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Full title: Managing forests for global and local ecosystem services: A case study of carbon, water and livelihoods from eastern Indonesia

## Highlights

- Trade-offs/synergies exist among ecosystem services in different scales.
- Forest carbon projects can have negative effects on watershed services.
- Watershed services from forests are very important to local communities.
- Forest carbon projects must consider further details within a land use class.

Full title: Managing forests for global and local ecosystem services: A case study of carbon, water and livelihoods from eastern Indonesia

## Abstract

Despite a recent increase of interest in global payment for ecosystem services (PES) mechanisms, there has been little comprehensive assessment of PES impacts on ecosystem services (ESs) at smaller scales. Better understanding of localized impacts of global PES can help balance ES deliveries for global benefits with those for meeting landscape and local level needs. Using a case study from eastern Indonesia, we assessed trade-offs and potential synergies between global PES (e.g. REDD+ for forest carbon) and landscape level ESs (e.g., water quantity, quality, regulation) and local ESs (e.g. forest products for food, energy, livelihoods). Realistic land use change scenarios and potential carbon credits were estimated based on historical land use changes and in-depth interviews with stakeholders. We applied a process-based hydrologic model to estimate changes in watershed services due to land use changes. Finally, local community's forest uses were surveyed to understand locally realized ESs. The results show empirical evidence that, without careful consideration of local impacts, a PES mechanism to protect global ESs can have negative consequences for local ecosystem services. We present management alternatives designed to maximize positive synergies between different ESs at varying scales.

**Keywords:** ecosystem services; carbon; REDD+; watershed services; livelihoods; Indonesia,

# 1 Introduction

Globally, tropical forests account for approximately 25% of all terrestrial carbon (Bonan, 2008). Deforestation is the largest source of carbon emissions from tropical developing countries (Pan et al. 2011). The 2015 UN climate change conference in Paris reconfirmed the importance of forests in global climate regulation. The agreement explicitly included the REDD+ mechanism[1] as part of the global climate regime, where tropical and sub-tropical countries could receive both public and private funding for reducing carbon emissions and conserving standing forests. Indonesia has the third largest tropical forest in the world, with one of the world's fastest rates of deforestation at more than 1,000 $km^2$ of forests (476 $km^2$ of primary forest) lost per year between 2000-2012 (Hansen et al., 2013; Margono et al., 2014). Indonesia has emerged as the major beneficiary of global negotiations to mitigate climate change through improved forest management (Simula, 2010). It has received the largest portion of REDD+ readiness commitments from the public sector ($757 million out of $2.8 billion total committed and dispersed from 2009 to 2014; Goldstein et al., 2015). In the private sector, carbon credits from protecting Indonesia's forests was 5.5% of all voluntary carbon transactions in 2015 (Hamrick and Goldstein, 2016).

Offering financial incentives for tropical developing countries to reduce deforestation and forest degradation can be a win-win-win solution for climate mitigation, ecosystem conservation and poverty alleviation (Pistorius, 2012). However, many previous studies have warned that international intervention in the form of Payments for Ecosystem Services (PES) can exacerbate internal social problems (Blom et al., 2010; Wunder, 2008). Failure to include consideration for local uses of resources in global PES design can

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[1] Reducing Emissions from Deforestation and Forest Degradation (REDD+) is an effort to offer financial incentives for developing countries to reduce emissions from forested lands. REDD+ projects include activities for (a) reducing emissions from deforestation, (b) reducing emissions from forest degradation, (c) while recognizing the role of conservation of forest carbon stocks, (d), sustainable management of forests, and (e) enhancement of forest carbon stocks (UN-REDD programme, 2017).

undermine rights of indigenous and local communities, exacerbate food and water insecurity (UN-REDD programme, 2017; Fazey et al, 2010), diminish ecological integrity and equity (Motel et al., 2009), and result in less than optimal outcomes for the ecosystem service targeted (Enrici and Hubacek, 2016; Skutsch et al. 2011). Despite a recent increase of interest in global PES mechanisms, there has been little comprehensive assessment of their impacts on localized ecosystem services (ESs) and livelihoods. Better understanding of the localized impacts is needed to find ways of balancing provision of an ES that provides benefits at the global scale, while meeting local needs for water, food, energy and livelihoods. Using a case study from eastern Indonesia, we present a detailed assessment of trade-offs and potential synergies among global ES (forest carbon), landscape-level regulating services (e.g. water) and localized provisioning services (e.g., forest products for food and energy). Specific research questions are: 1) what are realistic land management scenarios to recover forest area lost and improve forest conditions?; 2) how do these scenarios affect global, landscape and local ES provisions?; 3) how do global modelling results compare with local perception in assessments of ecosystem service change; 4) what are the management alternatives to maximize positive synergies among provisions of different ESs at varying scales?

## 2 Literature review: Ecosystem Services trade-offs and synergies

The Millennium Ecosystem Assessment (MA 2005) placed the term “ecosystem services” firmly in the policy agenda (MA 2005; Gómez-Baggethun et al., 2010). Since then, many have advocated the urgent need to incorporate sustainable provisioning of ESs into policies and planning for managing landscapes (e.g., Daily et al. 2009; de Groot et al, 2010). However, the flows of ESs are determined not only by ecosystem functions and processes (ES supply), but also by demands from various human actors (ES demand) in multiple-scales (Figure 1). Mouchet et al. (2014) advanced a typology to understand ES trade-offs by merging ecological and socio-economic considerations found in previous studies. Spatial and time lags of ESs (spatial and temporal trade-offs) can occur in both supply and demand sides, in terms of production and delivery (Rodrıguez et al. 2006) and benefits and costs (TEEB, 2010). Also targeting one ES can affect other ESs positively or negatively (among ESs synergies or trade-offs), and resilience of the ecosystem as a whole (reversible trade-off), as well as who “losers” and “winners” are among ES beneficiaries (beneficiaries trade-off) (Mouchet et al. 2014).

<Figure 1>

The forces of globalization are intensifying interactions among ES demand and supply over distances and cross-scales (Cash et al. 2006; Liu et al. 2015). Managing ESs and anticipating changes in their spatial, temporal and societal distributions are increasingly difficult as local events (e.g. land use change in tropics) can have global consequences (e.g. climate change) (e.g. Bruckner et al. 2015; Meyfroidt et al. 2013; Lambin et al. 2011; Seto et al. 2012). Spatially distributed beneficiaries of different ESs vary also in their social and economic status, which affect their ability to influence decision-making process (TEEB 2010). There have been several studies that addressed the spatial scale of managing ES (e.g., Hein et al., 2006; Willemen et al., 2012 – both in the Netherlands) and presented empirical evidence of trade-offs and synergies of different ES deliveries (e.g. González-Esquivel et al. 2015; Grossman 2015; Haines-Young et al., 2012; Maes et al. 2012; Mastrangelo and Laterra 2015; Mora et al. 2015; Turner etal. 2014 – in Europe and Latin America), However, those most affected by global PES, such as REDD+, are in tropical developing countries often lacking technical capacity for data collection, analysis and management (Goetz et al. 2014). With the growing significance of global carbon governance (Bierman, 2010), there is a critical need to understand how the economic and political scale of decision-making affects ESs at different scales. We chose three groups of ESs at global, landscape (watershed level) and local community scales to contribute to our current understanding about ES associations and potential effects of global PES schemes.

# 3 Methods

## 3.1 Study area

The case study area is Lombok island in Nusa Tenggara Barat (NTB) province, located in eastern Indonesia (Figure 2). According to a recent analysis of Landsat images, the forested area of Lombok decreased 28.6% from 1990 to 2010 (Bae et al., 2014). By comparison, Indonesia's national average forest loss is 20.3% during the same period (FAO, 2010). Lombok is also one of the most densely populated and impoverished areas in Indonesia. Seventy percent of the population of NTB province lives in Lombok, although the island only constitutes a quarter of the total land area of the province (708 persons/km$^2$, compared to 237 persons/km$^2$ for NTB and 132 persons/km$^2$ nationally, as of 2014, BPS-NTB, 2015). Economic opportunities are limited to agriculture (24% of Gross Domestic Product (GDP) and 43% of employment of the province) and the mining and quarrying sector (15% of GDP and 1.8% employment) (as of 2014, BPS-NTB, 2015). NTB is among the poorest provinces of Indonesia, based on the Human Development Index (HDI), a metric that combines average life expectancy, education level, and *per capita* income (65.19 compared to the national average 69.55 as of 2015, BPS, 2016).

<Figure 2>

Although forestry is a relatively small contributor to the wider economy of NTB (0.1% of GDP as of 2014, BPS-NTB, 2015), the forests in the northern part of the island, surrounding the Rinjani volcano complex, are an important source of subsistence and income to local communities. The forest also represents an important watershed, providing municipal water for the city of Mataram and irrigation for the major rice production regions throughout Lombok Island. The development of a program of payment for watershed services between municipal rate-payers and forest margin communities is one of the very first examples of PES systems in Indonesia (Diswandi, 2017; Pirard 2012; Prasetyo *et al.,* 2009). The program supports forestry or agroforestry projects proposed by community groups with funds collected from the downstream city's water use fees. A multi-stakeholder group (IMP, *Institusi Multi-Pihak*) consisting of representatives from the World Wildlife Fund, the district forest service, a local university, a mineral water company, the district government and Mount Rinjani National Park, selects and distributes funds for selected projects (Diswandi, 2017; Schweizer et al. 2016; Pirard, 2012).

### 3.2 Research approach

To assess the potential impacts of different land use change scenarios on ESs at different scales, we first identified alternative forest management scenarios that can be adopted by a future carbon PES scheme in Lombok. We then assessed the carbon, water and locally important services for food, energy and livelihoods impacts of these PES scenarios.

#### 3.2.1 Forest management scenarios

Forest carbon projects are designed to provide incentives to protect forests for the value of their standing carbon. Estimating carbon credits is essential for establishing the economic value of forest carbon projects. It includes two components: land-use and land-cover changes and the associated changes in carbon stock (VCS, 2012).

Future forest management scenarios were developed based on analysis of historical changes in land-use and land-cover, along with analysis of drivers of deforestation and forest degradation in the area. Detail of these changes have been reported in Bae et al., 2014 and Kim et al., 2016. Table 1 shows the changes in deforestation patterns in three 5-year periods (1995-2000; 2000-2005; 2005-2010). Land use classes[2] following deforestation were projected based on the satellite imagery footprint of the most recent historical land cover pattern (2005-2010). We focus on the area around the Rinjani volcano complex, where the majority of Lombok's remaining forests are located.

<Table 1>

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[2] Primary forest in this study is defined as mature or intact forest, where standing stock has reached stability. The forest is generally of native tree species, there are no clear indications of human activities, and the ecological processes are not significantly disturbed. Secondary forest is regenerated forest that has been disturbed by human activities or natural disasters. Secondary forest may include a natural forest with timber extraction, retaining artificial gaps in the canopy to 50-60%. This kind of forest includes agroforestry and community forests. Shrubland refers to land with woody vegetation where the dominant woody elements are shrubs, bushes and young generation trees, generally less than 5m in height. The latter appears usually after forest clear-cutting activities without crop cultivation. (Source: Bae et al. 2014).

When the Suharto regime fell in 1998, this socio-political shift caused an abrupt interruption of central government control of forest lands that encouraged massive forest encroachment that was common throughout Indonesia at the time (e.g., Resosudarmo, 2004). Figure 3 graphically illustrates the deforestation patterns during the three 5-year periods studied. Between 1995 and 2000, land use changes were driven by conversion of primary and secondary forests to shrubland, indicating no immediate cultivation after clearing of forest lands. After 2000, deforestation of primary forests decreased and some shrubland transitioned back to secondary forest. However, deforestation of secondary forest continued and secondary forest and shrubland are now being cultivated for dryland agriculture.

<Figure 3>

In addition to examining the historical patterns of land use changes, we conducted a series of in-depth interviews (January 2015) with key informants from provincial and local government forest agencies, as well as international and local NGOs, to better understand the varied contexts of forest management. Based on this information, we develop three land-use change scenarios that represent a range of possible reforestation and restoration outcomes. These scenarios are reported in Section 4.1.

### 3.2.2 Carbon assessment

To estimate the impacts of the projected future land use changes on carbon stocks, we used the area-weighted average of carbon stock for each carbon pool for forest and shrubland, based on field inventory (Table 2). The estimated changes of carbon stock are based only on land use class change in each scenario and do not incorporate other variations within land use classes. For all other land uses, the carbon stocks were assumed to retain the level of soil carbon in shrubland[3].

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[3] For carbon stock change, Verified Carbon Standard (VCS) guidelines state that the REDD+-related projects should account for the following carbon pools: above-ground living biomass of trees and non-trees, and wood products if harvested timbers are utilized to make long-lived wood products. Measuring and monitoring

<Table 2>

### 3.2.3 Hydrological modelling

We utilized a process-based hydrologic model, WaterWorld V 3.31, to project the hydrological impacts of the land-use change scenarios. WaterWorld is a spatially explicit, globally applicable model for calculating monthly water balance, runoff, water quality (including agricultural pollutants and soil erosion) and their spatial distributions under baseline and alternative land use change scenarios (Mulligan, 2013). WaterWorld V 3.31 uses globally available data sets from remote sensing, along with limited *in situ* precipitation data to reveal how forest restoration can affect water provisioning and regulating services (Mulligan 2013). WaterWorld V 3.31calculates water balance as a sum of wind driven rainfall, fog and snowmelt (not applicable in this case) minus actual evapotranspiration. Water infiltrates according to regional infiltration capacities (Gleeson et al., 2011), mediated by slope gradient and tree cover (lower gradient and greater tree cover lead to higher infiltration rates within the geology-controlled regional limits). Infiltration is calculated based on global permeability data using the lithology developed by Gleeson et al. (2011). The infiltration model takes the mean soil-conditioned hydraulic conductivity as the infiltration rate and increases it towards one standard deviation higher than the mean in each pixel as tree cover increases and slope decreases. Higher tree cover encourages infiltration, shallower slopes provide greater opportunity for it to occur. Infiltration is also limited by available porosity and declines in a linear fashion as the soil store fills. Infiltrated water joins subsurface base flow and travels much more slowly to streams than water running over the land surface. Infiltrated water flows downslope along subsurface flow lines dictated by surface topography and at rates dictated by the local infiltration rates of the soil that water is passing through. Infiltrated water may re-emerge as surface runoff anywhere downslope where soil conditions (subsurface flow rates) or water conditions (volume of water in relation to soil thickness mediated storage capacity)

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other carbon pools, such as living below-ground biomass and dead organic matter, are optional or not required.

dictate. This tends to occur most at the base of hillslopes and in channels where regolith thickness is less and thus water emerges at the surface, as baseflow. There is no separate deep groundwater model, WaterWorld models all subsurface moisture as a single per pixel unit. Tree cover also increases the rate of evapotranspiration and the rate of interception of fog, where it occurs. The model was applied to the current conditions in Lombok to produce information on the current hydrological ESs and also model their changes under different land use change scenarios. We also assessed local perception of watershed services linked with forest conditions through focus group discussions (FGD) and survey.

#### 3.2.4 Locally important ecosystem services for food, energy and livelihoods

To understand how local community members utilize and benefit from forest ecosystem services, in-person surveys were conducted at four locations (Figure 4). Survey locations were selected based on their proximity to forests with different designated functions, forest governance status, and permitted activities.

State forests in Indonesia are classified into three designated functional categories (ROI, 1999)[4] : 'Production Forest' for providing forest products; 'Protection Forest' for ecosystem protection, such as watershed and soil conservation; and 'Conservation Forest' for protecting biodiversity and ecosystem conservation. Production and Protection Forests in NTB province are managed by Forest Management Units (*Kesatuan Pengelolaan Hutan*, or KPH) that were created by the central government but are more or less decentralized (See Kim et al., 2016 and Sahide et al., 2016 for more complete information on the Forest Management Units). Conservation Forest is directly managed by the National Park (i.e. Conservation Forest Management Unit) under the central government authority. We selected one community adjacent to Production Forest (A), one near Protection Forest (B), and one near Conservation Forest (C), i.e., near the Rinjani National Park (Figure 4).

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[4] Indonesian Law Number 41/1999 distinguishes "forest" as an ecosystem dominated by trees and "forest area" defined as a particular area designated by the government. Thus, these administrative designations may not necessarily represent actual forest cover and particular forest conditions (Bae et al. 2014

<Figure 4>

We also included an additional community near a Protection Forest that recently gained official recognition as "Community Forest" (*Hutan Kemasyarakatan*, or HKm) (D). Community Forest is one of the legal mechanisms that communities can use to gain recognition for their usufruct rights (ROI, 2007). However, the legal process of establishing HKm is complicated, involving both local and central government agencies, and it can take years to gain formal approval (Intarini et al., 2015), which explains why less than 1% of Indonesia's forests are managed by communities with HKm status (Stevens et al. 2014)[5]. This particular community gained HKm status through intense facilitation supported by an international NGO (Flora and Fauna International) that initiated a REDD+ demonstration project in the area.

The various forest designations offer alternative levels of forest protection. As such, they differ in terms of the activities that local people are permitted to undertake in the forest. Table 3 provides a summary of permitted activities by forest designation.

<Table 3>

We conducted surveys across locations A, B, C, and D (January 2015) to assess the importance that community members attach to local forest ESs across the four locations. A list of locally important forest ESs was drawn up, following scoping focus group discussions with community members and local stakeholders. These services were then grouped into three groups of provisioning services and one regulating service:

- Naturally occurring non-timber forest products (NTFP), such as bamboo, honey and cattle feed;
- Agroforest products, such as various fruits and cash crops (e.g., coffee and cacao);
- Timber forest products, including fuelwood; and

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[5] The government of Indonesia declared a plan to dramatically increase community control of forests from 1.4 million hectares in 2014 up to 12.7 million hectares by 2019 and is currently identifying the areas suitable for community forests (Indonesia National Planning & Development Agency, 2015)

- Water regulation services.

Although cultural services of forests were also identified to be significant to these forest margin communities, it is difficult to measure those services and link them to forest conditions. Thus they were not explicitly investigated in our study. The survey questionnaire comprised five sections. First, we collected background information on the respondents, including their proximity to the forest. Next, we asked a general question on the extent to which the services they obtain from the forest sustains their needs and how this has changed over the past 5 years. The third and fourth sections respectively collected detailed information on the levels of consumption of provisioning and regulating services. Finally, we collected information on respondent's preferences for alternative future forest management options. The surveys were administered in-person by (trained) local enumerators, who conducted the surveys in the respondent's home in the local language. A sampling frame was developed for identifying respondents following consultation with community leaders and aimed to obtain a representative sample of community members. Survey data was analyzed separately for the four locations. After analyzing the data, we held a workshop with community members in each location to share our findings, elicit feedback on our preliminary results, and explore possible future options to more effectively manage the forests (March 2016).

# 4 Results

## 4.1 Land use change scenarios

Three future (30-year projection) land use change scenarios were developed based on spatial data on recent land use changes (2005-2010), combined with current forest management plans obtained from key informant interviews (January 2015). The scenarios included a Business-As-Usual scenario and two management scenarios aimed at improving forest condition.

### 4.1.1 Business-As-Usual (BAU) scenario

There has been little decrease of primary forests in the study area since 2000, although secondary forest and shrubland have changed to other land uses, primarily dryland agriculture. Under this scenario, these current trends in land use change would continue unabated, resulting in ~10% of currently forested land being converted to dryland agriculture. We used the latest available land-use data (2010) as the starting point for our simulations. The projected land use changes for the next 10 and 30 years are shown in Table 4

<Table 4>

### 4.1.2 Community Partnership (CP) scenario

Forest Management Units (KPHs) in Lombok currently use a spatial planning approach, in which the remaining primary forests are defined as core protected zones, and surrounding secondary forests are designated for community use. The agencies are developing programs to assure *de facto* usufruct rights for communities and allow agroforestry development through partnership agreements (*kemitraan*) in the secondary forest (Jang and Bae, 2014). The optimistic, yet realistic, scenario would be that this program will succeed at buffering encroachment into the primary forest, and the partnership agreements will expand to all forests around Mount Rinjani managed by KPHs. The resulting land use changes would increase the area of secondary forests to the 1995 level (i.e. before the period of rapid deforestation) with 50% of forest restoration occurring in the first 10 years. In this scenario, secondary forests would include well-managed agroforestry areas with forest cover converted from shrubland (32% increase of total forests in 30 years), while the area of primary forests would remain unchanged (Table 5).

<Table 5>

### 4.1.3. Forest Restoration (FR) scenario

This scenario presents the realistic upper limit of a reforestation scenario. It would require an intervention, for example a REDD+-type carbon project, that would lead to restoring all Lombok's forests to the 1995 levels with 50% of forest restoration occurring in the first 10 years. The resulting land use changes would include 7% increase of primary forest and 56% increase of total forest in 30 years (Table 6).

<Table 6>

## 4.2 Changes in carbon stock and potential carbon market values

Table 7 shows land use changes under two scenarios compared to the BAU scenario, as well as resulting total carbon stock changes. For example, secondary forests in Lombok, which contain an average of 181.1 metric tons of carbon per ha, are projected to increase by 24,060 ha in 10 years under CP scenario (from 65,462 ha under BAU to 89,522 ha under CP scenario). After combining changes in carbon stock with all land uses, total carbon stock under CP scenario would be a 4.0 million metric $tCO_2e$ increase for the first 10-year period, and a 6.9 million metric $tCO_2e$ over the thirty year project period. FR scenario will result in increase of 4.3 million metric $tCO_2e$ from BAU scenario REL for first 10 years and 7.6 million metric $tCO_2e$ over the 30 year project period.

<Table 7>

Carbon price (USD/ metric tCO2e) in voluntary carbon market varies by sources, although it is commonly higher for forest carbon. REDD+ projects for avoided planned deforestation ($1.9) and avoided unplanned deforestation[6] ($5.5) generally resulted in forest carbon offsets whose values were lower than those from sustainable agriculture/agroforestry

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[6] Carbon credits from REDD+ projects are based on different forms of avoided emission from planned (i.e. legally authorized and documented for conversion) and unplanned deforestation, as well as forest degradation (i.e. canopy cover remaining above the threshold for definition of forest and no change in land use).

($7.4), tree planting ($8.9) and improved forest management ($9.8) projects (average prices per metric $tCO_2e$ in 2014 from Goldstein et al., 2015). Even at the lower end of carbon price ($5) and emission reduction, we can expect at least $35 million of expected value generated for a 30-year forest carbon project in Lombok (Table 8). However, this amount indicates the carbon credit potential, not necessarily the actual payments required to start a project.

<Table 8>

## 4.3 Hydrological modelling results

WaterWorld V3.31 results predicted that CP and FR scenarios would result in decreased local annual water balance and runoff in most locations in Lombok due to increased evapotranspiration from tree cover. Figure 5 shows the changes in average surface water runoff and water balance under CP and FR scenarios. The differences between catchments reflect differences in the amount of tree cover change as well as the effects of varying fog frequency, rainfall totals and slope.

<Figure 5>

The WaterWorld metric for water quality is termed the human footprint on water quality (Mulligan, 2010; Mulligan, 2013) and indicates the impact of upstream land use on downstream water quality as a percent of water that fell as rain on human impacted land uses. Water quality was predicted to increase in the afforested areas because of reduced agricultural inputs, but reduced runoff through greater evapotranspiration can also translate to concentrated pollutants downstream from the remaining agricultural lands. Since most populations are at lower elevations (e.g. residents in the city of Mataram. For the location, see Figure 1) and most forest are at higher elevations, this can mean a minimal or negative effects from increasing forest cover on water quality to downstream beneficiaries. Moreover, although increased infiltration does lead to a greater fraction of water as subsurface flow, WaterWorld V3.31 shows the impact of reduced water balance is greater so dry season flows decrease as tree cover increases in this region. Overall, the water modeling showed no net benefits from recovering tree cover in terms of water supply and water quality downstream, except locally at a few remote very cloudy sites.

### 4.4 Local perceptions of forest ESs

To assess potential impacts of future land use change scenarios on provisioning services that sustain food, energy and livelihoods of local communities, we surveyed 408 individuals across the four forest locations. During the surveys, respondents were asked to report on their household's level of consumption of forest ESs obtained from the forest (NTFPs, agroforest products, and timber products), and their perceived market values of these ecosystem services (Section 4.4.1). We also asked respondents to indicate what services they would like to see being enhanced through future forest management actions (Section 4.4.2).

#### 4.4.1 Locally important provisioning services from forests

The majority (80%) of respondents reported that their household utilizes some forest ESs (Table 9). The community near the Protection forest (B) reported highest level of use (98% of respondents), followed by A near Production forest (86%), C near Conservation forest (81%), D Community forest (53%). Agroforest products were utilized most widely (69%), while smaller portions of respondents reported utilization of Natural NTFP (49%) and Timber (47%). The specific forest products utilized vary by locations: coffee (67%), banana (56%) and fern (49%) were most popular in A community; jackfruit (86%) and banana (82%) in the B community; fern (69%) and forage (58%) in C community; and coffee (35%) and Jackfruit (34%) in D Community forest. Fuelwood collection was higher in A near the Protection forest (79%), compared to other areas around where one-third of respondents reported collection. These variations are due to differences in permitted activities across different forest designations (See Table 3), as well as ease of access to markets and other socio-economic variables. For example, a previous study showed that domestic energy needs can be often met by deadwoods and branches collected in household gardens and fuelwood extraction from forests is highly correlated with opportunity to sell fuelwoods (Lee et al. 2015).

We also explored the economic value of the products collected from different locations. To calculate these values, reported volumes collected were multiplied by reported prices. When the price was missing but the respondent reported some level of extraction, the mean price was used. To get a conservative estimate of the values and avoid outliers skewing the data, we removed the top and bottom 10% of the value distribution. Average overall values of forest ESs utilized per household per year were highest in the Production forest ($141), followed by Community forest ($116), Protection Forest ($85) and Conservation forest ($46).

Table 9 provides further detail of the distribution of values by ESs by location. Highest values were found for Palm ($918 for 6% of Community forest users), Coffee ($262 for 67% of Production forest users and $64 for 35% of Community forest) and Durian ($81 for 13% of Community forest users and $75 for 33% of Production forest users). Timber products were largely restricted to fuelwood with relatively low value ($4/household/yr). Forest products most likely to be consumed by the household are: melinjo (94%), forage (91%), jackfruit (88%), taro (83%) and fern (83%), while cacao (92%) and palm (83%) were the products most likely to be sold. Our findings demonstrate that there was a significant variability in terms of forest uses by communities.

<Table 9>

### 4.4.2 Perceived importance of forest ESs

We asked respondents to indicate which services they would like to see improved by future forest management plans. Both water regulation (91% of respondents) and agroforest products (81%) were considered to be important by most respondents; a finding that is consistent across all four forest locations (Table 10). The over-riding importance placed on water regulation can be illustrated by a comment made by one respondent "[Other ecosystem services] are what we need to live, but water is life". The higher importance ranking of agroforest products may be explained by the fact that more people used and obtained higher values of services from agroforest products than the other forest ESs categories (Table 9). Natural NTFP (40%) and timber (27%) were considered to be less important. However, there were significant differences between locations in terms of the importance of these services. Natural NTFPs were considered important (67%) in the Conservation forests, while timber resources were considered important (76%) in the production forest. These differences in preferences reflect the activities that are permitted in the different types of forest. Analysis of the socioeconomic characteristics of respondents indicated that, generally, there was little difference between the socio-economics of the people living in the different forests.

<Table 10>

# 5 Discussion

## 5.1 Forest management, PES and the delivery of global and local services.

In this research, we explored the potential impacts of alternative land use change scenarios on ecosystem services across different scales from global to landscape and local levels. Our analysis identified two scenarios: a community partnership (CP) scenario which largely focused on increasing the area of secondary forest; and a forest restoration (FR) scenario which increased the area of both secondary and primary forest. In terms of global ES, it is clear that both of these scenarios can generate significant global carbon benefits: over a 30-year period the CP scenario was estimated to generate between $35 million to $69 million in carbon values, while the FR scenario would generate between $38 million and $76 million (at carbon price $5 to $10 per metric $tCO_2e$). Impacts of recovering primary and secondary forests on the ESs at landscape and local levels are less clear. The results from the global hydrological model, WaterWorld V3.31, employed here showed that the impacts of alternative scenarios on the delivery of watershed services are generally negative at the whole island scale. However, the community surveys showed that local community members strongly believe that declining of watershed services, especially water yield during dry season, is linked to historical events of deforestation and forest degradation.

In terms of local ESs, greatest benefits per household are found where communities are allowed to cultivate and utilize agroforest products (Table 9). Extraction of natural NTFP and timber is important to some, but generally are valued less. Estimation of an aggregate value of the local ESs in our study area is difficult due to overlapping land use classes and forest functions (Table 3) and also uncertainty of land tenure arrangements. For our analysis, we aggregated the average annual household value of forest ESs for each forest type with the number of households in our study area that have agriculture as their main occupation (Table 11). Our target population for this aggregation was the 23 sub-districts surrounding mount Rinjani. These sub-districts had a population of 1.313 million (with average household size of 3.57) as of 2010 and about 51.5% of population in the area reported agriculture as their main occupation, according to the latest census (BPS/NTB, 2012). The total value of locally provided forest ESs, we aggregate the average household values (Table 9) to the 51.5 % of households (Table 9). The value of local ESs delivered by forests of Lombok is currently estimated at $16 million to $18 million annually. Aggregated (undiscounted) over 30 years, the total value ranges from $486 million to $564 million.

<Table 11>

To allow a comparison of the carbon values (Table 8) with changes in values of locally provided forest ESs under different land use scenarios, we assume increase in forests in CP and FR scenarios (shown in Table 5 and 6) would be distributed to different forests according to the current ratio.[7]

<Table 12>

Although the predicted changes in locally provided forest ESs values associated with the CP or FR scenarios are approximate, we can demonstrate that these values are higher than the carbon values ($35.7 - $69m over 30 years for the Community Partnership scenario and $38- $76m for the Forest Restoration scenario).

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[7] Forests in the NTB province includes 20% production forest, 48% protection forest and 32%conservation forest.

Opportunity costs are the forgone economic benefits of alternative land use, in this case dryland agriculture. Communities in the area cultivate various crops, including maize, chili, cassava, peanuts, etc (Collins Higgins Consulting Group, 2012). Lombok is also one of the largest producers of tobacco in Indonesia (Lee et al. 2015). Profitability of dryland agriculture varies a great deal among different varieties of crops and year-to-year. For example, tobacco can go from a net profit to a net loss depending on weather conditions ($465 to $1,132/ ha under normal condition to -$371 to -$477/ha in a bad year e, g, 2002) (Keyser and Juita, 2005). Net revenue from maize in similar areas has been reported around $180/ha/yr (Da Silva and Murdolelono. 2010). Table 13 presents opportunity costs of carbon sequestration undiscounted and Net Present Value (NPV) with 10% discount rate over 30-year period per metric tCO2e with a range of per ha profitability (following the methodology described in White et al. 2010). Opportunity costs are lower than the current carbon price.

<Table 13>

Here we can draw a number of broad conclusions on the ES associations and potential effects of global PES scheme. First, the value of local ESs are potentially greater than that of global ES (carbon) and opportunity costs are low. Thus, carbon PES schemes (such as REDD+) need to be developed in a way to maximize synergies among global and local ESs. Carbon payments can provide the initial capital investment needed for creating nurseries and planting trees, but recovered forests can also provide income overtime for communities to maintain forests. Each community can develop a benefit-sharing mechanism under the partnership agreement *(kemitraan*) with KPHs or through Community Forest arrangement. For example, community D has started tree planting projects with REDD+ demonstration fund facilitated by an NGO (FFI/Indonesia). The species selection was negotiated with the community, and the result was mostly fruit trees planted. Second, higher benefits can be obtained by encouraging secondary forests (retaining artificial gaps in the canopy to 50-60%), while meeting community needs for NTFP, agroforest products and timber. Community partnership scenario is focusing on recovery of secondary forests, which is possible through agroforestry with forest covers. A previous study in the area shows that carbon stored in agroforestry land with significant forest cover (178 metric ton/ha, Markum et al. 2013), is similar to that in secondary forests (181 metric ton/ha, Table 2). Forest Restoration scenario included additional reforestation to recover primary forests. From the community point of view, primary forest does not generate significant economic revenues, although there may be cultural and religious significance that this study did not capture. Additional carbon payment expected from primary forest can motivate communities to recover primary forests for conservation purposes.

### 5.2 Data discrepancies: reconciling global modelling and local perceptions

A key debate in ecosystem service assessments relates to identifying what is the most appropriate source of data to measure ecosystem service change (TEEB, 2010). Evaluating watershed services is especially challenging because hydrological impacts can occur anywhere downstream of the site of service production (van Noordwijk et al. 2016). It is not easy to discern the roles of land use change from other influencing factors, such as climate variability, landscape-level changes, and spatial distribution of soil and vegetation types (Bruijnzeel, 2004). In this research, we used both global models (e.g. WaterWorld V3.31) and local knowledge (in-person surveys) to assess the impact of forest management on water regulation. Global models have a wide appeal in that they are usually based on the theoretically sound scientific knowledge and can be applied almost anywhere in the world at relatively low costs. In the absence of long term observation records, collecting local data may require surveys with local stakeholders/communities, which is often based on implicit and experiential knowledge rather than scientific evidence (Christie, 2012). In our research, we found discrepancies between these two data sources, particularly in terms of the predicted impact of forest management on water regulation services.

WaterWorld V3.31 showed that more tree cover decreases baseflow in both dry and wet seasons in most places due to increased evapotranspiration, while increasing baseflow in some places due to enhanced infiltration. This is supported by many studies that indicate higher evapotranspiration of trees than other cover types (Kaimowitz, 2004; Calder, 2001; Van Dijk et al., 2007). The overall effects of both scenarios were negative on watershed services. However, residents frequently reported contrasting views based on experience and observation. In surveys conducted in Lombok communities in 2002, residents reported that springs had gone dry in response to forest clearing (WWF 2002). According to Pirard (2011), 43% of the large springs surrounding Rinjani dried up in the decade 1992-2002, while approximately 30% of the Mount Rinjani was deforested during the same decade. Klock and Sjah (2012) reported that, during the previous twenty years, more than 400 springs dried up on Mount Rinjani, most likely from deforestation. The Jakarta Post (2014) reported that there are 107 springs currently utilized in Lombok, with many other sources not yet recorded by the government and under the control of local residents. In the above article, a local Village Head is quoted as emphasizing the function of forests as a sponge, absorbing water and releasing it gradually, thus enhancing water regulation and quality. Our community survey also confirm that water regulation was considered important to people living in the forest margins and the follow-up focus group discussions highlighted the strong local belief that retaining and enhancing forest cover protected water supply and water quality.

The prevailing scientific paradigm for linking forests to water has shifted since the early 1980ies when several reviews, both in the temperate zone and the humid tropics, show that there is little empirical evidence for forests storing excess water during wet periods and releasing it during dry periods, so called *sponge theory* (Bosch and Hewlett 1982; Ghimire et al. 2014a; Ilstedt et al. 2016). Since then, many studies supported *trade-off theory,* which means less water yields with increasing tree covers (Ilstedt et al. 2016). Deforestation, especially in the tropics, does contribute soil degradation and increase in impermeable surface, which lead to locally observed negative hydrological effects. However, there is limited evidence for reforestation increasing soil hydraulic conductivities (Ghimire et al. 2013; Ghimire et al. 2014a; Ghimire et al. 2014b). Moreover changes in water resources reflect not only the changes in ecosystem services (modelled here) but also the impacts of farmer behavior of water use and irrigation practices, which was not part of this study. Also, relying only on anecdotal data could lead to an erroneous conclusion regarding changes in spring discharge conditions caused by forest change. As noted above, illegal logging, encroachment and occupation reached its peak after the fall of the Suharto regime in 1998. Loss of forest cover notwithstanding, climate variation could have had a bearing on residents' perception of the effects of forest clearing. Long-term precipitation records shows that there are a great deal variations in precipitation during dry season among different locations and also years leading up to 1998 were dry, especially around the Mataram city in low elevation. Figure 6 shows average precipitation records from six weather stations around the city of Mataram and four weather stations near the survey locations around Rinjani Mountain. It is very possible that declining spring discharge was more directly related to climate than to land use change. Furthermore, the existence of the PES mechanism between the city of Mataram and the communities in their upper watershed area may have raised expectation of forest-margin communities that they may be able to be compensated for managing forest for watershed services that they provide. It may be especially true for the community D that gained Community Forest recognition and their forest represents important watershed for another city (city of Praya).

<Figure 6>

WaterWorld V3.31 simulates the impacts of forests versus other land uses on hydrological impacts based on high resolution remotely sensed data. It is a very detailed process model developed specifically for data poor mountainous and tropical environments. However, its results can not be field-validated without long-term spring discharge measurement data, and are not without limitations. Change in land cover and forest canopy structure have complex effects on fog input, rainfall interception, throughfall, stemflow, infiltration and runoff generation (Bruijnzeel et al., 2011; Dietz et al., 2006, Bruijnzeel et al., 2004). Some have argued that in contrast to other land use cover types, natural and recovered tropical rainforests throughout the world exhibit greater leaf litter, soil organic matter, and soil bioturbation by roots and fauna, as well as less soil surface sealing due to rainsplash, soil compaction by farm equipment, and impervious surface as part of infrastructure, all of which allow for enhanced infiltration and reduced soil erosion (Kumagai et al. 2009, Hairah et al. 2006, Bruijnzeel et al. 2004, Calder 2001, Mapa 1995). The net result of enhanced infiltration beneath recovered forest can be greater groundwater recharge, which can lead to improved dry season baseflow (Dias et al. 2015, Ogden et al. 2013, Peña-Arancibia et al. 2012, Bruijnzeel et al. 2006, Bruijnzeel et al. 2004). Forests do tend to increase evapotranspiration substantially compared with rain-fed agriculture and even higher infiltration rates cannot compensate for less water being available for infiltration and runoff. However, this basic assumption may be problematic in a tropical setting where atmospheric moisture is abundant; low vapor pressure deficit may result in reforestation having a negligible effect on evapotranspiration (Brauman 2012). Malder et al. (2013) argued that the data to formulate hydrological effects of land use change in global models are often generated outside the tropics with stable soil conditions and there is "complete lack of research on how forestation on degraded land affect hydrological functioning at the landscape scale." Empirical long-term spring discharge measurement data are needed to compliment and refine global models based on globally available datasets, in order to accurately evaluate land management practices that enhance watershed services (Wohl et al. 2012, Jose 2009, Locatelli and Vignola 2009).

What is clear from the above discussions is that there are number of factors that might affect the accuracy of both the global models and local opinions. Simply focusing on increasing tree covers can have negative impacts on watershed services and set up false expectations among local communities. For example, empirical studies in other seasonal dry tropics showed that reforestation with pine species in densely populated areas did little to increase soil hydraulic conductivities while increasing water uses of vegetation (compared to pasture) (Ghimire et al. 2013; Ghimire et al. 2014a; Ghimire et al. 2014b). Another study showed that hydrological benefits of reforestation can be maximized by considering the rates of evapotranspiration of different tree species, as well as tree size, age and density in planning reforestation projects (Ilstedt et al. 2016). Thus, global PES schemes must consider further details within a land use class (e.g. species selection, tree density, soil management, and landscape configurations) and measures to mitigate potential negative impacts.

### 5.3 Tradeoffs and synergies between global and local ecosystem services

Globally, simply ending the land use, passive restoration, has been shown to be more cost-effective than active restoration (Meli et al. 2017). However, in a densely populated region with complex social dynamics, protection of forest as carbon stock would be costly and ineffective (Skutsch et al. 2011). In both land use change scenarios, there is potential for developing forest carbon projects in the study area. Although on-site opportunity costs were low, social and indirect costs can be substantial (White et al., 2010). Most of the global forest carbon projects are financed as input-based projects, which often set a flat-rate payment per hectare under a contractual agreement of inputs to increase carbon stock (e.g., not cutting trees, tree planting or other management activities) (Wunder, 2008; Skutch et al., 2011). Input-based carbon projects allow the inputs (e.g. agreed management actions) to be negotiated between project proponents and local communities, which makes the projects less politically contentious and allows broader management goals to be addressed (Skutch et al., 2011). However, input-based projects would likely generate fewer carbon credits overall while making it difficult to trace carbon to project activities (Skutch et al., 2011). Lack of reporting on actual performance of existing projects, in terms of carbon sequestration, poses a serious problem for the future of global carbon financing (Fischer et al., 2016).

We previously advocated for an input-based mechanism with readiness activities for capacity building of both institutions and communities in the study area (Kim et al. 2016). The results of this study show that simply increasing tree cover is not enough for enhancing ES at all scales. Reforestation to increase carbon stock without considering the landscape as a whole can have negative impacts on watershed services (e.g. reduced runoff, and concentrated pollutants downstream from the remaining agricultural lands). In addition, implementing reforestation projects without consideration for local livelihoods can be detrimental to forest-margin communities. Thus the details of agreed-upon management actions would dictate the nature of association among different ESs.

Previous studies argued that global forest carbon projects are unlikely to succeed without addressing food, energy and water provisions at the local level (Minang and van Noordwijk, 2013; van Noordwijk et al., 2016). Indeed, the findings from our community study demonstrate that local people obtain a wide range of benefits from forests. Mixed agroforestry systems can be a key strategy for increasing the multi-functionality of land uses (Minang et al., 2014) as well as enhancing the diversity of local communities' livelihood options (Hoang et al., 2014). Potential values of agroforestry systems for integrating forests into a multifunctional landscape have been recognized, although the benefits may vary depending on practices and landscape configurations (Table 9; Dewi et al., 2013; Prabhu et al. 2015). It has been shown that intermediate tree cover can maximize ground water recharge in seasonally dry tropics (*optimal tree cover theory*)(Ilstedt et al. 2016). Impacts of agroforestry systems on the landscape's ability to provide watershed services vary depending on species selection of crops and shade trees and different cultivation practices employed (Condon et al. 2002; Thierfelder et al. 2009). For example, different tropical tree species have shown a wide range of production rates per cost of water loss by transpiration (Cernusak et al. 2007) and different root depths for promoting soil infiltration of rainfall (Ghestem et al. 2011). Local communities that we surveyed also recognized specific "watershed trees" e.g. Beringin (*Ficus benjamina*), where soils underneath were observed to be more moist, compared to other fast growing species, e.g. Sengon (*Albizia chinensis*). Also the amount of water needed to produce different agroforestry crops varies greatly. For example, coffee and cacao tend to have high water footprint (about 22,900 m$^3$/ton for coffee and 9,414 m$^3$/ton for cacao), compared to other crops (e.g. 514 m$^3$/ton for cassava) (Bulsink et al. 2009). Thus it is essential for forest carbon projects to consider the effects of increasing tree covers, along with species, size, and age distribution, on a range of ESs in the landscape and mitigate potential negative impacts. van Noordwijk et al. (2016) discussed several metrics for developing mitigation actions through agroforestry that can enhance different watershed services, including water yield, water flow and water quality, while improving local livelihoods. The plausible actions that can be incorporated into forest carbon projects include replacing fast growing tree plantations with low-evapotranspiration species and increasing presence of deep rooted trees while promoting litter layers and agricultural practices that increase infiltration and soil water content, enhancing sediment filter strips in fields and across landscape matrix, as well as protecting river banks, riparian zones and landslide-prone slopes, springs and sources of domestic water use.

It is clear from the community surveys that the value of forest ESs to local communities is significant but vary by locations. Although it is difficult to fully untangle the underlying reasons for this, these differences are reflective of different designated functions of forest, suitability of land for agroforestry, and the security of land tenure. Community partnership scenario focused on recovery of secondary forests through agroforestry to provide food, energy and livelihood options for local communities. However, the synergy among global, landscape and local ESs can be created only if the clear accountability can be established for maintaining the threshold of forest covers (for carbon accounting) with specific species selection and agroforestry practices to increase soil infiltration and water use efficiency (for watershed services). Although the Forest Restoration scenario adds recovery of primary forests, local communities may lack motivation for restoration activities for ecological benefits alone. Global PES, such as REDD+, can help establishing technical guidelines for agroforestry practices that maximize carbon and watershed benefits, as well as developing community monitoring schemes, while promoting ecological restoration of primary forest with added carbon values under Forest Restoration scenario.

## 6 Conclusions

In this paper, we assessed realistic forest management scenarios for reforestation in Eastern Indonesia and their effects on both global and local ES provisions. We have demonstrated that reforestation to increase carbon, i.e. global, ex-situ, ecosystem services, can have varying impacts on those ESs recognized locally. In particular, our results point to the significance of water regulation, agroforest products, and non-timber forest products to local communities. To create a sustainable local solution, we need to go beyond the zero-sum argument of livelihoods versus conservation. We demonstrated how global PES, such as REDD+, and landscape level PES, such as payment for watershed services, can help create, not dictate, such solution through agroforestry that meets global, landscape and local demands for ESs.

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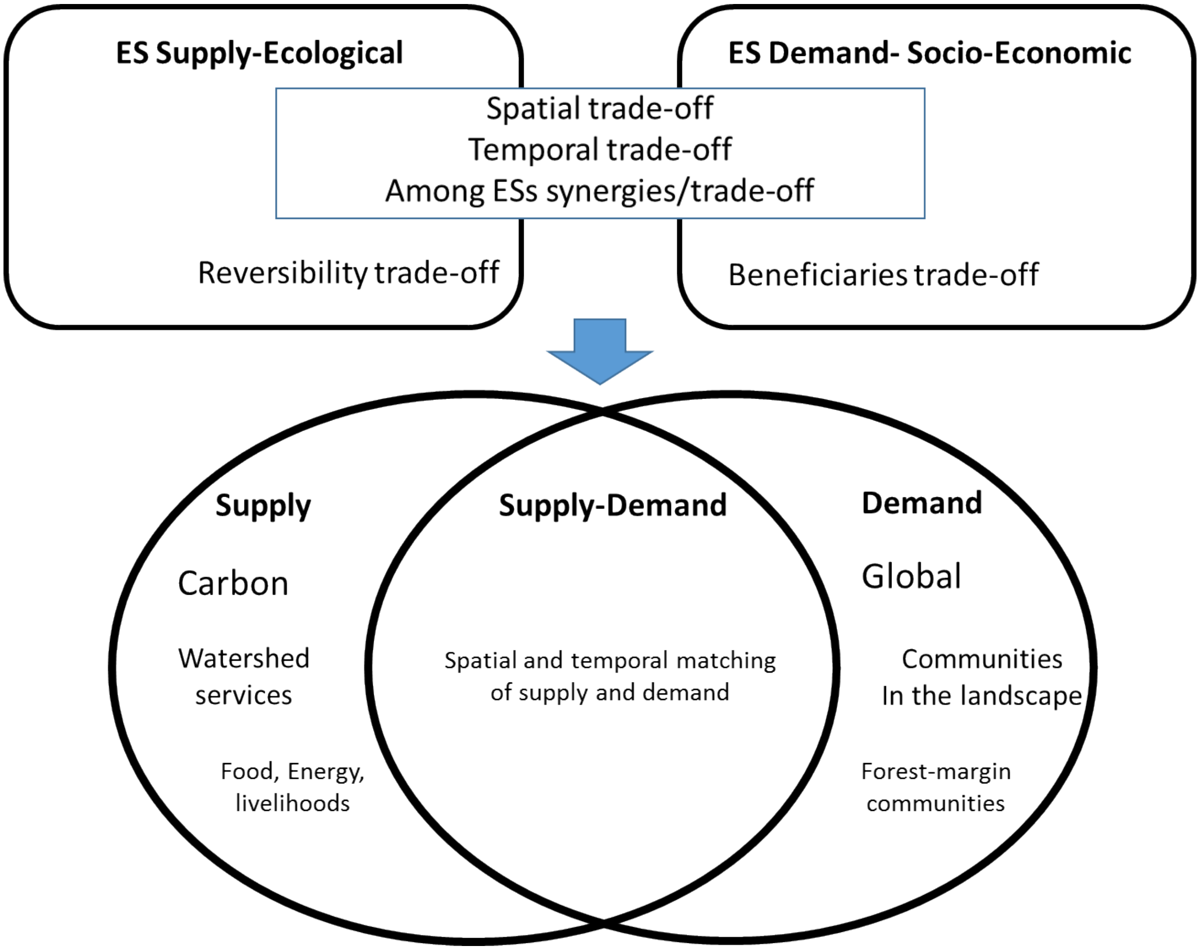


Figure 1: Conceptual framework to assess ecosystem services trade-offs (modified from Mouchet et al. 2014)

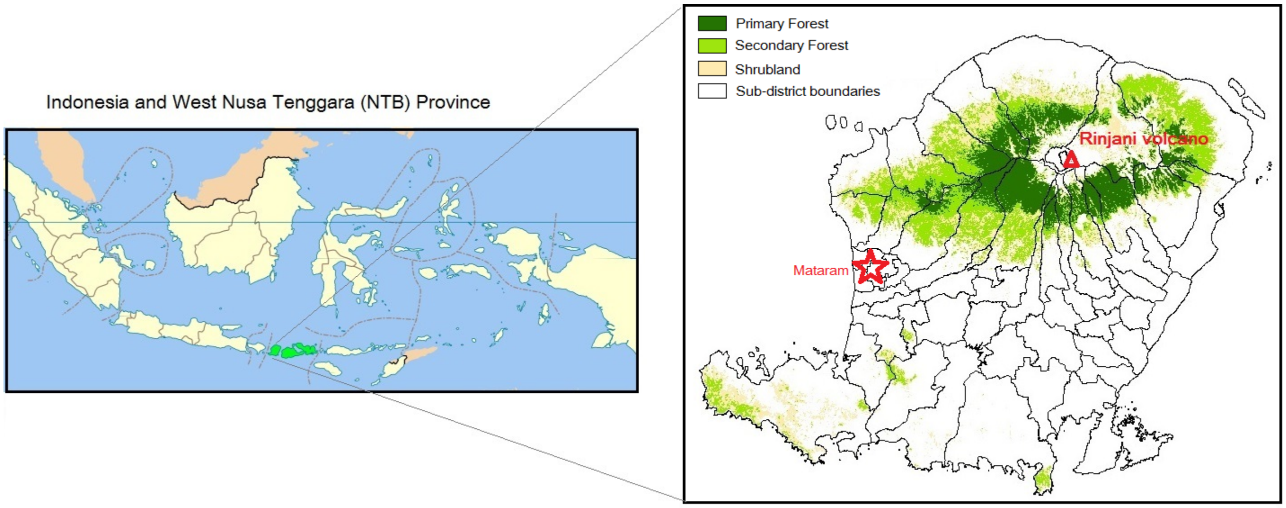
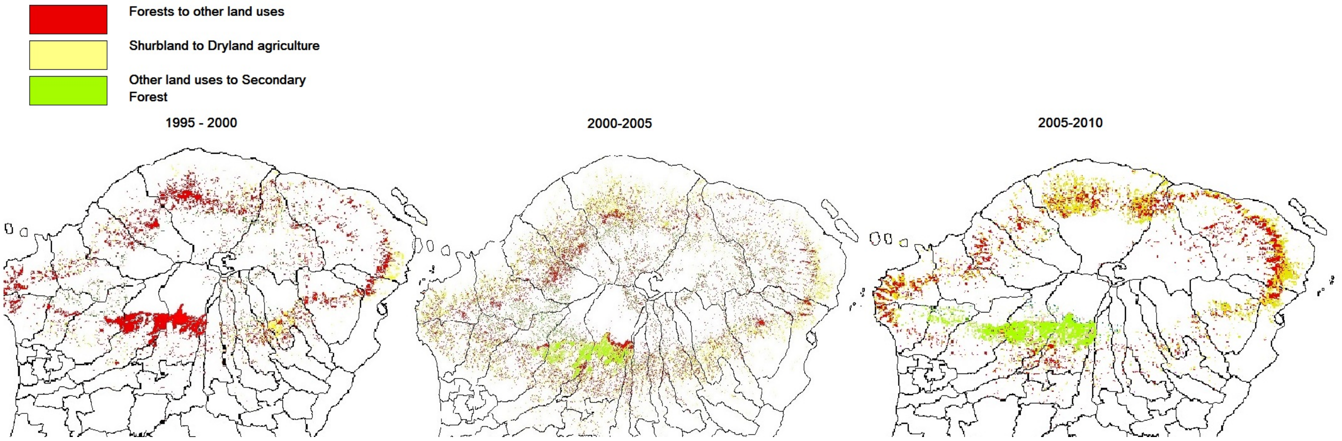


Figure 2: Map of West Nusa Tenggara province and the remaining forests in Lombok island (Source: National Institute of Forest Science, Republic of Korea)



1995 - 2000

2000-2005

2005-2010

Figure 3. Changes in forested area for three 5-year periods (Data source: National Institute of Forest Science, Republic of Korea).

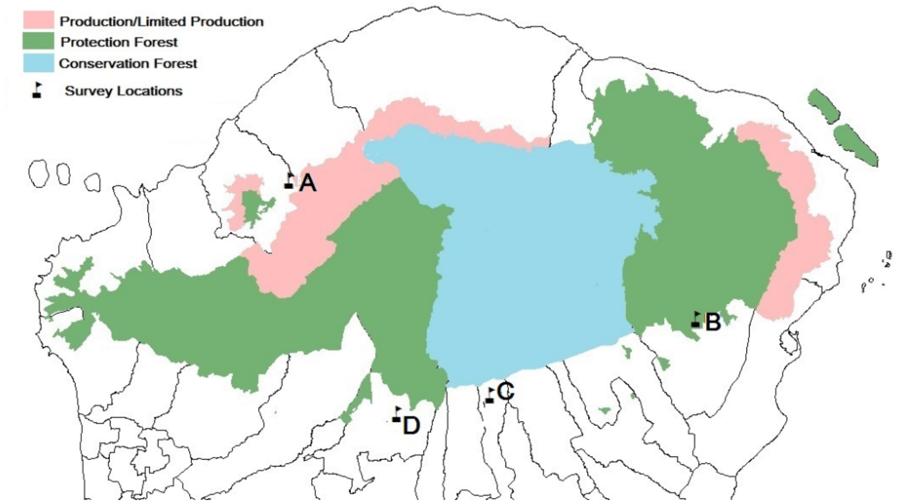


Figure 4: Survey locations (A, B, C, D) and designated forest functions.

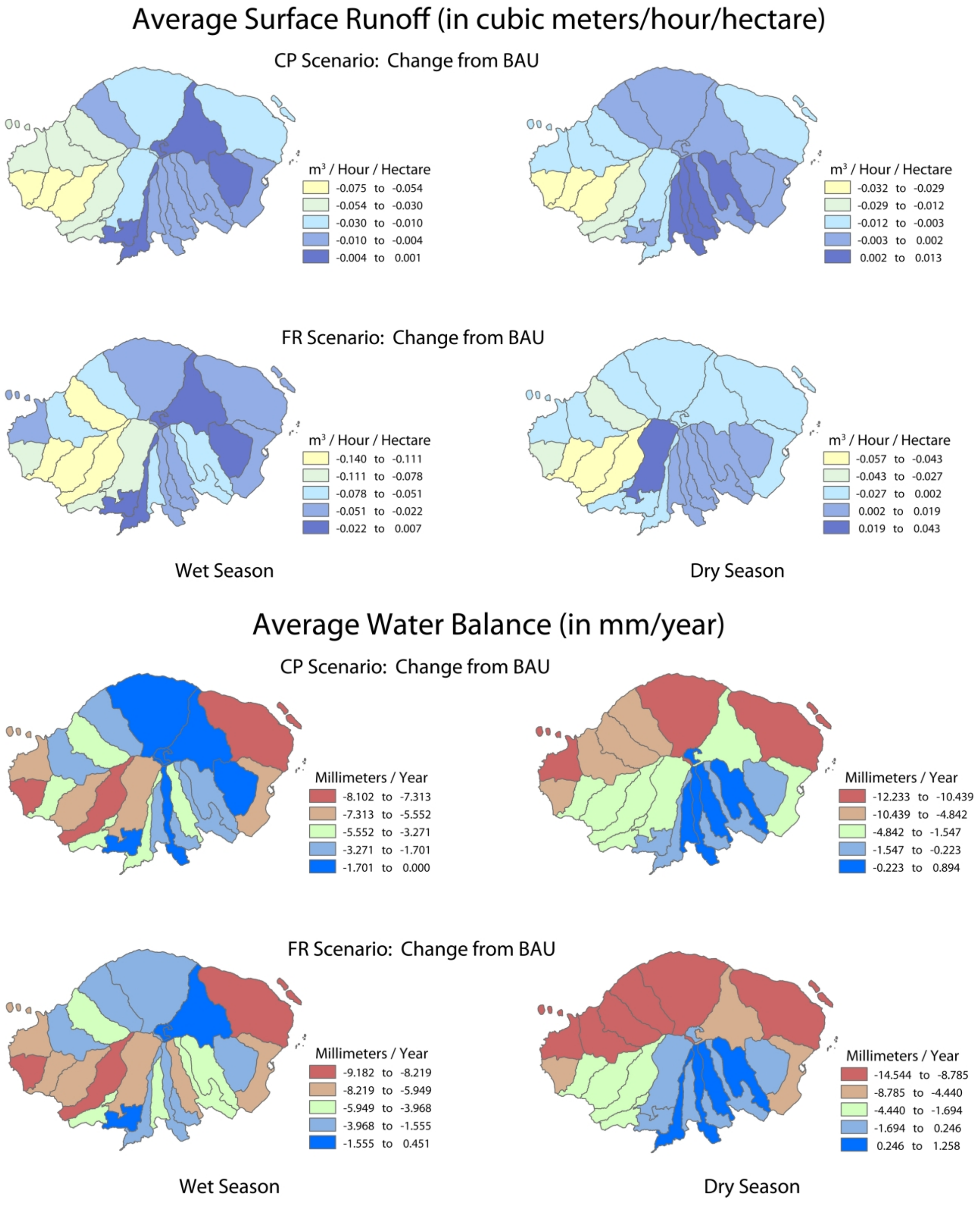


Figure 5: Changes in Average Surface Runoff ($m^3$/hour/ha) and Water Balance (mm/year) from recovery of secondary forests in Community Partnership (CP) scenario and recovery of secondary and primary forests in Forest Restoration (FR) scenario

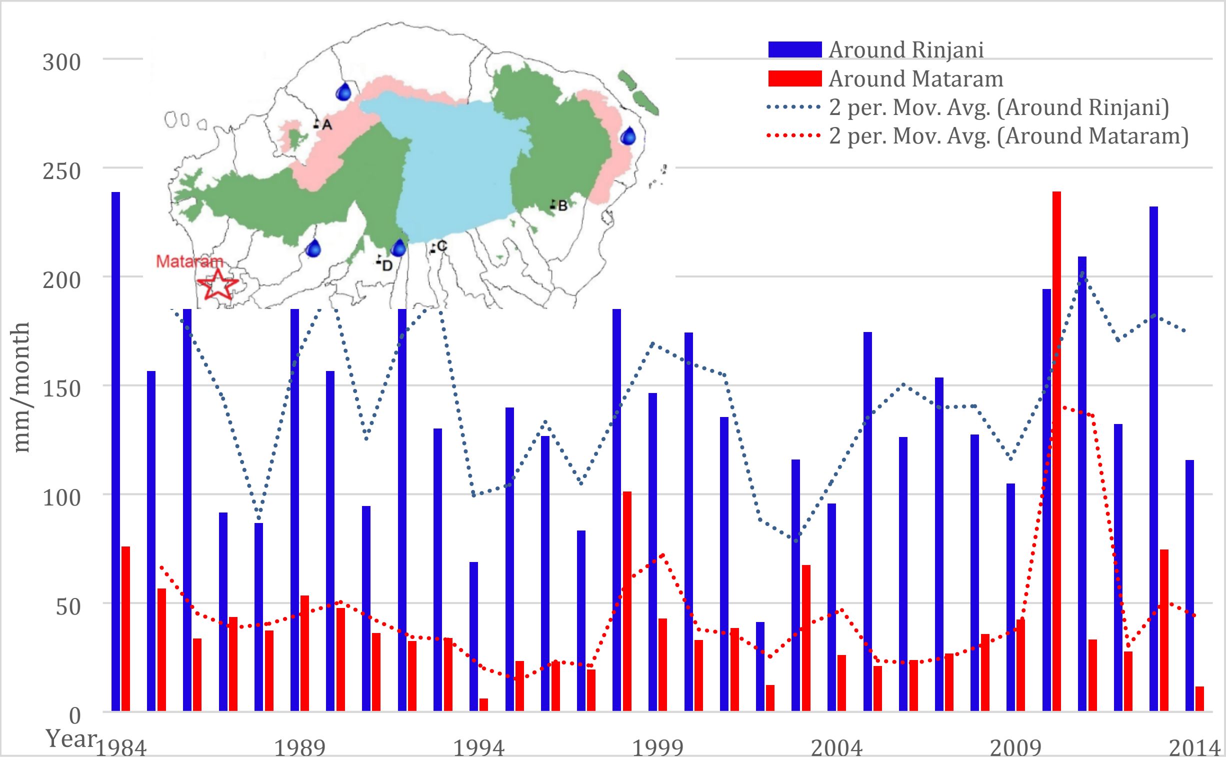


Figure 6: Precipitation records from 1984 to 2014 during dry season: average precipitation from four weather stations near the survey sites around Rinjani Mt and average of six weather stations around the City of Mataram, Lombok, Indonesia (Source: Information Board of Water Resources Province of NTB , 2016)

Table 1: Historical Land Use Changes in Lombok (Unit: ha; Source: Bae at al. 2014)

| Land Use Class | 1995 | 2000 | 2005 | 2010 | Changes 1995-2000 | Changes 2000-2005 | Changes 2005-2010 |
|---|---|---|---|---|---|---|---|
| Primary forest | 54,881 | 53,140 | 51,114 | 51,111 | -1,741 | -2,025 | -4 |
| Secondary forest | 105,064 | 77,452 | 69,752 | 67,258 | -27,612 | -7,700 | -2,494 |
| Shrubland | 12,767 | 33,627 | 42,052 | 34,419 | 20,859 | 8,425 | -7,633 |
| All other uses | 285,495 | 293,989 | 295,289 | 305,419 | 8,494 | 1,300 | 10,131 |

Table 2: Carbon stock by land use type (metric ton of carbon/ha ± standard deviation) (Source: Bae at al. 2014)

|  | Total | Living vegetation |  |  |  | Dead trees | Litters | Soils |
|---|---|---|---|---|---|---|---|---|
|  |  | Aboveground |  |  |  |  |  |  |
|  |  | Sub-total | Tree | Under growth | Below Ground |  |  |  |
| Primary forest | 206.6<br>(±76.66) | 109.9 | 108.6<br>(±59.89) | 1.3<br>(±1.15) | 29.7<br>(±16.12) | 18.3<br>(±26.05) | 1.7<br>(±1.25) | 47.0<br>(±17.52) |
| Secondary forest | 181.1<br>(±120.88) | 97.8 | 96.2<br>(±85.74) | 1.6<br>(±0.99) | 26.4<br>(±23.03) | 21.4<br>(±31.73) | 1.8<br>(±0.84) | 33.7<br>(±13.08) |
| Shrub land | 75.3<br>(±6.74) | 26.5 | 24.8<br>(±2.30) | 1.7<br>(±0.98) | 7.2<br>(±0.89) | 16.7<br>(±6.76) | 1.6<br>(±0.43) | 23.4<br>(±3.72) |

Table 3: Forest Classification and Permitted Activities (Source: Rosenbarger et al. 2013[1])

| Forest classification by function/ Permitted activities[2] | Timber Extraction | Cultivating medicinal/decorative plants, fungi, apiculture, swiftlet nests, capturing wildlife, cattle feed | Utilization of environmental services (water flow, ecotourism, biodiversity, environmental protection, carbon absorption and storage) | Extraction of non-timber forest products (rattan, bamboo, honey, resin, fruits, fungi) | Research, science, education, cultivation activities, cultural activities, and limited tourism |
|---|---|---|---|---|---|
| Production Forest (A) | Y[3] | Y | Y | Y | Y |
| Protection Forest (B, D[4]) |  | Y | Y | Y | Y |
| Conservation Forest (C) |  | Y[5] |  | Y[5] | Y |

[1] Compiled from: Government Regulation No. 6 of 2007, Minister of Forestry Regulation No. 13 of 2009, Minister of Forestry Regulation No. 37 of 2007, Minister of Forestry Regulation No. 49 of 2008.

[2] These activities can be legally allowed with permits granted by regent/mayor/governor or minister (depending on area jurisdictions). Although these activities reflect *de facto* uses, two communities in the study area (A and B) do not hold permits.

[3] There is no timber concession in the study area.

[4] The "Community Forest" status of community D means that the forest utilization permit (IUPHKm) was granted to this community for a period of 35 years.

[5] These activities are not allowed in Conservation Forest, but the community C is in "Traditional Zone", specially designated for very limited community uses for their livelihoods, including collecting cattle feeds.

Table 4: Potential Land Use Changes under the Business-As-Usual Scenario (ha)

| Land Use Class | Present | In 10 years | In 30 years |
|---|---|---|---|
| Primary forest | 51,111 | 51,111 | 51,111 |
| Secondary forest | 67,258 | 65,462 | 60,537 |
| Shrubland | 34,419 | 29,030 | 14,255 |
| All other land uses | 305,419 | 312,604 | 332,304 |

Table 5: Potential Land Use Changes under the Community Partnership Scenario (ha)

| Land Use Class | Present | In 10 years | In 30 years |
|---|---|---|---|
| Primary forest | 51,111 | 51,111 | 51,111 |
| Secondary forest | 67,258 | 89,522 | 105,064 |
| Shrubland | 34,419 | 33,675 | 12,767 |
| All other land uses | 305,419 | 283,899 | 289,265 |

Table 6: Potential Land Use Changes under the Forest Restoration Scenario (ha)

| Land Use Class | Present | In 10 years | In 30 years |
|---|---|---|---|
| Primary forest | 206.6 | 52,996 | 54,881 |
| Secondary forest | 67,258 | 89,522 | 105,064 |
| Shrubland | 34,419 | 33,675 | 12,767 |
| All other land uses | 305,419 | 282,014 | 285,495 |

Table 7: Land use and Carbon stock change under CP and FR scenarios

| Land Use Class | Carbon Stock (metric ton /ha) | Community Partnership scenario (change from BAU) (ha) | | Forest Restoration scenario (change from BAU) (ha) | |
|---|---|---|---|---|---|
| | | in 10 years | in 30 years | in 10 years | in 30 years |
| Primary forest | 206.6 | 0 | 0 | 1,885 | 3770 |
| Secondary forest | 181.1 | 24,060 | 44,527 | 24,060 | 44,527 |
| Shrubland | 75.3 | 4,645 | -1,488 | 4,645 | -1,488 |
| All other land uses | 23.4 | -28,705 | -43,039 | -30,590 | -46,809 |
| Total carbon stock change (metric tCO2e) | | 4,035,338 | 6,944,681 | 4,380,670 | 7,635,345 |

Table 8: Potential Undiscounted Total Market Values of Forest-sequestered Carbon in Lombok (USD millions).

| Carbon Price (USD/ metric tCO2e) | Carbon Value (in USD millions) | | | |
|---|---|---|---|---|
| | Community Partnership | | Forest Restoration | |
| | 10-year | 30-year | 10-year | 30-year |
| $5 | 20.18 | 34.72 | 21.90 | 38.18 |
| $7.50 | 30.27 | 52.09 | 32.86 | 57.27 |
| $10 | 40.35 | 69.45 | 43.81 | 76.35 |

Table 9: Level of use (% of respondents reporting collection from forests) and value of forest ESs (USD/household/yr)

| **Type of service** | **Forest ESs[1]** | **Production forest (A)** | | **Protection forest (B)** | | **Conservation forest (C )** | | **Community forest (D)** | | **All forests** | | **%** | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | **%** | **Value** | **%** | **Value** | **%** | **Value** | **%** | **Value** | **%** | **Value** | **Con-sumed** | **Sold** |
| **Natural NTFP** | Bamboo | 2 | 18.52 | 18 | 13.35 | 6 | 4.23 | 26 | 11.25 | 13 | 10.83 | 51 | 49 |
| | Forage | 5 | 31.11 | 15 | 39.21 | 58 | 44.39 | 10 | 26.67 | 22 | 40.49 | 91 | 9 |
| | Fern | 49 | 4.22 | 4 | 1.63 | 69 | 1.48 | 13 | 5.04 | 34 | 2.86 | 83 | 17 |
| | **Sub-total** | **50** | **8.41** | **32** | **20.21** | **81** | **27.14** | **33** | **14.37** | **49** | **18.18** | | |
| **Agroforest Products** | Jackfruit | 13 | 2.79 | 86 | 4.23 | 49 | 2.47 | 34 | 3.31 | 46 | 3.47 | 88 | 12 |
| | Durian | 33 | 74.80 | 7 | 38.27 | 8 | 16.89 | 13 | 81.63 | 16 | 66.46 | 60 | 40 |
| | Avocado | 17 | 8.63 | 29 | 18.45 | 43 | 5.42 | 3 | 18.04 | 23 | 10.20 | 44 | 56 |
| | Mangosteen | 3 | 18.89 | 0 | 0.00 | 0 | 0.00 | 1 | 18.52 | 1 | 18.80 | 44 | 56 |
| | Melinjo | 3 | 1.44 | 13 | 2.55 | 0 | 0.00 | 0 | 0.00 | 4 | 2.31 | 94 | 6 |
| | Cacao | 28 | 15.99 | 14 | 9.94 | 0 | 0.00 | 0 | 0.00 | 11 | 13.74 | 8 | 92 |
| | Coffee | 67 | 262.39 | 24 | 50.40 | 0 | 0.00 | 35 | 63.82 | 32 | 171.94 | 50 | 50 |
| | Banana | 56 | 14.95 | 82 | 15.01 | 0 | 0.00 | 23 | 13.66 | 42 | 14.89 | 36 | 64 |
| | Taro | 2 | 14.07 | 2 | 2.93 | 0 | 0.00 | 3 | 4.19 | 2 | 7.27 | 83 | 17 |
| | Palm | 0 | 0.00 | 0 | 0.00 | 0 | 0.00 | 6 | 918.52 | 1 | 918.52 | 17 | 83 |
| | Candlenut | 0 | 0.00 | 16 | 15.75 | 5 | 16.44 | 3 | 7.03 | 6 | 14.87 | 31 | 69 |
| | Other | 0 | 0.00 | 18 | 117.18 | 1 | 6.73 | 1 | 13.46 | 5 | 117.45 | 76 | 24 |
| | **Sub-total** | **84** | **142.86** | **96** | **49.04** | **57** | **14.15** | **40** | **103.89** | **69** | **77.70** | | |
| **Timber products** | Fuelwood | 35 | 7.17 | 80 | 3.59 | 36 | 2.99 | 35 | 5.92 | 48 | 4.56 | 87 | 13 |
| | Tools | 4 | 0.74 | 0 | 0.00 | 0 | 0.00 | 1 | 1.85 | 1 | 0.96 | 100 | 0 |
| | **Sub-total** | **37** | **6.66** | **79** | **3.59** | **37** | **2.99** | **34** | **5.40** | **47** | **4.41** | | |
| | **All forest ESs[2]** | **86** | **141.49** | **98** | **84.98** | **81** | **46.25** | **53** | **115.63** | **80** | **93.46** | | |

[1] No uses were reported for some NTFPs (e.g. langsat, and rattan) and timber products (materials for building and fencing).

[2] Total % of respondents whose household obtained some values from forest ESs; Mean aggregate value of services obtained from the forest (USD/household/yr).

Table 10: Importance of local forest ESs in future forest management plans by study location

| Forest service | Production forest | Protection forest | Conservation forest | Community forest | All respondents |
|---|---|---|---|---|---|
| | % of respondents stating that forest service was important | | | | |
| Natural non-timber forest products | 44 | 26 | 67 | 24 | 40 |
| Agroforest products | 92 | 70 | 1 | 86 | 81 |
| Timber forest products | 78 | 10 | 1 | 17 | 27 |
| Water regulation | 96 | 90 | 88 | 90 | 91 |

Table 11: Aggregate value of locally provided forest ESs

| | Value per year (USD/Household) [1] | Number of affected Households[2] | Value per year (million USD) | Undiscounted value over 30 years[3] (million USD) |
|---|---|---|---|---|
| Production forest | $121 | 44,104 | $6.2 | $187 |
| Protection forest | $83-$61 | 84,311 | $7.2-$9.7 | $241-$292 |
| Conservation forest | $38 | 61,044 | $2.8 | $85 |
| Total | | 189,460 | $16.2-$18.8 | $486 -$564 |

[1] $121 for Production Forest ($141 for 86% of the community utilizing forest products); $83 for Protection Forests ($85 for 98% of the community utilizing forest products) and $61 for Community Forests in Protection Forest ($115 for 53% of the community utilizing forest products) and $38 for Conservation Forest ($46 for 81% of the community utilizing forest products)

[2] Aggregated population of sub-districts near each designated forest function X 51.5% with agriculture as the main occupation based on the 2010 population census.

[3] Not accounting for population growth/discounting rate/forest product value change.

Table 12: Changes in value of locally provided forest ESs

| | Undiscounted value over 30 years[3] (million USD) | CP scenario[1] | | FR scenario[2] | |
|---|---|---|---|---|---|
| | | Forest area changes (%) | Changes in values (million USD) | Forest area changes (%) | Changes in values (million USD) |
| Production forest | $187 | 7.52 | $14.1 | 8.20 | $15.3 |
| Protection forest | $241-$292 | 18.05 | $43.5-52.5 | 19.68 | $47.4-57.5 |
| Conservation forest | $85 | 12.03 | $10.2 | 13.12 | $11.2 |
| Total | $486 -$564 | 37.6 | $67.8-76.8 | 41 | $73.9-84.0 |

[1] 44,527 ha or 37.6% increase in total forest area

[2] 48,297 ha or 41% increase in total forest area

Table 13: Opportunity costs of carbon sequestration (Value/metric tCO2e for 30-year)

| Profitability of Dryland Agriculture (USD/ha) | Community Partnership (Dryland Agriculture --> Agroforest: 44,527ha) | | Forest Restoration (Dryland Agriculture --> Agroforest: 44,527ha & 3,770 ha to primary forest) | |
|---|---|---|---|---|
| | Undiscounted | NPV with 10% discounting rate | Undiscounted | NPV with 10% discounting rate |
| $150 | $0.01 | $0.002 | $0.02 | $0.005 |
| $250 | $0.13 | $0.04 | $0.13 | $0.04 |
| $500 | $0.44 | $0.14 | $0.43 | $0.14 |
| $1,000 | $1.05 | $0.33 | $1.03 | $0.32 |
| $2,000 | $2.27 | $0.71 | $2.21 | $0.70 |

*Profitability of Dryland Agriculture/ha – ES value of Forest /ha ((Primary Forest: $54.58 = $2.8 million/51,111ha; Secondary/Agroforest: $144.22= $9.7 million/67,258ha); Primary forest = 206.6 metric tCO2e/ha; Secondary forest = 206.6 metric tCO2e/ha; Dryland Agriculture = Primary forest = 23.4 metric tCO2e/ha.