widely observed by farmers. Different villages in our study area have reported water problems, such as a reduction of water quantity particularly in dry seasons, “rusty taste” and high turbidity after rainfall event. Farmers link these problems to rubber expansion and agriculture (including rubber) activities in the catchment area. Correspondingly, they have taken some activities to “improve” drinking water quality by moving drinking water abstraction points from agriculture (e.g. rubber plantation) to forest areas. Real measures to guarantee a safe drinking water condition, such as water resource area maintenance and sufficient water treatment, are seldom implemented.

Centralized management is the typical traditional way to improve aquatic ecology at the watershed scale. Two widely adopted centralized management options are water protection zone and stream buffer zone establishment. Water protection zone is to assign different (normally three) priority areas to land within drinking water sources areas. Different land uses and activities are defined for each area. For instance, the core protection area defined as the area within a radius of 20 to 50 m around the drinking water abstraction point (Doerfliger et al., 1999). This area is strictly protected, namely having restrictions that land use should be forest and no use of fertilizers and pesticides is allowed. Stream buffer zone is man-made vegetated zone along the river, which functions as buffer between the stream and the impact (e.g. sediments, dissolved substances) stemming from the surrounding catchment (Castelle et al., 1993). Centralized management asks intense efforts and intervention from local government for zone and activity regulation; while mountainous area is normally marginalized from decision making process. This may explain the conflict between applied few water management measures and observed bad drinking water state. Based on investigations in villages in our study site, the drinking water situation was mostly categorized as "unimproved" and in few cases as "basic" according to the reference proposed by World Health Organization 2015 (Krauss, 2016).

Decentralized plot conservation may supply a better solution for water management in mountainous areas. Non-point pollution from agricultural chemicals (e.g. fertilizer, pesticide and herbicide) through water flow (e.g. surface runoff and groundwater) is the major contributor to surface water quality deterioration (Min & Jiao, 2002). Better agricultural management with less chemical application should highly improve the water quality in the whole region. It has been widely proved by different models and in different regions that on-site soil conservations are more efficient in reducing total sediment yield than off-site measures such as stream buffer zone and sediment retention ponds. Modelling results in our case study also confirmed that well-planned plot conservation measures could be sufficient to control increased sediment load caused by land use change. For our case, we recommended to conserve soil in rubber plantations by reducing herbicide application. This measure could further reduce chemical contamination of water bodies. Another advantage of decentralized plot conservation is the connection of soil and water. Centralized water management (e.g. water protection and stream buffer zone establishment) only focuses on water issue and raises land resource competition as well as upstream-downstream conflicts. In such a situation, lots of efforts are required to 1) educate and persuade local villagers the concept regarding establishing zones for water protection instead of economic benefit; 2) draw up subsidy policies to offset non-equivalent land use and management caused by different functional zones establishment. In summary, decentralized plot conservation efforts are farmer friendly measures as they are closely related to soil health and crop yield, while requiring less policy support compared to centralized watershed management. It can serve as a practical tool for watershed management in mountainous regions which are marginalized from decision making process.

Summary

Land use in Xishuangbanna, Southwest China, a typical subtropical rain forest region, has been dramatically changed over the past 30 years. Driven by favorable market opportunities, a rapid expansion of rubber plantations has taken place. This disturbs forests and land occupied by traditional swidden agriculture thus strongly affecting hydrological/erosion processes, and threatening soil fertility and water quality. The presented PhD thesis aimed at assessing farmer acceptable soil conservation strategies in rubber plantations that efficiently control on-site soil loss over an entire rotation time (25 – 40 years) and off-site sediment yield in the watershed. The study started with field investigations on erosion processes and soil conservation management options in rubber plantations (Chapter 2 and 3). Based on the field data, the physically based model "Land Use Change Impact Assessment" (LUCIA) was employed to assess long-term conservation effects in rubber plantations (Chapter 4) and scale effects on sediment yield in the watershed (Chapter 5).

Specifically, the first study aimed at assessing soil loss in rubber plantations of different ages (4, 12, 18, 25 and 36 year old) and relating erosion potential to surface cover and fine root density by applying the Universal Soil Loss Equation (USLE) model. This study adopted the space-for-time substitution for field experimental design instead of establishing a long-term observation. Spatial heterogeneity of soil properties (e.g. texture, organic carbon content) and topography (slope steepness and length) interfered erosion at different plantation ages. To meet this challenge, namely account for possible impacts of soil properties and slope on erosion, the empirical USLE model was applied in data analysis to calculate the combined annual cover, management and support practice factor CP, which represents ecosystem erosivity. Calculated CP values varied with the growth phase of rubber in the range of 0.006 - 0.03. Surface cover was recognized as the major driver responsible for the erosive potential changes in rubber plantations. The mid-age rubber plantation exhibited the largest erosion (3 Mg ha^{-1}) due to relatively low surface cover (40%-60%) during the rainy season, which was attributed to low weed cover (below 20%) and the low surface-litter cover favored by a high decomposition rate.

Based on the results of the first study, the second study focused on reducing soil loss in rubber plantations by maintaining a high surface cover through improved weed

management. Among the different weeding strategies tested, no-weeding most efficiently reduced on-site soil loss to 0.5 Mg ha^{-1}. However, due to the low farmer acceptance of the no-weeding option, we recommend reducing herbicide application to a single dose at the beginning of the rainy season (once-weeding) to better conserve soil as well as inhibiting overgrowth of the understory vegetation.

As the second experiment lasted only one-year, while rubber plantation is a perennial crop with a commercial lifespan of 25 – 40 years, the third study applied the LUCIA model to simulate the temporal dynamics of soil erosion in rubber plantations under different weeding strategies. The erosion module in LUCIA was extended to simulate both runoff and rainfall based soil detachment to better reflect the impact of the multi-layer structure of the plantation canopy. The improved LUCIA model successfully represented weed management effects on soil loss and runoff at the test site with a modelling efficiency (EF) of 0.5-0.96 and R^2 of 0.64-0.92. Long-term simulation results confirmed that “once-weeding” controlled annual soil loss below 1 Mg ha^{-1} and kept weed cover below 50%. Therefore, this weeding strategy was suggested as an eco- and farmer friendly management in rubber plantations.

Furthermore, LUCIA was applied at watershed level to evaluate plot conservation impact on sediment yield. Two neighboring sub-watersheds with different land cover were chosen: one a forest dominated (S1, control), the other with a mosaic land use (S2), which served to assess mono-conservation (conservation only in rubber plantations) and multi-conservation (conservation in maize, rubber and tea plantations) effects on total sediment yields. The model was well calibrated and validated based on peak flow (EF of 0.70 for calibration and 0.83 for validation) and sediment yield (EF of 0.71 for calibration and 0.95 for validation) measured from the two watersheds outlet points. Model results showed that improved weed management in rubber plantations can efficiently reduce the total sediment yields by 20%; while multi-conservation was largely able to offset increased sediment yields by land use change.

In summary, while exploring the dynamics of erosion processes in rubber plantations, a physically based model (LUCIA) was extended and applied to simulate weed management effects over an entire crop cycle (40 years) and implications at higher scale level (watershed sediment yield). Once-weeding per year was identified as an improved management to reduce on-site erosion and off-site sediment yield. But to fully offset increased sediment yield by land use change, a multi-conservation strategy should be employed, which not only focuses on new land uses, like rubber plantations, but also takes care of traditional agricultural types. A conceptual framework is proposed to further assess the specific sub-watershed erosion (e.g. sediment or water yield) effects in large watersheds by spatially combining process-oriented and data-driven (e.g. statistic based, machine learning based) models. This study also serves as a case study to investigate ecological issues (e.g. erosion processes, land use change impact) based on short-term data and modelling in the absence of long-term observations.

Zusammenfassung

Die Landnutzung in Xishuangbanna, Südwestchina, einer typischen subtropischen Regenwaldregion, hat sich in den letzten 30 Jahren dramatisch verändert. Getrieben von günstigen Marktchancen hat ein rapider Ausbau von Kautschukplantagen stattgefunden. Dies beeinflusst Wälder und Flächen, die durch traditionellem Brandrodungsackerbau bewirtschaftet werden, was starke Auswirkungen auf hydrologische Prozesse und Erosionsprozesse hat und die Bodenfruchtbarkeit und Wasserqualität bedroht. Die vorliegende Dissertation zielte auf die Bewertung von akzeptablen Bodenschutzstrategien für Landwirte in Kautschukplantagen ab, die den Bodenverlust innerhalb des Standortes während einer ganzen Rotationszeit (25 - 40 Jahre) und den Sedimentausstoß außerhalb des Standortes im Wassereinzugsgebiet effizient kontrollieren. Die Studie begann mit Felduntersuchungen zu Erosionsprozessen und Bodenschutz-Managementoptionen in Kautschukplantagen (Kapitel 2 und 3). Basierend auf den Felddaten wurde das physikalisch basierte Modell "Land Use Change Impact Assessment" (LUCIA) eingesetzt, um Langzeitschutzeffekte in Kautschukplantagen (Kapitel 4) und Skaleffekte auf den Sedimentausstoß im Wassereinzugsgebiet zu bewerten (Kapitel 5).

Konkret zielte die erste Studie darauf ab, den Bodenverlust in Kautschukplantagen unterschiedlichen Alters (4, 12, 18, 25 und 36 Jahre alt) zu untersuchen und das Erosionspotenzial mit der Allgemeinen Bodenabtragsgleichung (USLE) in Beziehung zur Oberflächenbedeckung und Feinwurzeldichte zu setzen. In dieser Studie wurde die „space-for-time substitution“ für experimentelle Feldforschung anstelle einer Langzeitbeobachtung übernommen. Räumliche Heterogenität der Bodeneigenschaften (z. B. Textur, organischer Kohlenstoffgehalt) und Topographie (Neigungssteilheit und -länge) beeinträchtigten die Erosion bei verschiedenen Pflanzungsaltern. Um dieser Herausforderung zu begegnen, nämlich mögliche Auswirkungen von Bodeneigenschaften und Gefälle auf die Erosion zu berücksichtigen, wurde das empirische USLE-Modell in der Datenanalyse, zur Berechnung der kombinierten jährlichen Bodenbedeckung, Management und support practice factor (CP), das die Ökosystem-Erosivität darstellt, verwendet. Berechnete CP-Werte variierten mit der Wachstumsphase von Kautschuk im Bereich von 0,006-0,03. Die Oberflächenbedeckung wurde als der Haupttreiber für Änderungen des erosiven

Potentials in Kautschukplantagen anerkannt. Die Kautschukplantage mittleren Alters wies aufgrund der relativ geringen Oberflächenbedeckung (40% -60%) während der Regenzeit die größte Erosion (3 Mg ha^{-1}) auf. Dies wurde auf einen geringen Unkrautbewuchs (unter 20%) und eine geringe Bodenbedeckung durch Oberflächenstreu, verursacht durch eine hohe Zersetzungsrate, zurückgeführt.

Basierend auf den Ergebnissen der ersten Studie konzentrierte sich die zweite Studie auf die Verringerung des Bodenverlusts in Kautschukplantagen, indem eine hohe Oberflächenbedeckung durch verbessertes Unkrautmanagement aufrechterhalten wurde. Unter den verschiedenen getesteten Unkrautbekämpfungsstrategien reduzierte „no-weeding“ den Bodenverlust vor Ort auf 0,5 Mg ha^{-1} am effizientesten. Aufgrund der geringen Akzeptanz der Unkrautbekämpfung durch den Landwirt empfehlen wir jedoch zu Beginn der Regenzeit („einmaliges Unkrautjäten“) eine Herbizidapplikation auf eine Einzeldosis zu reduzieren, um den Boden besser zu erhalten und das Überwachsen der Unterholzvegetation zu verhindern.
Da das zweite Experiment nur ein Jahr dauerte, während die Kautschukplantage eine mehrjährige Pflanze mit einer kommerziellen Lebensdauer von 25 bis 40 Jahren ist, wurde in der dritten Studie das LUCIA-Modell zur Simulation der zeitlichen Dynamik der Bodenerosion in Kautschukplantagen unter verschiedenen Strategien eingesetzt. Das Erosionsmodul in LUCIA wurde erweitert, um sowohl oberflächenabfluss- als auch niederschlagsbedingte Bodenerosion zu simulieren, um den Einfluss der mehrschichtigen Struktur des Plantagenschirms besser widerzuspiegeln. Das verbesserte LUCIA-Modell stellte erfolgreich die Auswirkungen des Unkrautmanagements auf den Bodenverlust und den Oberflächenabfluss am Versuchsstandort mit einer Modellierungseffizienz (EF) von 0,5-0,96 und R2 von 0,64-0,92 dar. Die Ergebnisse der Langzeitsimulationen bestätigten, dass "einmaliges Jäten" den jährlichen Bodenverlust unter 1 Mg ha^{-1} kontrollierte und die Unkrautabdeckung unter 50% hielt. Daher wurde diese Unkrautbekämpfungsstrategie als umwelt- und landwirtfreundliches Management in Kautschukplantagen vorgeschlagen.

Darüber hinaus wurde LUCIA auf Wassereinzugsgebietsebene angewendet, um die Auswirkung der Flächenerhaltung auf den Sedimentausstoß zu bewerten. Zur Bewertung der Auswirkungen auf die Gesamtsedimentmengen wurden zwei benachbarte Teileinzugsgebiete mit unterschiedlicher Landbedeckung ausgewählt. Für die Auswirkungen von Einzelschutz („mono-conservation“; Schutz nur in Kautschukplantagen) hat eine von Wald dominierende Landnutzung (S1, Kontrolle) gedient und für die Auswirkungen von Mehrfachschutz („multi-conservation“; Schutz in Mais-, Kautschuk- und Teeplantagen) eine Mosaiklandnutzung (S2).

Das Modell wurde gut kalibriert und validiert basierend auf dem Peak-Flow (EF von 0,70 für die Kalibrierung und 0,83 für die Validierung) und dem Sedimentertrag (EF von 0,71 für die Kalibrierung und 0,95 für die Validierung), die an den zwei Austrittsstellen des Wassereinzugsgebiets gemessen wurden.

Die Modellergebnisse zeigten, dass ein verbessertes Unkrautmanagement in Kautschukplantagen die gesamten Sedimentausbeuten um 20% reduzieren kann; während Mehrfachschutz weitgehend in der Lage war, erhöhte Sedimenterträge durch Landnutzungsänderungen auszugleichen.

Zusammenfassend wurde, während der Untersuchung der Dynamik von Erosionsprozessen in Kautschukplantagen, ein physikalisch basiertes Modell (LUCIA) erweitert und angewendet, um Unkrautmanagementeffekte über einen gesamten Erntezyklus (40 Jahre) und Implikationen auf höherer Maßstabsebene (Wasserscheidensedimentmenge) zu simulieren. Einmaliges Unkrautbekämpfung pro Jahr wurde als verbessertes Management identifiziert, um die Erosion vor Ort und den Sedimentaustrag außerhalb des Wassereinzugsgebietes zu reduzieren. Um den durch die Landnutzungsänderung erhöhten Sedimentausstoß jedoch vollständig ausgleichen zu können, sollte eine Mehrfachschutzstrategie angewandt werden, die sich nicht nur auf neue Landnutzungen wie Kautschukplantagen konzentriert, sondern sich auch um traditionelle landwirtschaftliche Typen kümmert. Ein konzeptueller Rahmen wird vorgeschlagen, um die spezifischen Erosionseffekte der sub-Wassereinzugsgebiete (z. B. Sediment oder Wasserausbeute) in großen Wassereinzugsgebieten durch räumliche Kombination von prozessorientierten und datengesteuerten (z. B. statistisch und machine-learning basierten) Modellen weiter zu bewerten. Diese Studie dient auch als Fallstudie zur Untersuchung ökologischer Fragen (z. B. Erosionsprozesse, Auswirkungen von Landnutzungsänderungen) auf der Grundlage von Kurzzeitdaten und Modellierung in Abwesenheit von Langzeitbeobachtungen.

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