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Impact of rubber tree dominated land-use on biodiversity and ecosystem services in the Greater Mekong Subregion

Dissertation

Submitted in fulfillment of the requirements for the degree
“Doktor der Agrarwissenschaften” (Dr. sc. agr.)

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Presented by
Inga Häuser, MSc.
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Examination committee

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Author's Declaration

I, Inga Häuser, hereby affirm that I have written this thesis entitled "Impact of rubber tree dominated land-use on biodiversity and ecosystem services in the Greater Mekong Subregion" independently as my original work as part of my dissertation at the Faculty of Agricultural Sciences at the University of Hohenheim.

All the authors in the quoted or mentioned publications in this manuscript have been accredited. No piece of work by any person has been included without the author being cited, nor have I enlisted the assistance of commercial promotion agencies. This thesis has not been presented into other boards for examination.

Place, Date Signature

Acknowledgement

It is my profound wish to thank my supervisor Prof. Dr. Joachim Sauerborn for his continuous moral and scientific support since I joined his workgroup in 2007. He always had an open door and encouraged my personal and scientific development at every time. I always valued his recommendations and opinion as he valued mine and I am pleased to say that he was the best boss I ever had.

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Overview of publications

In order to comply with the regulations for a cumulative doctoral thesis at the Faculty of Agricultural Sciences, several publications have been included into this work. As these publications have been edited to fit the regulations of different publishers, the style for quoting and the layout of the reference section may vary between chapters.

Chapter 2:

Häuser, I., Thellmann, K., Cotter, M. and J. Sauerborn (2015) Ecosystem services and biodiversity of rubber plantations – a systematic review. CAB Reviews 2015, 10, No. 037. doi: 10.1079/PAVSNNR201510037

Chapter 3:

Häuser, I., Martin K., Germer, J., He, P., Blagodatskiy, S., Liu, H., Krauß, M., Rajaona, A., Shi, M., Pelz, S., Langenberger, G., Zhu, C.-D., Cotter, M., Stürz, S., Waibel, H., Steinmetz, H., Wieprecht, S., Frör, O., Ahlheim, M., Aenis, T. and G. Cadisch (2015) Environmental and socio-economic impacts of rubber cultivation in the Mekong region: challenges for sustainable land-use. CAB Reviews 2015, 10, No. 027. doi: 10.1079/PAVSNNR201510027

Chapter 4:

Cotter, M., Häuser, I., Harich, F. K., He, P., Sauerborn, J., Treydte, A. C., Martin, K., Cadisch, G. (2016) Biodiversity and Ecosystem Services - A case study for the Assessment of Multiple Species and Functional Diversity Levels in a Cultural Landscape. Submitted to Ecological Indicators.

Chapter 5:

Václavík, T., Langerwisch, F., Cotter, M., Fick, J., Häuser, I., Hotes, S., Kamp, J., Settele, J., Spangenberg, J., Seppelt, R. (2016) Investigating potential transferability of place-based research in land system science. Environmental Research Letters 11(9): 095002.

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1 General introduction

1.1 The importance of rubber and its way to China

Rubber is a natural polymer with rubber-elastic properties, so-called elastomers which consist of polyisoprenes. They occur in many plants, although normally only in small quantities. A feasible harvest is only possible if plants contain latex tubes which are characteristic in several plant families, e.g. in Euphorbiaceae, Apocynaceae, Moraceae and Asteraceae. The most important source of rubber comes from the rubber tree *Hevea brasiliensis* Müll. Arg. that belongs to the family of Euphorbiaceae. Latex of the rubber tree contains only caoutchouc (polyisoprenes in the form of cis-1,4) which has high elastic properties (Rehm and Espig 1991). Caoutchouc can also be produced synthetically; in 2015 the ratio of natural versus synthetic rubber was 45% to 55% (International Rubber Study Group (IRSG)). However, especially the tyre industry relies heavily on natural rubber, since its elastic properties are unmatched until now, guaranteeing a constant or even rising demand in the future.

Hevea brasiliensis originates from the Amazon basin and until the beginning of the 20th century Brazil had the monopoly on rubber which had led to an economic upturn of the area around Manaus in Brazil. Its success story in South-East Asia began only after 1876, when Henry A. Wickham collected around 70,000 rubber seeds in Brazil and exported them. The seeds were first brought to London, where they germinated in Kew Royal Botanic Gardens (Great Britain). From there, the plants were shipped to British colonies, were only a few plants arrived alive and were spread from there to other destinations after propagation.

In the Amazon area, rubber trees were scattered in the forest which made tapping a labourious task. In contrast in South-East Asia trees were cultivated in plantations, making the harvest of latex much easier and quicker, resulting in surpassing Brazil's production which thereafter lost the lead of the global market. Because of the South-American leaf blight disease, caused by *Pseudocercospora ulei*, formerly known as *Microcyclus ulei* (Hora Júnior 2014) it was never possible to establish noteworthy plantations in South America. Until now, the disease did not spread to the paleotropics (Edathil 1986). Nowadays, rubber production mainly takes place in South-East Asia. In 2013, Thailand and Indonesia alone supplied nearly 60% of the global production which was close to 12 million tons (FAO - Statistics Division (FAOSTAT) 2016a). Figure 1 gives an overview of the top five producers of natural rubber in 2013. For the first time, China is part of the top five producers which is due to the fact that the production of Malaysia decreased between 2012 and 2013 from 0.92 million tons to 0.82 million tons. In consequence, China surpassed the Malaysian production with 0.86 million tons (FAO - Statistics Division (FAOSTAT) 2016b).

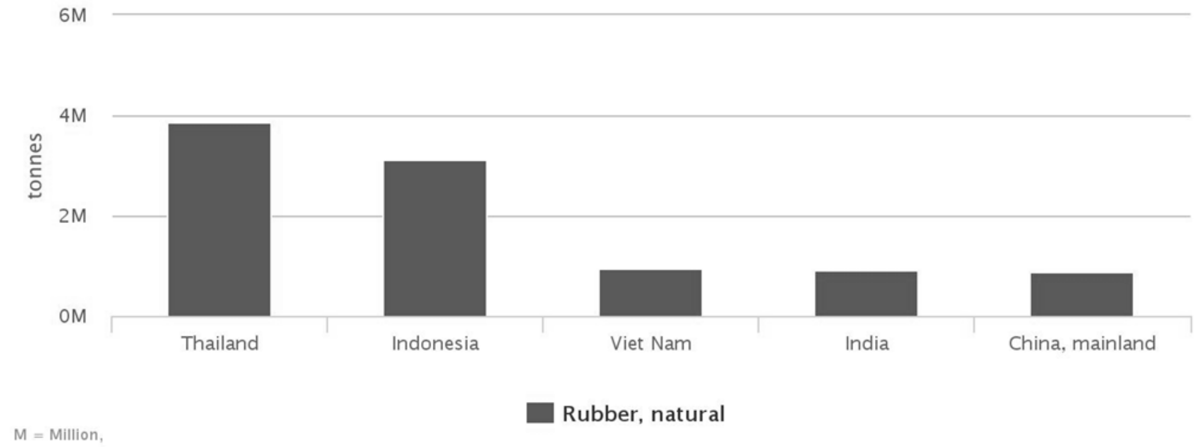

Figure 1 Production of top five producers of natural rubber in 2013 (FAO - Statistics Division (FAOSTAT) 2016a).

Li and Fox analyzed the distribution of rubber cultivation in the Greater Mekong Subregion (Li and Fox 2012). Traditionally, rubber is grown around the equator, from 10° South to 10° North, where rainfall is continuous the whole year round. In South-East Asia the countries of southern Thailand, Myanmar and Vietnam lie in this zone. But in the last decades the production area shifted as far as 22° North. Especially China invested in research aiming at growing rubber also in areas which are cooler and having a dry season. Their success encouraged other countries to also cultivate rubber. This resulted in a general shift of rubber cultivation to the north and now Cambodia, Laos and China also produce rubber. The authors also provided a map with traditional and non-traditional rubber production areas of the Greater Mekong Subregion, based on sub-national data from 2007 and 2009 (Figure 2). Given the relatively small area of China that is suitable for rubber production (the southern areas of the Provinces of Yunnan, Guangong and Hainan) (Liu et al. 2016) and the fact that they recently made it into the list of the top five producers, it is of interest to take a closer look at Chinas rubber production.

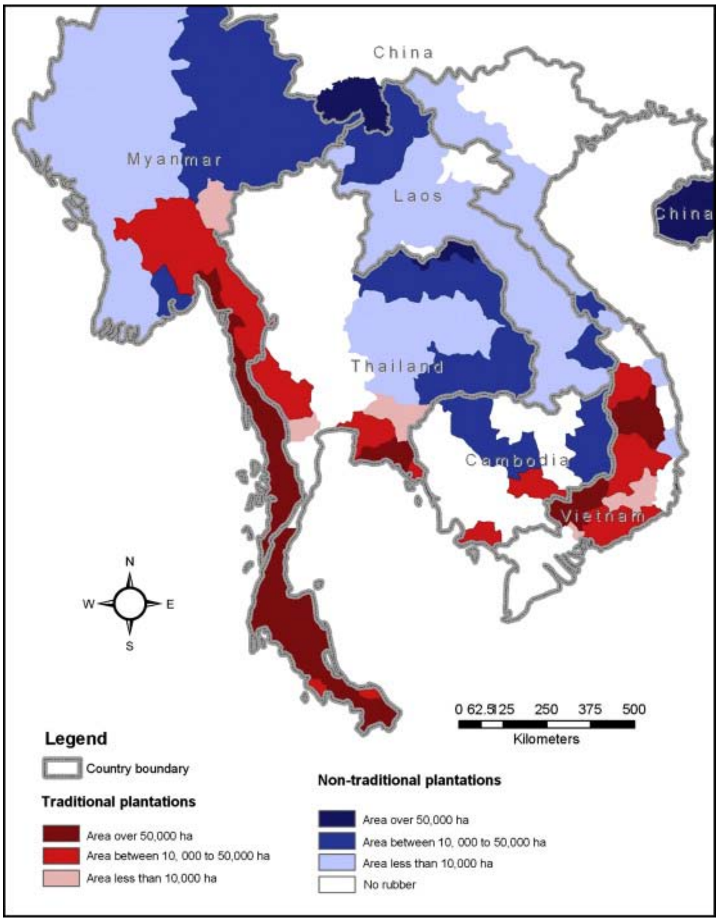

Figure 2 Traditional and non-traditional rubber tree-growing areas in mainland Southeast Asia (Li and Fox 2012).

Rubber and oil palm compete for the same production area, therefore the shift of rubber to the North is partly due to the rising demand for oil palm (*Elaeis guineensis)* products (for food and feed as well as for bio fuels). The global production area of oil palm nearly doubled in 15 years between 1999 and 2014 from 9.4 Mio ha to 18.6 Mio ha and in South-East Asia more than doubled from 4.8 Mio ha to 12.8 Mio ha (FAO - Statistics Division (FAOSTAT) 2016c). Since oil palm needs an even temperature between 24-28 °C it is climatically closer limited to the tropical regions between 10° South and 10° North and further restricted to altitudes below 500 m asl. (Rehm and Espig 1991). As a consequence, indirect land-use change promoted a replacement of rubber to the North (Saswattecha 2016).

In China, rubber cultivation was first tried in 1904 in Yingjiang, Dehong Dai and Jingpo Autonomous Prefecture, Yunnan Province where it failed (Chapman 1991). Only after the 1950ties it gained importance. The rise of rubber in China was first politically motivated. Especially after the rubber embargo by the USA in 1951 China pushed rubber cultivation in its own country to increase self-sufficieny. At the beginning, rubber production was carried out by state farms, but when these promoted the cultivation to smallholders who were mostly from the ethnic minorities Dai and Akha from the Dai Autonomous Prefecture of Xishuangbanna, Yunnan Province, they did not only pick up the innovation very quickly but developed an unforeseen agility of spreading the production even to neighbouring countries like Laos and Myanmar. The economic and social situation of these ethnic minorities who were considered as “backward” turned drastically from being very poor to gaining considerable wealth or even getting rich (Sturgeon 2010). In general, the whole region of Xishuangbanna in the Yunnan Province experienced an economic boost because of rubber-generated income between 1988 and 2003 (Liu et al. 2006). Figure 3 shows the increase in Chinese rubber production between 2003 and 2013 (FAO - Statistics Division (FAOSTAT) 2016d).

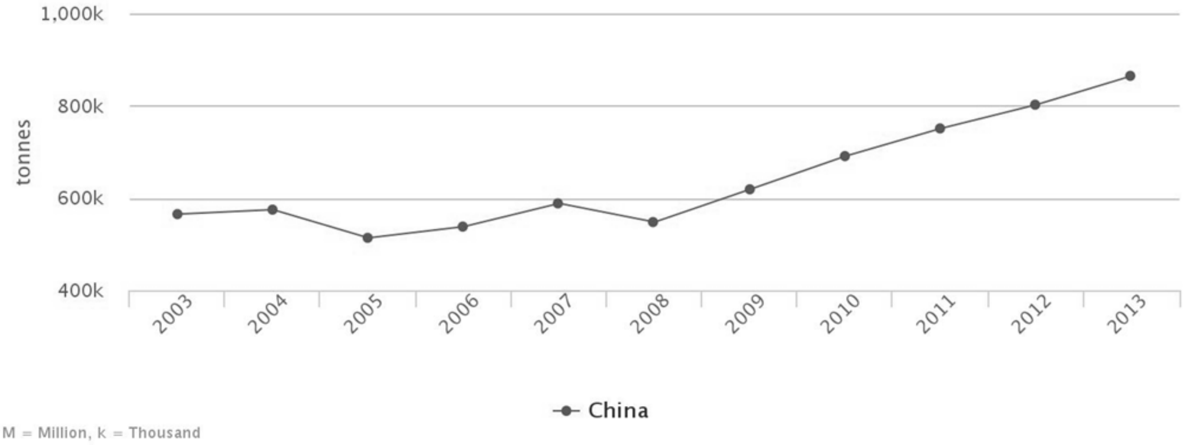

Figure 3 Production of natural rubber in China 2003-2013 (FAO - Statistics Division (FAOSTAT) 2016d).

The importance of rubber cultivation for China and how much research effort is dedicated to this topic can also be seen when looking at the number of publications dealing with rubber mapping efforts. Numerous maps are available for Xishuangbanna, the most southern tip of Yunnan Province that harbours the second largest rubber production in China with 37% of the Chinese production in 2011 (Chen et al. 2016). Unlike simple numbers from official statistics that do not give a geographical distribution, spatially explicit maps are a prerequisite for model approaches.

In 2007, Li et al. published a map series based on Landsat analyses for the years 1976, 1988 and 2003. Accuracy assessment for rubber classification in this study was 90%. The authors found a land share of rubber of about 11%, translating to roughly 2,100 km^2 (the area of Xishuangbanna is presumed with 19,200 km^2[1]). Official figures for rubber cultivation from 2004 are 1,730 km^2 (Ahlheim 2015). The second rubber map is from Xu et al. (2014), who found 4,200 km^2 of rubber in Xishuangbanna derived from RapidEye Imagery (which translates to roughly 22 % of land share), official data for the same period is 2,000 km^2. A second analysis based on the same satellite images was undertaken by Chen et al. (2016) with the aim to distinguish between young rubber (tree age below six years) and mature rubber (tree age above six years) with an accuracy result of 90% for rubber plantation identification. Based on MODIS, Enhanced Vegetation Index and Short-wave infrared images from 2010 Senf et al. (2013) achieved an overall accuracy of 73,5%, discriminating rubber, forest and non-forest areas. In 2015, Li et al (2015) published the latest map of rubber distribution in Xishuangbanna for the year 2014. This latest source (Li et al. 2015) states a land share of roughly 2,700 km^2 of mature rubber in the area (nearly 14% of land share). The overall accuracy for the assessment based on Landsat OLI images is 89.8% (Figure 4).

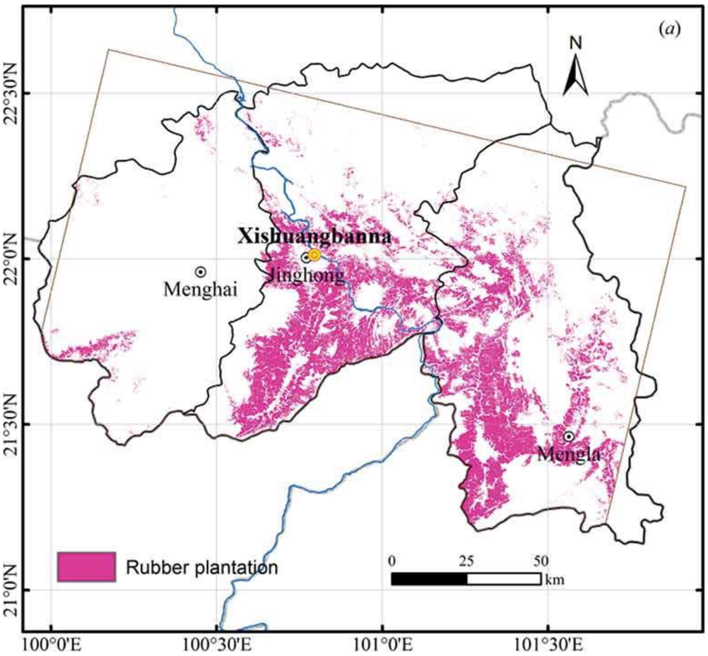

Figure 4 Mature rubber plantations in the Dai Autonomous Prefecture of Xishuangbanna, Yunnan Province, China in 2014 (Li et al. 2015).

In the beginning, rubber production in Xishuangbanna was carried out on moderate slopes and altitudes below 900 m asl. The aforementioned expansion of rubber in Xishuangbanna was possible because rubber cultivation expanded into higher elevations and on steeper slopes which are less suitable for rubber production due to topographic and climatic constraints (as a plant originating from the tropics, rubber is frost susceptible). Nevertheless, in the past three decades the upper limit shifted to higher elevations, in 1988 up to 1100 m asl., in 2002 to 1300 m asl. and in 2010 as high as 1,400 m asl. (Chen 2016). A similar trend is seen when regarding rubber plantations on different gradients. Between 2002 and 2010, the largest proportion of rubber plantations was on slopes of 18° and 20° respectively (Chen 2016). Both findings are confirmed by Liu et al. (2013).

The future of rubber production in China is less clear. On the one hand, trading possibilities with other South-East Asian countries increased in the recent past and nowadays 75% of the Chinese rubber demand is imported. Producer prices in China are considerably higher than in other important rubber producing countries (FAO - Statistics Division (FAOSTAT) 2016e) (Figure 5). This means that in China less suitable areas that produce lower yields may still be profitable for farmers whereas in other countries these lower yields would not be cost-effective anymore

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[1] For all calculations of land share in Xishuangbanna the area of 19.200 km^2 was used. Own area calculations based on Global Administrative Areas (GADM, www.gadm.org) and FAO GeoNetwork (www.fao.org/geonetwork) resulted in 19,222 km^2 and 19,177 km^2 respectively. Xu et al. (2014) calculates with an area of 19,164 km^2, which is in the same range.

because of the lower producer prices. Therefore, it is questionable how long rubber cultivation in China is viable. At least low-profitable sites will probably be abandoded in the near future (Yi and Cannon 2016). But while economic reasons may suggest a future downward trend of rubber cultivation in China, contrariwise climate change may lead to an opposite development. A study of Liu et al. (2015) suggests that global warming could change the suitability distribution of rubber cultivation, shifting rubber cultivation further north and hence increasing the area suitable for rubber tree growth in China.

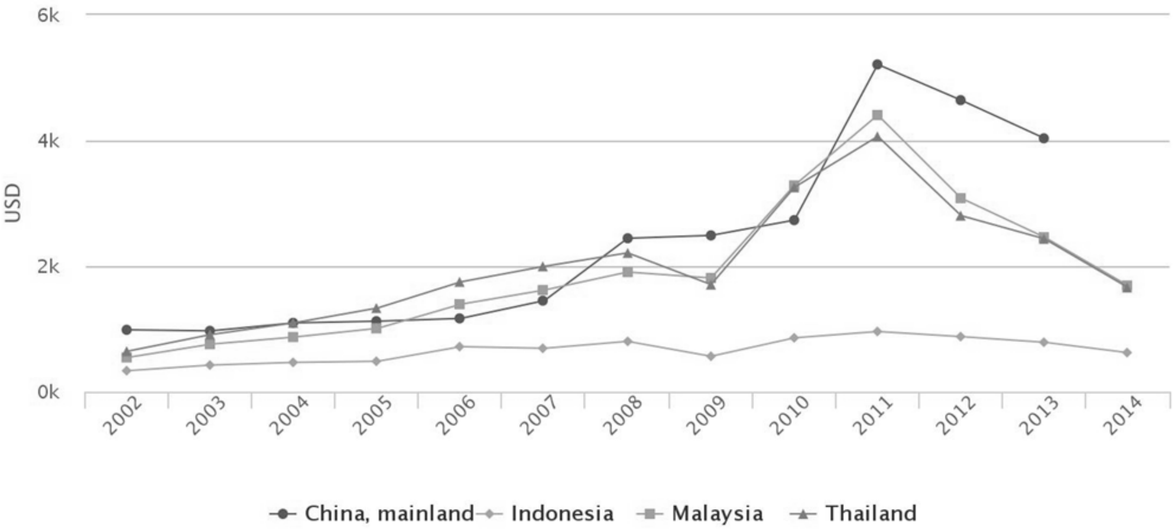


Figure 5 Producer prices for natural rubber of the top five producer countries in USD/tonne, 2002-2014 (FAO - Statistics Division (FAOSTAT) 2016e).

Figure 6 provides some impressions of the research area in the Naban River Watershed National Nature Reserve (NRWNNR).

Slope preparation (clear cut and establishment of terraces) before rubber planting. Google earth satellite image; left contour lines of rubber plantation; right forest. Secondary forest.

Mature rubber plantation in the lowlands, undergrowth vegetation removed. Freshly cut rubber tree with latex dripping into a collective bowl. Old cuts are clearly distinguishable. Traditional land-use, rice in the valley, maize on slopes.

Figure 6 Impressions of the research area**.** All pictures were taken in the Naban River Watershed National Nature Reserve (NRWNNR), Dai Autonomous Prefecture of Xishuangbanna, China. Pictures kindly provided by Sabine Baumgartner, except satellite image.

1.2 The concept of ecosystem services and its link to biodiversity

Rapid human population growth since the 1950s from around 2.5 billion to more than 7.5 billion people has led to a profound change in the extent of human's use of ecosystems. While humans have always relied on and shaped ecosystems for the fulfillment of their daily needs, the rapid increase in population led to a never seen requirement for food, feed, fiber and fuel, with large scale-changes for nature. In 2010, Pretty and more than 50 other authors collected "The top 100 questions of importance to the future of global agriculture" (Pretty et al. 2010). In their article, biodiversity, ecosystem services and conservation is one chapter in the natural resource input, stating that a "major challenge is to understand the best compromises between increasing food production while minimizing the negative impacts on biodiversity, ecosystem services and society".

The concept of ecosystem services is clearly anthropocentric; ecosystem services are defined as "the benefits people obtain from nature" (MEA 2005). Although this term is widely accepted it is relatively vague, leaving way to different interpretations of what exactly are benefits. To overcome this problem, Boyd and Banzhaf (2007) extended this definition to "final ecosystem services are components of nature, directly enjoyed, consumed, or used to yield human well-being". Ecosystem services are usually classified into three to four groups. The most important classification systems are: (i) Millennium Ecosystem Assessment (supporting, provisioning, regulating, cultural) (MEA 2005), (ii) Economics of Ecosystems and Biodiversity: Ecological and Economic Foundations (TEEB) (provisioning, regulating, cultural, habitat) (TEEB 2012)) and (iii) Common International Classification of Ecosystem Services (CICES) V4.3 (provisioning, regulation and maintenance, cultural) (Haines-Young and Potschin 2013).

Humans for agricultural purposes modify natural ecosystems in a way to maximize crop production, but it is already known for a long time that humankind benefits in numerous ways of different ecosystems. As early as 1983, Ehrlich and Mooney (1983) used the term "ecosystem service" for water purification, flood control, erosion prevention and continuous supply of firewood and timber. But it took some time until the concept found its way under this terminology into scientific mainstream, inducing a shift in scientific paradigm; only after the late 1990s publications (being a reliable indicator for topics scientists are working on) mentioning ecosystem services increased (Vihervaara et al. 2010). I will briefly summarize the most important milestones that moved forward this development in the next paragraphs.

In 1997, Costanza and twelve co-authors published an article in *Nature*, where they assessed the monetary value of 17 ecosystems in 16 biomes. Mostly based on willingness-to-pay approaches, they calculated the incredible sum of 33 trillion US dollars, nearly two times of the global gross domestic product of that time. Although the approach was criticized for a number of shortcomings (see Simpson (2011) for a thorough overview), it undoubtedly triggered an ongoing-discussion about the values of ecosystems.

In 2001, the United Nations commissioned a comprehensive study, the Millenium Ecosystem Assessment (MEA), that wanted to answer key questions concerning the state of ecosystems: How have they changed, why have they changed, how will the changes affect human well-being and how can we preserve them? Ecosystem services were defined and played a major role in this assessment. MEA was a response to governmental requests for information, deriving from four international conventions: The Convention on Biological Diversity, the Convention to Combat Desertification, the Convention of Wetlands and the Convention of Migratory Species. Not aiming at generating new knowledge, the goal was to collect, prepare and to communicate existing information, which included also expert judging in order to give answers to policy-relevant questions. The assessment was governed by a board consisting of multiple stakeholders with representatives of international institutions, governments, businesses, non-governmental organizations and indigenous people. More than 2000 authors and reviewers were involved during the four years between initiation till the publication of the synthesis report (MEA 2005).

Another international effort is the initiative of TEEB (The Economics of Ecosystems and Biodiversity), set off and funded mainly by the European Commission and the United Nations Environment Programme. The work was divided into three phases: the generation of an interim report which was presented in 2008 at the Conference of Parties (COP) of the Convention on Biological Diversity (CBD), the TEEB study reports, presented at the COP-CBD in 2010 and the on-going TEEB-country as well as TEEB sector/biome reports. The idea of this study was to underpin arguments for conserving ecosystems and biodiversity from an economic point of view by showing that the services they produce have a huge economic value even though they are largely provided for free and do not have a market value as such. It took up the idea that was already brought up by Costanza et al. (1997), but with a much higher research power and trying to overcome the critical points arisen from this first article. Although monetary valuation is a complex topic, information on how much ecosystems contribute to economy is urgently needed; ignoring monetary values would therefore be negligent. But policy makers must acknowledge the limitations of valuation methods and treat the gained information according to its feasibility (TEEB 2010a). The concept for TEEB has two major backbones: The first one is to calculate the economic value of all ecosystem services provided by a given ecosystem at a given ecological state. It is based on the concept of Total Economic Value (TEV) (Figure 7). The second one is to provide an insurance or option value that relies on the resilience capacity of a given ecosystem that supplies stable ecosystem services under variable environmental conditions. The background is the necessity to ensure that an ecosystem is not passing a tipping point, thereby avoiding regime shifts that would change an ecosystem irreversibly which in consequence would negatively impact the provided ecosystem services (TEEB) 2010a).

Since the work by Costanza et al (1997), MEA and TEEB the topic received increasingly more attention and the scientific publications dealing with this topic increased exponentially. After sustainability and biodiversity, also ecosystem services became another scientific buzzword, which went along with a shift in scientific paradigm (Vihervaara 2010). Nowadays, research on ecosystem services is well established in science.

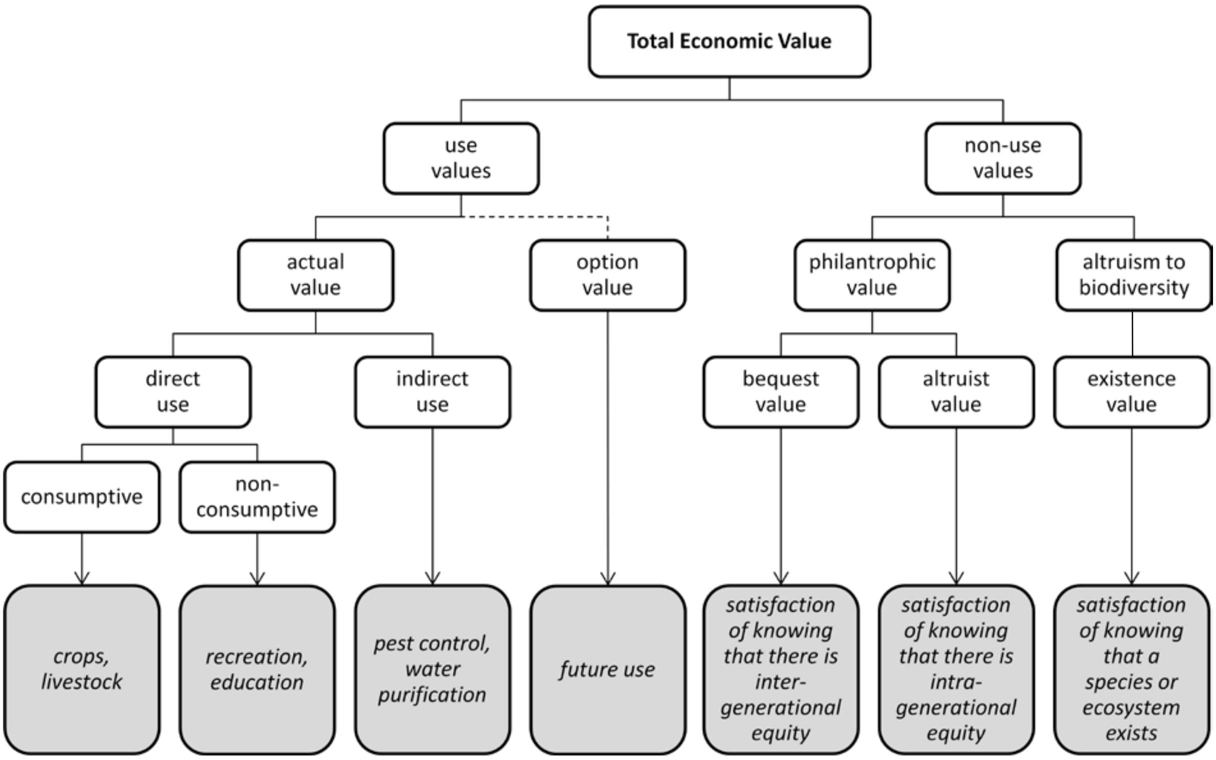


Figure 7 Value types within the Total Economic Value approach, adapted after TEEB (2010a).

Biodiversity is a shortened form of the words biological and diversity. According to the Convention on Biological Diversity (CBD) it is defined as: “the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems” (CBD 2007). Although it is until today impossible to accurately account for the role of biodiversity on ecosystem services in general it can be stated that maintaining functioning ecosystems that are able to provide multiple services rely on a broad biodiversity (TEEB 2010b).

1.3 The SURUMER project

This work is part of the joint research project “Sustainable Rubber Cultivation in the Mekong Region (SURUMER)”. In this project, a Sino-German research consortium works at the development of an integrative land-use concept in Yunnan Province, China. The work is funded by the German Federal Ministry of Education and Research (BMBF) from 2011 to 2016 under the grant number (FKZ) 01LL0919. Numerous subprojects from natural sciences, social sciences and economics are involved. As the project provides the framework for my dissertation, I will give a short overview of the project goals and the study region. The following paragraphs refer to the SURUMER web page (https://surumer.uni-hohenheim.de).

The aim of SURUMER is “to develop an integrative, applicable and stakeholder-validated concept for sustainable rubber cultivation in Yunnan”. To achieve this goal, multi-, inter- and transdisciplinary approaches are applied to identify trade-offs and synergies between ecosystem services and socio-economic goals. Figure 8 shows the thematic focal points of the different subprojects and how they are linked. In practical terms, the project strives to recommend an alternative land-use concept for rubber cultivation which is based on agro-ecological diversification, not only to improve the supply of bio-physical ecosystem services such as water regulation and purification, carbon sequestration and storage, but also to expand the portfolio of products to reduce the economic risk of farmers who are currently depending on rubber prices.

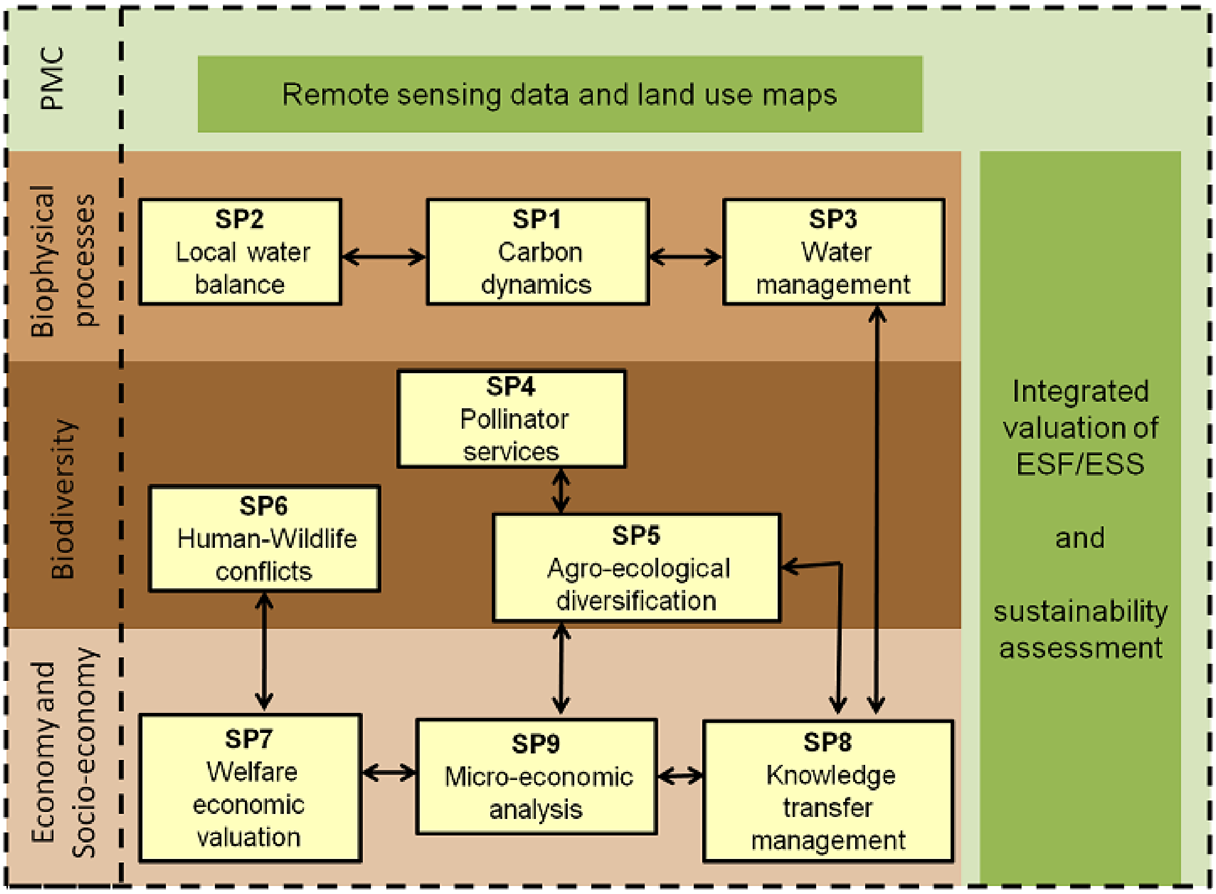


Figure 8 Overview of the different subprojects (SP) involved in SURUMER and how they and the Project Management and Coordination (PMC) are linked to develop an integrative, applicable and stakeholder-validated concept for sustainable rubber cultivation in Yunnan.

The main study area of SURUMER is the Dai Autonomous Prefecture of Xishuangbanna, the southern tip of Yunnan Province. The prefecture is a mountaineous region and borders Laos and Myanmar. The Mekong River passes from north to south. Figure 9 gives an overview about the location of the research sites of the different subprojcets. Xishuangbanna has a monsoon climate with a distinct cool dry season and hot rainy season (Figure 10). Field trials and data collection took mainly place in the Naban River Watershed National Nature Reserve (NRWNNR). The reserve comprises approximately 270 km^2 and is managed according to the United Nations Educational, Scientific and Cultural Organization (UNESCO) Man and Biosphere regulations. Figure 11 shows the land-use map of the NRWNNR in 2013.

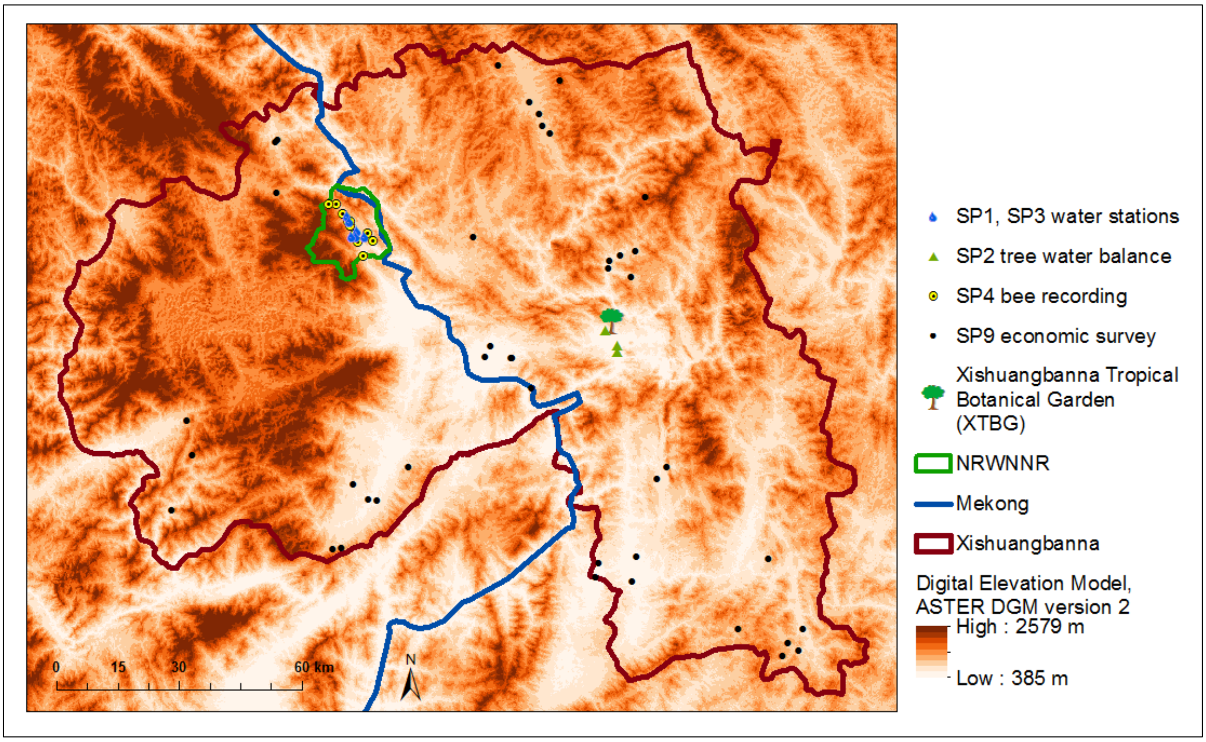
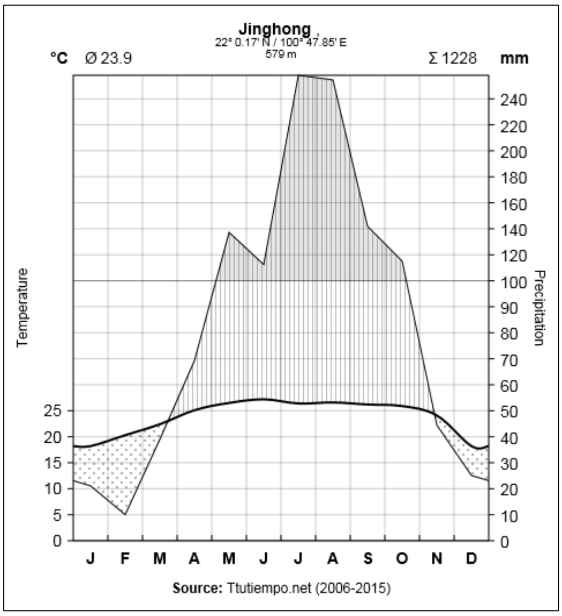


Figure 9 Overview of the SURUMER research sites in Xishuangbanna, Yunnan Province, PR China.



Source: Ttutiempo.net (2006-2015)

Figure 10 Climate graph of Jinghong, capital of Xishuangbanna, Yunnan Province, PR China. Weather data from Tutiempo Network (2016); Climograph created with PjotrC (https://dl.dropboxusercontent.com/u/17823304/Climograph/Climograph.html).

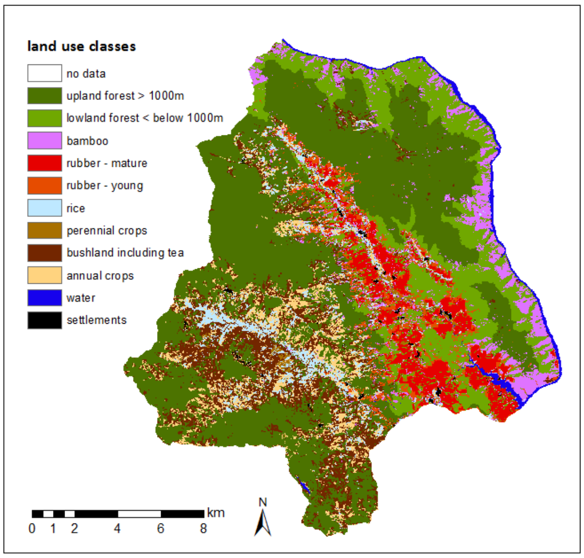


Figure 10 Land use map of the Naban River Watershed National Nature Reserve in 2013.

1.4 Objectives

The objectives of this dissertation are: (1) To give a comprehensive overview about the publications concerning ecosystem services and biodiversity in rubber cultivation systems. A major focus is on analyzing to what extent existing studies are able to inform land-use planners and decision-makers in general. (2) To provide an overview of various impacts by rubber cultivation on ecosystem services. (3) To develop a biodiversity indicator based on different species that can be included in existing biodiversity models to show the impact that different rubber cultivation scenarios will have on different species groups and (4) to test the transferability potential of the project results to other regions in the world.

1.5 Outline of thesis

Chapter two shows the analysis of the results of a thorough literature search that was carried out in two important databases for scientific articles concerning ecosystem services and rubber plantations. The resulting papers were analyzed regarding geographic origin of the studies, the number of assessed ecosystem services in each study and the type of ecosystem services studied. Based on these results it is possible to deduct conclusions on how good the existing studies are suited to inform land-use planners or decision-makers.

Chapter three starts with a review of the impact rubber cultivation has on the ecosystem services carbon storage and sequestration, provisioning and quality of water and provision of latex and therefore income generation. In a second step, possibilities to improve the rubber cultivation system are investigated.

Chapter four provides a detailed analysis of how biodiversity changes under different rubber cultivation scenarios for the Naban River Watershed National Nature Reserve. Therefore, data of various flora and fauna species were normalized to allow their integration into existing modeling software. Different groups of species were analyzed separately to show the differing impact on biodiversity if only certain species groups are considered (red list species versus species with a direct use value such as bees producing honey).

Chapter five completes the dissertation by examining the potential transferability of place-based research with the help of land system archetypes. These include multiple dimensions of land-use intensity and environmental and socio-economic conditions. The analysis was tested with twelve regional projects from a large joint research program focusing on sustainable land management, SURUMER being one of them. In a first step, the ‘project archetype’ was defined, based on a synthesis of global land system indicators. In a second step their transferability potential was tested by calculating statistical similarity of locations across the world.

An overall discussion of the different publications in chapter six concludes the thesis.

1.6 References

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2 Ecosystem services and biodiversity of rubber plantations – a systematic review

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Outline and overview

A thorough literature search in two important databases for scientific articles was conducted concerning ecosystem services and rubber plantations. The resulting papers were analyzed regarding geographic origin of the studies, the number of assessed ecosystem services in each study and the type of ecosystem services studied. Based on these results it is possible to deduct conclusions on how good the existing studies are suited to inform land-use planners or decision-makers.

Ecosystem services and biodiversity of rubber plantations – a systematic review

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Abstract

Humans benefit from ecosystems which provide free goods and services. The capacity of ecosystems to deliver such services is under constant stress since humans continue to alter the environment. Ecosystem service (ESS) approaches have the potential to pinpoint multiple benefits that humans derive from nature, including services without direct market value such as climate regulation, pollination and flood protection. Payments for Ecosystem Services (PES) schemes and land-use planning are two main fields of application that require reliable data about key ESSs provided by a given landscape. This review focuses on ESS studies on rubber (*Hevea brasiliensis*) cultivation and investigates whether the studies fulfil the mentioned requirements. We analysed 76 studies in respect to the regional origin, number of assessed ESSs by each study, distribution of ESSs concerning provisioning, regulating and maintenance and cultural ESSs and if the studies also included biodiversity. We found that there is still a huge gap between what topics scientists research (single or few ESSs that are easy to measure) and what information policymakers would need (results including multiple ESSs, ideally from different sections, which include less tangible ESSs such as cultural services). Of the analysed 76 publications only two fulfilled the requirements to inform policymakers and even they were not truly interdisciplinary. The main challenge remains to develop interdisciplinary studies which require joint research or collaborative projects with sufficient funding to fulfil the required task.

Keywords

Environmental services, Ecosystem functions, Land-use change, Agroecosystems, Literature review, Decision support

Review methodology

We searched the Web of Science (Thomson Reuters, USA) and Scopus (Elsevier, Amsterdam). We used the default search in both databases and combined each of the search terms ‘ecosystem service’, ‘ecosystem function’, ‘environmental service’, ‘environmental function’, ‘land-use change’ and ‘land-use change’ as a set expression with both ‘rubber’ and ‘*Hevea*’.

2.1 Introduction

Humans benefit from ecosystems which provide free goods and services. The capacity of ecosystems to deliver such services is under constant stress since humans continue to alter the environment. Since 1997, when ‘The value of the world’s ecosystem services and natural capital’ (Costanza et al. 1997) and ‘What are Ecosystem Services?’ (Daily 1997) were published, enhancing ecological understanding about ecosystem functions (ESFs) and services and the underlying role of biodiversity science has gained more importance, as shown in Vihervaara et al. (2010). Searching Web of Science for ‘ecosystem function’ or ‘ecosystem service’ in titles resulted in more than 4000 publications between 2010 and 2015, while Scopus yielded more than 2600 articles.

Even though the terms ecosystem functions and services are often used simultaneously, the definitions are clearly delimited from each other: ESFs are related to the structure (biophysical architecture) and the processes (e.g., decomposition and fluxes of nutrients) of an ecosystem (TEEB 2012). ESSs, on the other hand, are restricted to an anthropocentric view; they comprise benefits that humans derive from ecosystems (Costanza et al. 1997, Daily 1997, MEA 2005). The ecosystem service cascade from Haines-Young and Potschin (2010) shows the connection between ESFs and ESSs in detail with illustrated examples.

ESSs are usually classified into three to four groups. The most important classification systems are (i) Millennium Ecosystem Assessment (supporting, provisioning, regulating, cultural) (MEA 2005), (ii) Economics of Ecosystems and Biodiversity: Ecological and Economic Foundations (TEEB) (provisioning, regulating, cultural, habitat) (TEEB 2012)) and (iii) Common International Classification of Ecosystem Services (CICES) V4.3. (provisioning, regulation and maintenance, cultural) (Haines-Young and Potschin 2013).

An advantage of the ESS approach is that it pinpoints multiple benefits that humans derive from nature at a regional or global level, including services without direct market value, such as climate regulation, pollination and flood protection (Foley et al. 2005). Thus, the approach produces a more holistic picture than some more traditional concepts, such as marginal return calculations that ignore external effects. With the help of the ESS approach, biodiversity and environmental conservation could gain more importance in future planning and resource management (Portman 2013).

There are two main fields of application that require information about ESSs: Payments for Ecosystem Services (PES) schemes and land-use planning. An example of the previous is the detailed report ‘Payments for Ecosystem Services: Getting started’ from 2008 (Forest Trends, The Katoomba Group and UNEP 2008), where the whole process of developing a PES scheme is described. Very recently, another more vigorous call for integrating scientific knowledge into PES schemes was published (Naeem et al. 2015). For the latter, in 2009, Daily et al. (2009) proposed a framework to include results from ESS analysis into resource and land-use decisions. However, both concepts require reliable data about key ESSs provided by a given landscape. In most cases such knowledge is missing.

On a global scale, two comprehensive studies on ESSs have been published: MEA (MEA 2005) in 2005 and TEEB (TEEB 2012) in 2012. Both arose from international efforts and gave a global overview about the status of ESSs. While these studies give a comprehensive overview about multiple ESSs, the global scale entails a certain inaccuracy regarding regional differences. Therefore, such large-scale assessments are not suited for implementing PES schemes or for including ESSs without direct market value into land-use planning.

In 2002, Balmford et al. (2002) reviewed more than 300 papers ranging from local to regional scale in scope and found only five studies that compared natural with man-made ecosystems regarding total economic gains. Further, in all the studies included in the publication, the total economic gains from natural ecosystems were higher than in cultivated ecosystems.

In a more recent study from 2011, Seppelt et al. (2001) reviewed 153 publications on ESSs from the past 20 years. They found that more than 60% of the studies dealt with only few ESSs simultaneously (Costanza et al. 1997, Daily 1997, Vihervaara et al. 2010, TEEB 2012, MEA 2005). Apparently, there is still a gap between what type of information policymakers would need and what topics researchers study.

This review focuses on ESS studies concerning rubber (*Hevea brasiliensis*) cultivation. In the last two decades, rubber production increased by a factor of 1.5 in Southeast (SE) Asia (Indonesia, Malaysia, Myanmar, Thailand, Vietnam and Southern China (FAOSTAT 2013)) and has expanded into pristine areas. This situation is not only driven by an increased demand for natural rubber but also by the oil palm (*Elais guineensis*) boom. Especially in Malaysia, oil palm has replaced rubber plantations which in return have expanded into new areas (Li and Fox 2012). Plantations of, for example, oil palm or rubber can dominate whole landscapes in SE Asia. They generate high revenues for those owning the plantation, but the benefits are gained on the expense of natural ecosystems that have the potential to offer numerous ESSs and benefit many people. Hence, our aim is to analyse whether studies address multiple-key ESSs and fulfil the requirements to inform decision-makers.

2.2 Methods

Web of Science, operated by Thomson Reuters, USA and Scopus, managed by Elsevier, the Netherlands, are the largest citation databases for peer-reviewed science articles. Both cover about 55 million records (Thomson Reuters 2014, Elsevier 2014). Since ‘neither database is inclusive, but complements the other’ (Burnham 2006), we included both of them into our inquiry.

We applied the default search (Web of Science: topic, which comprises title, abstract, author keywords and keywords; Scopus: article title, abstract, keywords) and combined each of the search terms ‘ecosystem service’, ‘ecosystem function’, ‘environmental service’, ‘environmental function’, ‘land-use change’ and ‘land-use change’ as a set expression with both ‘rubber’ and ‘*Hevea*’.

In accordance with CICES V4.3. (Haines-Young and Potschin 2013), we considered ESSs as ‘the outputs of ecosystems ... that most directly affect the well-being of people’. CICES groups ESSs into three sections: provisioning, regulation and maintenance and cultural, which we used for our classification. To our knowledge, CICES’s list of ESSs with examples is the most detailed list currently available and it offers the best structure for classifying ESSs. Based on the search terms, 699 publications were found. They were analysed if they were determined to be relevant to our research question. For example, many publications dealt with rubber processing or rubber-based products and did not fulfil our criteria.

We conducted a three-step analysis: (i) exclusion based on title (401 studies), (ii) exclusion based on abstract (168 studies) and (iii) assessing the remaining 130 papers, from which we further excluded non-relevant publications (54 studies). Supplementary material A summarises the complete list of publications and the applied criteria for exclusion.

We analysed the remaining 76 studies in respect to their regional origin, number of assessed ESSs, distribution of ESSs concerning the three aforementioned sections and if the studies also considered biodiversity. Though CICES does not consider biodiversity as such an ESS (the only exception is the maintenance of habitats for plant and animal nursery and reproduction), ESSs and biodiversity are often studied together (TEEB 2012, MEA 2005, Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) 2015) since biodiversity influences the provision of many ESSs. Therefore, while we did not explicitly include biodiversity in our literature search, we include biodiversity in our results if it was studied.

2.3 Results

The 76 publications considered in our study, including the analysed criteria, are listed in Supplementary material B.

The largest share of research (81%) originated from rubber cultivation in SE Asia (Figure 1). Within this group, half of the studies were from SE China (either from Hainan Island or Yunnan Province). The other half was quite evenly distributed between Indonesia, Thailand and Malaysia.

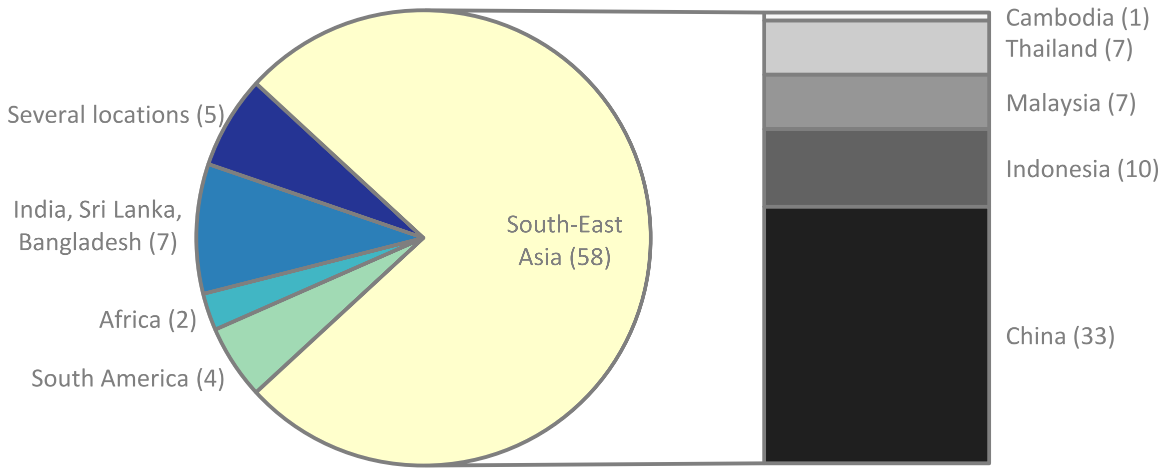

Figure 1 Geographic regions where studies were carried out (pie chart on the left). Numbers in brackets represent the absolute numbers of studies (n = 76). The country-specific results of studies originating from SE Asia are shown in the bar chart on the right.

The second largest share of publications (less than 10%) dealt with ESSs in rubber cultivations in Southern Asia, including India, Bangladesh and Sri Lanka. Latin America was represented by Brazil, Guatemala and two studies from Colombia, whereas only two studies were from Africa. Five publications (6%) included several study sites located in different countries or even on different continents.

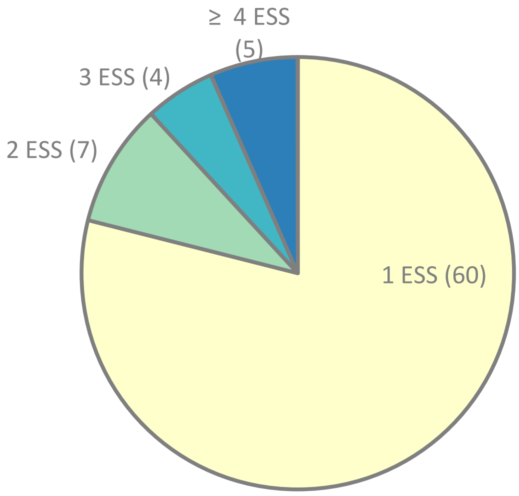

Figure 2 Number of publications dealing with one or more ESSs in rubber simultaneously. Numbers in brackets are the absolute numbers of studies (n = 76).

When the number of ESSs addressed in a study was investigated, it was discovered that in a majority of cases, authors focused on only one ESS (79%) (Figure 2). Of the eleven studies (14%) that researched two or three ESSs simultaneously, ten dealt with ESSs from different sections. Only five studies (less than 7%) included four or more ESSs. Of these five, one was published in 2008 by Hu et al. (Hu et al. 2008). They calculated changes of the monetary value of ESSs in a Chinese township (approximately 33 km^2) during a period of 18 years (1988–2006), when rubber production increased significantly in this area (from 12 to 45% of total land cover). Their results were based on a benefit transfer of the values published by Costanza et al. (1997), including 17 ESSs from all three sections (provisioning, regulation and maintenance, cultural). Another study that included four or more ESSs was published by Pfund et al. (2011) and conducted in three African countries (Cameroon, Madagascar and Tanzania) and two Asian countries (Indonesia and Laos). They investigated ten ESSs (from all three sections) with a participatory approach: villagers were interviewed to rank different land-use types concerning their provision of different ESSs. The study did not focus specifically on rubber plantations, but rubber was one of the assessed land-use classes in the Asian countries. Yet another publication by Sohel et al. (2014), conducted in a National Park in Bangladesh, demonstrated the spatial distribution of 22 ESSs (from all three sections) delivered by nine different land-use types (including rubber plantations) based on local expert knowledge (i.e., local experts ranked the relative supply of each ESS in participatory scoring exercises). Two other studies published in 2014 and based on the same research focusing on Colombia (Lavelle et al. 2014, Sanabria et al. 2014), analysed four soil-based ESSs of the 'maintenance and regulating' section using ants as an indicator of biodiversity.

In this paragraph, we report the various ESSs generated under rubber cultivation, their classification into the three sections defined by CICES (Haines-Young and Potschin 2013) and whether biodiversity was included in the study. The 76 analysed publications comprised a total of 121 ESSs plus 22 indicators of biodiversity, adding up to 143 indicators. Most of the ESSs investigated belonged to the 'regulation and maintenance' section (55%) (Figure 3). Of this group, more than one-third of the research studies (37%) analysed carbon storage/release (carbon stocks in soil, in below- and/or above-ground biomass). The second biggest share (21%) dealt with water-related ESSs and less than 10% with erosion control. The remaining 33% included, for example, greenhouse gas emissions other than carbon dioxide, flood mitigation, health (vector-borne diseases), soil quality and soil formation. The second largest share of studied ESSs was attributed to the 'provisioning' section (25%). In this section, 11 publications (14%) included rubber yield or income from rubber cultivation. Four publications studied cultural services – in total, six cultural ESSs (3%) were among the 121 ESSs studied. None of these dealt exclusively with cultural services, but also with ESSs from other sections. One publication assessed cultural services via benefit transfer values (Hu et al. 2008), while the other three studies interviewed farmers (Feintrenie et al. 2010) or assessed values with participatory approaches either with farmers (Pfund et al. 2011) or with local experts (Sohel et al. 2014). Fifteen per cent of the studied ESSs analysed biodiversity in rubber cultivation systems and in some cases it was compared with biodiversity in natural forests. Animals were studied more often than plants; the most popular group was invertebrates, such as nematodes, earthworms, beetles, hoverflies and bees, while the only analysed class of vertebrates was birds.

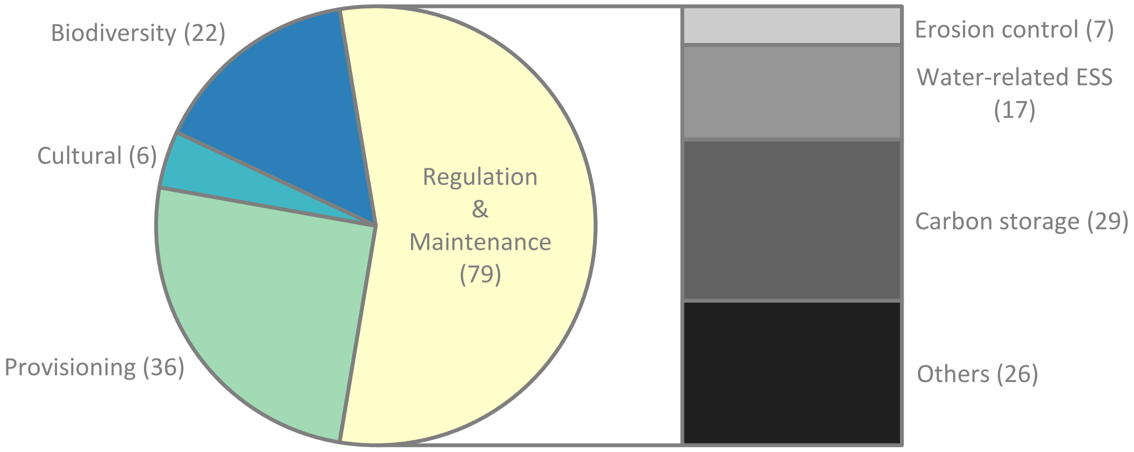

Figure 3 Sections of studied ESSs according to the classification system of CICES V4.3. and additionally the number of biodiversity-related indicators (pie chart on the left). In total, the 76 studies analysed 143 indicators (numbers in brackets are absolute numbers; n = 143). The distribution of the most studied section (regulation and maintenance) is shown in the bar chart on the right.

2.4 Discussion

At a first glance, the regional distribution of the analysed studies reflects the global distribution of rubber production as more than 80% of the studies originated from SE Asia which is the world's main rubber production region. However, of the 80%, more than half of the studies derived from China. Since rubber production in China is climatically restricted to the most southern parts (the Dai Autonomous Prefecture of Xishuangbanna in Yunnan Province and Hainan Island), the suitable area for cultivation is only about 53 000 km^2. In 2013, less than 7% of the global rubber yield was produced in China, whereas more than 70% was produced in Indonesia, Thailand, Malaysia and Vietnam (in descending order according to the portion of total rubber production) (FAOSTAT 2015). The number of Chinese studies is therefore disproportionately high.

There are several possible explanations for the high number of publications from China. Firstly, rubber cultivation started in China only in the 1980s (Chapman 1991); hence, the area is not a traditional rubber-growing area. The relatively quick transformation of vast areas of prime forests to rubber plantations could have resulted in both scientists and politicians to become interested in the study of the effects. Secondly, the research endowment in China is excellent with several outstanding research centres (Kunming Institute of Botany, Xishuangbanna Tropical Botanical Garden, both branches of the Chinese Academy of Sciences and Hainan University), so there are many scientists in the region to analyse land-use changes. Thirdly, studying ecosystems and valuing ESSs has already a relatively long tradition in China (Zhang et al. 2010). Lastly, the climatic conditions in China are sub-optimal for rubber production (since rubber is a plant of the humid tropics). Therefore, external large-scale effects derived from rubber cultivation might be unforeseeable and lead to higher public awareness.

Regarding the number of ESSs studied simultaneously in one region, there is clearly a huge gap between what topics scientists research (one or few ESSs) and what information policy-makers need for developing PES schemes and ESS-driven land-use planning (results including multiple ESSs, ideally from different sections). For as long as a majority of the studies concentrate on only one ESS, it is not possible to form a holistic picture. So far, it seems that studies are designed to answer questions in one discipline, which is obviously easier, quicker and cheaper to implement than multi-disciplinary approaches, but may hinder the main strengths of the holistic ESS approach. Using a desired interdisciplinary approach would require significant funding which is rarely available; politicians who decide on available funding for research frequently define research topics by opening thematic calls for proposals, which prevents scientists from choosing the research topic freely.

Of the five publications that included four or more ESSs only two provided sufficient information for policymakers. Pfund et al. (2011) concluded 'that a combination of social, economic and spatially explicit assessment methods is necessary to inform land-use planning'. Sohel et al. (2014) stated explicitly in their abstract that 'the results can be used to form the base for ESS-based landscape management and future conservation priorities in the area.' These two publications are outstanding exceptions in regard to the aim of the study, yet both of them assessed ESSs with interviews or participatory approaches. Again, there is no integration of different disciplines in the data acquisition that would be necessary if ESSs were measured in bio-physical terms.

From the other three publications including four or more ESSs, the one from 2008 (Hu et al. 2008) uses a benefit transfer approach, which means the data were not adapted to the regional conditions. The results could create awareness but they would not be useful for policy-making. The two recent studies from 2014 (Lavelle et al. 2014, Sanabria et al. 2014) were conducted in Colombia and constrained to soil ESSs.

So far, only few studies have included multiple ESSs, but there is hope that these approaches will gain more importance in the future. All of the aforementioned studies are quite recent (2008–2014), so one could assume that it took some time for scientists and sponsors to realize that these studies were needed.

When looking at the sections of ESSs that have been studied, it is obvious that the ESSs that are the easiest to measure, such as carbon stocks (24% of ESSs), have been studied most, whereas less tangible ESSs, such as cultural services, are researched least. Therefore, a holistic picture cannot be formed. In addition, more applied research needs to be conducted on understanding the ecological aspects of ecosystem services and the underlying role of biodiversity for the ESS approach to become more meaningful.

Astonishingly, only 14% of the publications included rubber yield or income from rubber production, which shows that a majority of the studies did not aim to compare total economic gains received from natural ecosystems (forests) with rubber production systems. However, this comparison is of major concern when it comes to land-use planning and establishing PES schemes.

Nearly one-third (29%) of the publications included biodiversity issues in their analyses, but the crucial step of studying the direct effects of biodiversity on ESSs is often missing. Some examples of such direct effects would be the impact of birds on seed dispersal, the role of beneficial organisms on pest control and the impact of bees on pollination.

A general shortcoming of our approach is that we limited our searches to literature published in English. If we had included other languages in the review, especially Chinese, we might have found a much higher number of publications. Further, perhaps the gap between studies conducted in China and the rest of the world would have been even more pronounced. Nevertheless, including publication in other languages would probably not have changed the overall picture concerning the incorporation of multiple ESSs. Based on our knowledge, there are no studies that have analysed the whole spectrum of different ESSs.

2.5 Conclusion/Summary

ESS approaches have the potential to form a holistic view of the services that ecosystems provide to humans in one region. This review on rubber cultivation showed that the majority of publications concentrated on single or few ESSs, which can provide only limited informative value. The results of the studies that focus on only few ESSs are therefore not what policy-makers would need for land-use planning or for developing PES schemes. Leaving the disciplinary path and developing interdisciplinary studies remains to be the main challenge. Overcoming the challenge will require joint research or collaborative projects which need sufficient funding to fulfil the required tasks.

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3 Environmental and socio-economic impacts of rubber cultivation in the Mekong region: challenges for sustainable land-use

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Outline and overview

This publication gives a thorough review of the impact rubber cultivation has on the ecosystem services carbon storage and sequestration, provisioning and quality of water and provision of latex and therefore income generation. In a second step, possibilities to improve the rubber cultivation system are investigated.

Environmental and socio-economic impacts of rubber cultivation in the Mekong region: challenges for sustainable land-use

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Abstract

More than 90% of the global natural rubber production originates from monoculture plantations in tropical Asia, especially from countries forming the Greater Mekong Subregion (GMS). Rubber cultivation is expected to further increase strongly in the near future, particularly at the expense of natural forests and is accompanied by various problems and threats to farmers and the environment. Implications on carbon balance and hydrological conditions as well as socioeconomic consequences referring to the situation in the GMS are reviewed. Results indicate considerable changes in ecosystem functions and services at different spatial and temporal scales with impacts on carbon stocks and sequestration, water quality and quantity, runoff and soil erosion. The long-term dependency on rubber as a single crop affects the socio-economic conditions and livelihood of the farmers and exposes them to economic and ecological hazards. Solutions for these interrelated problems require the development of alternative land-use systems and safeguarding important ecosystem functions and services on the one hand as well as providing economic viability on the other. Common suggestions include crop diversification and improved plantation management on the farm scale and alternative land-use strategies including conservation and restoration of forest on the landscape scale. Successful implementation of more sustainable concepts is only feasible within a socioeconomic framework, involving farmers and political decision-makers in the conceptualization process and the identification of trade-offs between ecological requirements and economic feasibility.

Keywords
Land-use change, Land-use scenario, Intercropping, Deforestation, Livelihood

Review Methodology
We used the Scopus bibliographic database for the current state of knowledge and 'rubber' as the basic keyword in combination with various other terms related to our review topic. We also considered relevant references from the articles obtained by this method. All authors are researchers in the presently (2012–2016) conducted German–Chinese joint project SURUMER (Sustainable Rubber Cultivation in the Mekong Region, https://surumer.uni-hohenheim.de) and contributed further information according to their specific background and literature sources.

3.1 Introduction

Natural rubber is an important primary product in the global economy and is found in many commonplace items. It is obtained from latex, the sap of the rubber tree (*Hevea brasiliensis* Muell. Arg). By far the biggest proportion (70%) of this natural resource is used in the tyre production (European Tyre and Rubber Manufacturers' Association (ETRMA) 2011, European Tyre and Rubber Manufacturers' Association (ETRMA) 2012). Considering all rubber-based products, the vehicle industry claims around three-quarters of the world production.

Although the rubber tree is native to Amazonia, more than 90% of the total natural rubber originates from tropical Asia (Food and Agriculture Organization, Statistical Division (FAOSTAT) 2013). The top five producing countries presently are Thailand, Indonesia, Malaysia, India and Vietnam. Demand for rubber increased enormously with the economic upturn in Asia and is expected to further increase strongly in the near future. In 2011, China used one third of the natural rubber produced worldwide – more than the consumption by the European Union member states, USA and Japan combined (European Tyre and Rubber Manufacturers' Association (ETRMA) 2012). Consequently, rubber cultivation has grown enormously within the last few decades, especially in the so-called Greater Mekong Subregion (GMS), comprising the countries bordering the Mekong River (Cambodia, Laos, Myanmar, Thailand, Vietnam and the Chinese province of Yunnan).

In Vietnam, the plantation area covered about 910 000 hectares (ha) by the end of 2012, including about one-third of trees too young for tapping (Association of Natural Rubber Producing Countries (ANRPC) 2010, FPT Securities Joint Stock Company 2013). This is twice the area compared with 2004 (Asia Commodities 2013). In the northeastern provinces of Thailand, the rubber cultivation area expanded from 42 000 ha in 2002 to 288 000 ha in 2011, an increase of 580%. In the same period, the forest area in this region declined by 18% and the area of agricultural land by 50% (Mongkolsawat and Putklang 2010). In Laos, about 140 000 ha of rubber were planted by 2008 and this area is expected to double within the next decade (Douangsavanh et al. 2008). In Cambodia and Myanmar, the cultivation area is expected to grow strongly in the near future (Li and Fox 2012). In Xishuangbanna, in the southern part of Yunnan Province (China), rubber cultivation area increased from 153 000 to 424 000 ha between 2002 and 2010, equivalent to 175%. This expansion mainly occurred at the expense of natural forests (Xu et al. 2014, Li et al. 2007). In 2012, the total harvested area of rubber in the GMS countries was more than 3.5 million ha[2]. Li and Fox (2012) estimated additionally more than 500 000 ha of young trees not yet producing rubber. If the present expansion of rubber continues, the cultivation area in the GMS could quadruple by 2050 (Fox et al. 2012). Expansion is also likely to shift rubber production further into higher altitude and latitude. New genotypes (clones) of rubber are able to tolerate dry periods and lower temperatures without important loss of latex yield. Plantations have now expanded to 27°N latitude, to elevations up to 1100m (Priyadarshan 2011) and into dry subhumid areas of the GMS (Clermont-Dauphin et al. 2013). Furthermore, projected impacts of climate change in Xishuangbanna indicate that the area conductive to rubber plantations, currently limited by climatic conditions, expands to approximately 75% of the total area (Zomer et al. 2013).

The large expansion of rubber cultivation and the additional yield expected from the developing young trees, however, lead to an increase in harvested rubber stocks, because the global yield is higher than the industrial demand. The global natural rubber market was oversupplied with a surplus of 220 000 tonnes (t) in 2011 and with 410 000 t in 2013. This trend is expected to continue. In consequence, global natural rubber prices declined (Global and China Natural Rubber Industry Report 2013-2016).

In the GMS, rubber is produced almost exclusively in monoculture plantations (unlike in Indonesia, where rubber is often part of mixed agroforestry systems). New plantations are usually established on bare soil after clearing the former vegetation, terracing is carried out on slopes. Latex harvest begins at a tree age of 7 years, maximizes at 20 years and typically ceases at around 35 years, leading to the end of the economic lifespan. Finally, the trees are cut and a new cultivation cycle starts.

3.2 Consequences of the rubber boom

Natural rubber is a renewable resource. This characteristic is often stressed by national and private companies, the rubber industry as well as by traders of rubber products (e.g. mattresses,

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[2] Based on FAOSTAT data of 2012, Area harvested (Food and Agriculture Organization, Statistical Division (FAOSTAT) 2012. The specifications for China include Hainan island with a plantation area of around 500 000 ha (Dong et al. 2012). For Laos, no FAOSTAT data are available, here the data of Douangsavanh et al. (2008) of 140 000 ha in 2008 were used.

toys and rubber wood furniture) to suggest that rubber cultivation is climate-smart and environmentally friendly. However, renewable does not necessarily mean sustainable. The shift from tropical forests and traditionally managed swidden fields to largescale rubber monoculture results in a loss of ecosystem services (Hu et al. 2008) and significant changes in ecological functions, socio-economic conditions and human welfare. In the following, important indications for these effects referring to the situation in the GMS are reviewed.[3]

3.2.1 Impacts on carbon balance

Deforestation and burning of natural tropical forests has significant impact on the global carbon cycle by decreasing the above- and below-ground carbon stocks and by increasing rates of carbon emissions to the atmosphere (Houghton and Hackler 1999). Deforestation contributes 12–15% of the total anthropogenic CO_2 emissions, from both biomass and soils (van der Werf et al. 2009). Li et al. (2008) estimated changes in biomass carbon stocks in Xishuangbanna (Southern Yunnan, China). They found that in the past, when the region was completely forested (1.9 million ha), the total carbon biomass would have been approximately 212 Tg. Owing to deforestation and forest degradation, the total carbon stock decreased to 81 Tg in 2003.

However, there are great uncertainties in carbon in the total ecosystem for several major land covers that are related to important land-use transitions (including rubber) in Southeast Asia (Ziegler et al. 2012). For example, there is a high variability in below-ground woody carbon. Data from naturally grown forest in the GMS and Malaysia range between 11 and 74 Mg C/ha and rubber plantations from GMS countries show 5–32 Mg C/ha in root biomass (Yuen et al. 2013). The changes in carbon balance by conversion of natural tropical forest into rubber plantations depend on the amount of carbon released by forest destruction and the amount of carbon sequestered by the plantations. We used 38 data sets on biomass accumulation in rubber to calculate a single graph (Figure 1). It shows a steady increase of carbon stock in young and mid-age rubber plantations. Integration of the fitted equation returns a time-averaged rubber biomass of 120 ± 40 Mg C/ha for 30 years after planting. Yang et al. (2013) calculated a timeaveraged rubber biomass of 97 Mg C/ha in a Xishuangbanna study site for a 25-year period. This result falls in the range of our calculated curve.

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[3] Impacts of rubber cultivation on biodiversity are reviewed in a separate article submitted to CAB Reviews (He P, Martin K. Effects of rubber cultivation on biodiversity in the Mekong Region).

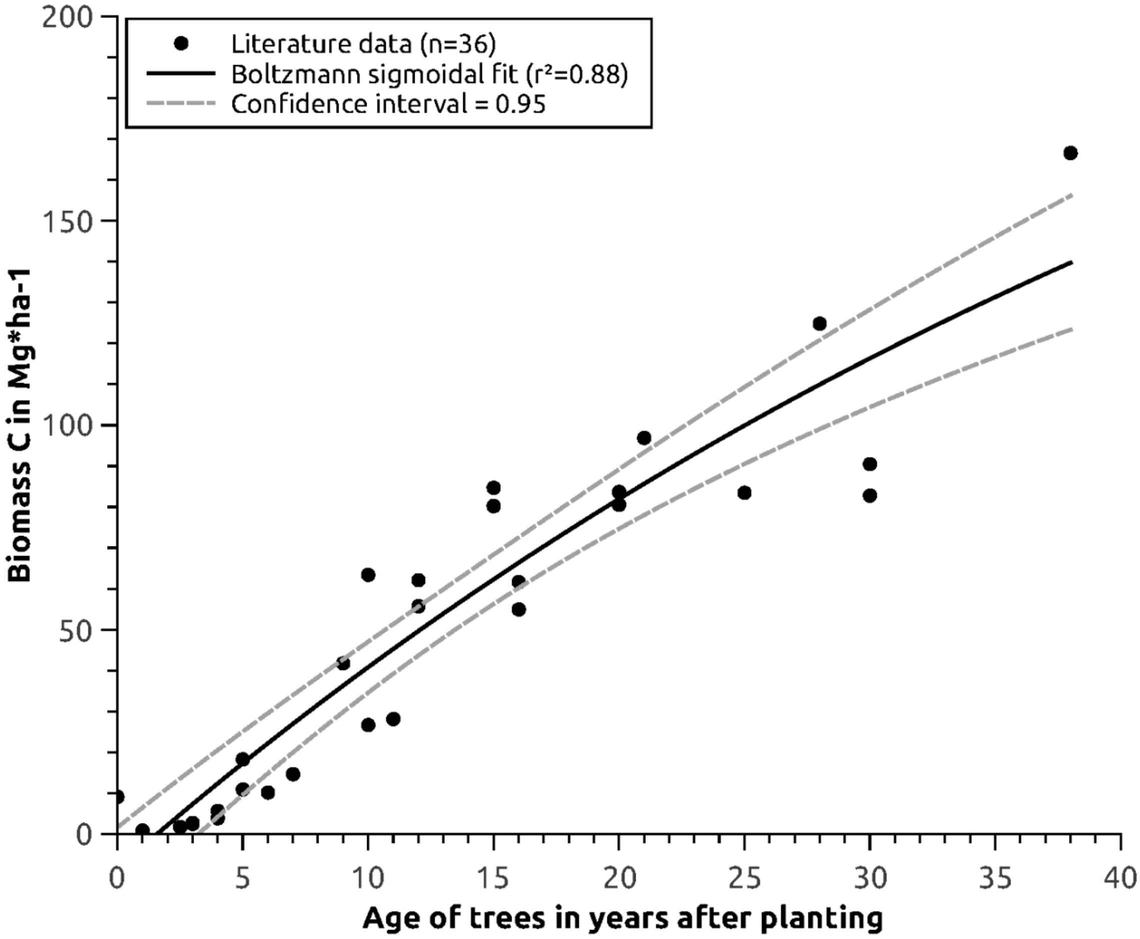

Figure 1 Above-ground biomass in rubber plantations. Sources Carmo et al. (2005), Yang et al. (2005), Oliveira et al. (2006), Cheng et al. (2007), Fernandes et al. (2007), Mandal and Islam (2008), Chantuma et al. 2012) and Saengruksawong et al. (2012).

For Southern Yunnan, Cotter et al. (2009) estimated that clearing of 1 ha of relatively undisturbed tropical seasonal rainforest releases about 438 t of CO_2. A rubber plantation in the same region below 800m sequesters approximately 192 t C/ha (equivalent to 703 tCO_2/ha) during its lifetime of 30 years (given a litter mass of 107 t C/ha and a latex output of 23 t C/ha). Consequently, a fully grown rubber plantation needs around 20 years to re-sequester the amount of CO_2 released by forest clearance. However, this balance ignores changes in the soil carbon pool, i.e. the amounts of carbon released by forest conversion and the soil carbon sequestration under rubber plantations. The extent of change in soil carbon pools strongly depends on mean annual precipitation and dominant soil clay mineralogy (Powers et al. 2011). Land-use change in the tropics from forest to plantation usually reduces total soil carbon stocks by roughly 5% on average (Powers et al. 2011).

For change of secondary forest into rubber plantations in Xishuangbanna, Yunnan, de Blécourt et al. (2013) found a reduction of nearly 20% of the initial soil carbon in a total soil depth of 1.2 m. In the topsoil (0–15 cm), the largest decrease was in the first 5 years following land-use change, when soil carbon stocks declined to approximately 80% of the original amount. The carbon stock reached a steady state after approximately 20 years at 68% of the original stock. Similarly, Yang et al. (2004) found soil carbon losses of 24% in a 3-year-old plantation and 21% in a 7-year-old plantation within a depth of 0–6 m. Zhang et al. (2007) confirmed that the soil carbon level decreases in the growth phase of the plantations, but found that it begins to rise again after an age of 26 years to reach its maximum at 40 years. This change in soil organic pools correlated negatively with latex yield.

Furthermore, there is evidence that conversion of secondary forest into rubber plantation significantly decreases soil microbial biomass carbon (Fang et al. 2011). According to Werner et al. (2006), 20-year-old rubber plantations show a lower CH_4 uptake and lower CO_2 emissions compared with primary and secondary forest sites. However, they suggest that the soil moisture and litter fall are important factors influencing carbon emissions which depend not only on climate, but also on rubber plantation age.

Quantification of rubber plantation carbon stocks and sequestration also provides a base for carbon trading options. Yi et al. (2014) estimated the carbon payments required to equal the potential rubber revenue for local farmers by comparing three land-use scenarios. They conclude that the prices in the carbon market would have to be considerably larger than they are currently to compete with the profitability of rubber.

Besides the conversion of forests, rubber plantations in the GMS also expand on the expense of various types of open land such as grassland, fallows and abandoned swidden fields. Preparation of such land-use types for rubber releases significantly less carbon from plant biomass into the atmosphere than forest. For Southern Yunnan, Cotter et al. (2009) estimated a release of 110 t/ha CO_2 from shrub-land, 19 t/ha from grassland and 438 t/ha from seasonal rainforest. For soil carbon, Powers et al. (2011) found that the establishment of perennial tree plantations on lands that were previously grazed or cropped increased soil carbon stocks, whereas the conversion of grasslands shows no change. Fallow or swidden land may include a wide range of different stages of natural succession between degraded grassland and secondary forest. Therefore, according to analyses of Ziegler et al. (2012), transitions from swidden agriculture to rubber plantations do not necessarily produce positive carbon outcomes.

Studies from Zhou et al. (2013) in Xishuangbanna indicate that in tropical seasonal rainforests carbon export via stream flow changes the carbon balance only modestly. The carbon stock in rubber plantations is affected by factors such as fertilization, the management of the undergrowth vegetation and site characteristics.

Large-scale rubber plantation in Xishuangbanna has increased the rate of soil loss by more than 50% (Zhou et al. 2014). Nuanmano et al. (2012) report erosion rates of about 100 Mg/ha/a from a rubber plantation area in Thailand. This rate can be considered as severe erosion. It is twice as much as shown from the erosion data summarized by Wiersum (1984), who states median values from below 1 Mg/ha/a for tree crops planted with cover crops or mulched to over 50 Mg/ha/a for clean weeded plantations with removed litter. Erosion in rubber plantations on sloping land depends on management practices and varies in a wide range. Terraces parallel to contours may alleviate soil organic carbon losses caused by the conversion of secondary forest to rubber plantation (de Blécourt et al. 2014) and may reduce soil erosion. Rubber planting induces not only soil carbon loss, but is also found to lead to extensive humification (Zhang et al. 2013), acidification and changes in the composition and quantity of nitrogen compounds (Cheng et al. 2007, Chen et al. 2012, Li et al. 2012).

3.2.2 Impacts on the hydrological cycle

With proceeding land-use, conversion of forest into rubber plantation, effects on climate as well as on water availability are reported. The average temperature of rubber-producing areas in Xishuangbanna increased significantly since 1960s, while the regions without rubber in Yunnan showed no change (He and Zhang 2005). There was also strong reduction in the number of fog days in Xishuangbanna since the 1950s (Wu et al. 2001, Zhou 2008). Observations in Xishuangbanna also indicate a reduced streamflow and dried up wells (Qiu 2009). Such circumstances suggest that rubber cultivation affects the local and regional water balance through the eco-physiological characteristics of rubber trees and by the plantation design and management.

Comparative studies in different catchments of Xishuangbanna by Tan et al. (2011) confirmed that evapotranspiration from rubber plantations is 15–17% higher than in primary rain forest and therefore considered rubber trees to act as ‘water pumps’. Tan et al. (2011) concluded that soil water storage during the rainy season is not sufficient to maintain the high evapotranspiration rates in rubber plantations, resulting in zero flow and water shortages during the dry season. Studies by Liu et al. (2013) showed that rubber trees extract their water mostly from the top 30 cm of the soil in the rainy season. During the late dry season, the depth of water uptake shifts to deeper soil levels. Rubber is a brevi-deciduous tree, because trees older than 3–4 years shed senescent leaves. After leaf shed, trees remain nearly leafless for up to 4 weeks. Whether this process is induced by drought (Carr 2012) or day length (Guardiola-Claramonte 2008) is not yet clear.

Guardiola-Claramonte et al. (2008) indicated that at a secondary forest site root water uptake is linked to water availability in the form of rain. In Xishuangbanna, native forest trees rehydrate after occasional rain events during the dry season, or shortly after the start of the rainy season (Guardiola-Claramonte et al. 2008). In the same region, leaf flushing in rubber occurs at the midst of the hottest and driest period, weeks before the rainy season starts. Flushing leaves during the dry season imply that the tree must have access to sufficient reserves of water for leaf expansion. Therefore, Guardiola-Claramonte et al. (2008) claimed that the additional stem potential needed for flushing is acquired through deep subsurface water uptake. Water storage depletion from the subsurface soil during the dry season increases water losses through evapotranspiration and reduces discharge from the catchment (Guardiola-Claramonte 2010). Carr (2012) found the validity of these assumptions difficult to reconcile. In a review on studies on water requirements of rubber, he found that few publications concerning water requirements of rubber trees exist, but they all show (maximum actual) evapotranspiration rates lower than might be expected for a tree crop growing in the tropics. Kobayashi et al. (2014) used sap flow measurements to study variations in transpiration rate in a rubber stand in Cambodia. Their results indicate that rubber trees actively transpire in the rainy season, but become inactive in the dry season. Kobayashi et al. (2014) argue that depletion in deep-soil moisture or stream desiccation due to large water uptake by rubber trees may partly be explained by the low ability of rubber trees to conserve the soil water, but high evapotranspiration could also be attributable to other water loss components, e.g. wet canopy evaporation, soil evaporation and transpiration of understory vegetation. To obtain a clear picture of the water budget of a rubber plantation and predict the sustainability of rubber cultivation with regard to its water use, processes at different scales, i.e. canopy, trees and leaves, need further investigations in a comprehensive manner (Kobayashi et al. 2014).

Another approach to explain the reduction in fog and streamflow is the diffuse reflection of light (albedo) from the canopy of rubber plantations, expressed through the ratio of the reflected solar radiation to the incoming solar radiation. Among other parameters, it depends on plant cover, i.e. leaf area index (Hales et al. 2004). Rubber plantations were found to have a lower leaf area index than secondary forest (Rusli and Majid 2014), leading to the assumption that the albedo of rubber is higher than natural forests. With a higher albedo, radiation transfer increases back to space, reducing clouds and rainfall (Gash and Shuttleworth 1991). Therefore, canopy characteristics rather than water use patterns of rubber plantations might account for the observed hydrological changes in the Xishuangbanna region.

Rubber plantations in Xishuangbanna show an increased surface water runoff (Mann 2009), resulting in a soil erosion rate that is 40 times higher than in tropical forests (Wu et al. 2001). Since rubber cultivation largely expands to steep elevations, this will increase the probability of landslides, the risk of destructive flooding of rivers and hydraulic stress for aquatic species. Increasing amounts of deposited sediment on the river bed reduces living space for macro-invertebrates and juvenile fish by clogging the pore space of the river bed. This results in a loss of fish habitats and in a reduction of biodiversity in the aquatic fauna (Noack 2012). Rubber cultivation areas could also affect downstream regions, including effects on hydropower projects planned or existing in the Mekong river basin (see Grumbine and Xu 2011).

The conversion of natural forest and traditionally managed swidden fields into rubber plantations also affects the quality and quantity of ground- and surface water. According to Tang et al. (2010), farmers in Xishuangbanna reported changes in the water resources in the last few years, especially a drop of the groundwater level and the villagers have to buy bottled drinking water (Tingting 2007). Many farmers approve that rubber cultivation is one of the factors causing potable water shortage (Zhou et al. 2011). In addition, rubber production in monocultures requires the use of high amounts of pesticides and chemical fertilizers (Qiu 2009, Ziegler et al. 2009). These agrochemicals enter the aquatic system by rainfall-induced wash-off, threatening water quality for humans and aquatic organisms.

3.2.3 Effects on socio-economic conditions and livelihood

In the GMS countries, three types of producers cultivate rubber: state-owned companies or farms, private entrepreneurs and smallholder farmers with portions varying by country. Smallholders comprise between 20 and 40% in Laos, Cambodia and Vietnam, 50% in China and about 90% in Myanmar and Thailand (Fox and Castella 2013). Free market and the lure of cash encouraged numerous private smallholders to give up their traditional land-use and turn to rubber over the last two decades.

Rubber cultivation can result in significant increases in household income and is hence a possibility to move households and communities out of poverty. Manivong and Cramb (2008) found positive net present values for investments in smallholder rubber production in northern Laos and Liu et al. (2006) observed a threefold increase in per capita income and expenditures over a period of 15 years due to rubber production in a township of Xishuangbanna, Yunnan. Farmers switching from swidden agriculture to rubber cultivation profited the most and ethnic minorities in Southern Yunnan even expanded rubber cultivation into neighbouring Laos (Sturgeon 2010).

In addition to smallholders, private entrepreneurs from China, Vietnam, Malaysia and Thailand invest heavily in rubber plantations in non-traditional rubber-growing areas of neighbouring Laos, Cambodia and Myanmar. Vietnam and Thailand also expanded in their own countries in areas where rubber is not yet grown (Fox et al. 2014). In Laos, up to 75% of the investment in rubber comes from foreign companies (Laungaramsri 2012).

Especially in Laos, Cambodia and Myanmar such companies either establish large-scale plantations under land concessions, or they use a contract-farming model with smallholders. In the first case, investors fully control the management, there is only low cooperation with local people, turning farmers into landless labourers. In the case of contract farming, farmers are still landowners and maintain their decision-making. According to the kind of contract, farmers either provide land and labour and the company provides seedlings and equipment or the company hires additional labour, sometimes the contracted farmer or workers of foreign origin. The benefits for the farmer are between 30 and 70% (Fox and Castella 2013). Overall, small-scale farmers are the backbone of natural rubber production in the GMS.

However, by deciding to grow rubber, farmers are committing themselves for decades to come and are thus dependent on a single product, which exposes them to further risks. Rubber is almost exclusively used for industrial purposes, so its demand depends strongly on the dynamics of the world economy. This means that there are bust-and-boom cycles in rubber prices, exposing farmers to income insecurity. In the period from 2011 to 2013, natural rubber futures prices in China plummeted by 41% and are expected to decline further (Global and China Natural Rubber Industry Report 2013-2016). With rubber tree plantations, other than with annual crops, farmers are not able to react with a shot-term production strategy on changing market situations. In addition, there are ecological hazards due to crop diseases, pests, unfavourable weather conditions or changes in climate.

Furthermore, Xu et al. (2005) concluded that rubber plantations in Yunnan eroded the capacity of farmers to manage ecologically diverse landscapes and to participate in market networks. The abandonment of traditional land-use practices in favour of a single crop may have severe implications for food and nutritional security of the rural population. Fu et al. (2010) stated that smallholder rubber producers suffer from livelihood vulnerability due to excessive rubber cultivation. Rural food security is predicted to become more tenuous in the Mekong region (Xu et al. 2014). This also includes the availability of natural resources, such as non-timber forest products, which rubber plantations do not provide.

Expanding rubber cultivation affects not only farmers in the rural areas, but also the urban population. In a survey conducted among local residents of Xishuangbanna (Ahlheim et al. 2012), it turned out that nearly 90% of respondents perceived an improved economic situation as a consequence of rubber cultivation in the region. At the same time, nearly 80% of respondents think that the environmental situation has deteriorated. Virtually all respondents consider observed problems such as reduction and pollution of water resources, loss of natural vegetation and species, soil erosion and alterations in weather and climate as consequences of rubber cultivation. The study also shows that many respondents would be willing to contribute financially to a project that would improve the situation by converting a part of the present rubber farmland back into forest. An average household would be willing to pay nearly 0.5% of the annual income, on average, to such a project (Ahlheim et al. 2012). Another study found that even among residents of Shanghai exists a non-negligible willingness to contribute financially (about 0.3% of the annual income) to the preservation of natural forest or a specific rare tree species in Xishuangbanna (Ahlheim et al. 2014), indicating a non-use value of the natural environment there.

3.2.4 Towards Sustainable Rubber Cultivation

The review of studies clearly indicates that increasing rubber cultivation in the GMS is accompanied by various problems and threats to farmers and the environment, though an increasing awareness of environmental deterioration leads to a change in values (Figure 2). This situation therefore requires the development of more sustainable land-use concepts. Generally, most concepts and studies are aiming at improved land-use and management and centre on the conceptual framework of ecosystem services, suggesting classification, indicator and assessment schemes (Fisher et al. 2009, van der Biest et al. 2014, Rodriguez-Loinaz et al. 2015, van Oudenhoven et al. 2012, La Rosa et al. 2013 and van Haaren and Albert 2011). However, most research on ecosystem services so far focused only on biophysical and valuation assessments of putative services and is not embedded in a social process for implementation (Cowling et al. 2008) and many problems concerning the practical implementation of concepts remain unsolved (Hermann et al. 2011, Kandziora et al. 2013 and van Haaren et al. 2014).

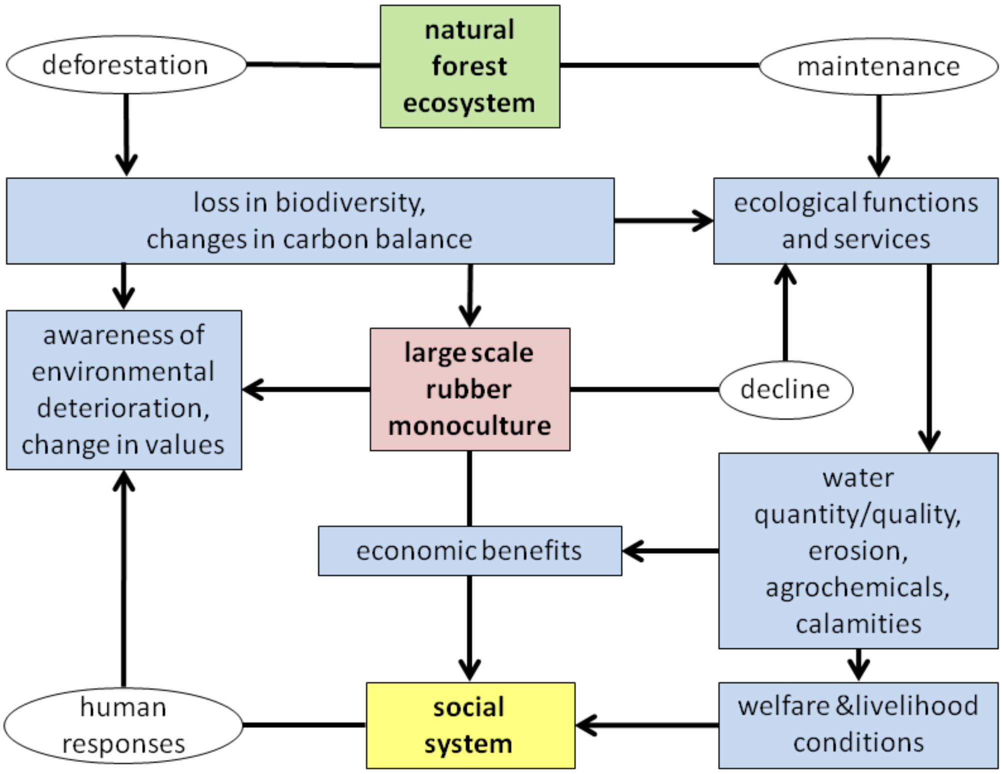

Figure 2 With evidence from this review, land transformation of forest into rubber monoculture triggers a shift in a variety of direct and indirect effects on ecosystem functions and services provided by the natural system. Changes in the local climate, the carbon and the hydrological cycle imply a higher liability of rubber cultivation to various ecological risks, affecting economic benefits as well as the welfare and livelihood conditions of the people. An increasing awareness of negative effects of rubber cultivation on the local environment among farmers and leading to changes in the value system among the population.

In consideration of this background, solutions for the specific problems of monoculture rubber cultivation should comprise: (1) the interdisciplinary analysis and quantification of ecological processes and services affected by rubber cultivation compared with natural forest conditions; (2) the development of alternative land-use strategies including the identification of trade-offs and synergies between safeguarding functions and services on the one hand and the socio-economic viability on the other; and (3) the identification of incentives of acceptance and implementation of the concepts by farmers and other stakeholders.

Different aspects of these challenges are recognized in various studies. Among the ecosystem services affected by rubber cultivation, regulation and quality of water is a major concern of the local people (Tang et al. 2010, Tingting 2007 and Zhou et al. 2011). A better understanding of the hydrological cycle in rubber dominated landscapes is necessary for the prediction of interrelated effects on the local climate and the fate of leached pesticides. To examine options for carbon trading schemes, more detailed information on carbon stocks and sequestration over time from rubber plantation is needed (Yi et al. 2014a).

Referring to alternative land-use strategies, a common suggestion to mitigate the ecological shortcomings of rubber monocultures as well as their economic risks is to practice intercropping and diversify farmers' product portfolio (Yi et al. 2014a, Xu 2006). Under present conditions, farmers prefer rubber monocultures with high returns over rubber intercropping with lower returns (Aenis et al. 2014, Zhang et al. 2015). The suggested development of rubber agroforestry systems or 'jungle rubber' particularly in locations with high ecological values for watershed protection and soil erosion reduction (Yi et al. 2014a) involves the same economic problems. In Indonesia, where 'jungle rubber' is common, rubber productivity is very low and farmers clearly prefer the shift to high-yielding monocultures (Williams et al. 2001). On the landscape scale, Yi et al. (Yi et al. 2014b) recommend the conversion of rubber plantations into forest in high elevations and on steep slopes as well as buffer zones along streams, but this would require payment for compensation.

Overall, the existing ecological and socio-economic problems of rubber cultivation in the Mekong region are widely acknowledged. Concordantly, suggestions for land-use change are based on system diversification and forest restoration and that both require economic incentives for the farmers. Although rubber prices presently show a downward tendency, the main obstacle to change is still the high economic attractiveness of rubber production coupled with too few alternative income sources. Beyond that, farmers need more education on economic risks of rubber monoculture production and on its ecological consequences for their environment and livelihood (Aenis et al. 2014). An implementation of alternative concepts also requires the strong involvement of policy makers at the national and provincial level and the general public by communication of concepts, costs and benefits of alternative land-use strategies (Xu et al. 2014, Aenis et al. 2014).

In consideration of these challenges, an approach to develop an integrative, applicable and stakeholder-validated concept for sustainable rubber cultivation is undertaken by the German-Chinese joint project SURUMER (Sustainable Rubber Cultivation in the Mekong Region, https://surumer.uni-hohenheim.de), conducted in Xishuangbanna 2012–2016. The basic project approach is the assessment and the quantification of major ecosystem processes and services in forest and rubber plantations, especially water balance and carbon dynamics. Along with results from economic analyses and valuations, these data are used to develop alternative land-use strategies.

To create economic and ecologically viable solutions, the SURUMER project aims at integrating ecologically suitable and economically valuable wild plants from the natural forest into rubber plantations. Candidates are species which are traditionally used as medicinal plants in that region (Ghorbani et al. 2011, Ghorbani et al. 2012). Among these, e.g. wild *Asparagus* species (Bucher et al. 2011) and *Paris polyphylla* (Zhang et al. 2012) are of high value in traditional Chinese medicine and have become rare due to overexploitation and the loss of forest areas. On landscape scale, different land-use scenarios on the future development of rubber cultivation are generated and analysed with multiple disciplinary and interdisciplinary modelling approaches, leading first to a bio-physical assessment of each scenario. In a second step, this assessment is supplemented with socio-economic appraisals on expected changes in household income and economic welfare of the rubber farmers. Scenario development involves different stakeholder groups including farmers, regional decision-makers and provincial policy levels. By catering the needs and wants of these groups, awareness of the consequences of different scenarios will be raised.

3.3 Conclusion

In the GMS, expanding rubber cultivation changes structure and function of natural ecosystems at different spatial and temporal scales and affects the socio-economic conditions and livelihood of the farmers in different ways. Solutions for these interrelated problems require not only a focus on ecosystem services according to common concepts (Haines-Young and Potschin 2012), but also entail a broader understanding of interlinked ecosystem functions. Solutions from the ecological point of view in designing experimental rubber cultivation systems to mitigate undesirable effects on ecosystem processes may not necessarily generate added value for the farmers. In order to create ecologically and economically sustainable rubber farming and management schemes, trade-offs between the ecological and economic requirements and expectations need to be identified. In addition, the challenge is to consider effects of spreading rubber cultivation on the landscape scale, where they directly affect people and their livelihood. Transferring scientific concepts into practical land-use requires a social–ecological systems approach and valuation must consider equally the social, bio-physical and economic dimensions within a multi-scale framework (Xu et al. 2014, Reyers et al. 2013 and Fontaine et al. 2014). Successful implementation is only feasible if relevant stakeholders (farmers and policy makers) of different levels are involved during the entire conceptualization process, ranging from a joint definition of development goals to evaluation, in particular the joint assessment of trade-offs between ecological requirements and economic feasibility.

Various substantial approaches and concepts for the development of sustainable rubber cultivation within a socio-economic framework have been developed in the recent years, with most research conducted in Xishuangbanna, Southern Yunnan. Taken together, these studies provide a promising base for practical implementation not only in that region, but also in other potential rubber cultivation areas across the Mekong region facing the same problems.

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4 Biodiversity and Ecosystem Services - A case study for the Assessment of Multiple Species and Functional Diversity Levels in a Cultural Landscape

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Outline and overview

This paper gives a detailed analysis of how biodiversity changes under different rubber cultivation scenarios for the Naban River Watershed National Nature Reserve. Therefore, data of various flora and fauna species were normalized to allow their integration into existing modeling software. Different groups of species were analyzed separately to show the differing impact on biodiversity if only certain species groups are considered (red list species versus species with a direct use value such as bees producing honey).

Biodiversity and Ecosystem Services - A case study for the Assessment of Multiple Species and Functional Diversity Levels in a Cultural Landscape

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Abstract

The expansion of large-scale plantations has a major impact on landscapes in the Tropics and Subtropics. Crops like soy bean, oil palm and rubber have led to drastic changes in land cover over the past decades, thereby altering ecosystem functions and services (ESS). These shifts in ESS such as climate regulation services, erosion and water cycles, biodiversity as well as soil fertility or the provisioning of raw materials have been assessed on a general scale through several models and software solutions (InVEST, ARIES, MIMES). However, suitable methods for the integration of biodiversity assessments in agricultural landscapes are scarce.

With this study, we introduce a methodology for incorporating multiple levels of species diversity into models to supplement the evaluation of ESS. We collected data sets from both published and unpublished sources on the distribution of vascular plants, selected pollinator groups, ground beetles, ungulates as well as amphibians, mammals, reptiles and birds in rubber-dominated landscapes, with a focus on our study sites in Southwest China and Thailand. Based on this information, we developed a common classification scheme that enables the integration of different facets of biodiversity (species diversity and functional diversity) to complete an interdisciplinary ESS assessment.

Species diversity data were normalized to allow a broad assessment of the impact of rubber cultivation on multiple levels of biodiversity. This resulted in a matrix of different land-use types and their suitability as habitat for the respective species groups. The findings were applied to two alternative land-use scenarios of our main research site in China to highlight the potential effects of land-use and management decisions on species and functional diversity. Our results showed clear indications. The conservation oriented scenario did score higher for habitat suitability in both total species (+5%) as well as IUCN Red List species (+6%) assessments compared to the current state or business as usual scenarios (-2% and -3%).

The process presented here allows for an application with established ESS/ESF software programs, in our case InVEST, using aggregated indices while still providing further information on the expected impact of the analyzed scenarios on specific species groups.

Keywords

Ecosystem Service Assessment, Biodiversity, InVEST, rubber, mainland Southeast Asia

4.1 Introduction

Although biodiversity is not an Ecosystem Service (ESS) in the narrow sense, it is inextricably linked to ESS because biodiversity influences their provision (MEA 2005). Several analytical tools exist to allow for replicable and quantifiable ESS analyses, such as InVEST (Integrated Valuation of Ecosystem Services and Trade-offs), ARIES (Artificial Intelligence for Ecosystem Services, 2016a, b) and MIMES (Multiscale Integrated Model of Ecosystem Services), which all belong to the group of independently applicable and generalizable landscape-scale models (Bagstad et al. 2013a).

InVEST is provided by the Natural Capital Project (The Natural Capital Project 2015) and has a modular approach. Its tools are based on deterministic production functions; two of them cover biodiversity issues: habitat quality and habitat risk assessment. The habitat model of InVEST has already been used in different scientific analyses (e.g. Dhakal et al. (2014), Terrado et al. (2016)). ARIES includes not only the provisioning of ESS and possible trade-offs, but also the flow and the use of ESS. The current release includes eight ESS and even though there is no explicit model on biodiversity, the link between biodiversity and ESS is stated frequently. Although InVEST and ARIES differ greatly in their methodology, a comparison of the two tools yielded similar results for the ESS carbon, water and scenic view sheds for a case study in Arizona (Bagstad et al. 2013b). MIMES aims at supporting sustainable and ecosystem-based management planning with a special focus on incorporating concepts to capture the dynamic character of coupled human and natural systems. Biodiversity is included in the modeling process as being part of the biosphere, but neither explicitly mentioned as output nor included in the research questions so far presented in the literature (Boumans et al. 2015).

Out of these three possibilities we chose InVEST for our analyses because it has the most detailed routine for the evaluation of biodiversity within the framework of available ESS Assessment tools. By combining aspects of habitat suitability with the potential to analyze alternative scenarios of land-use decision, it was the most suitable tool to meet our research questions. In addition it is open-source, well-documented and the models have a high generalizability (Bagstad et al. 2013a).

Human activities have a strong impact on the distribution of habitats, land cover and landscape patterns throughout the world. Deforestation, transformation, degradation and intensification impact the quality and quantity of ESS provided by a given landscape; the potential as habitat for species being one of these (Ahrends et al. 2015). The expansion of rubber (*Hevea brasiliensis)* in mainland Southeast Asia (MSEA) is providing a suitable case study to asses some of these impacts (Ziegler et al. 2009), as a range of studies have covered the implications for single species groups, but so far a general assessment is amiss.

The shift from low input agriculture to intensified systems additionally reduces functional and species diversity and consecutively, the availability of potentially useful habitat types for species originating from natural forest. Rubber plantations generally harbored less than half of the original fauna and flora species richness compared to natural forest. Most studies therefore found that many of these species are unable to exist permanently in rubber plantations. There is clear evidence that the existence of natural forest area is essential for the conservation of large proportions of native forest species in rubber-dominated landscapes (He and Martin 2015).

4.1.1 Study sites

The research area to which our model is applied is the Naban River Watershed National Nature Reserve (NRWNNR, 22°08'N 100°41'E) in Xishuangbanna, Yunnan Province, PR China. The nature reserve covers 271 km^2 and its elevation ranges from 500 m to 2300m a.s.l., covering the watershed of the Naban River, a tributary of the Mekong River. It features an especially high diversity of natural vegetation types, as well as a variety of land-use systems due to the topographically and ethnically diverse background of the region (Zhu 2005). In Xishuangbanna, economic development and biodiversity conservation compete for the same land. So far, economic goals have dominated: from 1976 to 2003, 140,000 ha of tropical rainforest were replaced by rubber (Li et al. 2005). Since then, forest clearing has continued.

Data on mammal diversity were mainly collected in the Province Surat Thani in southern Thailand, in and around the Tai Rom Yen National Park (TRY). TRY is located in the East of the province and covers an area of about 400 km^2 with an elevation range from around 100 to 1200m a.s.l.. The protected area was created in 1991 and includes evergreen forest, partly characterized by limestone formations (DNP 2013), as well as cultural landscapes dominated by rubber plantations. This region is a traditional rubber cultivating area (Li & Fox 2012) that lies within the transition zone of the Indo-Burma and Sundaland biodiversity hotspots, home to more than 300 mammal species (Myers et al. 2000).

4.2 Material and Methods

4.2.1 General Approach

With this study we aim to introduce a methodology for incorporating multiple levels of species diversity into models to supplement the evaluation of ESS. We have combined original data from field work within our study sites with complementary data on biodiversity from detailed literature review studies. In order to make these results comparable we conducted a normalization process and integrated this data into the spatially explicit model habitat suitability of InVEST.

4.2.2 Own studies on site

Floristic inventory data were analyzed considering the occurrence of rare and endemic species according to the IUCN (International Union for Conservation of Nature) red list of species and the value of a vegetation type for human use, expressed through the number of medicinally usable plants. A total of 18,901 m^2 of land in NRWNNR were surveyed based on 610 plots, with different sizes (1m^2 for rice paddies to 400m^2 for forest plots) for different land-use types (see table 1). Some 1,252 species from 635 genera and 158 families were identified. More details (sampling structure, size bias and categorization) are described in Cotter et al. (2014).

We assessed the presence of ungulates using camera traps, transect surveys and spoor plots in two different protection zones in 2010, for detailed data collection see Treydte et al. (2013). Mammal diversity data were collected in the years 2013 and 2014 through camera traps, transect surveys and interviews with local farmers living in the periphery of the TRY National Park in Thailand. In Thailand, assessments were undertaken inside the natural forest, in the adjacent farmland and at the forest-farmland boundary. A total of 180 randomly selected farmers were interviewed around the TRY National Park about wildlife species and number of individuals that came to the farmland. Camera traps were repeatedly installed at 30 locations along the forest edge around the protected area. Further, 21 locations were sampled several km into the forest, along wildlife trails. Cameras remained at least two weeks in one location. In addition, data were collected on 25 one-km trail-transects within the forest and along 25 complementary transects up to one km into the farmland, in perpendicular distance to the forest border. All mammal species were considered but for small species such as mice, rats or squirrels identification to species level was often not feasible and most animals recorded were assigned to the overall taxonomic group of ‘rodents’, resulting in a non-proportional low species number for this group. Other groups for which we recorded sufficient species were ungulates, carnivores and primates. Various small species with a weight of less than 7 kg belonging to different taxonomic groups were compiled in the group ‘other small mammals’, e.g. treeshrews (*Tupaia glis*), pangolins (*Manis javanica*) and colugos (*Galeopterus variegates*).

4.2.3 Literature research

By using Scopus bibliographic database, we investigated published articles dealing with animal biodiversity in rubber plantations in comparison to natural habitats and other land-use types when indicated. For data analysis, we selected studies for two criteria: (a) precise and comparative information on species numbers in rubber plantations and natural habitats, (b) studies located in MSEA, assuming that the ecological conditions and land-use types in this region are largely comparable to the situation in the target area of NRWNNR. Therefore, we did not include studies from other regions such as India and Indonesia, as well as studies conducted in rubber agroforestry systems (“jungle rubber”). Accordingly, we considered a total of 10 studies, including soil nematodes, different taxa of insects, spiders, frogs, bats and birds.

4.2.4 Data integration

The data sets from field and literature studies were subjected to a normalization process aiming at transforming species (group) numbers to comparable and computable values, named habitat scores, between 0 and 1. The highest reported species number was used as reference value (1), while the species numbers reported for other land-use classes were transformed into a fraction of the reference value (e.g. plant species numbers; see Table 2: Forest 796, Rubber 518, rice paddies 146 resulted in normalized species indices of Forest 1, Rubber 0.65, Rice paddies 0.18.). These indices were aggregated into a habitat suitability matrix that served as input into our InVEST scenario evaluation. Reference values for the aspect of habitat sensitivity to different threats needed for an InVEST assessment were derived from literature review and decided upon during expert panel meetings.

We analyzed (1) red list species (flora and mammals) for the biodiversity aspect of nature conservation (red list species), (2) an overall biodiversity score including all species for an overall picture (all species) and (3) the combination of bees (honey provision) and medicinal plants to show from which species humans benefit directly (human use). For all groups spatially explicit results were modeled with the habitat quality model of InVEST 3.2.0 for the land-use map of 2007 and two scenarios for 2025.

4.2.5 Case study scenarios

The original land-use map for our study site in Southwest China was created from 2007 data (Wehner et al. 2014). In order to test our approach we applied our methodology to two scenarios for the year 2025, the 'Business as usual' scenario extrapolating the trend of further expanding rubber plantations and the 'Go green' scenario increasing the available forest area. These scenarios have been developed to evaluate current land-use composition and to establish possible future trajectories of land-use change in the region (Wehner et al. 2014) using CLUE-S modeling approaches combined with socio-economic modeling techniques (Gibreel et al. 2014) and combined ecological assessment and economic optimization approaches (Cotter et al. 2014).

We assessed habitat suitability based on our collected data sets with InVest in order to assess the impact the different land-use scenarios would have on biodiversity as an ESS. Further studies will report on the integration of this assessment into our integrated multidisciplinary assessment approach.

Table 1 Distribution of different land-use classes between scenarios. Values are given in %.

%	Forest	Rubber	Rice	Agriculture	Bushland
Land-use 2007	69.5	9.5	3	15.3	2.2
Go green 2025	78	6.3	3.3	7.5	4.8
Business as usual 2025	67.6	9.6	3.3	15.2	3.1

4.3 Results

All groups of vertebrates and invertebrates listed in Table 2 showed lower species numbers in mature rubber plantation compared to natural forest, latter representing the original type of vegetation in MSEA. The proportional abundance of species in these two land-use categories varied widely according to taxa and study. In most cases, species numbers were roughly 30-50% lower in rubber plantation compared to forest.

The results show that hoverflies occur in much higher species numbers in agriculture and rice fallows than in forest and rubber plantations (Meng et al. 2012a). Species numbers of ground beetles in forest and agricultural land-use classes are nearly identical (but much lower in rubber plantations; Meng et al. 2012b). On the other hand, higher species numbers of longhorn beetles and birds were found in forest compared to rice and agriculture (Meng et al. 2013, Peh et al. 2006). Generally, relatively high numbers of species in agricultural habitats are explained by the occurrence of additional species that are adapted to open land conditions and do not originate from forest.

Small vertebrates such as frogs, birds and rodents (Aratrakorn et al. 2006, Peh et al. 2006, Behm et al. 2013, Li et al. 2013) showed a lower diversity in rubber-dominated land-use systems than in natural forests but generally occurred in higher numbers than larger vertebrates such as carnivores or ungulates across all land-use classes. The latter rely on natural forest for shelter, whereas the reduced cover in plantations makes them easy targets for hunters (Luskin et al. 2014). The diversity and activity of insectivorous bats is greatly reduced in rubber plantations because insects are less than half as abundant as in natural forest habitats (Phommexay et al. 2011).

We used the results of Table 2 to model habitat suitability scores for three different groups of species within the NRWNNR. Information on the input data for the model is available in the Supplementary Material. The linear scale shows increasing habitat suitability from zero to one (red to dark green, respectively). For the 2007 land-use map and both scenarios habitat suitability is worst for red list species and best for species used by humans, with in-between scores for the aggregated biodiversity index of all species. When comparing the two scenarios with the land-use map of 2007 within species groups the ‘Business as usual’ scenario slightly decreases habitat suitability, whereas the ‘Go green’ scenario slightly increases habitat suitability. Landscape habitat scores (sum of all pixel habitat scores) for the ‘Business as usual’ scenario vary between 97-99%, for the ‘Go green’ scenario between 103 and 106%. Again, the magnitude decreases from red list species through all species to human use species. We also created a benchmark scenario to see how much habitat suitability would behave over a longer time span by converting all land-use classes except settlements to forest. Using the resulting landscape habitat score as 100% benchmark the landscape habitat scores for the three land-use maps for 2007 is 79%, 83% and 90% for red list species, all species and human use species respectively.

4.4 Discussion and conclusions

With this case study, we were able to show that our approach of combining multi-species data supplemented with literature data (allowing for a broader assessment of the impacts of land-use decisions on biodiversity) can be used as input into established ESS assessment frameworks. The habitat suitability for the species groups studied has changed over the course of the scenarios as expected. Due to the relatively narrow (or realistic) definitions of the scenarios, these changes have been less pronounced as we had expected. Especially when considering the indices from the category “human use” we see less of an impact between the scenarios. This can be explained when looking at the distribution of these species groups within the land-use classes (Table 2). Medicinal plants for example are still commonly distributed within rubber plantations, albeit with different species (generalists, ruderal flora). A shift between rubber and forest thus has a relatively low impact on the numbers of species with medicinal use. In addition, wild bee species and ground beetle species are present both with forest dwelling species and species adapted to open-land or succession states found in young rubber. This is hinting towards one of the weaknesses of this method: the decision-making on “what” to analyze and “how” to group them. When taking a look at total species numbers within land-use classes, the tropical forests are ranking top. Considering the landscape level, a mosaic of tropical forests with their high diversity of endemics and a variety of agricultural activities offers a higher number of potential habitats and so a higher, but differently composed species numbers due to the newly introduced farmland species. With this in mind, it seems to be a very reasonable step to include sub-assessments such as “Red List species” or “endemics” into the evaluation in order to cover not just quantity, but also “quality” of species diversity.

Table 2 Results of species numbers for flora, vertebrates and invertebrates per land-use class (absolute numbers and proportions). Habitat scores are calculated as the proportion of species found in a specific land-use class related to a benchmark value, which is the land-use class with the highest absolute number of species found (100%). Studies from different countries are listed separately (e.g. ungulates, birds).

	Forest		Rubber		Rice		Agriculture		Bushland		
	n	%	n	%	n	%	n	%	n	%	sources
Flora											
Red list	91	*100*	20	*22*	1	*1*	6	*7*	3	*3*	Own, 2009
Endemic	126	*100*	45	*36*	2	*2*	15	*12*	3	*2*	Own, 2009
Invasive	7	*35*	20	*100*	20	*100*	18	*90*	14	*70*	Own, 2009
Medicinal	209	*100*	193	*92*	60	*29*	92	*44*	55	*26*	Own, 2009
Total species	796	*100*	518	*65*	146	*18*	255	*32*	124	*16*	Own, 2009
Habitat scores^{1,2}	**1**		**0.65**		**0.18**		**0.32**		**0.16**		
Vertebrates											
Red List mammals	10	*100*	5	*50*							Own, Thailand, 2013
Ungulates	6	*100*	2	*33*							Own, Thailand, 2013
Ungulates	6	*100*	3	*50*							Own, China, 2010
Primates	6	*100*	3	*50*							Own, Thailand, 2013
Carnivores	10	*100*	3	*30*							Own, Thailand, 2013
Rodents	7	*100*	5	*71*							Own, Thailand, 2013
Mammals < 7 kg	5	*100*	3	*60*							Own, Thailand, 2013
Bats (insectivores)	26	*100*	13	*50*							Phommexay et al. 2011
Frogs	18	*100*	11	*61*							Behm et al. 2013
Birds	108	*100*	41	*38*							Aratrakorn et al. 2006
Birds	53	*100*	43	*81*							Li et al. 2013
Birds	62	*100*	46	*74*					31	*50*	Peh et al. 2006
Habitat scores^{1,3}	**1**		**0.56**								
Invertebrates											
Wild bee	35	*100*	25	*71*	26	*74*	30	*86*			Meng et al. 2012a
Hoverflies	12	*36*	2	*6*	33	*100*	31	*94*			Meng et al. 2012a
Ground beetles	59	*92*	34	*53*	64	*100*	53	*83*			Meng et al. 2012b
Longhorn beetles	193	*100*	77	*40*	33	*17*	82	*42*			Meng et al. 2012b
Soil Nematode taxa	42	*100*	28	*67*							Xiao et al. 2014
Spiders	193	*100*	95	*49*							Zheng et al. 2015, averages
Habitat scores^{1,4}	**1**		**0.49**		**0.29**		**0.37**				
All Species											
Habitat scores^{1,5}	**1**		**0.57**		**0.16**		**0.23**				

[1] Habitat scores are calculated as the proportion of species found in a specific land-use class related to a benchmark value, which is the land-use class with the highest absolute number of species found (100%).

[2] Based on total species found. The benchmark for flora is for all plant groups forest, except for invasive species (highest species numbers found in rubber and rice).

[3] Based on the sum of the listed species groups. The benchmark for vertebrates is for all groups forest.

[4] Based on the sum of the listed species groups. The benchmark for invertebrates is for all groups forest, except for hoverflies (highest species numbers found in rice).

[5] Mean values of overall habitat scores of flora, vertebrates and invertebrates.

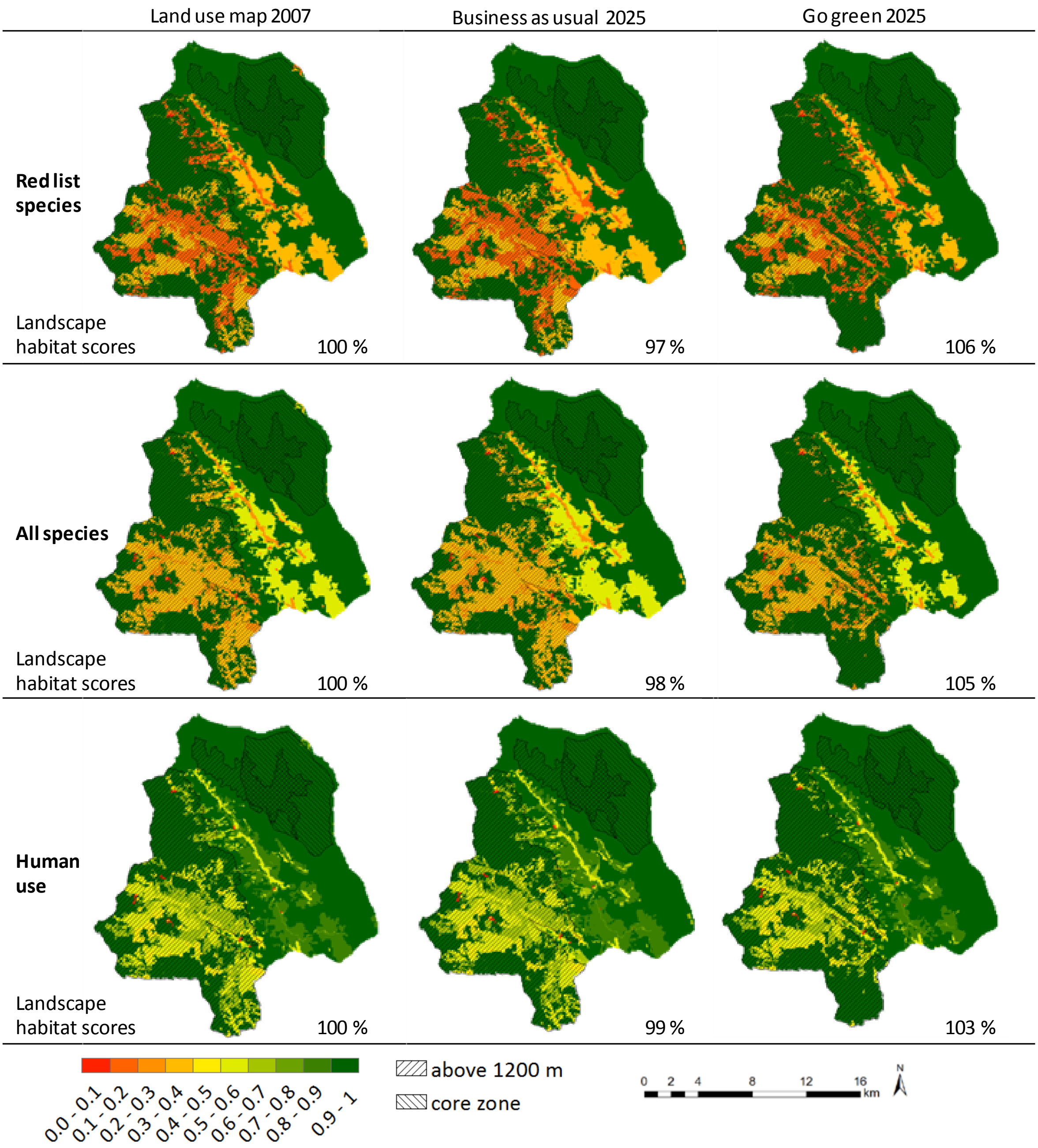

Figure 1 Displays the modeled habitat scores for the land-use map of the year 2007 and the two scenarios for 2025, 'Business as usual' and 'Go green'. Values close to zero designate areas with no or a low habitat suitability for the respective species groups (red list species, all species and species with a human use value), values close to one designate areas with a high habitat suitability (forest was used as reference for highest habitat suitability). Hatched areas cannot be converted to rubber plantations either because they are above 1200m a.s.l. or legally protected (core zone of the Man and Biosphere Programme).

4.4.1 Rubber as habitat

For this study we assumed that rubber management practices throughout MSEA have similar impacts on biodiversity. We want to show a potential approach for combining multiple sources of data in order to allow for a broader assessment of the potential impacts that certain decisions may have on ESF and ESS. To do so, we had to make compromises on the level of details included, a situation that most scientists face in data-scarce environments. While rubber served as a good example to test our approach, management practices within the plantations can have an impact on habitat quality that we could not cover within our case study. Another aspect to be considered is the difficulty to include different succession stages in rubber plantations. Over the cultivation period the rubber system offers habitat for multiple and successive communities of species, from open land and forest remnant species in the establishment phases up to (moderately) shade tolerant species usually found in secondary forest systems. In the four insect studies for example it was stated that young rubber sites generally harbor higher insect species numbers than mature rubber plantations due to the richer structural diversity and the existence of pronounced ground cover vegetation. The lower heterogeneity in mature rubber plantations also affects food availability for several larger wildlife species.

4.4.2 Upscaling and transferability

The method presented in this paper is quite robust to scale effects. With the inclusion of the normalization process into the method we aim to never assess specific species diversity alone, but the effects of its integration into a given landscape matrix. As such, data from rather small scale assessment can be compared to data from larger scale assessments. At the same time this approach can easily be applied to a wide range of study sites, as the techniques used are not restricted to tropical environments or certain agricultural practices. Data collection and literature review has certainly to be adapted to the given situation and backgrounds, but tools such as InVEST have shown to be usable for multiple spatial levels.

Based on studies similar to ours, an extended framework for the incorporation of biodiversity related aspects into InVest, MIMES or ARIES could be attempted, allowing for a role more prominent and better understood in ESS assessments worldwide.

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5 Investigating potential transferability of place-based research in land system science

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Outline and overview

This publication proposes a new approach to examine the potential transferability of place-based research with the help of land system archetypes. These include multiple dimensions of land-use intensity and environmental and socio-economic conditions. The analysis was tested with twelve regional projects from a large joint research program focusing on sustainable land management, the project Sustainable Rubber Cultivation in the Mekong Region (SURUMER) being one of them. In a first step, the 'project archetype' was defined, based on a synthesis of global land system indicators. In a second step their transferability potential was tested by calculating statistical similarity of locations across the world.

Investigating potential transferability of place-based research in land system science

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Abstract

Much of our knowledge about land use and ecosystem services in interrelated social-ecological systems is derived from place-based research. While local and regional case studies provide valuable insights, it is often unclear how relevant this research is beyond the study areas. Drawing generalized conclusions about practical solutions to land management from local observations and formulating hypotheses applicable to other places in the world requires that we identify patterns of land systems that are similar to those represented by the case study. Here, we utilize the previously developed concept of land system archetypes to investigate potential transferability of research from twelve regional projects implemented in a large joint research framework that focus on issues of sustainable land management across four continents. For each project, we characterize its project archetype, i.e. the unique land system based on a synthesis of more than 30 datasets of land-use intensity, environmental conditions and socioeconomic indicators. We estimate the transferability potential of project research by calculating the statistical similarity of locations across the world to the project archetype, assuming higher transferability potentials in locations with similar land system characteristics. Results show that areas with high transferability potentials are typically clustered around project sites but for some case studies can be found in regions that are geographically distant, especially when values of considered variables are close to the global mean or where the project archetype is driven by large-scale environmental or socioeconomic conditions. Using specific examples from the local case studies, we highlight the merit of our approach and discuss the differences between local realities and information captured in global datasets. The proposed method provides a blueprint for large research programs to assess potential transferability of place-based studies to other geographical areas and to indicate possible gaps in research efforts.

5.1 Introduction

Understanding the interactions between people, land use and the environment is a central challenge for land system science (Rounsevell *et al* 2012). Much of our knowledge on land systems and the goods and services they provide is derived from place-based research and local assessments of ecosystem services (ESS). Place-based research of land use typically takes the form of case studies rooted in a particular place and context (van Vliet *et al* 2015). As a bottom-up approach, it is used to characterize the drivers and consequences of land use and its change in a specific location. Case studies have been used to estimate world's potentially available cropland (Lambin *et al* 2013), to reveal the complexity of coupled human and natural systems (Liu *et al* 2007), to assess the role of protected and managed forests for the long term maintenance of forest cover in the tropics (Porter-Bolland *et al* 2012), or to identify opportunities for enhancing the relevance of ESS assessments for decision making (Förster *et al* 2015).

The generalization and transferability of results from place-based research, however, is inherently limited because the drivers and processes of land use are complex, and their outcomes are contingent upon specific geographical context, including prevailing social, economic, political and cultural conditions. This limitation is especially true for land systems, i.e. social-ecological systems (SESs; Ostrom 2007), in which the interactions of different agents can be mediated through direct and indirect linkages and feedbacks with the physical environments (Letourneau *et al* 2012). Unlike studies with controlled research design, case studies collect empirical evidence on land-use phenomena in their real-world context and rely on non-random selection of sites whose unique characteristics facilitate meeting specific research goals. Consequently, the types and levels of land-use intensity, the environmental conditions, the social and political settings, and the spatio-temporal scales may vary substantially across sites (van Vliet *et al* 2015).

The outcome of place-based research is thus an evolving model that accounts for observed properties and behavior of the studied land system but also allows formulating hypotheses applicable to previously unstudied areas that have similar properties (Billick and Price 2010). Here we assume that similarity of land systems constitutes the potential for transferability, i.e. the more similar two sites are in terms of land use, environmental and socioeconomic conditions, the higher the probability that methods, results and conclusions from a project site prove applicable at a similar site. However, where these geographical sites with similar properties are located is typically unknown or not part of the research agenda. Therefore, given the variable design and multidisciplinary nature of place-based research, there is a need to better link the findings of the many case studies conducted and assess their relevance beyond the study areas.

Biomes, ecoregions and landscape typologies may provide a starting point for such efforts, but the applicability of biogeographical frameworks is limited because they do not incorporate human land use or reduce it to a single dimension of disturbance (Martin *et al* 2014). The use of integrative models of human-environment interactions has increased over the last years, after various global datasets on crop yields (Monfreda *et al* 2008), fertilizer use (Potter *et al* 2010) and other land-use intensity indicators became available (Kuemmerle *et al* 2013). For example, the anthrome framework was used to map the rate of landscape transformation over centuries (Ellis *et al* 2010) or to describe the current distribution of conservation efforts at the global scale (Martin *et al* 2014). New classifications of land systems were developed for their use in Integrated Assessment Models, in order to examine environmental consequences of interactions between economic, social and biophysical systems (Letourneau *et al* 2012, van Asselen and Verburg 2012). In addition, initiatives such as GLOBE (Ellis 2012; http://globe.umbc.edu) emerged to facilitate synthesis of case studies by providing an online database and tools for assessing the global relevance of land-use case studies based on their geographical context (Magliocca *et al* 2015).

Most recently, the concept of land system archetypes (LSAs) was developed in response to the calls for frameworks that incorporate multiple dimensions of land-use intensity in SESs (Václavík *et al* 2013). As agricultural intensification, including ecological intensification (Pywell *et al* 2015), is likely to continue in the future, it is becoming clear that a wider spectrum of land-use intensity metrics needs to be considered (Erb *et al* 2013, Kehoe *et al* 2015). LSAs utilize a wide range of such metrics and offer an alternative view on land systems by integrating various measures of land-use intensity in the context of prevailing environmental and socioeconomic conditions. The framework is well suited to increase the global relevance of place-based research because it provides a first step for classifying land systems with similar properties as those represented in the investigation sites.

Here, we adapt the existing framework of LSAs (Václavík *et al* 2013) and propose a new approach to examine potential transferability of place-based research. We apply our approach to twelve regional projects of the German Sustainable Land Management (SLM) Program, a large-scale funding initiative that provides a platform for research of sustainable land-use across four continents, with the focus on deriving sustainability transformation strategies. For each case study, we define the 'project archetype', i.e. the unique land system present in the study area, based on a synthesis of global land system indicators. Assuming that similarity in land-use intensity and environmental and socioeconomic conditions among regions is a basic pre-condition for transferability, we address the following questions: (1) Where are the areas to which the research methods, results and conclusions of local case studies from the SLM Program can be potentially transferred? (2) Are there regions across the world that are under- or over-represented by the research within the SLM Program? Using specific examples from selected regional projects, we highlight the merit and applicability of our approach, describe the differences between local realities and information captured in global datasets, and discuss the optimal strategies for improving transferability in the future. We also discuss the criteria that determine and limit transferability of place-based research, thus testing our hypothesis that similarities constitute transferability potentials.

5.2 Materials and Methods

5.2.1 Case studies

We analyzed twelve case studies that are part of the SLM Program, funded by the German Federal Ministry for Education and Research (Eppink *et al* 2012). Their objective is to improve the understanding of interacting ecological and socioeconomic systems, and to foster transformations towards more SLM (see table 1 for an overview of project focus, research questions and adopted measures). The projects have similarities in common drivers of change, such as population growth, developments in economic markets and climate change. There is also a distinct overlap in the ecosystem services considered, such as food production, fresh water supply and climate regulation. The projects are conducted in 13 countries across four continents with a wide range of conditions that define the underlying LSAs (figure 1). The spatial scale of the projects ranges from a few hundreds to several hundred thousands of square kilometers.

Table 1 Regional projects within the German Sustainable Land Management Program focus on various aspects of land use and ecosystem services across four continents. Documents and videos summarizing each case study can be accessed at the program's website (http://modul-a.nachhaltiges-landmanagement.de/en/projects/).

Project/area	Focus	Research question	Measures
Carbiocial: 119 170 km²	Carbon sequestration, biodiversity and social structures in Southern Amazonia: models and implementation of carbon-optimized land management strategies	Optimize land management to minimize negative feedback by land use change in the frame of climate change and socio-economic development	Decision support systems for carbon-optimized land use by region-specific modeling of land use impact
CC-LandStraD: 357 021 km² Analyzed focus area: 6310 km2	Interdependencies between land use and climate change: strategies for a sustainable land management in Germany	How can be a sustainable land use in 2030 in Germany? How to model the interactions between land use and climate change in Germany?	Strategies for sustainable land management and ist contribution to climate change mitigation in Germany
COMTESS: 600 km²	Sustainable coastal land management: trade-offs in ecosystem services	Which strategies can promote a sustainable management of vulnerable coastal landscapes?	Adapted water management strategies including a sustainable agricultural land use under changing hydrological conditions
INNOVATE: 377 000 km²	Interplay among multiple uses of water reservoirs via innovative coupling of substance cycles in aquatic and terrestrial ecosystems	Which governance options promote sustainable ecosystem services and economic viability under climate change conditions?	Decision support systems based on constellation analysis for land and water use based on modeled land and water use scenarios
KULUNDA: 93 230 km²	How to prevent the next global dust bowl? Ecological and economic strategies for sustainable land management in the Russian steppes: a potential solution to climate change	How degradation and desertification processes can be mitigated by development and implementation of adequate sustainable land management practises?	Adopted agricultural management and tillage operation for advanced steppe restoration
LEGATO: 1575 km²	Land-use intensity and ecological engineering: Assessment tools for risks and opportunities in irrigated rice based production systems	How to advance long-term sustainable development of intensive land use systems, against risks arising from multiple aspects of global change, by quantifying the dependence of ecosystem functions (ESF) and the services (ESS) they generate in agricultural systems in South East Asia?	Implementation of ecological engineering, organic farming for landscape scale management and sustainable intensification
LUCCi:12 350 km²	Land use and climate change interactions in the Vu Gia Thu Bon river basin, Central Vietnam	Which role does land use play for GHGemissions? Which strategies for sustainableland and water management can cope with climate change impacts?	Implementation of land-use planning and water management strategies for mitigation ofGHGemissions in agriculture and forests based on regional climate change scenarios by bio-economic optimization model
SASCHA: 1200 km²	Sustainable land management and adaptation strategies to climate change for the Western Siberian grain belt	How to mitigate the negative impacts of agricultural landuse change on ecosystem services and biodiversity in Western Siberia?	Modeled future land-use scenarios; toolkits for monitoring change and land-use planning; written guidance and training for policymakers
SuLaMa:7500 km²	Participatory research to support sustainable land management on the Mahafaly plateau in south-western Madagascar	How to reconcile biodiversity conservation and the maintenance and enhancement of ecosystem services with economic land management?	Participatory determination of strategies for implementing a jointly developed sustainable land management plan

Table 1 (Continued)

Project/area	Focus	Research question	Measures
SuMaRiO: 650 000 km²	Sustainable management of river oases along the Tarim River, China	How to support oasis management along the Tarim River (TR) under conditions of climatic and societal change?	Enhanced water management and land management particularly with regard to ecology on basis of scientific results and their application in a decision support system tool
SURUMER: 265 km²	Sustainable Rubber Cultivation in the Mekong Region - Development of an integrative land-use concept in Yunnan Province, China	How does the current practice of rubber management affect Ecosystem Functions and Services (ESF/ESS)? And how can the system be improved towards sustainability?	Development and dissemination of improved rubber management schemes, stakeholder involvement, diversification of production
The Future Okavango:430 000 km²	Scientific support for sustainable land and resource management in the Okavango Basin	How to improve land use and resource management with scientific knowledge?	Development of tools (scenarios, storylines, DSS) and strategies for sustainable land use and river basin management

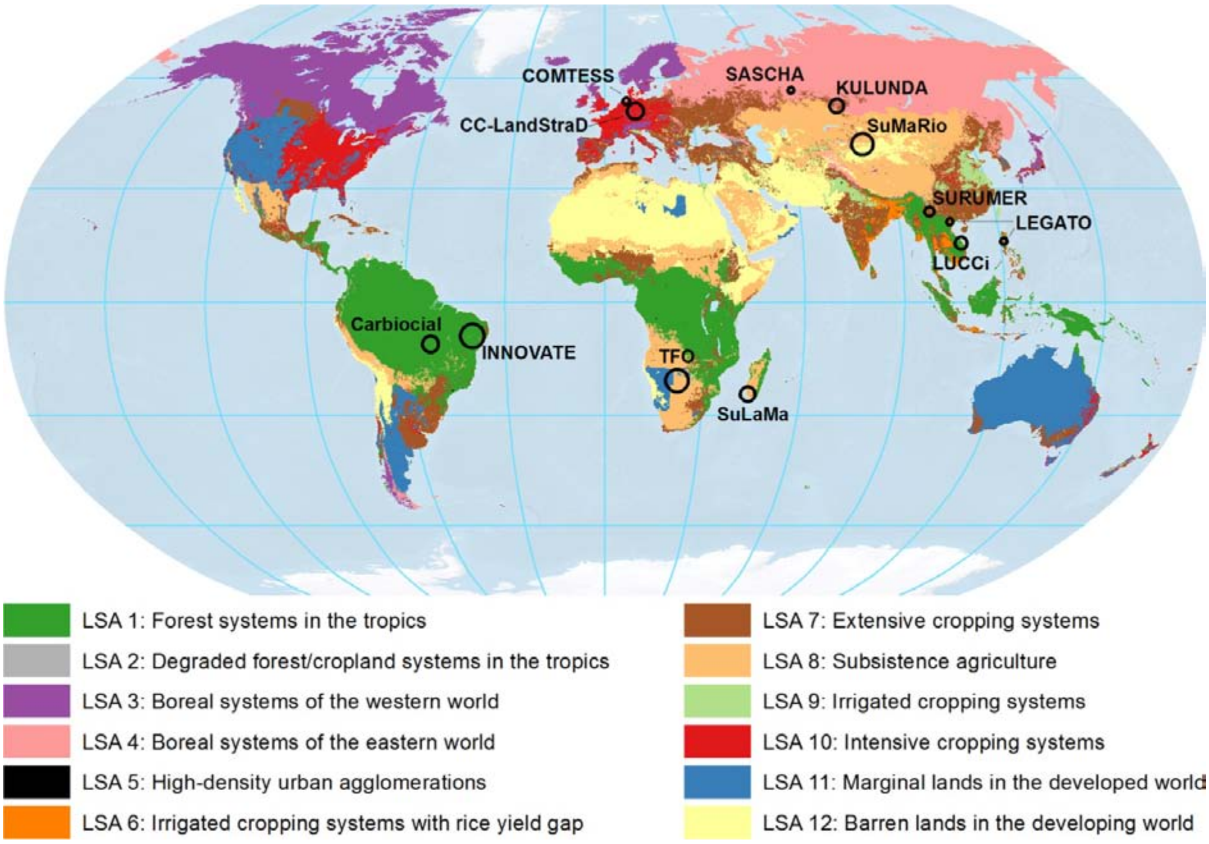

Figure 1 Geographical locations of investigated regional projects and their distribution in global land system archetypes defined by Václavík *et al* (2013) based on clustering of similar land-use intensity, environmental and socioeconomic conditions. The size of the project symbol is relative to the project's study area. The geographic locations of project study areas were obtained from the Geoportal of the SLM Program at: http://geoportal-glues.ufz.de/.

5.2.2 Data

We considered the same set of 32 indicators of land-use intensity, environmental conditions and socioeconomic situation as previously used to define global LSAs (see table A1 in the supplementary material). Details on the datasets and the indicator selection are provided in Václavík *et al* (2013). In summary, we compiled 15 land use datasets that measure different aspects of agricultural intensity in terms of inputs, outputs and system metrics (sensu Kuemmerle *et al* 2013). For input metrics, we chose maps of cropland and pasture cover (Klein Goldewijk *et al* 2011) and also calculated their changes over the last 50 years to account for temporal trends. In addition, we considered the extent of areas equipped for irrigation (Siebert *et al* 2007) and the levels of nitrogen (N) fertilizer input (Potter *et al* 2010). For output metrics, we included crop yields for wheat, maize and rice (Monfreda *et al* 2008), because these crops represent approximately 85% of global cereal production (Hafner 2003) and are grown in most of the considered regional projects. For system-level metrics, we selected yield gaps for wheat, maize and rice (IIASA/FAO 2012), the human appropriation of net primary production (HANPP; Haberl *et al* 2007), and soil erosion caused by water and tillage (Van Oost *et al* 2007).

To represent environmental conditions, we used five uncorrelated bioclimatic variables from the CliMond database (Kriticos *et al* 2012) accompanied by climate anomalies interpolated from NOAA's long-term measurements of land surface temperatures (Menne *et al* 2009). For biophysical factors that reflect productivity and growth conditions of ecosystems, we included datasets on normalized difference vegetation index (NDVI) (Tucker *et al* 2005) and soil organic carbon (Batjes 2006). Vertebrate species richness for mammals, birds, reptiles and amphibians were derived from expert-based range maps (BirdLife International 2012, IUCN 2012) as a biodiversity indicator reflecting both natural conditions and long-term effects of land management (Green *et al* 2005, Phalan *et al* 2014). For economic indicators of land systems, we used three indices provided by the Food and Agriculture Organization (FAO) at a national level: gross domestic product (GDP), the proportion of GDP resulting from agriculture and the capital stock in agriculture. As socioeconomic factors and indicators of human pressure on land, we used gridded data on population density (CIESIN 2005), the world governance index of political stability (Kaufmann *et al* 2010) and the global map of accessibility that measures travel time to major cities and market places (Uchida and Nelson 2009). We are aware that the geographical scope of the different parameters is not identical, and that taking data aggregated at national or regional scale can mask significant deviations in the research sites; one example is discussed below.

The requirement was that these datasets are available for the entire terrestrial surface of the world, so that transferability potentials can be investigated beyond the study areas of our case studies. The land-use data were derived for circa the year 2005, the time period where such datasets are richest at the global scale. Prior to the final variable selection, we inspected Pearson correlations between all variables in order to avoid redundancy in the input information (see table A2 in the supplementary material). Our final set of input indicators included only those with $|r| < 0.7$ (Dormann *et al* 2013). All data were aggregated to the spatial resolution of 5 arc-min (~9.3 × 9.3 km at the equator).

5.2.3 Analysis of transferability potentials

We estimated transferability potentials for the twelve regional projects by calculating the statistical similarity of all 5 arc-min pixels across the world to the unique land system present in each project study area. We assumed that if the project study area overlaps with a specific LSA (Václavík *et al* 2013), then its research is potentially relevant for other geographical regions that belong to the same archetype. However, the original global classification, based on a self-organizing map clustering of the same variables as used here, is relatively coarse and thus high variability in land-use intensity and other conditions exists within the individual archetypes (Václavík *et al* 2013). Also, the availability, resolution and quality of underlying data vary across the world, suggesting that the precision of an archetype definition is not always comparable across regions. Therefore, we extended the original archetype framework and adopted a three-step approach to quantify the degree of similarity between given case studies and other regions around the world (figure 2).

First, we analyzed the conditions in each project as reflected by the considered variables and determined the 'project archetype', i.e. the unique land system in the study area. Prior to the analysis, we checked the data for skewed distributions and removed extreme outliers. Because of their differing units, we normalized all variables to zero mean and unit variance, so the results can be interpreted in terms of how much and in which direction the project conditions deviates from the global average. We defined the *project archetype* as the combination of the means A_i for all variables i = 1,..., 32 calculated as:

$$A_i = \frac{1}{p}\sum_{n=1}^{p} x_n \tag{1}$$

with x being the normalized value of each variable and p being the total number of cells in the regional project. Second, we calculated statistical similarity of the project archetype (represented by each grid cell within the project) to each global grid cell in the multi-dimensional space defined by considered variables, assuming higher transferability potentials in locations with similar land systems (figure 2). As a measure of similarity, we used an absolute distance D, calculated as:

$$D = \frac{1}{g \times p \times v}\sum_{i=1}^{v}\sum_{n=1}^{p}\sum_{m=1}^{g}\left|x_{i,n} - x_{i,m}\right| \tag{2}$$

with x being the normalized value of variable i, g being the number of global grid cells, p being the number of cells within a regional project and v being the number of considered variables. Third, using the inverse of distance D, we mapped the gradient of transferability potentials for each project in the geographical space (figure 2). For better visualization, we divided the gradient of transferability potentials into four equal classes, with the lowest 25% distance interval representing 'high' transferability potential and the highest 25% distance interval representing 'no' transferability potential.

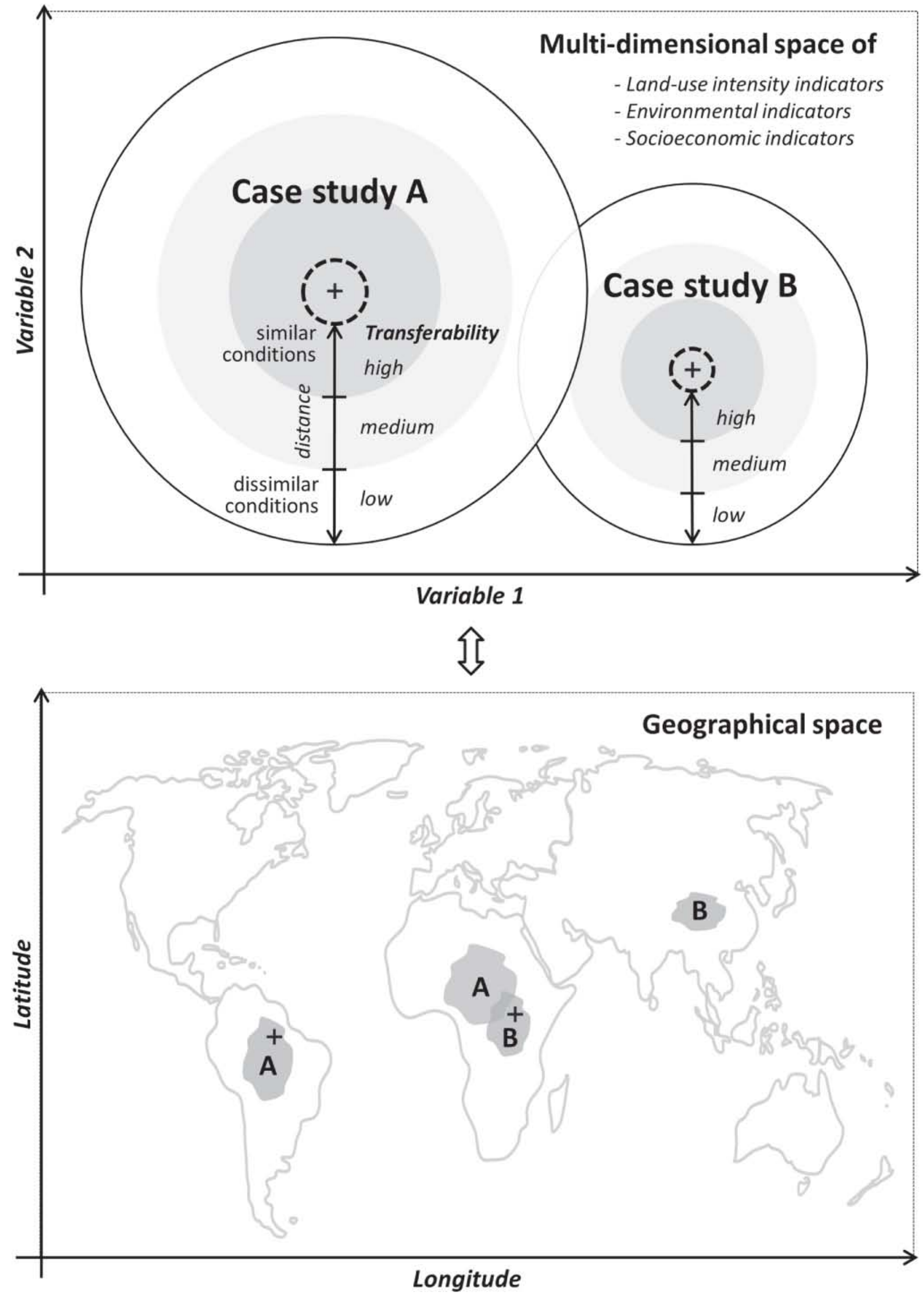

Figure 2 Conceptual diagram of identifying and mapping potential transferability of place-based research. The upper rectangle represents a multidimensional space defined by land-use intensity, environmental and socio-economic indicators. The crosses denote the 'project archetypes', i.e. the mean conditions in the study areas of two hypothetical case studies; the circles denote the range of conditions present in the case study areas, with different shading representing similarity of conditions. The distance does not represent a geographical distance but a statistical measure of similarity of the considered variables. This distance can be mapped in a geographical space (lower rectangle), here showing the 'high' level of similarity (i.e. transferability potential) for each case study, with crosses denoting the location of the hypothetical study areas. Land systems similar to the project archetypes may differ in size or overlap both in the multi-dimensional and geographical space.

We assumed that the variability in underlying conditions, which is likely to be higher for projects with larger study areas, may affect the total area estimated as having high transferability potential. Therefore, we used ordinary least square (OLS) regression analysis to examine the relationship between the total variability of conditions in the study area (calculated as the sum of standard deviations for all variables) and the extent of the 'high' transferability level. All analyzes were conducted in R version 3.2.0 (R Development Core Team 2011), using the libraries 'rgdal' (Keitt *et al* 2011) and 'vegan' (Oksanen *et al* 2013).

Finally, we chose one project, SASCHA (table 1, figure 1), for which we refined the analysis of transferability potentials with finer-scale data from its study area in Western Siberia. To illustrate the potential effects that differences in global versus local data may have on the final analysis, we replaced the values of six original variables (from datasets with a global extent) with those for the same variables from local datasets. Local datasets of cropland area (ha), pasture area (ha), N fertilizer use in agriculture (kg ha^{-1}), wheat yield (t ha^{-1}) and human population density were obtained from the Territorial Authority of the Federal State Statistics Service of the Tyumen Region (TyumStat 2015a, 2015b). The data were available at a district-level resolution for the entire province of Tyumen (160 000 km^2), which has 22 districts with variable climate, socio-economic conditions, suitability for agriculture, cropland and pasture extent and land-use trends (Kühling *et al* 2016). The global GDP estimate for Russia was replaced by an official local estimate (the 'regional domestic product') for the Tyumen province (RosStat 2015).

5.3 Results

Each case study was characterized by a unique project archetype defined by a specific combination of land-use intensity, environmental and socioeconomic conditions (figure 3). The results identifying a gradient of transferability potentials for each of the twelve case studies are shown in figure 4. The mapped levels of transferability potentials varied regionally, often exhibiting spatial clustering of highly similar conditions around the project sites (e.g. for CC-LandStraD, COMTESS, LEGATO). In contrast, highly similar conditions were found for a number of projects in locations that are geographically distant from the study sites (e.g. for CarBioCial, KULUNDA, SASCHA). This corresponds with the original archetype classification which identified relatively large areas of similar land systems across the tropics and the boreal biome.

The top 25% of the calculated transferability potentials (the 'high' level in figure 4) contained areas with extents ranging from 138 140 km^2 to 6572 616 km^2 for SURUMER and CarBioCial, respectively (figure 3). A spatial overlay of these areas with high transferability potentials (top 25%) highlighted the parts of the world, for which the research of the SLM Program is most relevant (figure 5), but it also revealed areas that are likely under-represented by the SLM Program, e.g. in North America, Central Africa and the Middle East. In contrast, several regions were predicted as having high transferability potentials for more than one project. For example, large spatial overlaps of high transferability potentials exist for CC-LandStraD and COMTESS in Western Europe, for SASCHA and KULUNDA in Western Siberia and for CarBioCial and INNOVATE in central South America.

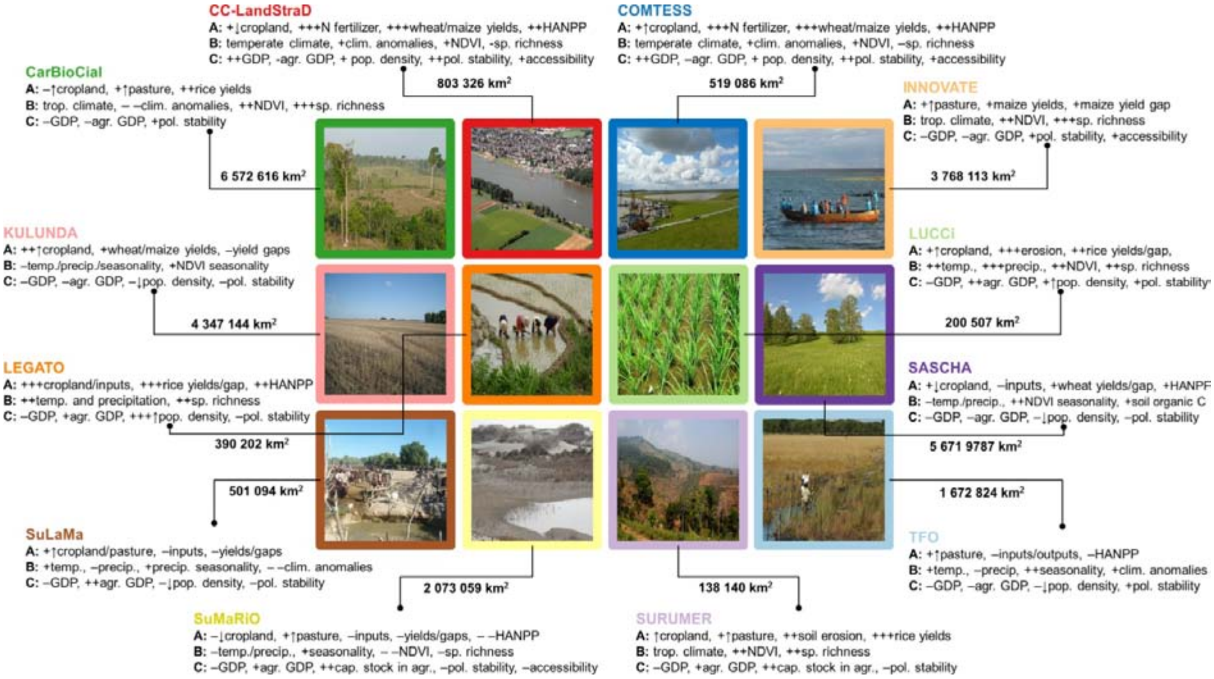

Figure 3 Project archetypes of the Sustainable Land Management Program. Summarized description of (A) land-use intensity indicators, (B) environmental conditions, (C) socioeconomic factors that characterize each project. The + and − signs show whether the factor is above or below global average (+ is up to 1 s.d., ++ is 1–2 s.d., +++ is >2 s.d.). The ↑ and ↓ signs signify increasing/decreasing trends within the last 50 years. The numbers in km² show the total areas of regions identified as having a high level of transferability potential (the top 25% of the gradient).

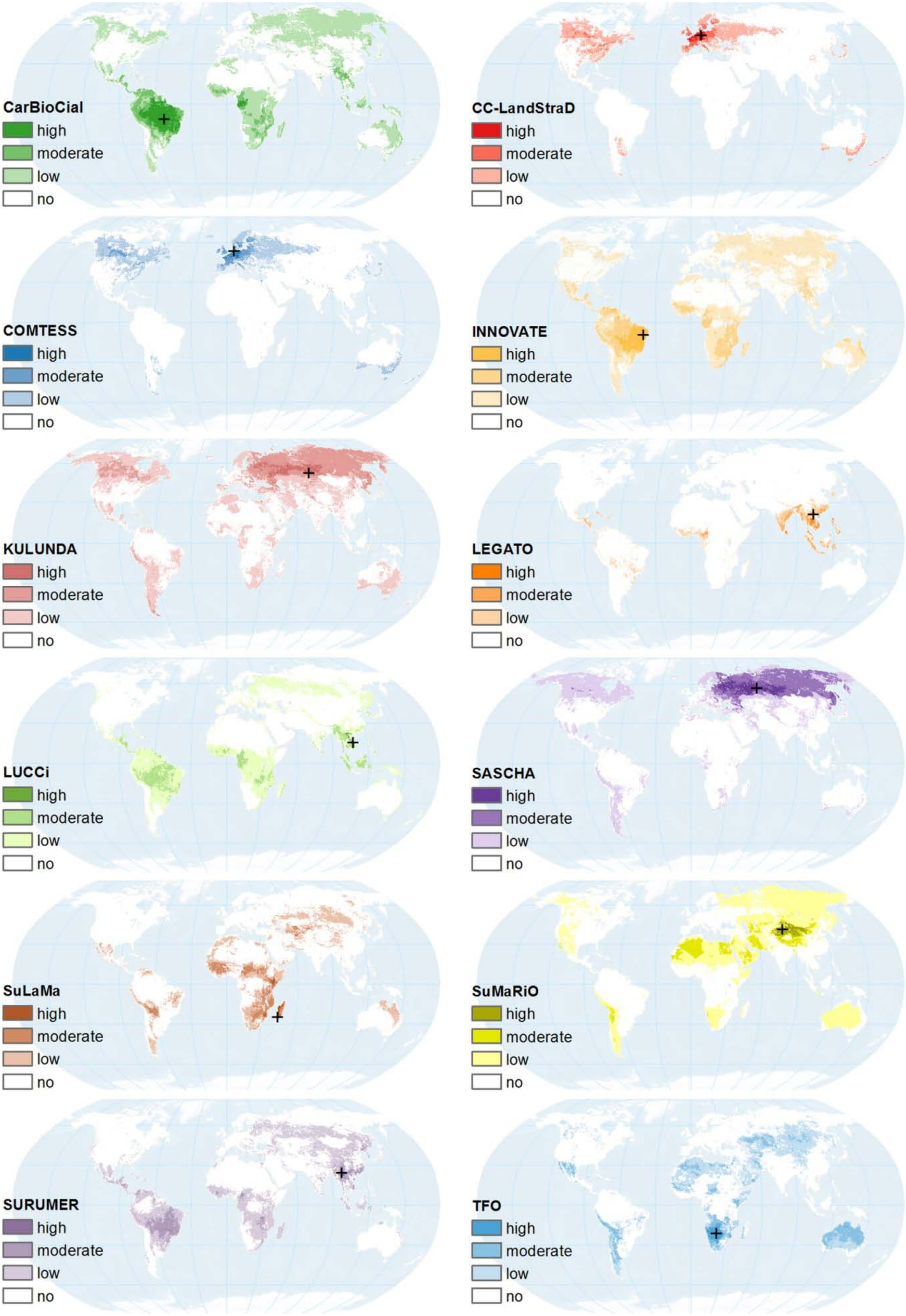

Figure 4 Mapped transferability potentials for the 12 regional projects based on all 32 variables. Areas with conditions similar to those in the regional projects are assumed to have higher transferability potentials. The gradient of transferability potentials is divided in four equal classes, ranging from high to no transferability potentials. For all RPs the same threshold is used, so the levels of transferability potentials and their spatial extents are comparable among the projects. The black cross in each map denotes the location (centroid) of each project's study area.

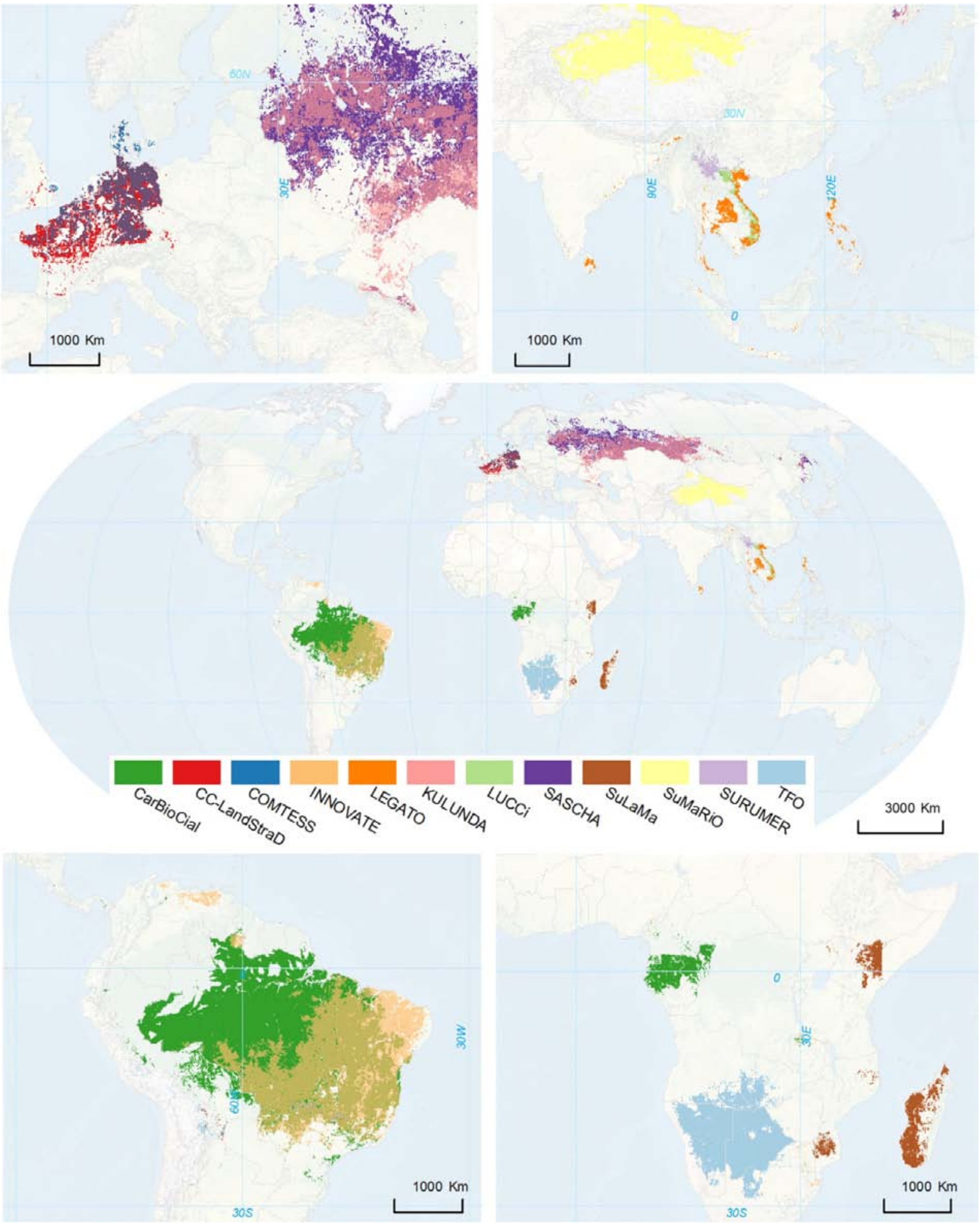

Figure 5 Areas with high transferability potential (top 25% of the transferability gradient), highlighting regions for which the research of the SLM Program is most relevant. The areas with no color, e.g. in North America, Central Africa and the Middle East, are under-represented in the Program's research efforts. In contrast, several regions are simultaneously covered by multiple projects. In the upper left inset, there is a large spatial overlap between CCLandStraD and COMTESS in central Europe (with COMTESS extending more to the coastal areas in the north), and between SASCHA and KULUNDA in Asia. In the lower left inset, there is an overlap in transferability potentials for CarBioCial and INNOVATE.

The differences and overlaps in project transferability potentials were also apparent when inspecting the combination of variable values that characterize each regional project (figure 6). For example, CC-LandStraD and COMTESS had similar values for most variables but differed slightly in cropland area and yields (see also figure A1 for non-standardized values in the original units). While several projects fell into the same global LSA (figure 1) defined by Václavík *et al* (2013), their project archetypes may still differ largely, suggesting a high regional diversity of conditions in the globally defined land systems. Even the conditions within the project study areas varied substantially for some case studies (figures 6 and A1). However, no significant relationship was found between the total variability of conditions in the study area and the extent of the 'high' transferability potential (OLS, $R^2 = 0.163$, $p = 0.194$).

The refined analysis for SASCHA revealed similar patterns of transferability potentials across the Western Siberian grain belt compared to the analysis based on values of global datasets. However, the area estimated to have a 'high' transferability potential decreased to about 48% (2742 136 km^2) of the area identified in the original analysis (figure 7). The global estimates of cropland area proportion closely matched the local statistics from the region (approx. 16% of cropland cover in both global and local datasets). However, the global values for realized wheat yield, GDP and population density were slightly lower than those collected from local sources. Larger differences between global and local data occurred for pasture area (difference of 12%) and N fertilizer (difference of 31 kg ha^{-1}).

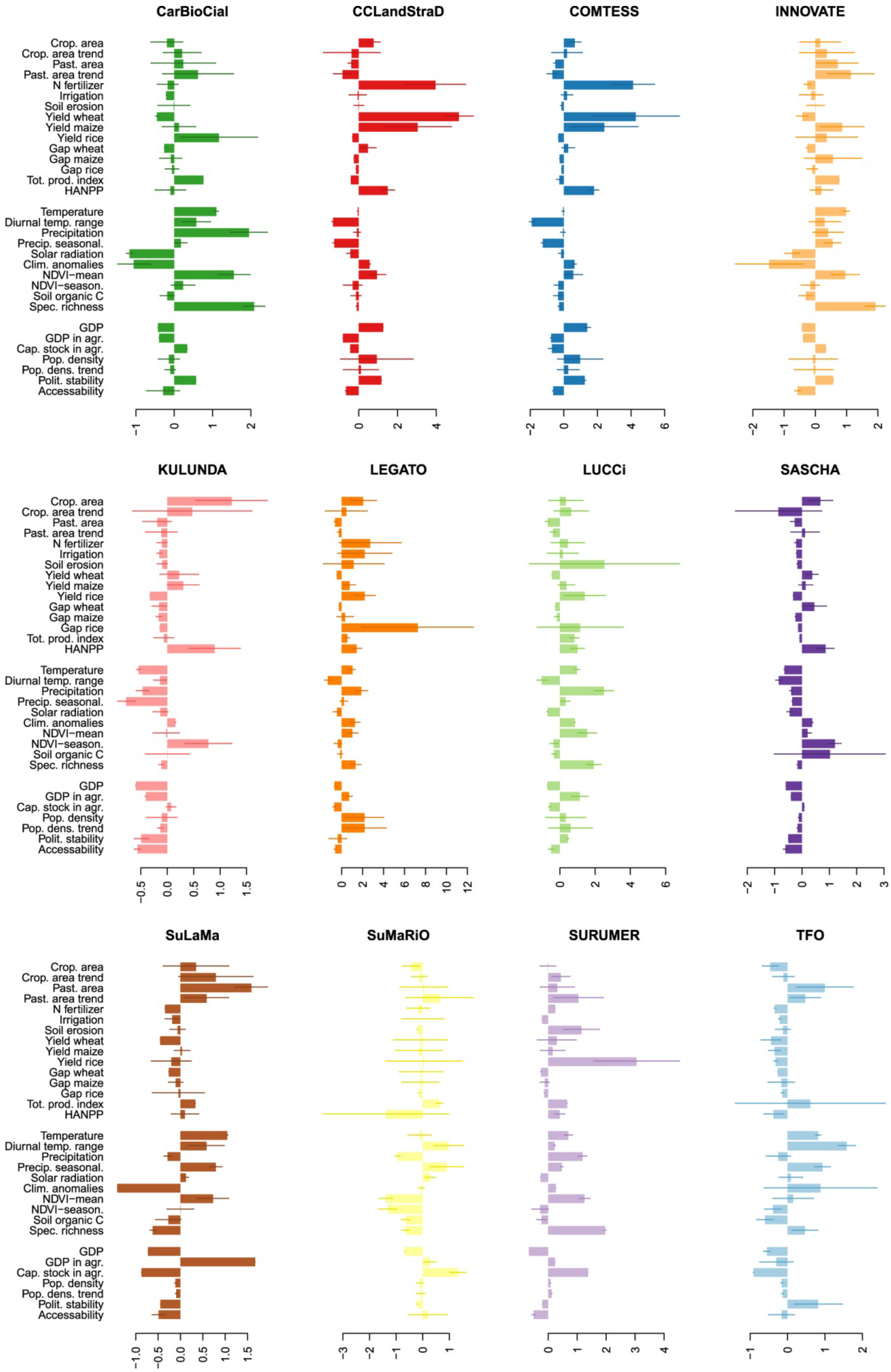

Figure 6 Detailed characterization of land systems in regional projects, showing the combination of normalized variable values. Zero on the *x*-axis is the global mean, so the graphs show whether and how much an indicator is above or below the global mean. The bars represent mean values of the conditions in each study area; the whiskers represent standard deviation (variability of the indicator) within the study area.

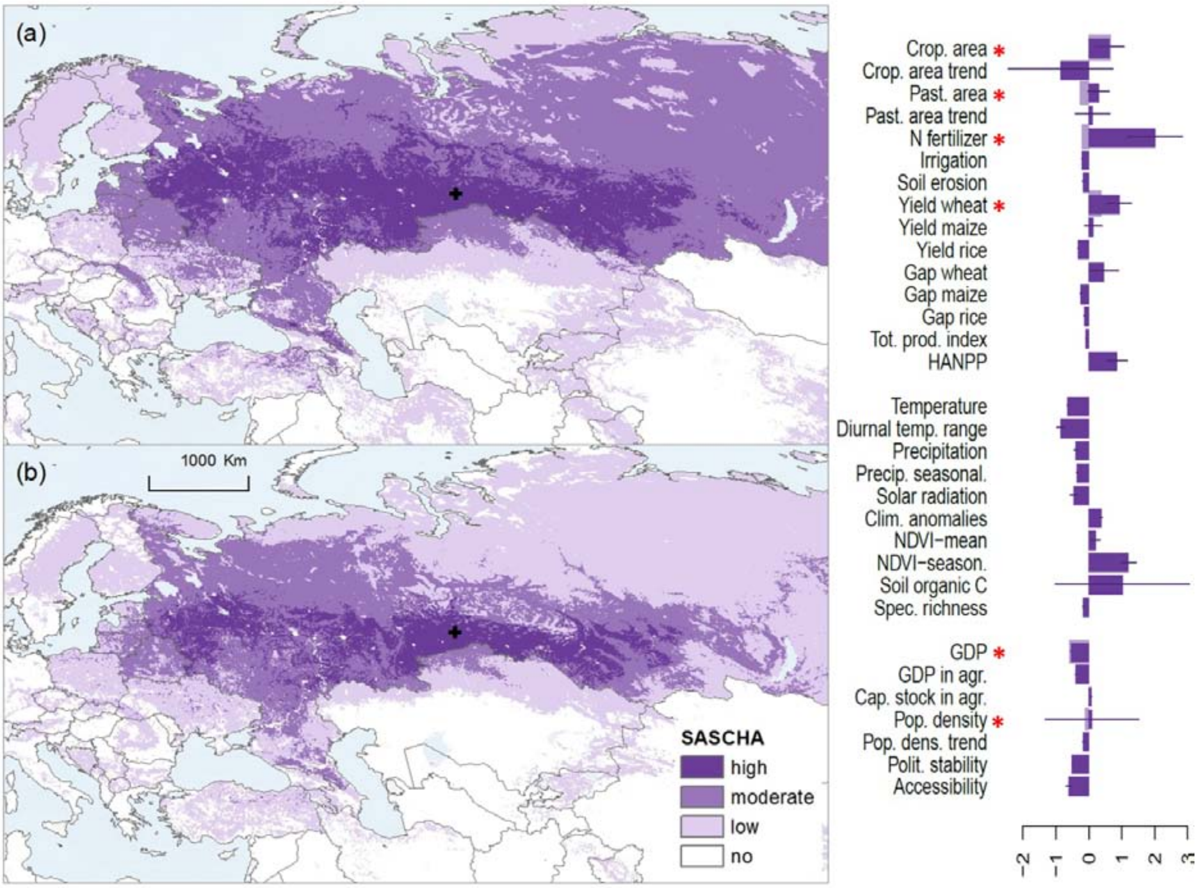

Figure 7 Transferability potential for the SASCHA regional project based on (a) global variables only and (b) refined with finer-scale datasets from the region. The chart on the right shows the combination of normalized variable values in the study area captured by the considered datasets. Variables for which local data were available are marked with red asterisks; their original values captured by the global datasets are displayed as light purple bars for comparison. The black cross in the maps denotes the project location.

5.4 Discussion

Our results show that there are areas beyond the projects' study sites that have similar land systems as those identified in the twelve regional projects of the SLM Program. While the degree of mapped transferability potentials was highly variable in different parts of the world, it was typically clustered around the project sites (figure 4). This pattern suggests that considered land-use intensity, environmental and socioeconomic conditions are spatially dependent (i.e. autocorrelated) and that calculated statistical distance partially corresponds to geographical distance. On the other hand, high transferability potentials for some case studies were found in regions that are relatively far from the project sites. This was typical for projects where variable values were close to the global mean or where the project archetype was driven by large-scale environmental or socioeconomic conditions. For example, areas with high transferability potentials for KULUNDA and SASCHA were identified across the entire Eurasian steppe belt. This is likely due to the similar biophysical conditions (climate, soils) along the latitudinal ecozones of Eurasia (Degefie *et al* 2014, Kamp *et al* 2015). Similarly, the socioeconomic conditions are rather comparable across Russia due to a strongly centralized political and economic system.

The spatial overlay of the high transferability potentials for all projects (figure 5) highlighted not only regions for which research of the SLM Program is most relevant but revealed also 'white spots' that constitute archetypes of low transferability potentials due to relatively large differences among the projects in the underlying land system conditions. This shows that even when a project falls within a certain LSA, its transferability potential does not necessarily extend across the entire range of this land system because the globally defined LSAs still host a high diversity of conditions. This also confirms our assumption that the original archetype framework needed to be refined to allow a reasonable analysis of transferability potentials of regional case studies. However, we did not confirm our assumption that the level of variability in considered conditions within the project's study area affects the total area estimated as high transferability potential. The non-significant relationship between the two factors can be caused by the complexity of considered land system indicators, the spatial distribution of their values across the world, the selection of the threshold used to define the 'high' transferability potential level (i.e. the top 25% of the distance gradient), but also by the relatively small sample size ($n = 12$).

In contrast, we found several regions with spatial overlap of high transferability areas for multiple projects. The similarity in project archetypes of these case studies can be in large part attributed to the close proximity of their study sites. For example, the relative closeness of CC-LandStraD and COMTESS, both with study sites in northern and western Germany, resulted in 62% overlap of their areas with high transferability potential. Nevertheless, even at this scale and based on global datasets of land system indicators, our analysis was able to detect relatively small local differences between the project characteristics. CC-LandStraD that has the aim to analyze contributions of land management in Germany to climate mitigation has its study sites chosen to represent land systems of a developed high-tech country in a temperate climate zone and reliable political structures (Fick *et al* 2014). Indeed, this is reflected in the results that identified areas with high potential for transferability in large parts of Western Europe, especially Germany and France but also parts of Central Europe. In contrast, COMTESS that focuses on developing land use strategies to promote sustainable management of vulnerable coastal landscapes (Karrasch *et al* 2014) has its high transferability potentials situated more to the north. In addition to covering large portions of Germany and France, the estimated areas with high transferability potential extend to coastal areas of Belgium, the Netherlands and Denmark.

The refined analysis of transferability potentials for SASCHA revealed a dependency of the results on the resolution and accuracy of the considered input data (figure 7). Surprisingly, the global datasets captured the regional realities (represented by data from regional statistics) considerably well, although the indicators of pasture areas and N fertilizer were underestimated in the global datasets. This led to an overall decrease of the estimated transferability potentials in the refined analysis (e.g. lower transferability potentials in Ukraine due to differences in yields and socioeconomic conditions), but the general pattern remained largely similar, covering most of the Western Siberian grain belt (Kühling *et al* 2016). Similarly for SURUMER, which focuses on sustainable rubber cultivation in the Mekong Region, only few out of the 32 global variables had values that did not closely match the local reality (Häuser *et al* 2015). Of the land-use intensity indicators, both cropland area and the use of N fertilizer were underestimated. Recent data published by Xu *et al* (2014) point towards a total share of 22% of rubber, paddy rice and upland maize cultivation in the SURUMER wider research area, as opposed to about 10% of cropland area indicated in the global dataset. Also, the maize yields appear to be underestimated considering the availability of industrial fertilizer and presence of an agricultural extension system in China (Hu *et al* 2009). The FAO data for China (FAOSTAT 2010) indicate maize yields of 5.2 t ha^{-1}, as opposed to less than 2 t ha^{-1} given by the global dataset. Soil organic carbon content may be also underestimated as the vast majority of rubber plantations are situated on soils cleared recently from forest (Häuser *et al* 2015).

We used a comprehensive set of global land system metrics with the highest resolution currently available. However, despite considerable improvements in global-scale geospatial datasets (Verburg *et al* 2011), the main sources of uncertainty remain in the quality of input data and the availability of socio-cultural information in a globally standardized format. The quality of datasets is affected by many factors, such as the reliability of ground-based inventories, processing techniques of remotely sensed records, positional accuracy, spatial scale of data aggregation or the difficulties in quantification and standardization (Kuemmerle *et al* 2013). Some land use indicators (e.g. yield gaps, N fertilization) are also based on hybrid maps that link remote sensing or ground-based measurements with outputs of mechanistic models, therefore errors in the base data can propagate onto derivative maps (Verburg *et al* 2011). This also explains why many available land use indicators tend to be correlated, although in our analysis we included only those with limited redundancy (table A2). Although we embraced a wide range of variables on land-use intensity as well as environmental and socioeconomic conditions, numerous gaps exist in the availability of important land system indicators. For instance, information on mechanization, farm size, pesticide use, labor intensity, shifting cultivation or forest logging is lacking or is unavailable in adequate quality for many regions. Furthermore, information on culture, governance and policies are notoriously difficult to capture in spatially explicit datasets (Otto *et al* 2015).

On the other hand, our approach is not limited to the selected sets of indicators but allows including any data that are appropriate for a given case study and research question. For example, we included yields for three major crops as output metrics of land-use intensity because together they are representative for the majority of global cereal production. However, data for many other crops are now available. For instance, oil palm and soybean plantations are of major concern for conservation due to their expansion in the tropics (Gasparri *et al* 2013, Wilcove *et al* 2013); they can be used in the analysis in addition to or instead of the current three crops. Our approach also allows giving preference to specific variables or sets of variables. Figure 8 provides an example of the transferability potential analysis for LEGATO calculated separately for land-use intensity, environmental and socioeconomic conditions. For instance, when the environmental conditions are considered to be the sole criteria for defining transferability potentials, the results identify large portion of Southeast Asia as having similar land systems determined by broad-scale gradients of climate, soil and natural productivity of ecosystems (figure 8(b)). When land-use intensity is selected as the main criterion, the areas with high transferability potentials are restricted to a scattered pattern of intensive irrigated cropping systems in the Philippines, Vietnam, Thailand and Indonesia (figure 8(a)). The combination of all sets of variables then leads to the final pattern determined by the overall similarity of land systems to the project archetype (figures 4 and 5).

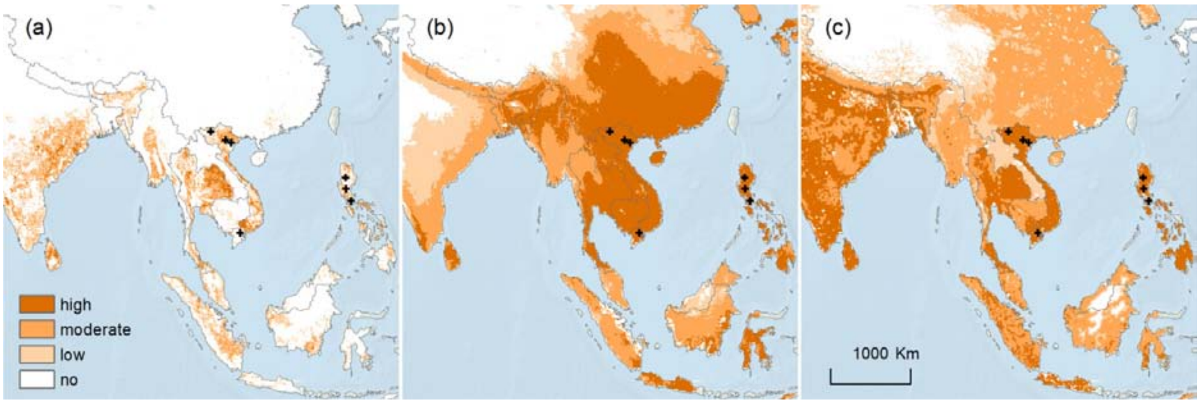

Figure 8 Transferability potential for the LEGATO regional project based on (a) land-use intensity indicators, (b) environmental indicators, and (c) socioeconomic indicators. The black crosses denote locations of the project's seven study landscapes in the Philippines and Vietnam.

Transferability, however, is a complex issue and testing which specific results or land management recommendations may be transferable into which regions, and how they can be implemented, requires a separate comprehensive analysis for each case study, based on project-specific hypotheses and fine-scale methods and data. For example, LEGATO investigates how to advance long-term sustainable development of irrigated rice agro-ecosystems, quantifying a range of ecosystem services, from provisioning services of rice production, through regulating services of pollination and biocontrol, to cultural services of identity and sense of place (Settele *et al* 2015). Different sets of criteria would have to be considered to test transferability of findings for different ecosystem services. The transfer of results regarding rice production requires accounting not only for the land-use metrics considered in this study but also for soil characteristics (e.g. concentration of silicon), dynamics of soil biota, varieties of rice planted or co-production of other goods in the rice paddies, such as fish and molluscs (Klotzbücher *et al* 2015, Schmidt *et al* 2015). The transfer of results regarding biological control of pests depends on the functional similarity of local food webs (species compositions, population densities, growth rates) that occur in landscapes with comparable habitat heterogeneity and is restricted to areas where the costs of pesticide application are high enough for farmers to be motivated to search for methods using biocontrol, or where governments intervene to enforce principles of ecological engineering (Spangenberg *et al* 2015). The results regarding cultural services cannot be easily transferred outside of the cultural context defined by the semiotic system of local communities, including religious views, belief systems, traditions and rituals (Spangenberg *et al* 2014). These complex issues illustrate that rather than offering a way to test local-scale transferability of specific findings *per se*, our approach provides a starting point to identify broad-scale regions with potential transferability of place-based research by calculating envelopes that define the general boundaries of projects' relevance outside of their study areas.

5.5 Conclusions

Place-based research in local and regional case studies has been central to understanding land use as a result of dynamic interactions within SESs that operate across spatial and temporal scales (Rounsevell *et al* 2012). Any generalization of place-based research is challenging because results depend on host of factors unique to the study system. Needed are ways of extracting general insights from the idiosyncrasies of place, so they can be applied to previously unstudied systems (van Vliet *et al* 2015). In this paper, we addressed this challenge by assessing the geographical relevance of case studies and investigated their potential transferability beyond the geographical context in which they are conducted. We adapted the previously developed concept of LSAs because land systems serve as an efficient platform for integrating different perspectives and dimensions of land use research (Verburg *et al* 2015). Our analysis of transferability potentials contributes to the development of globally relevant knowledge creation and sharing in land system science, and advances the discussion on how applicable the most up-to-date global datasets are for characterizing regional-scale findings. The proposed method can serve as a blueprint for large-scale research programs to assess potential transferability of place-based studies to other geographical areas and to indicate possible gaps in research efforts. Such assessments will be ultimately helpful to better understand and enhance the transparency of the biophysical and socioeconomic background on which decision-makers develop and evaluate SLM strategies.

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6 General Discussion

The expectations of ecosystem service approaches for biodiversity and environmental conservation are high, some environmentalists even claim that this is the last great hope to integrate conservation into planning and resource management (Portman 2013). For sure, these approaches have the potential to show from a holistic point of view what different services ecosystems provide and how they can benefit humans. So it should be possible to include this knowledge into a system of decision-making for a better use of ecosystems for society as a whole, leaving the path from maximizing individual short-term gains to making fair use of nature's resources intra- and inter-generational. Daily et al. (2009) described the necessary steps for this process which should be ideally a continuous process (figure 1). *Decisions* on land-use planning and management will always impact *ecosystems*. It is the task of science to show and quantify these changes which can either be ex post for decisions that were already implemented or ex ante before decisions are executed. How these changes in ecosystems will change the production of *services* is another question science should aim to answer. Measuring indicators, developing production functions and modeling are adequate methods from natural sciences to address this topic. With the help of economic and cultural models these biophysical units need to be translated into *values* in the next step. Changing *institutions* is a crucial part, whereupon the term in this framework refers mainly to customs and traditions and only in the second instance to formal institutions like governmental laws. Achieving institutional change is one of the biggest challenges but important to influence the next step from institutions to decisions. Different incentives or deterrents can initiate a change in behavior and therefore decisions: direct or indirect monetary rewards, legal sanctions, guilt and remorse or approval by the community are different possibilities; there are various options to influence decisions of individuals or communities. After running through the framework loop, ideally it begins again with analyzing how these newly accomplished alternative decisions will impact ecosystems.

This dissertation focuses on the part of the above mentioned decision loop that deals with ecosystems and services and what needs to be in between. One of the general problems arising from this framework is that robust scientific concepts are needed to fulfill the demands for decision-making, which take some time to be developed. There are a number of questions still to be solved. Up to 2010 Seppelt et al. (2001) reviewed 153 publications about ecosystem service assessments concerning the use of important factors that were: data source (primary or secondary data), model type (model, look-up table or other[4]), indicator (bio-physical, categorical, monetary or other), system border definition (bio-physical, administrative, both and other), considered uncertainty (quantitative, qualitative, none

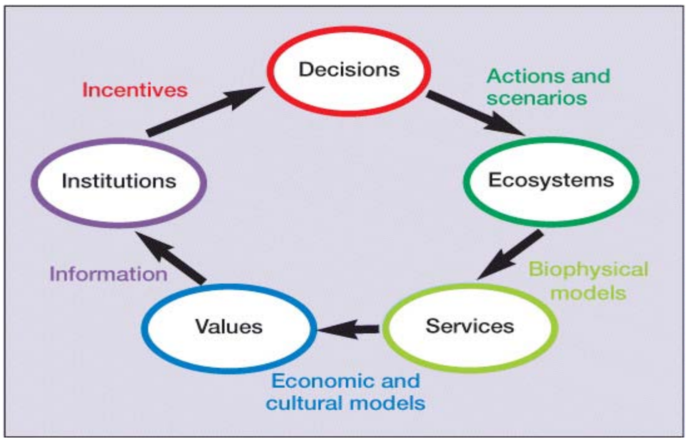

Figure 1 Integration of ecosystem services into decision-making (Daily et al. 2009).

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[4] other refers in this list always to cases with insufficient information

or other), results validated (yes, no, other), ecosystem services in isolation (no, yes or other), scenarios used (yes (political, behavioral, demographic or climate change scenarios), no or other), stakeholder involvement (yes or no) and specific recommendations given (yes or no). They came to the conclusion that most of the ecosystem service studies did not do well in responding to the above posed scientific questions.

The following paragraphs will discuss the challenges that were relevant for this study of biodiversity and ecosystem services in rubber.

The number of services analysed

The review on rubber cultivation (see chapter two) showed that the majority of publications concentrated on single or few ESSs, which provides only limited information for decision-makers who need a holistic picture considering multiple ecosystem services from the different groups of provisioning, regulating and cultural services to make sound decisions. The results of the studies that do not cover a range of ecosystem services, preferably from the different groups are therefore not what policy-makers need for land-use planning or for developing Payment for Ecosystem Service (PES) schemes. Leaving the disciplinary path and developing interdisciplinary studies remains to be a major challenge. Overcoming it will require joint research or collaborative projects. In the review of chapter three we looked at different ecosystem services for a broad picture of the consequences that rubber cultivation inflict on the Greater Mekong Subregion, ranging from income generated by rubber (provisioning service), carbon stocks, run-off, water quality and erosion (regulating services). In consideration of this background, solutions for the specific problems of monoculture rubber cultivation should comprise: (1) the interdisciplinary analysis and quantification of ecological processes and services affected by rubber cultivation compared with natural forest conditions; and (2) the development of alternative land-use strategies including the identification of trade-offs and synergies between safeguarding services on the one hand and the socio-economic viability on the other; and (3) the identification of incentives of acceptance and implementation of the concepts by farmers and other stakeholders. To facilitate the next steps in the decision loop from services to decisions via values and institutions we recommend including true stakeholder-participation (farmers as well as local politicians) at an early stage of the ecosystem service approach, where scientists not only inform, but taylor their research to the needs of their stakeholders.

The use of proxys based on land-use

The model of InVEST (Integrated Valuation of Ecosystem Services and Trade-offs) that was used for the biodiversity analysis in the Naban River Watershed National Nature Reserve, Dai Autonomous Prefecture of Xishuangbanna, China, in chapter four relies on land-use maps; ecosystem services are modeled using proxys for different land-use classes, resembling a kind of spatial value transfer. Since data on the provision of actual ecosystem services in a given landscape is normally not available, this method is a possibility for data scarce environments. But, as in all models, the users should always bear in mind that a model aims at reflecting reality as close as possible, but certainly has its limitations. Eigenbrod et al. (2010) compared the provision of three ecosystem services (biodiversity, carbon storage and recreation) spatially explicit in England. One map displayed available primary data. For the second map, the authors calculated mean values for the three analyzed ecosystem services for each land-use class and displayed the results according to the land-use classes. These two maps were devastatingly different; meaning that for all proxy-based derived maps for ecosystem services caution is advised when interpreting spatially explicit results. Especially when aiming at identifying ecosystem service hotspots (hotspots being areas with either high values for one ecosystem service or above-average values for a bunch of ecosystem services) the results of this method should only be used to identify areas where additional primary data collection of actual ecosystem services should be carried out. For our study on biodiversity (see chapter four) we used our own local primary data of the region wherever possible to derive locally adapted proxys for our models, but the above mentioned short-coming persists. Detailed primary data gathering in the areas of investitgations would be a better (or even the best) solution, but due to financial, personnel and time constraints this is normally not possible. Another option would be to differentiate land-use classes further into different age groups (for rubber) or different management groups (low-input versus high input). But again, this would increase the effort for data gathering.

The relationships between different ecosystem services

In general, relations between ecosystem services can either be neutral (they do not influence each other), synergistic (a rise in one ecosystem service entails a rise in another) or traded off against each other (a rise in one ecosystem service entails a drop in another). Bennett et al. (2009) described the different possible changes in two services following a driver of change: is the interaction of ecosystem services strong or weak, mono- or bi-directional, is the impact of the driver influencing only one ecosystem services or both? They conclude that there is still a number of questions to be answered to improve our understanding of these relationships.

A new possibility of analyzing relations was developed by Jopke et al. (2015). Instead of testing for linear correlations between ecosystem services, they used bagplots to analyze relationships of pairs of ecosystem services. Their spatially displayed results for Europe reveal that although a relation between two services might in general be positive-negative, thereby being classified as a trade-off, this must not necessarily be the case in all areas. Mapping crop capacity versus soil fertility revealed considerable areas in Poland and Germany where these two services both show high values. This underlines again the importance of primary data.

The analysis of relationships as such is already a complex task and as seen above may sometimes yield contradictory results. But since they form the basis for the next discussion points, science should further strive to explore these relations.

The prevailing use of independent models instead of dynamic models

Analysing or modeling single ecosystem services independently from one another ignores the different relationships and possible interdependencies of ecosystem services. To better understand ecosystems, these are crucial questions which are until now often neglected. The use of dynamic models where ecosystem services are coupled would be a solution to address this problem.

There are a number of analytical tools available that allow replicable and quantifiable ecosystem service analyses, such as InVEST, ARIES (Artificial Intelligence for Ecosystem Services) and MIMES (Multiscale Integrated Model of Ecosystem Services), which all belong to the group of independently applicable and generalizable landscape-scale models (Bagstad et al. 2013).

However, the first two so far rely on independent models for each ecosystem service without feedback loops. Since many of the ecosystem services influence each other this is a far-reaching simplification, reducing the explanatory power of these models. Only in 2015, global and regional results of the dynamic model MIMES were described and scientifically published, (Boumans et al. 2015). This model aims at supporting sustainable and ecosystem-based management planning with a special focus on incorporating concepts to capture the dynamic character of coupled human and natural systems. In this model, the different spheres of lithosphere, hydrosphere, biosphere, atmosphere and anthrosphere are coupled, thereby capturing exchange, interaction and feedbacks between the different spheres. Additionally, different forms of capital are included: built, human, social and natural capital. Production functions for ecosystem services, demand profiles for the different services and impact functions of economic production on the structure and function of ecosystems are further key elements of the model. The incorporation of scenarios allows to model consequences for different alternatives. The downside of this very promising model is that data requirements for such a dynamic model are very high and far beyond what data is usually available outside of the US and Europe.

The necessity of trade-off analysis

Knowledge about the aforementioned relationships between different services is also crucial for trade-off analysis. Trade-offs are the most challenging relations for maximizing human well-being, therefore the topic needs further attention.

In general, trade-offs arise from management choices, changing the type, magnitude and relative mix of services that are provided from different ecosystems. They can be classified into three categories: spatial scale, temporal scale and reversibility (Rodriguez et al. 2006). Spatially, the effects of the trade-offs can either occur at a local scale (e.g. deforestation changing groundwater table levels), at a regional scale (e.g. high inputs of fertilizers in an upstream watershed to improve food provision changing water quality downstreams) or at a global scale (e.g. rising CO_2 levels due to large scale deforestation excalerating climate change that causes sea levels to rise which eventually increase flood risks for low lying coastlines). Temporal scales refer to the time when the trade-offs occur, from immediate changes (e.g. diverted water upstream for irrigation is no longer available downstream) to intermediate changes (e.g. the above mentioned fertilizer example) to long-term changes (e.g. with moderate extraction rates, fossil aquifers may last several generations until they are depleted). The last category refers to reversibility. Trade-offs can either be reversible (e.g. reforestation to mitigate climate change) or irreversible (e.g. when tipping points of ecosystems are trespassed and the system entered a new state). Trade-offs across these three categories occur in all possible combinations, thereby forming a complex system.

In a given region, it is possible to analyze trade-offs between rivaling pairs of ecosystem services. Changing land-use will impact both ecosystem services, which means that the increase in one service will inevitably decrease the second service and vice versa. This allows to generate so called efficiency frontiers (figure 2). Efficiency here means, that neither of the two ecosystem services is “wasted”. There exists a real trade-off in the sense that raising the production of one ecosystem service inevitably causes the reduction of the other. With the help of these frontiers it is possible to optimize efficiency in a landscape to a given goal. For example: If a landscape represents the situation in point C and if policy makers decide that ecosystem service 1 is more

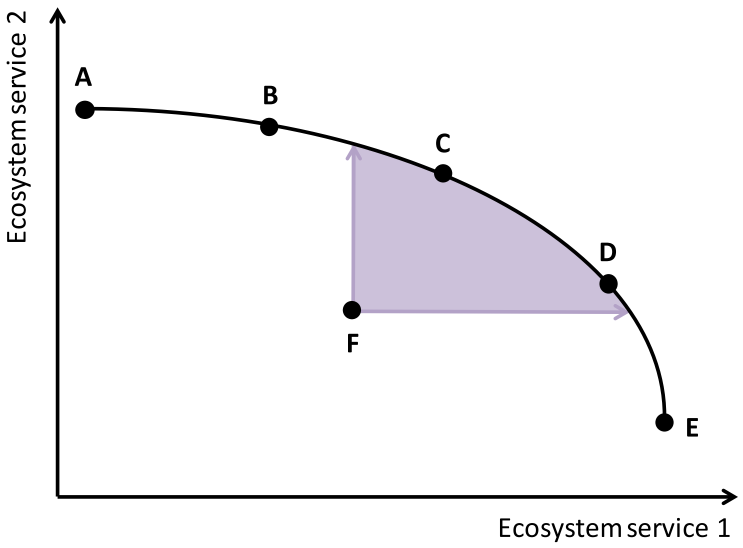

Figure 2 Efficiency frontier of two ecosystem services.

important than ecosystem service 2, they might decide that a better solution for their given problem is in point D. The frontier line between A and E shows the inherent trade-off between ecosystem service 1 and ecosystem service 2, point A representing a landscape that maximizes ecosystem service 2 on the expense of ecosystem service 1 and *vice versa* for point E. All points on the efficiency frontier in between these two extremes represent land-use options which trade-off ecosystem 1 and ecosystem 2, respectively, but show all efficient options. Comparing real landscapes with these efficiency frontiers may reveal that the efficiency frontier is not yet reached (point F) and showing possibilities to increase overall ecosystem provision in this landscape (each point in the purple area would be an improvement in comparison to point F, efficient solutions lie again on the efficiency frontier). These efficiency frontiers can also be calculated for more than two ecosystem services if the abscissa is substituted by a universal indicator under which different ecosystem services can be combined, e.g. using monetary values (for a detailed example see Polasky et al. (2008), where a landscape is optimized for biodiversity and economic returns from agriculture and managed forestry). Nevertheless, this approach does not solve the general problem of which ecosystem service should be maximised for an overall optimal solution, this decision remains in the hands of policy-makers.

Management practices have the potential to mitigate or even reverse a trade-off, as shown in the previous example of Jopke et al. (2015) (crop capacity and soil fertility were not traded off against each other in Germany and Poland, but in the rest of Europe). The importance of management practices as influencing factor is also underpinned by several studies reviewed by Cavender-Bares et al. (2015). This is a shortcoming of our biodiversity study dealing with flora (vascular plants), vertebrates (mammals and birds) and invertebrates (bees, beetles, hoverflies, nematodes and spiders) where we could neither include different management options of rubber cultivation (amount of pesticides used, intercropped or not) because of the lack of data, nor different succession stages in rubber plantations. Over the cultivation period the rubber system offers habitat for multiple and successive communities of vascular plants, vertebrates and invertebrates, from open land and forest remnant species in the establishment phases up to (moderately) shade tolerant species usually found in secondary forest systems. For an optimized landscape one rule should therefore be to create a beneficial mosaic of different rubber stages.

Even when limiting oneself to analyzing trade-offs it is still a multi-layered problem. Trade-offs occur between ecosystem services, but they are seen from an anthropocentric point of view: how do humans benefit, so these trade-offs also have a social dimension. Different stakeholders may value certain ecosystems services differently at a given time. Human societies in general value provisioning services higher than regulating and cultural services. When new areas are settled they are first altered to fulfill basic human neads, especially to produce food (Rodriguez et al. 2006). Cavender-Bares et al. (2015) showed in a review the correlation between wealthy and poor countries respectively (based on gross national income) and biodiversity (indicators are bird diversity, vascular plant diversity and mammal diversity). Biodiversity in wealthy countries is low, whereas it is high in poor countries. But poor countries will value local provisioning services that either fill their stomach or generate income higher than conservation of biodiversity, a global service valued high by wealthy countries. This dilemma shows that especially trade-offs that occur over large spatial or temporal scales need international mechanisms or incentives (e.g. Payments for Ecosystem Services PES) to globally protect the threatened ecosystem services (Rodriguez et al. 2006).

The transferability of results

The transferability of results gained in a specific region to other regions is always an issue of interest. Even if ecological conditions in another region might be similar it cannot be taken for granted that the results can be transferred to this region. Especially differing socio-economic conditions might oppose the transferability. To address this problem, all projects of the Sustainable Land Management Project, sponsored by the German Federal Ministry for Education and Research were tested for transferability. To include not only ecological indicators, but also socio-economic indicators the system of land archetypes was used (see chapter five). The results for the SURUMER project showed, that only few of the 32 global indicators did not closely match the local reality. The transferability potential for the GMS was quite high and therefore emphasizes the importance of the project results for this huge region. Other areas with high scores of statistical similarity where the transferability potential is assumed to be high were identified in Africa (where rubber production is also carried out) and in South-America where large-scale rubber production is limited by the South-American leaf blight disease, caused by *Pseudocercospora ulei*.

The monetary dimension

In the decision-making loop presented at the beginning of this chapter, the next step is from services to valuation. This is the link between natural sciences and economics, where biophysical values need to be transformed into monetary values. Although this step is not always included and for some ecosystem service approaches biophysical values without further monetary valuation might be sufficient, for several approaches valuation is very important, e.g. for developing PES schemes, environmental-economic accounting and green Gross Domestic Product (GDP).

A number of methods for monetary valuation are available. Direct market values are quite easy to assess, whereas indirect market values are more complicated to measure. Examples for indirect market values are avoided cost (e.g. how much is saved by a wetland that provides flood control for downstream housings, thereby avoiding property damage), replacement cost (e.g. how much does it cost to build a water treatment plant to clean the water after a pollution), travel cost (e.g. how long do people travel and how much do they pay for this travel to visit a certain national park) and hedonic pricing (e.g. how much do people pay more to buy a house with ocean view). Another method is the contingent valuation system, which uses questionnaires to test willingness to pay for a hypothetical scenario (de Groot et al. 2002). All these methods have their specific strengths and weaknesses. Monetary valuation is adding another component with a number of assumptions (ranging from how to derive values to what discount rate to assume) that makes the research even more complex and less possible to give reliable predictions because it further increases uncertainty. The authors of The Economics of Ecosystems and Biodiversity (TEEB) explicitly note that placing "a monetary value on biodiversity and ecosystem services are fraught with complications, only some of which currently can be addressed" (TEEB 2010).

The inclusion of biodiversity in the ecosystem service framework

Although biodiversity is not an ecosystem service in the narrow sense, it is inextricably linked to the concept because it influences their provision (MEA 2005). There are different ways of how biodiversity can be incorporated into ecosystem service assessments: (i) From a very narrow ecosystem service perspective that focuses on services for human-well being it is possible to include biodiversity directly as provisioning service for hunting and gathering. This is easy to implement, but undermines the importance of biodiversity and considers only a very small fraction of non-human life. We analyzed this part of biodiversity in chapter four by studying "human use" species separately. (ii) In a bit broader sense as a regulating and maintaining service as included in the Common International Classification system (Haines-Young and Potschin 2013), where biodiversity is directly linked to certain services: one being micro-organisms for bio-remediation and the other being habitat protection for maintaining nursery populations and species that are important for pollination, seed dispersal, pest and disease control. (iii) In the broad sense as it is defined in MEA (2005): biodiversity is an underlying driver for ecosystem functions that result in ecosystem services. In this sense it can (a) be included in dynamic modeling as a driver, which would be the best way to capture the importance of biodiversity but also be the most complex one to realize or (b) it can be analyzed along with other ecosystem services and these results can be included in a second step in a trade-off analysis, which is much more common and also the one we used in chapter four.

Our study showed that habitat suitability for the species groups studied (overall biodiversity consisting of vascular plants, vertebrates and invertebrates; "human use species" consisting of wild bees and plants for traditional Chinese medicine; "red list species" consisting of red list vascular plants and red list mammals) changed over the course of the scenarios. Due to the relatively narrow (or realistic) definitions of the scenarios, these changes were only marginal but it is still possible to reason an interesting point for discussion. Especially when considering the indices from the category "human use", which is an ecosystem service as described in the previous paragraph under (i), we see less of an impact between the scenarios. This can be explained when looking at the distribution of these species groups within the land-use classes. Medicinal plants for example are still commonly distributed within rubber plantations, albeit with different species (generalists, ruderal flora). A shift between rubber and forest thus has a relatively low impact on the numbers of species with medicinal use. In addition, wild bee species are present with forest dwelling species and species adapted to open-land or succession states found in young rubber. This is hinting towards one of the weaknesses of this method: the decision on "what" to analyze. A concentration on biodiversity for human use gives a different picture compared to analyzing red list species. If not only quantity, but also quality of biodiversity is of interest, then separate analyses for red list species or endemics should be carried out.

This leads to a very fundamental critique of the ecosystem service concept in general and the inclusion of biodiversity in particular, the ongoing dispute about the proper objective for conservation, which is divided into two groups (Polasky et al. 2012). One group shares the opinion that mankind should conserve nature (including ecosystems and biodiversity) for its intrinsic values (traditional approach of conservationists) and the other one argues that biodiversity and ecosystem services are fundamental in contributing to human well-being, thereby arguing from an anthropocentric point of view. For a detailed scholarly discussion see (Reyers et al. 2012).

Concluding remarks

As has been shown, there are still many open questions and fields that need further research. Consequently, it is difficult to address all of the above mentioned challenges at a time. Although research is progressing and there are promising methods that could solve at least a number of the problems mentioned in this chapter it will still take more time and more effort until a profound understanding is gained and until science is able to provide the necessary data for decision-making, a precondition for integrating conservation into planning and resource management.

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7 Appendix

7.1 Ecosystem services and biodiversity of rubber plantations – a systematic review, Supplementary material A

Complete list of references

Database 1: WebofKnowledge
Database 2: Scopus

Search terms 1:
Ecosystem service* AND rubber
Ecosystem service* AND hevea
Ecosystem function AND hevea
Ecosystem function AND rubber
Search terms 2:
environmental service AND rubber
environmental service AND hevea
environmental function AND hevea
environmental function AND rubber
Search terms 3:
land-use change AND rubber
"land-use change" AND rubber
land-use change AND hevea
"land-use change" AND rubber

Relevance

1	exclusion based on title
2	exclusion based on abstract
3	exclusion based on paper
0	publication included

Cause of rejection

1	no reference of rubber tree
2	no rubber in title/abstract
3	does not fit to research question
4	generation of land-use maps
5	not in English
6	only land-use change
7	not available
8	rubber and oil palm not seperated
9	only cited ESS (no data)
10	abstract only, no ESS data

Reference	Database	Search terms	Rele-vance	cause of rejection
(1) 2nd International Conference on Environmental Science and Material Application, ESME 2014. Advanced Materials Research 2014;908.	2	1, 2	1	1
(2) 2013 International Conference on Materials Science, Machinery and Energy Engineering, MSMEE 2013. Advanced Materials Research 2014;853.	2	2	1	1
(3) 3rd International Conference on Energy and Environmental Protection, ICEEP 2014. Advanced Materials Research 2014;962-965.	2	2	1	1
(4) 5th KKU International Engineering Conference 2014, KKU-IENC 2014. Advanced Materials Research 2014;931-932.	2	3	1	1
(5) 4th International Conference on Engineering Materials, Energy, Management and Control, MEMC2014. Advanced Materials Research 2014;977.	2	2	1	1
(6) 2013 3rd International Conference on Materials and Products Manufacturing Technology, ICMPMT 2013. Advanced Materials Research 2013;834-836.	2	2	1	1
(7) World Environmental and Water Resources Congress 2013: Showcasing the Future - Proceedings of the 2013 Congress. World Environmental and Water Resources Congress 2013: Showcasing the Future - Proceedings of the 2013 Congress; 2013.	2	2	2	1

Reference	Database	Search terms	Rele-vance	cause of rejection
(8) 2013 4th International Conference on Information Technology for Manufacturing Systems, ITMS 2013. Applied Mechanics and Materials 2013;421.	2	2	1	1
(9) 2013 International Conference on Energy Research and Power Engineering, ERPE 2013. Applied Mechanics and Materials 2013;341-342.	2	2	1	1
(10) 4th international Conference on Manufacturing Science and Engineering, ICMSE 2013. Advanced Materials Research 2013;712-715.	2	2	1	1
(11) 2nd International Conference on Civil Engineering and Transportation, ICCET 2012. Applied Mechanics and Materials 2013;253-255(PART 1).	2	2	1	1
(12) 3rd International Conference on Textile Engineering and Materials, ICTEM 2013. Advanced Materials Research 2013;821-822.	2	2	1	1
(13) 2013 2nd International Conference on Function Materials and Nanotechnology, FMN2013. Advanced Materials Research 2013;771.	2	2	1	1
(14) 13th International Multidisciplinary Scientific Geoconference and EXPO, SGEM 2013, volume 2. International Multidisciplinary Scientific GeoConference Surveying Geology and Mining Ecology Management, SGEM; 2013.	2	3	1	1
(15) 13th International Multidisciplinary Scientific Geoconference and EXPO, SGEM 2013, volume 1. International Multidisciplinary Scientific GeoConference Surveying Geology and Mining Ecology Management, SGEM; 2013.	2	3	1	1
(16) 2nd International Conference on Green Building, Materials and Civil Engineering, GBMCE 2013. Applied Mechanics and Materials 2013;368-370(1).	2	3	1	1
(17) Global Conference on Civil, Structural and Environmental Engineering, GCCSEE 2012 and the 3rd International Symposium on Multi-field Coupling Theory of Rock and Soil Media and Its Applications, MCTRSM 2012. Advanced Materials Research 2012;600.	2	2	1	1
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(682) Zhai D-, Xu J-, Dai Z-, Cannon CH, Grumbine RE. Increasing tree cover while losing diverse natural forests in tropical Hainan, China. Regional Environmental Change 2014;14(2):611-621.	2	1	3	4,6
(683) Zhang Nian-Nian, Chen You-Qing, Lu Zhi-Xing, Zhang Wei, Li Ke-Li. Species diversity, community structure difference and indicator species of leaf-litter ants in rubber plantations and secondary natural forests in Yunnan, southwestern China. Acta Entomol Sin 2013 NOV 20 2013;56(11):1314-1323.	1	1, 2	0	0
(684) Zhang H, Zhang G-. Landscape-scale soil quality change under different farming systems of a tropical farm in Hainan, China. Soil Use Manage 2005 MAR 05;21(1):58-64.	1	2	0	0
(685) Zhang J, Cao M. Tropical forest vegetation of Xishuangbanna, SW China and its secondary changes, with special reference to some problems in local nature conservation. Biol Conserv 1995;73(3):229-238.	2	3	3	3
(686) Zhang M, Schaefer DA, Chan OC, Zou X. Decomposition differences of labile carbon from litter to soil in a tropical rain forest and rubber plantation of Xishuangbanna, southwest China. Eur J Soil Biol 2013;55:55-61.	2	3	0	0
(687) Zhang M, Zou X-. Comparison of soil C and N in rubber plantation and seasonal rain forest. Chinese Journal of Applied Ecology 2009;20(5):1013-1019.	2	3	0	0
(688) Zhang M, Fu X, Feng W, Zou X. Soil organic carbon in pure rubber and tea-rubber plantations in South-western China. Trop Ecol 2007 2007;48(2):201-207.	1, 2	3	0	0

Reference	Database	Search terms	Rele-vance	cause of rejection
(689) Analysis on operating performance of RTV coated insulators used in dry area. EIC 2014 - Proceedings of the 32nd Electrical Insulation Conference; 2014.	2	2	1	1
(690) Zhang Y, Zheng B, Ma Y, Li H, Liu W, Cao M. Population, policy and profit in tropical deforestation in Xishuangbanna, China. Global Change and Forestry: Economic and Policy Impacts and Responses 2010:77-85.	2	3	3	10
(691) Zhang Z, Van Coillie F, De Clercq EM, Ou X, De Wulf R. Mountain vegetation change quantification using surface landscape metrics in Lancang watershed, China. Ecol Ind 2013;31:49-58.	2	3	2	6
(692) Study on reliability of room temperature vulcanization silicone rubber and conductive composite silicone rubber reinforced by silica. Proceedings - International Symposium on Advanced Packaging Materials; 2011.	2	2	1	1
(693) Shadow price of CO2 emissions and potential cost saving from carbon emissions trading in Shanghai manufacturing sector. Proceedings of the 26th International Conference on Efficiency, Cost, Optimization, Simulation and Environmental Impact of Energy Systems, ECOS 2013; 2013.	2	2	1	1
(694) Zhou X, Ning P, Huang J, Lu X. Evaluation for forest water conservation value in Naban river watershed National Nature Reserve. 2011 International Conference on Electrical and Control Engineering, ICECE 2011 - Proceedings; 2011.	2	1	0	0
(695) Zhou X, Wang Z, Yu B, Seitz L. Effects of large-scale rubber farm on soil erosion and river bed material in the upper Mekong Basin. Proceedings of the International Conference on Fluvial Hydraulics, RIVER FLOW 2014; 2014.	2	3	0	0
(696) Zhou ZG, Lei M, Chen JY. Impact of rubber modifier on the rheological properties of asphalt. Advanced Materials Research 2014;997:465-470.	2	2	1	1
(697) Ziegler AD, Phelps J, Yuen JQ, Webb EL, Lawrence D, Fox JM, et al. Carbon outcomes of major land-cover transitions in SE Asia: Great uncertainties and REDD+ policy implications. Global Change Biol 2012;18(10):3087-3099.	1, 2	1, 2, 3	0	0
(698) Zimmerman MJ, Waldron MC, Barbaro JR, Sorenson JR. Effects of low-impact-development (LID) practices on streamflow, runoff quantity and runoff quality in the Ipswich River Basin, Massachusetts: A summary of field and modeling studies. US Geological Survey Circular 2010(1361):6-41.	2	3	1	1
(699) Zomer RJ, Trabucco A, Wang M, Lang R, Chen H, Metzger MJ, et al. Environmental stratification to model climate change impacts on biodiversity and rubber production in Xishuangbanna, Yunnan, China. Biol Conserv 2014;170:264-273.	2	3	3	3, 9

7.2 Ecosystem services and biodiversity of rubber plantations – a systematic review, Supplementary material B

Relevant list of references

Supplementary material B, relevant list of references

Region code	
1	America
2	SE-Asia
3	Africa
4	Bangladesh, India, Sri Lanka
5	several locations

Section code	
1	Provisioning ESS
2	Regulation & Maintenance ESS
3	Cultural ESS
bio	Biodiversity

Country code (if region code = 2)	
1	Thailand
2	Malaysia
3	Indonesia
4	China
5	Cambodia

Reference	**Region code**	**Country code**	**No. of ESS or biodiversity indicator studied**	**Rubber yield or income from rubber included**	**Section code**	**Ecosystem Service or Biodiversity indicator**
(57) Aratrakorn S, Thunhikorn S, Donald PF. Changes in bird communities following conversion of lowland forest to oil palm and rubber plantations in southern Thailand. Bird Conservation International 2006;16(1):71-82.	2	1	1	no	bio	bio (birds)

Reference	Region code	Country code	No. of ESS or biodiversity indicator studied	Rubber yield or income from rubber included	Section code	Ecosystem Service or Biodiversity indicator
(59) Araujo QR, Comerford NB, Ogram AV, Al-Agely A, Santos Filho LP, Santos JG. Soil carbon and physical property changes in Brazilian Coastal Tableland soils with land-use following deforestation. Agrofor Syst 2005;63(2):193-198.	2	1	1	no	2	carbon
(62) Ayanu YZ, Trung Thanh Nguyen, Marohn C, Koellner T. Crop production versus surface-water regulation: assessing tradeoffs for land-use scenarios in the Tat Hamlet Watershed, Vietnam. International Journal of Biodiversity Science Ecosystem Services & Management 2011 SEP 2011;7(3):231-244.	2	4	2	yes	2 1	water rubber
(84) Belcher B, Rujehan, Imang N, Achdiawan R. Rattan, rubber, or oil palm: Cultural and financial considerations for farmers in Kalimantan. Econ Bot 2004;58(SUPPL.):S77-S87.	2	3	1	yes	1	financial benefits
(93) Bhumiratana A, Sorosjinda-Nunthawarasilp P, Kaewwaen W, Maneekan P, Pimnon S. Malaria-associated rubber plantations in Thailand. Travel medicine and infectious disease 2013 2013 Jan-Feb;11(1):37-50.	2	1	1	no	2	health
(133) Chiti T, Grieco E, Perugini L, Rey A, Valentini R. Effect of the replacement of tropical forests with tree plantations on soil organic carbon levels in the Jomoro district, Ghana. Plant Soil 2014 FEB 2014;375(1-2):47-59.	3		1	no	2	carbon
(144) Cotter M, Martin K, Sauerborn J. How do "Renewable Products" impact biodiversity and ecosystem services - The example of natural rubber in China. Journal of Agriculture and Rural Development in the Tropics and Subtropics 2009;110(1):9-22.	2	4	1	no	2	carbon
(145) Cotter M, Berkhoff K, Gibreel T, Ghorbani A, Golbon R, Nuppenau E, et al. Designing a sustainable land-use scenario based on a combination of ecological assessments and economic optimization. Ecol Ind 2014 JAN 2014;36:779-787.	2	4	3	yes	bio 1 1	rubber food medicines

Reference	Region code	Country code	No. of ESS or biodiversity indicator studied	Rubber yield or income from rubber included	Section code	Ecosystem Service or Biodiversity indicator
(154) de Blecourt M, Brumme R, Xu J, Corre MD, Veldkamp E. Soil Carbon Stocks Decrease following Conversion of Secondary Forests to Rubber (Hevea brasiliensis) Plantations. PLoS One 2013 JUL 19 2013;8(7):e69357-Article No.: e69357.	2	3	1	no	2	carbon
(155) de Blecourt M, Haensel VM, Brumme R, Corre MD, Veldkamp E. Soil redistribution by terracing alleviates soil organic carbon losses caused by forest conversion to rubber plantation. For Ecol Manage 2014 FEB 1 2014;313:26-33.	2	4	1	no	2	carbon
(163) Dinesh R, Chaudhuri SG, Sheeja TE. Soil biochemical and microbial indices in wet tropical forests: Effects of deforestation and cultivation. Journal of Plant Nutrition and Soil Science 2004 February 2004;167(1):24-32.	4		1	no	2	carbon
(173) Du J, Yang X-, Zhang H, Yu G-. Quantitative distribution of earthworms and its relationships with environmental factors in tropical secondary forest and rubber plantation in Xishuangbanna. Chinese Journal of Ecology 2008;27(11):1941-1947.	2	4	1	no	bio	bio (earthworms)
(187) Fang L, Yang X-, Du J. Effects of land-use pattern on soil microbial biomass carbon in Xishuangbanna. Chinese Journal of Applied Ecology 2011;22(4):837-844.	2	4	1	no	2	carbon
(189) Feintrenie L, Schwarze S, Levang P. Are local people conservationists? Analysis of transition dynamics from agroforests to monoculture plantations in Indonesia. Ecology and Society 2010;15(4).	2	3	2	yes	1 3	economic returns, sentimental attachment
(221) Gharibreza M, Raj JK, Yusoff I, Othman Z, Tahir WZWM, Ashraf MA. Land-use changes and soil redistribution estimation using 137Cs in the tropical Bera Lake catchment, Malaysia. Soil Tillage Res 2013;131:1-10.	2	2	1	no	2	erosion

Reference	Region code	Country code	No. of ESS or biodiversity indicator studied	Rubber yield or income from rubber included	Section code	Ecosystem Service or Biodiversity indicator
(239) Guardiola-Claramonte M, Fox JM, Giambelluca TW, Troch PA. Changing land-use in the Golden Triangle: Where the rubber meets the road. Sustainability Science for Watershed Landscapes 2010:235-250.	2	4	1	no	2	water
(240) Guardiola-Claramonte M, Troch PA, Ziegler AD, Giambelluca TW, Durcik M, Vogler JB, et al. Hydrologic effects of the expansion of rubber (Hevea brasiliensis) in a tropical catchment. Ecohydrology 2010;3(3):306-314.	5		1	no	2	water
(241) Guardiola-Claramonte M, Troch PA, Ziegler AD, Giambelluca TW, Vogler JB, Nullet MA. Local hydrologic effects of introducing non-native vegetation in a tropical catchment. Ecohydrology 2008 APR 2008;1(1):13-22.	5		1	no	2	water
(244) Guo Fang-fang, Li Yong-mei, Li Zhao-li, Wang Zi-lin. Effects of land-use change on soil organic carbon and microbial biomass carbon and nitrogen in Naban River watershed, Yunnan Province of Southwest China. Shengtaixue Zazhi 2012 OCT 2012;31(10):2473-2478.	1		1	no	2	carbon
(247) Haggar J, Medina B, Aguilar RM, Munoz C. Land-use change on coffee farms in southern guatemala and its environmental consequences. Environ Manage 2013;51(4):811-823.	2	3	2	no	bio 2	bio (trees) carbon
(267) Hu H, Liu W, Cao M. Impact of land-use and land cover changes on ecosystem services in Menglun, Xishuangbanna, Southwest China. Environ Monit Assess 2008 NOV 2008;146(1-3):147-156.	2	4	17	no	4 x 1 10 x 2 2 x 3 bio	***1** (see below)
(282) Ishizuka S, Tsuruta H, Murdiyarso D. An intensive field study on CO2, CH4 and N2O emissions from soils at four land-use types in Sumatra, Indonesia. Global Biogeochem Cycles 2002;16(3):22-1.	2	3	1	no	2	greenhous gas emission

Reference	Region code	Country code	No. of ESS or biodiversity indicator studied	Rubber yield or income from rubber included	Section code	Ecosystem Service or Biodiversity indicator
(283) Ishizuka S, Iswandi A, Nakajima Y, Yonemura S, Sudo S, Tsuruta H, et al. The variation of greenhouse gas emissions from soils of various land-use/cover types in Jambi province, Indonesia. Nutr Cycling Agroecosyst 2005 JAN 2005;71(1):17-32.	2	4	1	no	2	greenhous gas emission
(290) Jessy MD, Prasannakumari P, Abraham J. Carbon and nutrient cycling through fine roots in rubber (hevea brasiliensis) plantations in India. Exp Agric 2013;49(4):556-573.	4		2	no	2 2	carbon soil formation
(292) Jiang Ju-sheng, Wang Ru-song. Hydrological eco-service of rubber plantations in Hainan Island and its effect on local economic development. J Environ Sci (China) 2003 September 2003;15(5):701-709.	2	4	1	no	2	water
(305) Karunakaran N. Shift to rubber cultivation and consequences on environment and food security in Kerala. Journal of Rural Development 2013;32(4):395-408.	4		2	no	1 2	groundwater depletion, soil formation
(319) Kongsager R, Napier J, Mertz O. The carbon sequestration potential of tree crop plantations. Mitigation Adapt Strat Global Change 2013;18(8):1197-1213.	3		1	no	2	carbon
(331) Kwa BH. Environmental change, development and vectorborne disease: Malaysia's experience with filariasis, scrub typhus and dengue. Environ Dev Sustainability 2008;10(2):209-217.	2	2	1	no	2	health
(338) Lavelle P, Rodriguez N, Arguello O, Bernal J, Botero C, Chaparro P, et al. Soil ecosystem services and land-use in the rapidly changing Orinoco River Basin of Colombia. Agriculture Ecosystems & Environment 2014 MAR 1 2014;185:106-117.	1		4	no	bio 2 2 2	*2 (see below)
(343) Lehébel-Péron A, Feintrenie L, Levang P. Rubber agroforests' profitability, the importance of secondary products. Forests Trees and Livelihoods 2011;20(1):69-84.	2	3	1	yes	1	rubber

Reference	Region code	Country code	No. of ESS or biodiversity indicator studied	Rubber yield or income from rubber included	Section code	Ecosystem Service or Biodiversity indicator
(355) Li H, Ma Y, Aide TM, Liu W. Past, present and future land-use in Xishuangbanna, China and the implications for carbon dynamics. For Ecol Manage 2008 FEB 20 2008;255(1):16-24.	2	4	1	no	2	carbon
(356) Li S, Zou F, Zhang Q, Sheldon FH. Species richness and guild composition in rubber plantations compared to secondary forest on Hainan Island, China. Agrofor Syst 2013 OCT 2013;87(5):1117-1128.	2	4	1	no	bio	bio (birds)
(359) Li Y, Xia Y, Lei Y, Deng Y, Chen H, Sha L, et al. Estimating changes in soil organic carbon storage due to land-use changes using a modified calculation method. IForest 2014;8:45-52.	2	4	1	no	2	carbon
(373) Liu W, Hu H, Ma Y, Li H. Environmental and socioeconomic impacts of increasing rubber plantations in Menglun township, southwest China. Mountain Research and Development 2006 AUG 2006;26(3):245-253.	2	4	1	yes	1	rubber
(381) Mande HK, Abdullah AM, Aris AZ, Nuruddin AA. A Comparison of Soil CO2 Efflux Rate in Young Rubber Plantation, Oil Palm Plantation, Recovering and Primary Forest Ecosystems of Malaysia. Polish Journal of Environmental Studies 2014 2014;23(5):1649-1657.	2	2	1	no	2	carbon
(391) Meng L, Martin K, Liu J, Burger F, Chen J. Contrasting responses of hoverflies and wild bees to habitat structure and land-use change in a tropical landscape (southern Yunnan, SW China). Insect Science 2012 DEC 2012;19(6):666-676.	2	4	1	no	bio	bio (hoverflies)
(392) Meng L, Martin K, Weigel A, Liu J. Impact of rubber plantation on carabid beetle communities and species distribution in a changing tropical landscape (southern Yunnan, China). J Insect Conserv 2012 JUN 2012;16(3):423-432.	2	4	1	no	bio	bio (beetles)
(393) Meng L, Martin K, Weigel A, Yang X. Tree Diversity Mediates the Distribution of Longhorn Beetles (Coleoptera: Cerambycidae) in a Changing Tropical Landscape (Southern Yunnan, SW China). PLoS One 2013 SEP 19 2013;8(9):e75481-Article No.: e75481.	2	4	1	no	bio	bio (beetles)

Reference	Region code	Country code	No. of ESS or biodiversity indicator studied	Rubber yield or income from rubber included	Section code	Ecosystem Service or Biodiversity indicator
(409) Moreira A, Fageria NK, Garcia Y Garcia A. Soil fertility, mineral nitrogen and microbial biomass in upland soils of the Central Amazon under different plant covers. Commun Soil Sci Plant Anal 2011;42(6):694-705.	1		1	no	2	carbon
(418) Nakagawa M, Momose K, Kishimoto-Yamada K, Kamoi T, Tanaka HO, Kaga M, et al. Tree community structure, dynamics and diversity partitioning in a Bornean tropical forested landscape. Biodivers Conserv 2013 JAN 2013;22(1):127-140.	2	2	1	no	bio	bio (trees)
(440) Alos Palsar for biomass assessment on rubber and teak plantations in peninsular Malaysia. 34th Asian Conference on Remote Sensing 2013, ACRS 2013; 2013.	2	2	1	no	2	carbon
(453) Peh KS-, Sodhi NS, de Jong J, Sekercioglu CH, Yap CA-, Lim SL-. Conservation value of degraded habitats for forest birds in southern Peninsular Malaysia. Divers Distrib 2006 SEP 2006;12(5):572-581.	2	2	1	no	bio	bio (birds)
(459) GIS application for assessing the effects of land-use change on surface runoff and soil erosion in phatthalung watershed, Southern Thailand. 31st Asian Conference on Remote Sensing 2010, ACRS 2010; 2010.	2	1	2	no	2 2	water, erosion
(464) Pfund J-, Watts JD, Boissière M, Boucard A, Bullock RM, Ekadinata A, et al. Understanding and integrating local perceptions of trees and forests into incentives for sustainable landscape management. Environ Manage 2011;48(2):334-349.	5		10	no	7 x 1 2 x 2 3	***3** (see below)
(476) The impact of expanding rubber tree plantation on soil erosion in the mekong sub basin. 34th Asian Conference on Remote Sensing 2013, ACRS 2013; 2013.	2	1	1	no	2	erosion
(483) Rao E, Xiao Y, Ouyang Z, Zheng H. Spatial characteristics of soil conservation service and its impact factors in Hainan Island. Shengtai Xuebao/ Acta Ecologica Sinica 2013;33(3):746-755.	2	4	1	no	2	erosion

Reference	Region code	Country code	No. of ESS or biodiversity indicator studied	Rubber yield or income from rubber included	Section code	Ecosystem Service or Biodiversity indicator
(500) Rodrigo VHL, Stirling CM, Silva TUK, Pathirana PD. The growth and yield of rubber at maturity is improved by intercropping with banana during the early stage of rubber cultivation. Field Crops Res 2005 January 14, 2005;91(1):23-33.	4		1	yes	1	rubber
(509) Saha SK, Nair PKR, Nair VD, Kumar BM. Carbon storage in relation to soil size-fractions under tropical tree-based land-use systems. Plant Soil 2010 MAR 2010;328(1-2):433-446.	4		1	no	2	carbon
(510) Sajikumar N, Remya RS. Impact of land cover and land-use change on runoff characteristics. J Environ Manage 2013.	4		1	no	2	water
(514) Sanabria C, Lavelle P, Fonte SJ. Ants as indicators of soil-based ecosystem services in agroecosystems of the Colombian Llanos. Applied Soil Ecology 2014 DEC 2014;84:24-30.	1		4	no	bio 2 2 2	bio (ants) carbon water greenhous gas emission
(519) Sasaki N, Yoshimoto A. Benefits of tropical forest management under the new climate change agreement-a case study in Cambodia. Environmental Science and Policy 2010;13(5):384-392.	2	5	1	yes	1	revenue from latex yield
(520) Satakhun D, Gay F, Chairungsee N, Kasemsap P, Chantuma P, Thanisawanyangkura S, et al. Soil CO2 efflux and soil carbon balance of a tropical rubber plantation. Ecol Res 2013 NOV 2013;28(6):969-979.	2	1	1	no	2	carbon
(550) A watershed-scale assessment of present and future carbon stock: GIS application in Khlong Yai watershed of Thailand. 28th Asian Conference on Remote Sensing 2007, ACRS 2007; 2007.	2	1	1	no	2	carbon
(565) Sohel MSI, Ahmed Mukul S, Burkhard B. Landscape's capacities to supply ecosystem services in Bangladesh: A mapping assessment for Lawachara National Park. Ecosystem Services 2014.	4		22	no	11 x 1 9 x 2 2 x 3	*4 (see below)

Reference	Region code	Country code	No. of ESS or biodiversity indicator studied	Rubber yield or income from rubber included	Section code	Ecosystem Service or Biodiversity indicator
(582) Tan-Soo J-, Adnan N, Ahmad I, Pattanayak SK, Vincent JR. Econometric Evidence on Forest Ecosystem Services: Deforestation and Flooding in Malaysia. Environmental and Resource Economics 2014.	2	2	1	no	2	flood mitigation
(592) Thiollay J. The role of traditional agroforests in the conservation of rain and forest bird diversity in Sumatra. Conserv Biol 1995 1995;9(2):335-353.	2	3	1	no	bio	bio (birds)
(621) Modelling human-landscape system dynamics to support reward mechanisms for agro-biodiversity conservation. iEMSs 2012 - Managing Resources of a Limited Planet: Proceedings of the 6th Biennial Meeting of the International Environmental Modelling and Software Society; 2012.	2	3	3	yes	1 2 bio	latex yield carbon bio (trees)
(623) Villamor GB, Le QB, Djanibekov U, van Noordwijk M, Vlek PLG. Biodiversity in rubber agroforests, carbon emissions and rural livelihoods: An agent-based model of land-use dynamics in lowland Sumatra. Environmental Modelling and Software 2014;61:151-165.	2	3	3	yes	1 2 bio	latex yield carbon bio (trees)
(624) Villamor GB, van Noordwijk M. Social role-play games Vs individual perceptions of conservation PES agreements for maintaining rubber agroforests in Jambi (Sumatra), Indonesia. Ecology and Society 2011;16(3):27.	2	3	2	yes	1 bio	revenues from rubber yields, bio
(629) Wahren A, Berkhoff K, Herrmann S, Feger K-. Building an integrated modeling framework for assessing land-use change and its consequences for areal water balance in mountainous Southwest China. Advances in Geosciences 2010;27:71-77.	2	4	1	no	2	water
(640) Wauters JB, Coudert S, Grallien E, Jonard M, Ponette Q. Carbon stock in rubber tree plantations in Western Ghana and Mato Grosso (Brazil). For Ecol Manage 2008 APR 20 2008;255(7):2347-2361.	5		1	no	2	carbon

Reference	Region code	Country code	No. of ESS or biodiversity indicator studied	Rubber yield or income from rubber included	Section code	Ecosystem Service or Biodiversity indicator
(647) Wenjie L, Wenyao L, Hongjian L, Wenping D, Hongmei L. Runoff generation in small catchments under a native rain forest and a rubber plantation in Xishuangbanna, southwestern China. Water and Environment Journal 2011;25(1):138-147.	2	4	1	no	2	water
(663) Xiao HF, Tian YH, Zhou HP, Ai XS, Yang XD, Schaefer DA. Intensive rubber cultivation degrades soil nematode communities in Xishuangbanna, southwest China. Soil Biology & Biochemistry 2014 SEP 2014;76:161-169.	2	4	1	no	bio	bio (nematodes)
(664) Xie J, Li Z-, Li Y-, Guo F-. Evaluation of soil quality under different land-use types in Naban River watershed, Yunnan Province of Southwest China. Chinese Journal of Applied Ecology 2011;22(12):3169-3176.	2	4	1	no	2	soil quality
(666) Xu Zheng-hui, Hu Gang, Yu Xin-wen. Biomass and ecological function of ant communities in the tropical rain forest of Xishuangbanna, China. Zool Res 1999 Dec., 1999;20(6):441-445.	2	4	1	no	bio	bio (ants)
(672) Yang Jing-Cheng, Huang Jian-Hui, Pan Qing-min, Han Xing-Guo. Spectroscopic characteristics of soil organic matter in different tropical ecosystems in Xishuangbanna, southwest China. Zhiwu Shengtai Xuebao 2004 September 2004;28(5):623-629.	2	4	1	no	2	carbon
(676) Yi Z-, Wong G, Cannon CH, Xu J, Beckschäfer P, Swetnam RD. Can carbon-trading schemes help to protect China's most diverse forest ecosystems? A case study from Xishuangbanna, Yunnan. Land-use Policy 2014;38:646-656.	2	4	3	yes	1 2 bio	rubber carbon bio (seed plants)
(681) Zhai D-, Cannon CH, Dai Z-, Zhang C-, Xu J-. Deforestation and fragmentation of natural forests in the upper Changhua watershed, Hainan, China: implications for biodiversity conservation. Environ Monit Assess 2014;187(1).	2	4	1	no	bio	bio (forests)

Reference	Region code	Country code	No. of ESS or biodiversity indicator studied	Rubber yield or income from rubber included	Section code	Ecosystem Service or Biodiversity indicator
(683) Zhang Nian-Nian, Chen You-Qing, Lu Zhi-Xing, Zhang Wei, Li Ke-Li. Species diversity, community structure difference and indicator species of leaf-litter ants in rubber plantations and secondary natural forests in Yunnan, southwestern China. Acta Entomol Sin 2013 NOV 20 2013;56(11):1314-1323.	2	4	1	no	bio	bio (ants)
(684) Zhang H, Zhang G-. Landscape-scale soil quality change under different farming systems of a tropical farm in Hainan, China. Soil Use Manage 2005 MAR 05;21(1):58-64.	2	4	1	no	2	soil quality
(686) Zhang M, Schaefer DA, Chan OC, Zou X. Decomposition differences of labile carbon from litter to soil in a tropical rain forest and rubber plantation of Xishuangbanna, southwest China. Eur J Soil Biol 2013;55:55-61.	2	4	1	no	2	carbon
(687) Zhang M, Zou X-. Comparison of soil C and N in rubber plantation and seasonal rain forest. Chinese Journal of Applied Ecology 2009;20(5):1013-1019.	2	4	1	no	2	soil degradation
(688) Zhang M, Fu X, Feng W, Zou X. Soil organic carbon in pure rubber and tea-rubber plantations in South-western China. Trop Ecol 2007 2007;48(2):201-207.	2	4	1	no	2	carbon
(694) Zhou X, Ning P, Huang J, Lu X. Evaluation for forest water conservation value in Naban river watershed National Nature Reserve. 2011 International Conference on Electrical and Control Engineering, ICECE 2011 - Proceedings; 2011.	2	4	1	no	2	water conservation
(695) Zhou X, Wang Z, Yu B, Seitz L. Effects of large-scale rubber farm on soil erosion and river bed material in the upper Mekong Basin. Proceedings of the International Conference on Fluvial Hydraulics, RIVER FLOW 2014; 2014.	2	4	1	no	2	erosion
(697) Ziegler AD, Phelps J, Yuen JQ, Webb EL, Lawrence D, Fox JM, et al. Carbon outcomes of major land-cover transitions in SE Asia: Great uncertainties and REDD+ policy implications. Global Change Biol 2012;18(10):3087-3099.	5		1	no	2	carbon

Impact of rubber tree dominated land-use on biodiversity and ESS in the GMS

Section	*1
1	Food production
1	Raw material
1	Genetic resources
1	Water supply
2	Gas regulation
2	Climate regulation
2	Disturbance regulation
2	Water regulation
2	Erosion control
2	Soil formation
2	Nutrient cycling
2	Water treatment
2	Pollination
2	Biological control
3	Recreation
3	Cultural
bio	Habitat/refugia

Section	*2
2	carbon
2	water
2	greenhouse gas emission
bio	soil macro invertebrates

Section	*3
1	food
1	medicinal products
1	construction
1	firewood
1	tools, basketry
1	marketed items
2	water
2	conservation
3	rituals
-	others

Section	*4
1	crops
1	livestock
1	fodder
1	capture fisheries
1	aquaculture
1	wild foods
1	timber
1	wood fuel
1	energy (biomass)
1	biochemicals/medicine
1	fresh water
2	local climate regulation
2	global climate regulation
2	flood protection
2	groundwater recharge
2	air quality regulation
2	erosion regulation
2	nutrient regulation
2	water purification
2	pollination
3	recreation & aesthetic values
3	intrinsic value of biodiversity

7.3 Biodiversity and Ecosystem Services - A case study for the Assessment of Multiple Species and Functional Diversity Levels in a Cultural Landscape, Supplementary material – Habitat Quality Modeling, InVEST 3.2.0

For detailed information on the model please refer to InVEST User Guide, Chapter Habitat Quality (www.naturalcapital.org). For the model runs, we used the following input data.

Current and future land cover

Our main research area is the Naban River Watershed National Nature Reserve (NRWNNR) located in the Dai Autonomous Prefecture of Xishuangbanna (Figure 1).

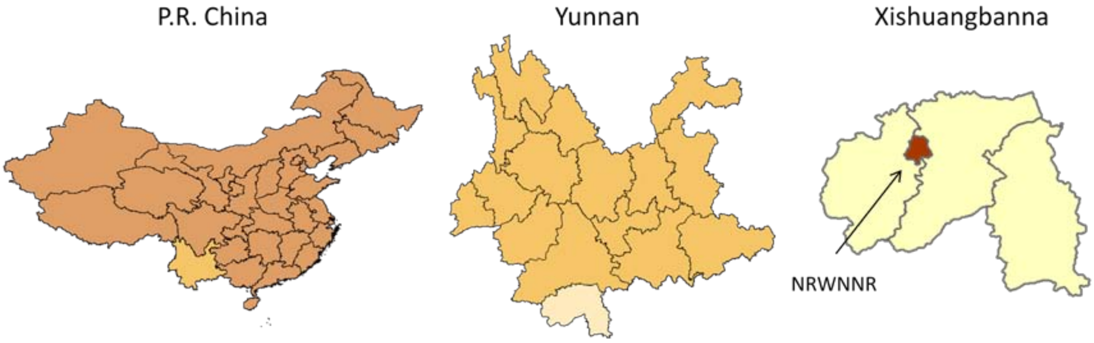

Figure 1 Location of Research area. From left to right: PR China with highlighted Province of Yunnan, Province of Yunnan with highlighted Dai Autonomous Prefecture of Xishuangbanna, Dai Autonomous Prefecture of Xishuangbanna with highlighted research area (Global Administrative Area, http://www.gadm.org/).

The habitat quality model requires a current land cover which was available from the Project Living Landscapes China (Lilac) from 2007 (Wehner et al. 2014). As future land covers we used the two scenarios Business as usual (Gibreel et al. 2014) and Go Green (Cotter et al. 2014), both calculated for the year 2025, which were also available from Lilac (Figure 3 and Figure 4). For all three maps, land-use classes were simplified by reclassification to fit the current research question (Figure 2-4).

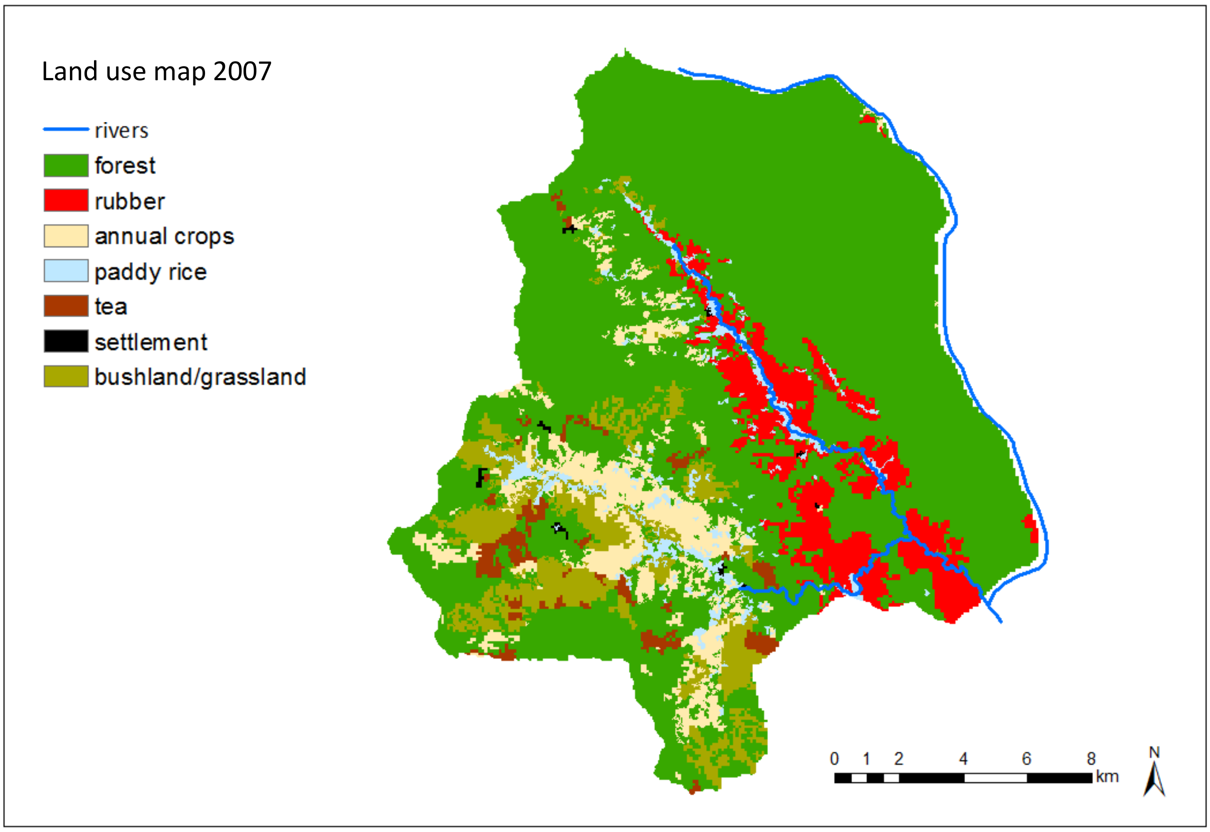

Figure 2 Current land-use map

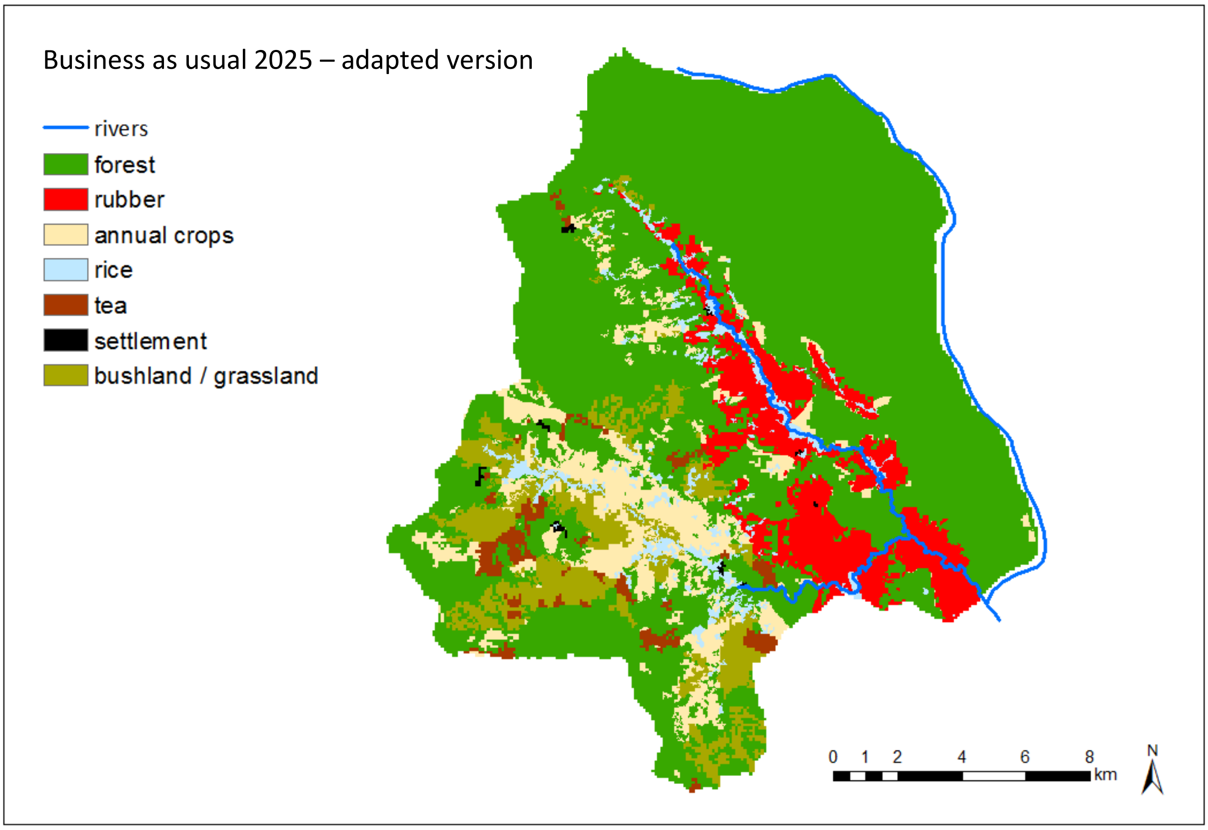

Figure 3 Scenario 1, Business as usual

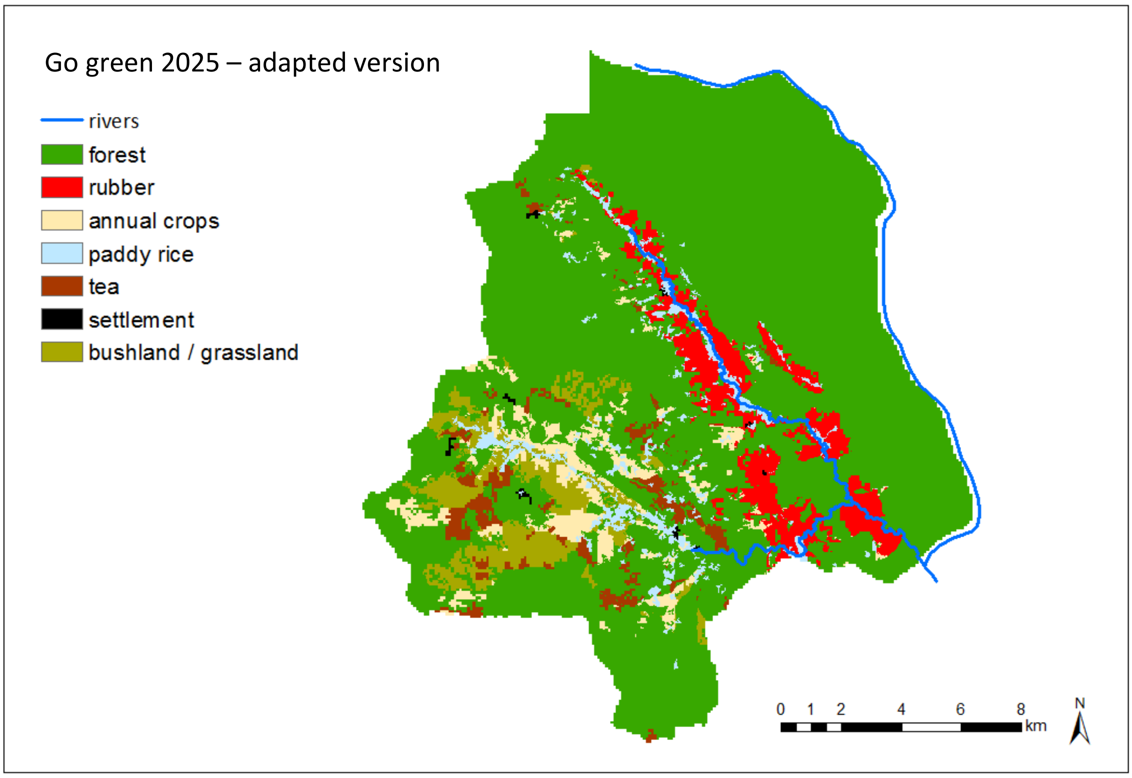

Figure 4 Scenario 2, Go Green

Accessibility

Landscapes that are protected either legally or by geographic features can be included in the modeling process. Since the NRWNNR is managed according to the Man and Biosphere (MAB) Programme, the area is divided into a core zone, where all access is prohibited (input value 0), a buffer zone with limited access (input value 0.5) and an experimental zone with no restrictions (input value 1) (Figure 5) (modified from Naban River Watershed National Nature Reserve Bureau 2014).

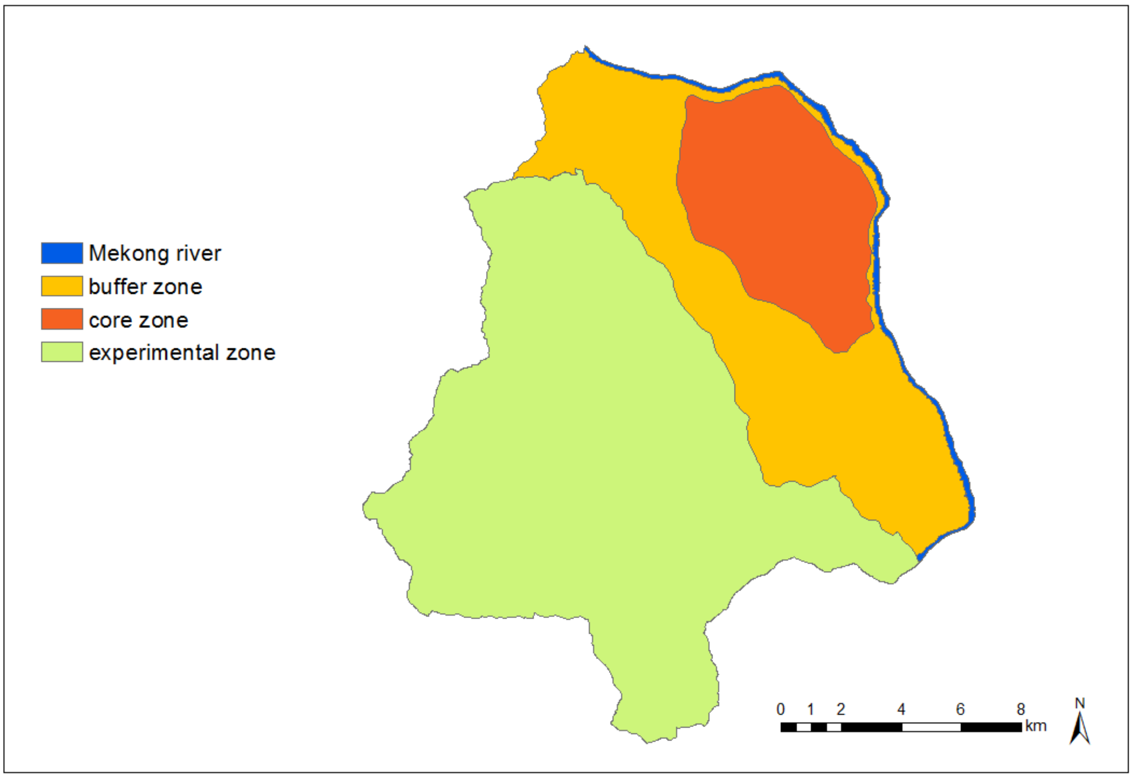

Figure 5 Delineation of the different MAB zones in the research area

Habitat scores for land-use classes

There are basically two options in InVest to model habitat quality. The first one is a simple consideration of which land-use types are habitat or non-habitat in general, irrespective of different species. The underlying assumption is an island-ocean model, therefore, all natural habitats are considered as habitat and all managed land-use classes as non-habitat. The second one is an assignment of relative habitat suitability scores to each land-use. The underlying assumption is that species also "use" managed landscapes, although these might not be suited to provide all necessary functions (feed, suitability for reproduction) to allow occupation but they might be used temporarily. Since different species use different land-use types as "intermediate" habitat, this approach is generally not meant for biodiversity in general, but for single species. We used a third approach for our analysis. From personal work, detailed data on presence/absence of different species from different groups (plants, vertebrates, invertebrates) in different land-use classes are available. From these absolute data we derived relative habitat suitability scores for respective species groups. Thereby we are able to show how important different land-use types are for different groups.

Since we did not have complete data for all land-use classes (neither for our own data, nor from literature), we had to complete the missing data with values of similar land-use classes (Table 1).

Table 1 Complete table with input habitat scores

land-use class	red list habitat score	all species habitat score	human use habitat score
forest	1	1	1
rubber	0.36	0.57	0.82
annual crops	0.18	0.33	0.65
paddy rice	0.16	0.26	0.51
tea	0.12	0.28	0.49
settlement	0.1	0.1	0.1
bushland / grassland	0.32	0.38	0.56

Threats

Edge and fragmentation effects in InVEST are included by the inclusion of threats that can degrade a neighboring habitat. We included four threats into our analysis: three land-use classes (rubber, annual crops and settlements) and roads. Threat data for the three land-use class were extracted from the respective maps. The road map used includes major roads in the area (Figure 6) (modified from Naban River Watershed National Nature Reserve Bureau 2014).

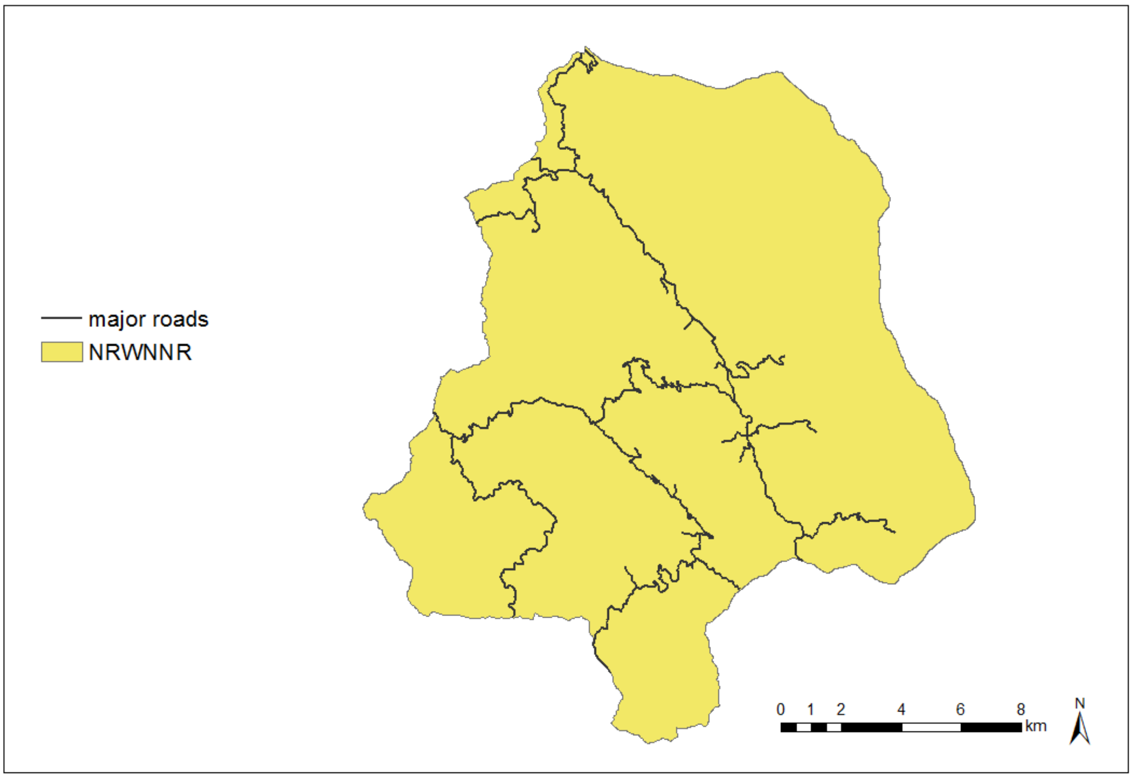

Figure 6 Roadmap of the Naban River Watershed National Nature Reserve (NRWNNR)

For all threats three variables need to be defined: the maximum distance to what the threat has an effect (in km) (dist), if the threat effect decreases linearly or exponentially (decay) and how strong the effect is in relation to other threats (weight). Values for these variables were defined separately for the different species groups flora, vertebrates and invertebrates by expert panels. Values were averaged for the different species groups analyzed, all values for the aggregated biodiversity index (all species), the values of flora and vertebrates, for the red list species (red list) and the values of flora and invertebrates for the species with direct value for humans (human use). Table 2 shows the final input values for the different model runs.

Table 2 Values for threat data

	red list			all			human use		
Threat	**dist**	**decay**	**weight**	**dist**	**decay**	**weight**	**dist**	**decay**	**weight**
rubber	0.1	expon	0.35	0.1	expon	0.27	0.1	expon	0.3
annual crops	0.1	expon	0.3	0.1	expon	0.3	0.1	expon	0.3
settlements	1	expon	1	1	expon	1	1	expon	1
roads	0.1	expon	0.7	0.1	expon	0.5	0.1	expon	0.45

Sensitivity of habitats

The last requested input data are values on how sensitive the different habitats are to the threats rubber (rub), annual crops (ancr), settlements (settl) and roads (road). These values were also derived from expert panel meetings for the species groups flora, vertebrates and invertebrates and averaged (same procedure as for threat data) for the analyzed groups red list, all species and human use (Table 3). Values close to 1 represent high sensitivity to a threat, whereas a value of 0 represents full insensitivity.

Table 3 Values for sensitivity data

	red list				all species				human use			
land-use class	**rub**	**ancr**	**settl**	**road**	**rub**	**ancr**	**settl**	**road**	**rub**	**ancr**	**settl**	**road**
forest	0.65	0.85	1	0.95	0.7	0.6	1	0.8	0.65	0.45	1	0.75
rubber	0	0.15	0.85	0.7	0	0.1	0.87	0.63	0	0.15	0.85	0.65
annual crops	0	0	0.4	0.4	0.1	0	0.5	0.33	0.15	0	0.35	0.2
paddy rice	0	0.05	0.4	0.4	0	0.07	0.47	0.33	0	0.05	0.3	0.2
tea	0	0	0.35	0.2	0	0	0.27	0.17	0	0	0.05	0.05
settlement	0	0	0	0	0	0	0	0	0	0	0	0
bushland grassland	0.3	0.5	0.65	0.5	0.23	0.33	0.5	0.4	0.05	0.15	0.25	0.25

References

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Gibreel TM, Herrmann S, Berkhoff K, Nuppenau E, Rinn A (2014). Farm types as an interface between an agroeconomical model and CLUE-naban land change model: Application for scenario modelling. Ecological Indicators 36:766-778 DOI 10.1016/j.ecolind.2013.09.009

Naban River Watershed National Nature Reserve Bureau (ed.) (2014). Naban river watershed national nature reserve (NRWNNR) - atlas of ecological baseline results. Naban River Watershed National Nature Reserve Bureau, Jinghong

Wehner S, Herrmann S, Berkhoff K (2014). CLUENaban-A land-use change model combining social factors with physical landscape factors for a mountainous area in southwest china. Ecological Indicators 36:757-765 DOI 10.1016/j.ecolind.2013.09.027

7.4 Investigating potential transferability of place-based research in land system science, Supplementary material

Table 1 Datasets used for classification of land system archetypes

Archetype factor	Spatial resolution	Unit	Source
Land-use intensity factors			
Cropland area	5 arc-minutes	km^2 per grid cell	(Klein Goldewijk et al 2011)
Cropland area trend	5 arc-minutes	km^2 per grid cell	(Klein Goldewijk et al 2011)
Pasture area	5 arc-minutes	km^2 per grid cell	(Klein Goldewijk et al 2011)
Pasture area trend	5 arc-minutes	km^2 per grid cell	(Klein Goldewijk et al 2011)
N fertilizer	0.5 arc-degrees	kg ha^{-1}	(Potter et al 2010)
Irrigation	5 arc-minutes	Ha per grid cell	(Siebert et al 2007)
Soil erosion	5 arc-minutes	Mg ha^{-1} year^{-1}	(Van Oost et al 2007)
Yields (wheat, maize, rice)	5 arc-minutes	t ha^{-1} year^{-1}	http://www.gaez.iiasa.ac.at/
Yield gaps (wheat, maize, ric	5 arc-minutes	1000 t	http://www.gaez.iiasa.ac.at/
Total production index	national level	index	http://faostat.fao.org/
HANPP	5 arc-minutes	% of NPP0	(Haberl et al 2007)
Environmental factors			
Temperature	5 arc-minutes	°C × 10	(Kriticos et al 2012)
Diurnal temperature range	5 arc-minutes	°C × 10	(Kriticos et al 2012)
Precipitation	5 arc-minutes	mm	(Kriticos et al 2012)
Precipitation seasonality	5 arc-minutes	coeff. of variation	(Kriticos et al 2012)
Solar radiation	5 arc-minutes	W m^{-2}	(Kriticos et al 2012)
Climate anomalies	5 arc-degrees	°C × 10	http://www.ncdc.noaa.gov/cmb-faq/anomalies.php#grid
NDVI – mean	4.36 arc-minutes	index	(Tucker et al 2005)
NDVI – seasonality	4.36 arc-minutes	index	(Tucker et al 2005)
Soil organic carbon	5 arc-minutes	g C kg^{-1} of soil	(Batjes 2006)
Vertebrate species richness	calculated from range polygons	# of species per grid cell	http://www.iucnredlist.org/technical-documents/spatial-data
Socioeconomic factors			
Gross Domestic Product	national level	$ per capita	http://faostat.fao.org/
Gross Domestic Product in ag	national level	% of GDP	http://faostat.fao.org/
Capital Stock in agriculture	national level	$	http://faostat.fao.org/
Population density	2.5 arc-minutes	persons km-2	(CIESIN 2005)
Population density trend	2.5 arc-minutes	persons km-2	(CIESIN 2005)
Political stability	national level	index	http://www.govindicators.org
Accessibility	0.5 arc-minutes	minutes of travel time	http://bioval.jrc.ec.europa.eu/products/gam/index.htm

Table 2 Correlation matrix of input variables calculated from a random sample of 10 000 points. Bolding denotes significant correlations at the 0.05 level. Shading denotes correlations above 0.5 or below -0.5.

Variables	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27	28	29	30	31	32
1 Cropland area	1.00																															
2 Crop. area trend	0.31	1.00																														
3 Pasture area	**0.01**	0.09	1.00																													
4 Pasture area trend	0.05	**0.63**	0.42	1.00																												
5 N fertilizer	**0.54**	0.05	-0.02	0.03	1.00																											
6 Irrigation	0.42	**-0.01**	**0.00**	0.05	0.48	1.00																										
7 Soil erosion	0.17	0.11	0.06	0.04	0.06	**0.02**	1.00																									
8 Yield wheat	0.42	0.02	0.02	0.04	**0.55**	0.29	0.04	1.00																								
9 Yield rice	0.35	0.05	0.04	0.09	0.38	0.40	0.10	0.18	1.00																							
10 Yield maize	0.45	0.04	0.07	0.06	0.47	0.29	0.07	**0.59**	0.36	1.00																						
11 Yield gap wheat	0.48	**0.04**	**0.00**	0.03	0.45	**0.57**	0.03	0.43	0.19	0.34	1.00																					
12 Yield gap rice	0.34	0.05	-0.05	**0.01**	0.28	0.48	0.06	0.04	0.32	0.09	0.08	1.00																				
13 Yield gap maize	0.45	0.09	0.03	0.05	0.39	0.28	0.08	0.27	0.22	0.41	0.30	0.13	1.00																			
14 TPI	0.04	0.13	0.16	0.22	**0.01**	0.04	0.05	-0.05	0.12	**0.00**	**0.01**	0.07	**0.01**	1.00																		
15 HANPP	**0.68**	0.16	0.09	0.07	0.47	0.24	0.14	0.41	0.31	0.42	0.33	0.25	0.32	0.04	1.00																	
16 Temperature	0.28	0.13	0.38	0.17	0.13	0.14	0.11	0.07	0.27	0.21	0.08	0.15	0.14	0.16	0.25	1.00																
17 Diur. temp. range	0.03	0.12	0.44	0.19	-0.06	0.03	-0.02	-0.03	-0.06	0.03	**0.01**	-0.04	0.05	0.20	**0.01**	**0.51**	1.00															
18 Precipitation	0.16	0.06	-0.08	0.02	0.10	0.04	0.23	**0.02**	0.34	0.16	-0.03	0.18	0.10	0.07	0.20	0.38	-0.25	1.00														
19 Precip. seasonal.	0.04	0.10	0.25	0.16	**-0.01**	0.10	**-0.01**	-0.16	0.03	-0.09	**-0.02**	0.08	0.05	0.23	**-0.01**	0.37	**0.54**	-0.21	1.00													
20 Solar radiation	0.14	0.07	0.32	0.12	0.07	0.13	-0.03	0.13	-0.05	0.07	0.18	0.02	**0.02**	0.06	0.10	0.40	**0.57**	-0.39	0.36	1.00												
21 Climate anomalies	0.10	**0.02**	0.09	0.04	0.25	0.09	**0.00**	0.20	0.13	0.13	0.12	**0.02**	0.07	0.04	0.12	-0.07	0.14	-0.28	0.05	0.19	1.00											
22 NDVI-mean	0.26	0.14	0.04	0.11	0.17	0.06	0.15	0.24	0.28	0.32	0.08	0.10	0.17	0.10	0.33	0.46	0.02	**0.67**	-0.17	-0.21	-0.10	1.00										
23 NDVI-seasonality	**0.01**	0.06	-0.29	**-0.02**	**0.00**	-0.04	**-0.01**	0.03	-0.04	**-0.01**	**0.01**	-0.03	0.02	**0.02**	0.05	-0.34	-0.25	0.12	-0.25	-0.48	-0.09	0.34	1.00									
24 Soil org. C	-0.05	0.02	-0.20	-0.02	-0.02	-0.04	**-0.02**	**0.00**	-0.04	-0.04	-0.03	**0.00**	-0.03	-0.03	-0.02	-0.25	-0.21	0.02	-0.18	-0.25	-0.02	0.09	0.39	1.00								
25 Species richness	0.24	0.14	0.14	0.14	0.09	0.07	0.17	0.04	0.32	0.22	0.02	0.13	0.15	0.22	0.25	**0.62**	0.11	**0.65**	**0.02**	-0.17	-0.23	**0.65**	0.07	-0.07	1.00							
26 GDP	-0.06	-0.07	-0.12	-0.15	0.05	-0.07	-0.07	0.18	-0.14	0.14	-0.05	-0.09	**-0.01**	-0.30	-0.03	-0.28	-0.16	-0.09	-0.36	**-0.01**	0.13	-0.08	**-0.01**	0.07	-0.26	1.00						
27 GDP in agr.	0.14	0.09	0.20	0.10	**-0.01**	0.07	0.08	-0.10	0.10	-0.03	0.04	0.10	0.08	0.28	0.11	0.45	0.29	0.15	0.43	0.14	-0.13	0.15	-0.15	-0.13	0.28	**-0.51**	1.00					
28 Cap. stock in agr.	0.13	-0.04	-0.02	**0.00**	0.20	0.09	-0.02	0.19	0.10	0.30	0.07	**0.00**	0.14	0.03	0.15	-0.13	0.10	**-0.01**	-0.09	-0.08	0.14	0.10	0.19	0.06	0.02	0.43	-0.27	1.00				
29 Pop. density	0.29	-0.05	-0.02	0.04	0.38	0.35	0.07	0.23	0.28	0.20	0.27	0.24	0.18	0.02	0.28	0.13	-0.04	0.11	0.04	0.03	0.08	0.11	-0.03	-0.03	0.11	-0.07	0.07	0.06	1.00			
30 Pop. density trend	0.35	-0.03	-0.01	0.05	0.43	0.41	0.09	0.19	0.33	0.18	0.28	0.34	0.22	0.04	0.30	0.16	-0.02	0.14	0.08	0.03	0.08	0.11	-0.04	-0.03	0.13	-0.11	0.11	0.07	0.68	1.00		
31 Political stability	-0.14	-0.07	**-0.10**	-0.09	-0.01	-0.11	-0.07	0.08	-0.10	0.02	-0.07	-0.11	-0.05	-0.17	-0.13	-0.37	-0.32	-0.08	-0.39	-0.11	0.12	-0.12	-0.05	0.05	-0.22	**0.67**	**-0.57**	0.08	-0.09	-0.13	1.00	
32 Accessibility	-0.17	0.02	-0.14	**0.01**	-0.11	-0.08	-0.04	-0.14	-0.11	-0.15	-0.09	-0.06	-0.09	**-0.02**	-0.19	-0.37	-0.28	-0.06	-0.12	-0.18	-0.07	-0.30	-0.12	-0.03	-0.21	0.14	-0.14	-0.09	-0.08	-0.09	0.24	1.00

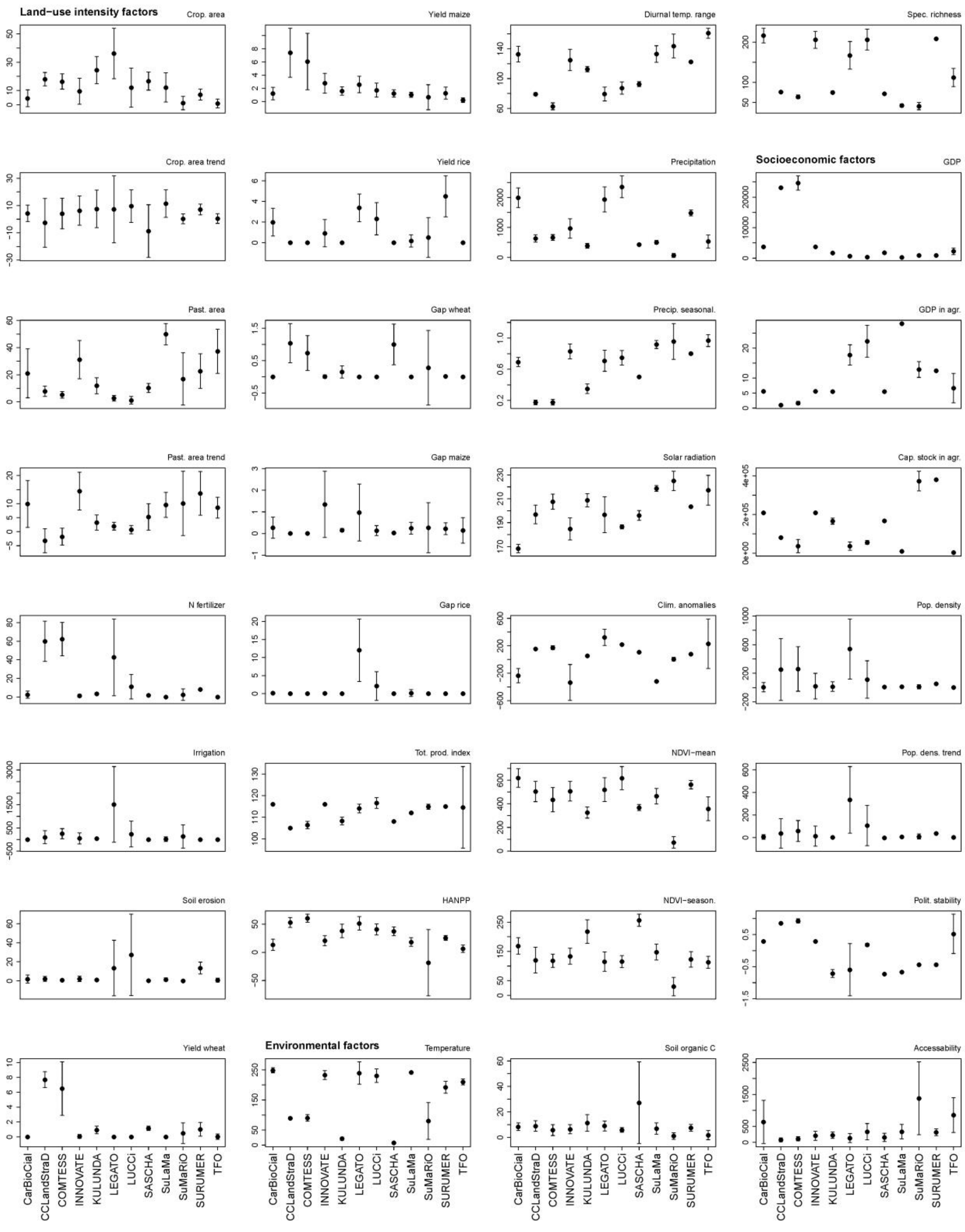

Figure 1 Comparison of land-use intensity indicators, environmental conditions and socioeconomic factors that characterize land systems in each regional project. Dots represent mean values; whiskers represent standard deviations. For variable units, see Table 1.

Summary

The present dissertation analyzes the impact of rubber tree (*Hevea brasiliensis*) dominated land-use on biodiversity and ecosystem services in the Greater Mekong Subregion (GMS). Although originating from South-America natural rubber is mainly cultivated in South-East Asia. This region is now one of the most important producers of natural rubber. Traditionally, rubber was cultivated between 10° South and 10° North, where rainfall is continuous the whole year round. But in the last decades the production area shifted as far as 22° North, hence expanding further into the GMS. This expansion of rubber plantations affects different ecosystem services such as carbon storage, availability and quality of water and threatens biodiversity in this highly biodiverse region.

In order to analyze these impacts the first task was to give a comprehensive overview about publications concerning ecosystem services and biodiversity in rubber cultivation systems. A thorough literature review showed that the majority of publications concentrated on single or few ecosystem services, which does not match the demands of decision-makers. In order to make sound decisions for land-use planning or developing Payments for Ecosystem Services Schemes a holistic view including multiple services that ecosystems provide to humans is needed. Leaving the disciplinary path and developing interdisciplinary studies remains to be the main challenge which requires joint research that need sufficient funding to fulfill the required tasks.

In order to fill this knowledge gap, the second task was to analyze the impacts of rubber on the ecosystem services carbon storage, soil erosion, water availability and water quality and economically and socially related ecosystem services, such as income and livelihood security. Although there are still great uncertainties about carbon storage in different land-use systems there are hints that the carbon storage in rubber plantations is lower than in natural forests, which are the original vegetation in the GMS when not only considering carbon stored in aboveground biomass (including latex), but also changes in soil carbon pools and soil microbial biomass carbon. Concerning erodibility, rubber plantations increase the soil erosion risk compared to natural forests (magnitude depends on management options such as weeding and terracing). The hydrological cycle is also affected by rubber production. After its large-scale introduction in Xishuangbanna, Yunnan Province, one of the main rubber producing areas in China, the average air temperature increased significantly, the number of fog days decreased and there are reports about reduced streamflow and dried up wells. Villagers report a drop in groundwater level which is causing potable water shortages. All of this suggests that rubber cultivation affects the local and regional water balance. Since rubber production in monocultures requires the use of huge amounts of pesticides and chemical fertilizers which enter the aquatic system by rainfall-induced run-off, water quality for humans and aquatic organisms is as well threatened. Undoubtedly, from an economic point of view the introduction of rubber resulted in significant increases in household income and is hence a possibility to move households and communities out of poverty. However, by deciding to grow rubber, farmers are committing themselves for decades to come and are thus dependent on a single product, which exposes them to further risks. Rubber is almost exclusively used for industrial purposes, so its demand strongly depends on the dynamics of the world economy. Boom-and-bust cycles in rubber prices expose farmers to income insecurity. In comparison to annual crops,

farmers are less able to react with a short-term production strategy on changing market situations with a rubber plantation (other than deciding not to tap when prices are too low, thereby saving tapping costs). In addition, there are ecological hazards due to crop diseases, pests, unfavorable weather conditions or changes in climate. The abandonment of traditional land-use practices in favor of a single crop may as well have severe implications for food and nutritional security of the rural population. The review of studies clearly indicates that increasing rubber cultivation in the GMS is accompanied by various problems and threats to farmers and the environment. Therefore, the development of more sustainable land-use concepts is required. Concordantly, suggestions for land-use change are based on system diversification and forest restoration, which both require economic incentives for farmers. Although rubber prices presently show a downward tendency, the main obstacle to change is still the high economic attractiveness of rubber production coupled with too few alternative income sources. Beyond that, farmers need more education on economic risks of rubber monoculture production and on its ecological consequences for the environment and their livelihood.

The next part of this dissertation deals with the development of a biodiversity indicator (based on selected flora and fauna species) that can be included in existing biodiversity models to show the impact that different rubber cultivation scenarios will have on various species groups. A combination of multi-species data supplemented with literature data (allowing for a broader assessment of the impacts of land-use decisions on biodiversity) was used and included flora (vascular plants), vertebrates (mammals and birds) and invertebrates (wild bees, hoverflies, beetles, soil nematodes and spiders). The resulting indicator was used as input into an established ecosystem service assessment framework (Integrated Valuation of Ecosystem Services and Tradeoffs – InVEST). Finally, a current land-use map from 2007 was compared with a go-green-scenario (reduced area for rubber cultivation) and a business-as-usual scenario (increased area for rubber cultivation). The habitat suitability for the species groups studied changed over the course of the scenarios as expected; the go-green scenario increased overall biodiversity whereas the business-as-usual scenario decreased overall biodiversity. In order to get a more detailed picture, sub-groups of biodiversity were analyzed as well. First, a sub-group of "human use species" category was assessed which included only species which are directly used by humans (wild bees for honey production and plants for Traditional Chinese Medicine), to include the anthropocentric point of view. Second, the sub-group "red list species", consisting of red list plants and mammals was analyzed separately to include the conservational aspect. The results showed that when considering the indices from the category "human use species" there is less of an impact between the scenarios, because medicinal plants for example are still commonly distributed within rubber plantations, albeit with different species (generalists, ruderal flora). Wild bee species are also present both with forest dwelling species and species adapted to open-land or successional stages found in young rubber. This leads to an important conclusion: not only the decision on "what" is analyzed (biodiversity), but also "how" it is analyzed (groups of different species) considerably influences the results. From this point of view, sub-assessments like "red list species" or "endemics" are important to include in order to cover not just "quantity", but also "quality" of species diversity.

To complete this dissertation the transferability of this place-based research to other regions of the world was tested to assess the relevance beyond the study area. To do this the system of land archetypes was used, which utilizes a wide range of land-use intensity metrics and incorporates simultaneously environmental and socio-economic conditions. In a first step, the 'project archetype' was defined, based on a synthesis of global land system indicators. For the project Sustainable Rubber Cultivation in the Mekong Region (SURUMER), only few of the 32 global indicators did not closely match the local reality. The transferability potential for the GMS was quite high and therefore emphasizes the importance of the project results for this huge region. Other areas with high scores of statistical similarity where the transferability potential is assumed to be high were identified in Africa (where rubber production is also carried out) and in South-America where large-scale rubber production is limited by the South-American leaf blight disease, caused by *Pseudocercospora ulei*.

The presented results were embedded into a wider context and discussed concerning different weaknesses of the used methods (land-use based proxies, independent models) and the necessity of trade-off analyses which includes a discussion about monetary valuation of ecosystem services. There are still many open questions and fields that need further research. It will still take more time and more effort until a profound understanding is gained and until science is able to provide the necessary data for decision-making, a precondition for integrating conservation into planning and resource management.

Zusammenfassung

Die vorliegende Dissertation analysiert den Einfluss von steigendem Kautschukanbau (*Hevea brasiliensis*) hinsichtlich Biodiversität und Ökosystemdienstleistungen in der Greater Mekong Subregion (GMS, Wassereinzugsgebiet des Mekong, umfasst die Länder China (Provinz Yunnan und die Autonome Region von Guangxi Zhuang), Kambodscha, Laos, Myanmar, Thailand und Vietnam). Obwohl der Kautschukbaum ursprünglich aus Südamerika stammt wird er heute hauptsächlich in Süd-Ost Asien angebaut. Traditionell wurde Kautschuk in der Region zwischen dem 10° Südlicher Breite und dem 10° Nördlicher Breite angebaut, in der das ganze Jahr über kontinuierlich Regen fällt. In den letzten Jahrzehnten verschob sich der Anbau allerdings bis zum 22° Nördlicher Breite und hielt somit weiten Einzug in die GMS. Die Ausbreitung der Kautschukplantagen beeinflusst die Bereitstellung verschiedener Ökosystemdienstleistungen wie Kohlenstoffspeicherung, Wasserverfügbarkeit und –qualität und beeinträchtigt die Biodiversität dieser Region.

Um diese Einflüsse zu untersuchen war die erste Aufgabe die Erstellung eines umfassenden Überblicks über Veröffentlichungen bezüglich Ökosystemdienstleistungen und Biodiversität in Kautschuk-Anbaugebieten. Eine umfassende Literaturübersicht zeigte, dass die Mehrzahl der Publikationen nur eine einzige oder sehr wenige Ökosystemdienstleistungen behandelt. Dies entspricht nicht den Anforderungen von Entscheidungsträgern, die für fundierte Entscheidungen hinsichtlich Landnutzungsplanung oder die Entwicklung von Zahlungen für Ökosystemdienstleistungen (Payments for Ecosystem Services, PES) ein holistisches Bild brauchen, das mehrere Ökosystemdienstleistungen simultan betrachtet, die von betroffenen Ökosystemen bereit gestellt werden. Den disziplinären Weg zu verlassen und interdisziplinäre Studien zu entwickeln bleibt die größte Herausforderung für die Wissenschaft. Um diese Aufgabe erfüllen zu können sind Verbundprojekte mit ausreichender Finanzierung und Laufzeit notwendig.

Um einen Beitrag zu einer mehr holistischen Sichtweise zu leisten, war die zweite Aufgabe die Analyse des Einflusses von Kautschuk auf die Ökosystemdienstleistungen Kohlenstoffspeicherung, Bodenerosion, Wasserverfügbarkeit und -qualität, sowie die ökonomisch und sozial verwandten Ökosystemdienstleistungen wie Einkommen und Existenzsicherung. Obwohl es bezüglich der Kohlenstoffbindung in verschiedenen Landnutzungsformen noch große Unsicherheiten gibt, verdichten sich die Hinweise darauf, dass in Kautschukplantagen weniger Kohlenstoff gespeichert ist, als in natürlichen Wäldern (der ursprünglichen Vegetation in der GMS), wenn nicht nur der Kohlenstoff betrachtet wird, der in oberirdischer Biomasse (inklusive Latex) gespeichert ist, sondern auch Änderungen in abiotischen und biotischen gebundenen Kohlenstoff im Boden mit einbezogen werden. Das Risiko von Erodibilität von Böden in Kautschukanlagen ist im Vergleich zu natürlichen Wäldern höher (das Ausmaß wird dabei u.a. von Managementmaßnahmen wie Art der Unkrautbekämpfung und Terrassierung bestimmt). Der Wasserkreislauf wird von Kautschukplantagen ebenfalls beeinflusst. Nach der großflächigen Einführung von Kautschuk in Xishuangbanna, Provinz Yunnan, eines der wichtigsten Anbaugebiete in China, stieg die mittlere Lufttemperatur signifikant, die Anzahl der Nebeltage sank und es wird von geringerem Oberflächenabfluss der Flüsse und ausgetrockneten Brunnen berichtet. Dorfbewohner bezeugen außerdem einen sinkenden Grundwasserspiegel, der zu Trinkwasserknappheit führt. All dies legt nahe, dass der Kautschukanbau den lokalen und

regionalen Wasserhaushalt beeinflusst. Die Wasserqualität für Menschen und wasserlebende Organismen ist durch die Kautschukproduktion ebenfalls beeinträchtigt, da diese den Einsatz von großen Mengen an Pestiziden und synthetischen Düngern erfordert, die durch Regen induzierte Auswaschung in das aquatische System gelangen. Zweifelsfrei hat die Einführung von Kautschuk eine signifikante Steigerung des Farmeinkommens bewirkt. Deshalb bietet der Kautschukanbau die Möglichkeit, einzelne Haushalte und ganze Gemeinden aus der Armut zu führen. Allerdings legen sich die Bauern mit der Kautschukproduktion auf Jahrzehnte fest und sind somit abhängig von einer einzelnen Kultur, was ein erhöhtes Risiko bedeutet. Naturkautschuk wird fast ausschließlich für industrielle Zwecke verwendet, so dass die Nachfrage stark von der Dynamik der globalen Wirtschaft abhängt. Schwankende Konjunkturzyklen führen bei den Bauern zu Einkommensunsicherheiten. Im Vergleich zu einjährigen Kulturpflanzen können die Bauern weniger flexibel mit kurzfristigen Produktionsänderungen auf sich ändernde Marktsituationen reagieren (außer der Entscheidung keinen Latex zu zapfen und somit Arbeitskosten zu sparen, wenn der Preis sehr tief fällt). Weitere ökologische Gefahren bestehen durch Pflanzenkrankheiten und –schädlinge, ungünstige Witterung oder Klimaveränderung. Die Aufgabe von traditionellen Landnutzungsformen zu Gunsten einer einzigen Kulturpflanze kann zusätzlich ernste Folgen für die Ernährungssicherung der Landbevölkerung haben. Die Auswertung der Studien zeigte deutlich, dass der zunehmende Kautschukanbau in der GMS von verschiedenen Problemen und Bedrohungen für die Bauern und die Umwelt begleitet wird. Die Entwicklung von nachhaltigen Landnutzungskonzepten ist daher unabdingbar. Aufgrund der beschriebenen Ergebnisse basieren Vorschläge für eine nachhaltigere Landnutzung auf Diversifizierung der Plantagen und die Wiederaufforstungen von Wäldern, beides erfordert aber ökonomische Anreize für die Bauern. Obwohl die Kautschukpreise im Moment in einem Abwärtstrend sind ist die hohe ökonomische Attraktivität von Kautschuk, gekoppelt mit den wenigen alternativen Einkommensmöglichkeiten, immer noch das größte Hindernis für einen Wandel. Darüber hinaus sollten die Bauern hinsichtlich ökonomischer Risiken von Kautschukmonokulturen und deren ökologischen Konsequenzen für die Umwelt und ihre Lebensgrundlage geschult werden.

Ein weiterer Teil der Dissertation behandelt die Entwicklung eines Biodiversitäts-Indikators (basierend auf ausgewählten Arten von Flora und Fauna), der in bestehenden Biodiversitäts-modellen verwendet werden kann. Der Indikator soll dazu dienen den Einfluss von verschiedenen Kautschuk dominierten Landnutzungsintensitäten auf unterschiedliche Gruppen von Arten aufzuzeigen. Eine Kombination aus eigenen Daten basierend auf mehreren Arten wurde mit Literaturdaten vervollständigt (um eine umfassende Abschätzung des Einflusses von Landnutzungsentscheidungen auf Biodiversität zu erlauben). Der Datensatz umfasste Flora (Gefäßpflanzen), Vertebraten (Säuger und Vögel) und Invertebraten (Wildbienen, Schwebfliegen, Käfer, Bodennematoden und Spinnen). Der resultierende Indikator wurde als Eingabe in ein bestehendes Modell zur Bewertung von Ökosystemdienstleistungen verwendet (Integrated Valuation of Ecosystem Services and Tradeoffs – InVEST). Abschließend wurde eine Landnutzungskarte aus dem Jahr 2007 mit zwei Szenarien verglichen, einem „go-green“ Szenario (reduzierte Fläche für Kautschukanbau) und ein „business-as-usual“ Szenario (vergrößerte Fläche für Kautschukanbau). Die Habitateignung für die untersuchten Artengruppen änderte sich in den Szenarien wie erwartet, das „go-green“ Szenario erhöhte die allgemeine Biodiversität, wohingegen das „business-as-usual“ die allgemeine Biodiversität verringerte. Um ein differenzierteres Bild zu erhalten wurden zusätzlich Untergruppen analysiert. Die erste

Untergruppe „Arten für den menschlichen Gebrauch“ beinhaltete nur die Arten, die der Mensch direkt nutzt (Wildbienen für die Honigproduktion und Pflanzen für Traditionelle Chinesische Medizin), um einen anthropozentrischen Standpunkt mit einzubeziehen. Die zweite Untergruppe „Rote Liste Arten“, bestehend aus Pflanzen und Säuger die auf der roten Liste verzeichnet sind, wurde getrennt untersucht, um Naturschutzaspekte mit einzubeziehen. Die Ergebnisse zeigten, dass es bei einer Beschränkung auf die Arten „für den menschlichen Gebrauch“ weniger Unterschiede zwischen den Szenarien gibt, da die Medizinalpflanzen auch in Kautschukplantagen vorkommen, wenn auch mit anderen Arten (Generalisten und Ruderalpflanzen). Wildbienenarten waren sowohl mit waldlebenden Arten, als auch mit Offenlandarten vertreten, die auch in frühen Sukzessionsstadien im Kautschuk leben können. Dies führt zu einer wichtigen Schlussfolgerung: Allein die Entscheidung „Was“ untersucht wird ist nicht ausreichend (Biodiversität), sondern auch das „Wie“ (Gruppierung von verschiedenen Artgruppen) ist wichtig, da es erhebliche Auswirkungen auf die Ergebnisse hat. Aus diesem Blickwinkel ist die weiterführende Untersuchung von Untergruppen wie „Rote Liste Arten“ oder „endemische Arten“ wichtig, um nicht nur die „Quantität“, sondern auch die „Qualität“ von Artenvielfalt zu erfassen.

In einem letzten Aspekt wurde die Übertragbarkeit der Projektergebnisse auf andere Regionen der Welt untersucht, um die Relevanz des Kautschukanbaus auch für Gebiete außerhalb des eigentlichen Untersuchungsgebietes zu ermitteln. Dafür wurde das System der „Land-Archetypen“ benutzt, das eine große Auswahl von Landnutzungsintensitäten verwendet und gleichzeitig ökologische und sozio-ökonomische Gegebenheiten mit einbezieht. In einem ersten Schritt wurde der Projekt-Archetyp definiert, basierend auf der Zusammenführung von globalen Indikatoren mit deren Hilfe Landnutzung und Landeigenschaften beschrieben werden können. Für das Verbundprojekt Nachhaltiger Kautschukanbau in der Mekong-Region (Sustainable Rubber Cultivation in the Mekong Region (SURUMER) wichen nur wenige der 32 untersuchten globalen Indikatoren von den realen lokalen Werten ab. Das Übertragbarkeitspotenzial für die GMS war relativ hoch und unterstützt damit die Wichtigkeit der Projektergebnisse für diese große Region. Andere Gebiete mit hohen Werten für statistische Ähnlichkeit, in denen das Übertragbarkeitspotenzial als hoch angesehen wird, wurden in Afrika identifiziert (wo Kautschukanbau ebenfalls praktiziert wird) und in Süd-Amerika, wo großflächiger Kautschukanbau allerdings durch die Südamerikanische Blattfallkrankheit beschränkt wird, die durch den Pilz *Pseudocercospora ulei* hervor gerufen wird.

Die in der Dissertation vorgestellten Ergebnisse wurden in einen größeren Kontext eingebettet und bezüglich verschiedener Schwachpunkte der verwendeten Methoden (landnutzungsbasierte Proxys, unabhängige Modelle) diskutiert. Die Notwendigkeit einer Zielkonflikt-Analyse wurde hervorgehoben, die auch die Diskussion um monetäre Bewertung von Ökosystemdienstleistungen beinhaltet. Es bleiben viele offene Fragen und Bereiche, die näher untersucht werden müssten. Bis ein tiefgreifendes Verständnis gewonnen wird und bis die Wissenschaft in der Lage sein wird die notwendigen Daten für Entscheidungsfindungen zu liefern, die die Voraussetzung für die Integration von Natur- und Umweltschutz in Planung und Ressourcenmanagement ist, werden noch mehr Zeit, Kosten und Mühen notwendig sein.

Curriculum vitae

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