Impact of land use and land cover changes on carbon storage in rubber dominated tropical Xishuangbanna, South West China

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ABSTRACT
Land use and land cover (LULC) play a significant role in carbon regulation. South-China accounts for ~65% of China’s carbon sink. In Xishuangbanna (South-China), rubber is expanding rapidly creating an urgent need to understand and monitor LULC change and how spatial variation affects carbon storage (CS). This is vital for the formation and implementation of better land use management practices. We studied LULC changes of 22-year period; addressing how these changes have affected the CS. We quantified LULC changes between 1988 and 2010 using remote sensing methods and calculated CS changes using InvES.T. Results showed that between 1988 and 2010, the rate of deforestation accelerated to 203.2 km² yr⁻¹ and ~23% of forest were lost. Conversion of natural forest to rubber was responsible for 78% of this deforestation. Rubber expansion rate was 153.4 km² yr⁻¹. Changes to LULC drove a temporal CS reduction 0.223 Tg C/km². Local stakeholders have strong economic interest in converting land to more profitable plantations. Government efforts is required to control land conversion through new policies and incentives to retain natural forest. Assessment of specific potential land use change will be required to avoid promoting the conversion of high carbon storage land uses to low carbon storage land uses.

INTRODUCTION
Land use changes and rubber expansion in Southeast Asia, China and Xishuangbanna

In the tropics the highest rates of deforestation and forest degradation have been recorded in Southeast Asia. In Southeast Asia’s humid tropics, there has been widespread expansion of large monocultures of oil palm (Elais guineensis) and rubber (Hevea brasiliensis) plantations, across areas previously covered by native lowland forests (DeFries, Houghton, and Hansen et al. 2002). This region hosts a high concentration of endemic species, which could be obliterated by habitat destruction. By 2100 A.D., Southeast Asia could potentially lose 13% to 42% of its species, of which at least 50% are endemic (Koh and Wilcove 2008).

Land use change can adversely affect ecosystems and the ecosystem services which they provide. These include carbon sequestration, soil quality, water regulation, and climate regulation (Defries, Foley, and Asner 2004; Laganiere, Angers, and Pare 2010; Sauer et al. 2012; Deng, Wang, and Li et al. 2014; Deng and Shangguan 2016). Global climate regulation is perhaps the most important ecosystem service, including sequestration or emission of greenhouse gases (Perrings et al. 2011). Lal (2004a) estimated that the total soil carbon pool is four times greater than the biotic pool and three times greater than the total atmospheric pool. Although there have been some previous carbon storage experiments in Xishuangbanna, these have been limited to specific study areas, which are not necessarily representative of how overall carbon storage could have changed across the entirety of Xishuangbanna (Nizami, Yiping, and Liqing et al. 2014; Yang et al. 2015; Yang, Blagodatsky, and Lippe et al. 2016). Therefore, it is...
essential to understand carbon fluxes and storage in terrestrial carbon sinks in such highly biodiverse areas as Xishuangbanna. Temporal mapping of land use changes and ecosystem services is a promising technique to approach these issues.

China’s economy has developed at an unprecedented rate (Gong 2014). A large proportion of China’s land area is devoted to agriculture and plantations. The developing economy has produced a high demand for rubber, leading to mass conversion of land use to rubber plantations (Sarathchandra, Dossa, and Ranjitkar et al. 2018). Rubber now provides an important cash income to Xishuangbanna’s rural villagers, who lack alternative cash income sources (Yi, Cannon, and Chen et al. 2014a).

The rapid expansion of rubber plantations could have serious repercussions. Rubber plantations have low carbon storage compared to the natural forest ecosystems that they are replacing (Li and Fox 2012; De Blécourt, Brumme, and Xu et al. 2013). The loss of natural habitats is especially likely to cause species extinctions if it occurs in biodiverse ecosystems with a high number of endemic species. Our study area of Xishuangbanna is part of the Indo-Burma biodiversity hotspot (Myers, Mittermeier, and Mittermeier et al. 2000). Xishuangbanna’s natural forests are particularly biodiverse. Although they comprise only 0.2% of China’s landmass, they harbor over 25% of its fauna and flora (Liu, Linderman, and Ouyang et al. 2001).

China’s domestic demand for rubber far exceeds its production. China has become a net importer of rubber, accounting for 30% of global rubber consumption (FAO 2010). This high demand has driven land use conversion to rubber plantations, even in areas with high biodiversity, which provide essential ecosystem services. Such loss of ecosystem services is likely to undermine the productivity of the very land uses that have displaced them (Li, Aide, and Ma et al. 2007).

Early studies of the expansion of monoculture plantations and agriculture in tropical and subtropical landscapes predicted a decrease in biodiversity, ecosystem services and above- and below-ground carbon stocks (Guo and Gifford 2002; Sodhi et al. 2004; Bunker, Declerck, and Bradford et al. 2005; Xu and Melick 2007), although they did not directly measure these outcomes. In this study we tested their predictions by examining land use changes in Xishuangbanna over the past two decades and how these have affected carbon storage capacity.

Implementation of the United Nations Framework Convention on Climate Change (UNFCCC) requires reporting on the changing carbon storage capacity of forest land, cropland, and grassland areas. For such reporting, we need to quantify the potential of different land uses to sequester and store carbon (Zhiyanski, Gikov, and Nedkov et al. 2016a). Ecosystem service modeling of carbon storage is one means to approach this task. This requires simulation of supply, use and demand for ecosystem services, based on ecological and socio-economic data (Burkhard and Maes 2017).

Previous studies in Xishuangbanna have addressed the potential consequences of converting rain forest to rubber plantations (Xu et al. 2005; Noordwijk et al. 2008; Ziegler et al. 2009; Tan et al. 2011; Li and Fox 2012; Meyer et al. 2015). However, Tan et al. (2011) stated that the ecological and environmental consequences of land use conversions remain uncertain. Studies in other tropical areas found reductions in biomass (above- and below-ground), litter and soil carbon stocks (Schorrha, Angelo, and Teixeira et al. 2002; Nair, Kumar, and Nair 2009) and suggested that agroforestry would be a possible approach to restoring carbon sequestration in deforested lands. Yu et al. (2014) and Yang, Blagodatsky, and Lippe et al. (2016) evaluated the carbon storage capacities of different vegetation types, but we found no studies taking a holistic view of how land use changes as a whole affected carbon storage in Xishuangbanna. Given the importance of carbon storage in regulating climate, it is clear that more studies are needed regarding the impact of land use change on carbon storage. Consideration is also required of how far reversing land use change could also reverse current loss of carbon storage capacity.

A proper understanding of a system is essential for successful management; a principle that is true for ecosystem services. Research is necessary to guide and support the management of natural resources, ecosystems and socio-ecological systems (Sarathchandra, Kambach, and Ariyaratna et al. 2018). Warren-Thomas, Dolman, and Edwards (2015) stated that by 2024 an estimated additional 4.3–8.5 million ha of rubber plantations will be required to satisfy the increasing demand for rubber. Such expansion would threaten a significant portion of existing forest areas, therefore it is essential to measure ecosystem services if we are to understand how ecosystems contribute to human well-being. This is vital to forming and supporting land use policies which would have a positive impact on natural resource management.

In our current era, the basic resources for both humans and nature are threatened by both global and regional pressures (IPBES 2019). Consequently, principles for measuring and valuation of ecosystem services are attracting attention for their potential to assist the resolution of environmental dilemmas (Nerlich, Graeff-Honninger, and Claupein 2013). Appropriate methods for measuring or valuation of ecosystem services depend on variety of factors, such as the type of ecosystem service and the study region (Malinauskaitė et al. 2021). Ecosystem services are generally evaluated in terms of their biophysical, economic and socio-cultural contributions. For example, socio-cultural valuation can be
based on literature reviews, stakeholder mapping, observations, interviews or preference surveys. (Bélisle, Wapachee, and Asselin 2021). Economic valuations of ecosystem services can use methods including benefit transfer (Zhou, Wu, and Gong 2020), choice experiments and market data (Hynes, Chen, and Vondola et al. 2021; Kieslich and Salles 2021).

It is especially important not to ignore the roles of stakeholders and economic forces. Currently, most models that forecast ecosystem services and biodiversity losses as a result of habitat loss implicitly assume that passive sampling can be extrapolated from larger to smaller habitats. However such scenario projections typically ignore demographic effects (Chase, Blowes, and Knight et al. 2020), market incentives and government policies that can have crucial impacts on the direction and extent of changes. Considering these factors, this paper was designed with the following objectives: (1) to analyze the changes in land use and land cover across the study area over specific study periods of 1988–2010; (2) to compare changes in land use and calculated carbon storage among the beginning, middle and end of the study period; (3) to evaluate the effect of land use conversion on total carbon storage of the studied landscape; and (4) to assess the implications for policy formation to reduce the potential carbon storage losses that could arise from further land use change.

**Materials and methods**

**Study site**

Xishuangbanna Dai Autonomous Prefecture (19,164 km²) is located in Yunnan Province, in Southwestern China, bordering Laos and Myanmar on the upper course of the Mekong River (Lü et al. 2010). Most of its land area is categorized as mountainous, with an elevation range of 475 m to 2,428 m (Li et al. 2008). Xishuangbanna records annual precipitation of 1,493 mm. It has a typical tropical monsoon climate with a rainy season between May and October and a dry season between November and April. Due to its climate and geography, Xishuangbanna is considered to be a transition zone, which has the highest level of biodiversity in China (Hongmao et al. 2002; Lü et al. 2010).

Xishuangbanna supports five types of natural forest: tropical seasonal rain forest (at elevations < 900 m), tropical montane rainforest (700 m – 1,500 m), monsoon forest (< 900 m), monsoon forest located on limestone (< 800 m) and subtropical evergreen broad-leaved forest (1,000 m – 1,500 m). Each forest type hosts a unique assemblage of faunal and floral diversity (Zhang and Cao 1995).

**Data acquisition, processing and analysis**

The administrative data, including boundaries of the study area and remote sensing data used for this study were obtained from the Center for Mountain Ecosystem Studies, Kunming Institute of Botany, Chinese Academy of Sciences, and from two previous studies (Xu, Grumbine, and Beckschäfer 2014b; Chen, Yi, and Schmidt-Vogt et al. 2016). However, neither of these studies had addressed how the total carbon storage of the landscape has been affected during the recent period of intensive rubber expansion. We briefly summarize our methodology here.

**Remote sensing data of land use types**

Land use types were mapped based on a Landsat TM image and a Landsat ETM+ image (path/row number: 130/45, acquisition dates: 2 February 1988 and 28 March 2002, spatial resolution of 30 m). We used this study window period because few published studies have focused on this time period of escalated rubber expansion.

Radiometric and geometric rectification had already been applied to these images (LIT products) by the imagery supplier. For 2010, a published map by Xu, Grumbine, and Beckschäfer (2014b) was used and the corresponding RapidEye (RE) satellite images were reanalyzed, using a refined classification scheme that enables rubber plantations and other land use types to be distinguished. During January and February of 2010, the 48 RapidEye (RE, ortho product level 3A, spatial resolution of 5 m) scenes were captured from TM, ETM+, and RE images that were registered in Universal Transverse Mercator Projection (UTMg WGS 84). We used high-resolution RE imagery because it allows superior identification of different vegetation types, via its use of red edge spectral bands in combination with other spectral bands. We thereby developed a land cover classification scheme for the identification of forests, farmlands, rubber, shrubs, banana, barren land, tea, water and construction (Table 1).

This classification was quantified empirically via verification based on the different land use types shown in Google Earth Quickbird imagery. We obtained 460 GPS points in the study area between 2009 and 2011. Out of these 460 GPS points, we recorded land cover information for 308 points. Image classification was done with the software eCognition 8.0 (Trimble, U.S.), using membership function and nearest neighbor classifiers. Associated threshold values of certain object features were determined and implemented in class membership functions. The object-based classification using the membership function classifier was also applied for identification of land cover in Landsat images. Training areas were acquired from ground-truthed GPS points and high-resolution satellite
images obtained from Google Earth (DigitalGlobe, U.S.). RE images were classified scene by scene and subsequently merged. Freely accessible DigitalGlobe archives played an important role in assessing the accuracy of our classifications (Olofsson, Foody, and Herold et al. 2014). We randomly selected 71 land use reference points for accuracy assessment for 2001 and 2002. These were generated based on high-resolution images from Google Earth. For verification of RE image classification for the year 2010, we used an additional combination of 361 points that had been obtained from the field and from Google Earth in 2009 and 2010 (Chen, Yi, and Schmidt-Vogt et al. 2016).

Land use types in 2010 derived from RE images were resampled to 30 m to match the resolution of RE images from 1988 and 2002. This resampling of the 2010 layer for rubber plantations from 5 m to 30 m resulted in a change of less than 1% in the calculated

**Table 1. Land use transition matrix for the study area between 1988–2010 (area in km²).**

| 1988/2010 | Banana | Construction | Farmland | Forest | Barren land | Roads | Rubber | Shrub | Tea | Water | Total |
|-----------|--------|--------------|----------|--------|-------------|-------|--------|-------|-----|-------|-------|
| Banana    | 0      | 0            | 0        | 151.6  | 0           | 0     | 1.4    | 23    | 0   | 176   |
| Construction | 0    | 0            | 0        | 126.4  | 19.4        | 0     | 12.8   | 43.6  | 0.8 | 203.1 |
| Farmland   | 0      | 0            | 856.3    | 1056.8 | 0           | 0     | 3.2    | 173   | 0   | 2089.3|
| Forest     | 0      | 0            | 252.8    | 10,391.2 | 0         | 0     | 0.3    | 49.8  | 0.1 | 10,694.2|
| Barren land| 0      | 0            | 69.2     | 0.9    | 23.8        | 0     | 58.4   | 6.1   | 23.3 | 184.2 |
| Roads      | 0      | 0            | 0.9      | 2.1    | 2.6         | 0     | 0.6    | 0.2   | 1.3 | 7.7   |
| Rubber     | 0      | 0            | 263.7    | 3142.8 | 0           | 0     | 799.3  | 53.6  | 35.4 | 4294.8|
| Shrub      | 0      | 0            | 61.3     | 3.3    | 0           | 0     | 929    | 0     | 0   | 946.9 |
| Tea        | 0      | 0            | 1.3      | 46.1   | 0           | 0     | 0      | 78.9  | 0   | 126.3 |
| Water      | 0      | 0            | 1.4      | 0      | 0           | 0     | 0      | 0.2   | 93.8 | 95.4  |
| Total      | 0      | 0            | 1506.9   | 14,921.2 | 45.8        | 0     | 871.1  | 993.6 | 429 | 97.2  | 18,817.9 |
total rubber plantation area (Chen, Yi, and Schmidt-Vogt et al. 2016). Prior to analysis, all maps had been projected to a common grid system; the Universal Transverse Mercator (UTM) WGS84 reference system. All map spatial analyses were done in ArcGIS 10.2.

Quality of information derived from remotely sensed data should be determined by an accuracy assessment. This is most reliable when used along with ground reference data or data derived from aerial photographs at or near the time of the satellite overpass (Tao et al. 2015). We used the Kappa coefficient to evaluate the accuracy of the classification methods which we used to produce land use/land cover maps (see supplementary materials for details). Thereafter, the raster layers of land use from 1988, 2002, and 2010 were used to analyze land use conversion for each specific time period and to calculate carbon storage capacity.

Calculating carbon storage across Xishuangbanna
To determine the parameters for the levels of carbon storage for our calculations, we considered the carbon present in each land use type. Ten land use categories were identified in the study region (Table 2). For current carbon storage (total weight of carbon stored per square kilometer, Mg C/km²) values for each land use category, we referred to previous carbon related studies in Xishuangbanna (Jia, Zheng, and Zhang 2006; Li et al. 2008; Guirui et al. 2010; Yi, Cannon, and Chen et al. 2014a; Yang et al. 2015; Yang, Blagodatsky, and Lippe et al. ; Lu et al. 2016). We applied the carbon storage and sequestration model available in InVEST (Integrated Valuation of Ecosystem Services and Tradeoffs) to estimate temporal carbon storage. InVEST is preferable to other models for its ability to demonstrate ES results spatially in a visualized map. Moreover, InVEST consists of sub models which can be selected by users according to their specific needs and precision of available data (Nelson, Ennaanay, and Stacie et al. 2015; Fang, Bai, and Jiang et al. 2020).

The InVEST carbon storage model partitions carbon storage into four components: above-ground, below-ground, soil, and dead organic matter. Grid land cover data is used to provide the input to the model. It calculates the total carbon in each pool and provides an estimate of total carbon storage across the whole landscape (Sharp, Chaplin-kramer, and Wood et al. 2015; Zhiyanski, Gikov, and Nedkov et al. 2016a). We used the 3.1.1 version of InVEST tools, at a resolution of 30 m × 30 m. We opted to include carbon storage in all 4 pools. This was intended to avoid the bias associated with studies that only estimate above ground biomass – which will generally underestimate total carbon storage (Sierra, Del Valle, and Orrego et al. 2007).

Calculating land use change and its impact on carbon storage
We mapped land cover data for Xishuangbanna from the years 1988, 2002 and 2010. We calculated the annual rate of land use change separately for each land use type from equation 1 (Sarathchandra, Dossa, and Ranjitkar et al. 2018). This equation assumes linear conversion rates. To account for temporal variation in land use conversion, we analyzed separately the data from 1988 to 2002 and from 2002 to 2010.

\[
LU_{change} = \frac{(A_t - A_0)}{(t_2 - t_1)}
\]

Where:

- \( LU \) change = rate of land use change per year (km² year⁻¹),
- \( t_1 \) = year of first data recording
- \( t_2 \) = year of second data recording
- \( A_0 \) and \( A_t \) = area of land use type (km²) at times \( t_1 \) and \( t_2 \)

The contribution of each land use type to total deforestation was measured using the following formula:

\[
\text{Deforestation contribution} = \frac{\text{Individual LU change}}{\text{Total forest cover change}} \times 100
\]

Moreover, we quantified the LULC dynamics by conducting map to map comparisons using overlay function of ArcMap 10.6 in ArcGIS. Here we obtained a two-way cross-matrix and used it to describe the main land use changes in the study area. The quantity of conversion from one LULC type to another over the study period was determined by conducting cross tabulation analysis on a pixel-by-pixel basis (Falahatkar, Soffianian, and Khajeddin et al. 2011; Zhang, Li, and Chen 2011; Zhang, Zheng, and Feng et al. 2017; Butt et al. 2015; Hu, Batunacun, and Zhuang 2019).

We then mapped the carbon storage in each land cover and land use type and evaluated how changes in vegetation cover have affected the total carbon storage capacity of Xishuangbanna. We calculated carbon storage for each land use type using linear regression (Lian 2015):
**Table 2.** Main land use changes and their mean elevation in Xishuangbanna between 1988 and 2010.

| Land use   | Land cover 1988 (km²) | Land cover 2002 (km²) | Land cover 2010 (km²) | Absolute change in land cover 1988–2010 (km²) | Rate of land use change (km² y⁻¹) | Relative change in land cover (%) | Elevation (m) (Mean ± SE) |
|------------|-----------------------|-----------------------|-----------------------|----------------------------------------------|----------------------------------|----------------------------------|-------------------------------|
| Banana     | 0.0                   | 0.0                   | 176.0                 | 176.0                                        | 8.0                              | 0.9%                            | 872.2 ±8.2                   |
| Construction | 0.0                  | 0.0                   | 203.1                 | 203.1                                        | 9.2                              | 1.1%                            | 1214.0 ±1.14                 |
| Farmland   | 15,069                | 1811.3                | 1901.7                | 82.4                                         | 26.4                             | 3.1%                            | 1299.3 ±11.8                 |
| Forest     | 15,129.4              | 13,650.3              | 10,658.4              | −4227.0                                      | −192.2                           | −23.3%                           | 1422.3 ±12.5                 |
| Barren land| 45.9                  | 50.3                  | 184.2                 | 138.3                                        | 6.3                              | 1.1%                            | 1357.5 ±12.0                 |
| Roads      | 0.0                   | 0.0                   | 7.7                   | 7.7                                          | 0.4                              | 0.01%                           | 935.0 ±8.5                   |
| Rubber     | 871.1                 | 1897.6                | 4294.8                | 3423.7                                       | 155.6                            | 18.2%                           | 1203.0 ±11.3                 |
| Shrub      | 993.6                 | 958.1                 | 946.9                 | −46.6                                        | −2.1                             | −0.3%                           | 1239.8 ±8.2                  |
| Tea        | 428.5                 | 399.6                 | 126.3                 | −302.2                                       | −13.7                            | −1.4%                           | 1102.5 ±14.2                 |
| Water      | 97.2                  | 97.1                  | 95.4                  | −1.8                                         | −0.1                             | −0.01%                          | 881.7 ±26.2                  |
**Figure 2.** Carbon storage change for six land use types during 1988, 2002 and 2010.

\[
y_n = M t_n + b \quad \text{[Eq. 3]} \\
\text{Where:} \\
\begin{align*}
y_n & \quad = \text{predicted total carbon storage at time } n \\
t_n & \quad = \text{time period} \\
M & \quad = \text{slope of the regression line} \\
b & \quad = \text{the intercept of } y \\
\end{align*}
\]

\[
SE_{\text{line}} = (\bar{y} - (Mt_1 + b))^2 + (\bar{y} - (Mt_2 + b))^2 \ldots (\bar{y} - (Mt_n + b))^2 
\quad \text{[Eq. 4]}
\]

Where:

\( SE_{\text{line}} \) = the standard error of the regression line

To determine how much of the variation of \( y \) is due to variation in time \( n \), we calculated the total variation in \( y \) (Lian 2015):

\[
SE_{\bar{y}} = (\bar{y} - \bar{y})^2 + (\bar{y} - \bar{y})^2 \ldots \ldots (\bar{y} - \bar{y})^2 
\quad \text{[Eq. 5]}
\]

Where:

\( \bar{y} \) = mean value of \( y \)

\( SE_{\bar{y}} \) = standard error of the mean of \( y \)

We calculated the percentage of the total variation in \( y \) which was not due to variation in \( n \) (time). The \( R^2 \) value gives the residual proportion of the variation which is not described by the regression of \( n \) with \( y \):

\[
R^2 = 1 - \frac{SE_{\text{line}}}{SE_{\bar{y}}} \quad \text{[Eq. 6]}
\]

Where:

\( R^2 \) = the coefficient of determination of the regression line.

\( \frac{SE_{\text{line}}}{SE_{\bar{y}}} \) = proportion of carbon storage due to \( y \)

Finally, carbon storage change during the study period was calculated using

\[
\Delta C = \sum \left( \frac{C_{t_2} - C_{t_1}}{t_2 - t_1} \right) 
\quad \text{[Eq. 7]}
\]

Where:

\( \Delta C \) = carbon storage change (tons/year)

\( C_{t_1} \) = carbon storage at time \( t_1 \)

\( C_{t_2} \) = carbon storage at time \( t_2 \)

**Results**

**Accuracy assessment**

The accuracy of the land use types identification of maps produced for 2002 were 88.7% and the overall accuracy for all land use land cover classes for 2010 was 87.5%, with a Kappa coefficient of 0.82.

**Deforestation and associated land use change**

In 1988, 78.9% of Xishuangbanna was covered by natural forest, followed by farmlands 9.5%, natural shrubs 5% and rubber 4.5%. By 2010, 23.3% of forest cover had been destroyed. During the same period, land covered by rubber plantations increased by 18.2%, followed by farmland 3.1%. All other natural land cover types had reduced in area by 2010, including shrubs by 0.3% and water by 0.01% (Figure 2).

The transition matrix for the study area (Table 1) shows how each LULC category had changed from 1988 to 2010. The planting of rubber monocultures was responsible for 70.4% of this deforestation,
followed by agriculture (23.2%), construction of buildings and roads (2.8%) and banana plantations (3.3%). This rapid expansion of rubber plantations has largely replaced diverse tropical seasonal rainforests (Fig. S1). These had previously occupied the areas that are most suitable for rubber cultivation (Warren-Thomas, Dolman, and Edwards 2015).

Total forest cover in Xishuangbanna had reduced by 203.2 km² y⁻¹. Meanwhile, rubber plantations had expanded by 153.4 km² y⁻¹. Further major land use changes included an increase in banana plantations and construction. Despite a high demand for tea throughout Asia, the land area devoted to tea cultivation diminished by 71% between 1988 and 2010 (Table 2).

Our spatial analyses demonstrated that, since 1988, rubber had been planted at progressively higher elevations. This upward range expansion had contributed to the reductions in the range of forests and other natural land uses. Previously, inaccessible high elevation habitats had been comparatively safe from conversion to rubber plantations.

**Changes in carbon storage**

In 2010, lowland forests were the land use type which had the highest mean carbon storage, at 119.9 Mg C/ km² (Figure 2). These lowland forests include tropical rainforest, tropical seasonal rainforest and mixed forest with bamboo.

Carbon storage in banana plantations, farmlands and rubber plantations was calculated to have increased over time, due to increases in their land use cover. In contrast, for both natural land use types (forest and shrubs) and tea plantations, carbon storage decreased over time.

In 2010, rubber plantations held a mean carbon stock of 17.3 Mg C/km². The lowest carbon stock was for tea plantations. In 1988, forests held a mean carbon stock of 156.3 Mg C/km² while rubber and tea plantations held 7.1 Mg C/km² and 0.57 Mg C/km² respectively. The highest individual land cover carbon storage capacity for these three time periods was for natural forest, followed by rubber plantations, banana plantations, shrubs and tea plantations.

**Carbon storage changes caused by land use change during 1988 –2010**

Carbon storage calculations for Xishuangbanna during 1988 to 2010, based on the InVEST modeling tool, showed an overall loss of carbon storage in the region during this period (Figure 2).

Xishuangbanna’s total carbon storage in 1988 was 0.61 Tg C. By 2002, this had fallen to 0.47 Tg C, and to 0.41 Tg C by 2010 (Figure 3). This represents a decrease of 0.2 Tg C, over a 22-year period. The average annual decrease between the years 1988 to 2010 was 0.01 Tg C year⁻¹. These reductions in carbon storage will have
resulted in a corresponding degree of increased carbon emissions.

Discussion

Deforestation and drivers of land use change

During our study period, our results show that Xishuangbanna suffered heavy deforestation; losing 23.3% of its natural forest area. Meanwhile, rubber plantations were established across 18.2% of its land area. Expansion of rubber plantations has generally driven loss of natural forest (Qiu 2009; Xu, Grumbine, and Beckschäfer 2014a).

Historical records of changing land ownership in China give supporting evidence of widespread changes in land use. In the 1950s, the Chinese government established large-scale state-owned rubber plantations in Xishuangbanna. In the early 1980s, China introduced the “Household Responsibility System.” This enabled smallholder farmers to plant rubber (Chapmen 1991). During the 1960’s, monoculture rubber plantations began to replace Xishuangbanna’s lowland forest. By the 1990’s, swidden agricultural lands were also being converted to rubber plantations. This situation was worsened by the land use definitions used in government policy documents and national statistics, which did not distinguish between rubber and natural forest (Van Noordwijk, Tata, and Xu et al. 2012).

Because China’s policies grouped all natural forest types and plantations together as a single land use category, incentives for afforestation were granted to any green cover type regardless of whether these represented economic plantations or recovery of natural habitat. Thus, a policy intended to promote protection and reestablishment of natural forests actually promoted conversion of natural forest to rubber.

In the early 2000’s, income from rubber plantations boosted average rural incomes in Xishuangbanna by 10-fold (Richard, 2009). By 2009, rubber yield prices had reached approximately 15,000 RMB ha⁻¹ year⁻¹, whereas farmers could only earn 2,000–3,000 RMB ha⁻¹ year⁻¹ from tea or rice (Qiu 2009). This provided a strong economic incentive for the rapid expansion of rubber in Xishuangbanna. However, the pursuit of these economic incentives came at the cost of serious negative environmental impacts (Ziegler et al. 2009).

Our analyses also demonstrated other important changes to land use practices during our study period, including the introduction of short-term cash crops and infrastructure development (roads and other construction). Expansion of road networks has facilitated the transport of rubber and bananas from plantations to markets. Banana trees are fast-growing, giving a short-term profit, which has encouraged farmers to replace rice paddy fields with banana plantations.

Carbon storage changes triggered by the conversion of land use: consequences and mitigation

Between 1988 and 2010, we found that the carbon storage in Xishuangbanna decreased by 0.223 Tg C km⁻². Although the total percentage of cultivated and forest land cover did not drastically change, conversion between specific land use types was responsible for this overall loss of carbon storage. Changes in the carbon storage capacity were determined by which land use types were converted to rubber. For example, conversion of other agricultural crops (which have a low carbon storage capacity), would cause less loss of carbon storage capacity than large scale conversion of natural forest (which has the highest carbon storage capacity) to rubber plantations.

Previous studies indicate that the primary mechanism by which land use conversion affects carbon storage capacity is by reducing total plant biomass (Jiang, Apps, and Peng et al. 2002; Schrotha, Angelo, and Teixeira et al. 2002; Mao, Wang, and Dai 2009; Guirui et al. 2010).

Differences in total carbon storage between the base year of 1988 and our LULC map of 2010, correspond to land cover change during this period. The clearest contributing factor was a reduction in natural forest cover from 79% to 56%. Other variables that would have contributed to changes in the carbon storage capacity of these ecosystems would be: changes in the soil conditions, harvesting of timber, fires, tree maturity, average annual precipitation and temperature (Wu, Li, and Zhou et al. 2018). Previous studies of China’s carbon sequestration potential have recorded that Southern China accounts for > 65% of China’s total carbon sink (Fang et al. 2007; Guirui et al. 2010). It will therefore be of special importance to implement suitable policies in Southern China to avoid or minimize the conversion of land uses with a higher carbon storage capacity to those with a lower carbon storage capacity.

Land use change comprises both changes in overall land cover of each land use type and in species composition (Meyer et al. 2015). For example, different tree species within natural forest have different growth rates, in addition to differing above- and below-ground biomass, and thus differing abilities to act as carbon sinks (Chi, Guo, and Fang et al. 2017). Agricultural crops also differ in their carbon storage capacity (Cantarello, Newton, and Hill 2011). Perennial
crops are likely to store more carbon as above-ground biomass. Whether annual crops act as carbon sinks or sources could depend on agricultural practices, including tillage methods (Udawatta and Jose 2012; Kim, Kirschbaum, and Beedy 2016).

Keeping organic carbon in terrestrial sinks will be crucial to limiting climate change. Forests, grasslands, peat swamps, and other terrestrial ecosystems collectively store much more carbon than does the atmosphere (Lal 2004b). However, current shifts in global land use are reducing terrestrial carbon storage – turning previous carbon sinks into potential carbon sources (Nair, Kumar, and Nair 2009). Potential feedbacks between soil processes and increasing temperatures could further exacerbate the situation (Smith 2004).

The Intergovernmental Panel on Climate Change (IPCC) has stated that it will be impossible to keep global temperatures at safe levels unless there is also a transformation in human activities in food production and land management. Humans currently exploit 72% of the planet’s ice-free surface. Agriculture, forestry and other human land uses jointly account for almost a quarter of greenhouse gas emissions (IPCC 2019). As human populations increase, this situation will worsen, unless active steps are taken to protect land uses that act as carbon sinks and avoid conversion to land uses that act as carbon sources. Action is needed at a landscape level, to keep carbon sequestered for longer in the form of organic biomass. Globally, forests act to capture and retain large quantities of carbon (Espinoza-Domínguez et al. 2012). Therefore, conversion of deforested land to agroforestry has been proposed as an effective means to enhance carbon sequestration (Nair, Kumar, and Nair 2009) while diversifying income generation for vulnerable farming sectors. In such agroforestry ecosystems, above-ground species diversity affects their effectiveness in carbon storage and sequestration. Therefore, to maintain higher carbon storage capacity, agroforestry practices should be selected that support higher plant species diversity. This is of particular importance in tropical forests, where there is an especially strong relationship between higher plant species diversity and higher carbon storage (Liu, Herbert, and Hashemi et al. 2006; Henry, Tittonell, and Manlay et al. 2009; Laban, Metternicht, and Davies 2018).

4.3 Improving land use management practices for rubber dominated landscapes: recommendations based on the global scientific literature

As the landscape of Xishuangbanna is being increasingly converted to rubber plantations, it is essential to develop better land use management systems which maintain ecosystem health and support the local economy. Xishuangbanna has one of the most fertile soils in China, making it suitable for a range of agroforestry applications, including coffee, tea and cocoa. Agroforestry practices and intercropping species should be selected that enhance specific ecosystem services. Carbon storage potential differs between agroforestry systems. In Mexico, coffee agroforestry and intercropping was found to have higher carbon storage potential than coffee monocultures (Espinoza-Domínguez et al. 2012; Hergoualc’h, Blanchart, and Skiba et al. 2012). In Xishuangbanna, if coffee plants were grown under shade trees (Albizia), there was increased above- and below-ground biomass (Dossa et al. 2008). Thus, one means to mitigate the negative impact of rubber plantations in Xishuangbanna would be to promote the use of rubber as the shade tree for coffee.

Some other land use types have even lower carbon storage than rubber plantations, so it might be damaging to promote the widespread conversion of land already planted with rubber to other crops. In Ghana, the aboveground carbon sequestration potential of rubber plantations (214 t C ha$^{-1}$) was found to be higher than that of orange (76 t C ha$^{-1}$), cocoa (65 t C ha$^{-1}$) and palm (45 t C ha$^{-1}$) plantations (Kongsager, Napier, and Mertz 2013). More experiments on carbon sequestration are required in Xishuangbanna in order to select the best intercropping species for agroforestry, but intercropping of rubber with cocoa appears to be particularly promising. One study of rubber – cocoa intercropping found a total carbon storage of 91.5 Mg C ha$^{-1}$. Of this, 84.7 Mg C ha$^{-1}$ were stored in rubber trees, 5.22 Mg C ha$^{-1}$ in cacao trees and 1.67 Mg C ha$^{-1}$ in litter (Cotta, Jacovine, and Hn De et al. 2008). If further research supports these findings, then we would suggest that intercropping of rubber trees with coffee and cacao trees could simultaneously achieve the multiple goals of increasing carbon sequestration, enhancing ecosystem services, and diversifying crops. If these intercropping systems are further combined with crops that are in high demand locally, it would reduce the risk to farmers posed by the failure of any single crop. Crop diversification is an effective means to mitigate against disease, flooding, and drought, as well as against the potential effects of future climate change (Read, Beerling, and Cannell et al. 2001).

Economic mechanisms can be harnessed to avoid further loss of natural ecosystems in Xishuangbanna (Yi, Wong, and Cannon et al. 2014b). This could be achieved through mechanisms such as market-based ecosystem payments and carbon financing. Carbon pricing and trading schemes could compensate farmers for any loss of income arising from crop market price fluctuations. Such initiatives could be used to encourage local farmers and other relevant parties to preserve existing natural
land uses and discourage further conversion between land use types.

One major limitation of our study is that we inferred carbon measurement values from other studies. As soil organic carbon and below-ground biomass can be highly variable, this approach has the potential to under- or over-estimate total carbon stocks (Yang, Blagodatsky, and Lippe et al. 2016). Future carbon related studies in Xishuangbanna could be made more robust if extensive soil sampling was conducted, giving more comprehensive understanding of below-ground carbon stocks, and how these are impacted by land use change.

Conclusion

Rapid deforestation and land cover changes in Xishuangbanna have primarily been driven by conversion of natural forest to monoculture rubber expansion. Additional contributors have been the expansion of other recently introduced short term cash crops, such as banana.

Because large trees dominate carbon storage in natural forests (Mildrexler, Berner, and Law et al. 2020), and trees are generally larger in natural forests than plantations, such loss of natural forest cover would hugely reduce carbon storage capacity. We showed that Xishuangbanna’s rapid land use change has also reduced its overall ability to store carbon. Revised land use management and conservation policies are needed that disincentivize land use conversion, especially from high carbon storage land categories to low carbon storage land categories. The impact of new policies should be monitored over time, using appropriate methods to assess land use change and associated changes in ecosystem services. It is essential that policies are integrated with information on species and the predicted effects of climate change (Zomer, Xu, and Wang et al. 2015).

One potential strategy to mitigate the impact of rubber monoculture expansion, would be to introduce intercropping practices into existing plantations. Species should be selected for intercropping that are economically viable, provide ecosystem services, and increase the potential carbon sequestration of rubber plantations.

Disclosure statement

No potential conflict of interest was reported by the author(s).

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