SALIENCE, CREDIBILITY AND LEGITIMACY IN LAND USE CHANGE MODELING:

MODEL VALIDATION AS PRODUCT OR PROCESS?

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The endless cycle of idea and action,

endless invention, endless experiment,

brings knowledge of motion, but not of stillness;

knowledge of speech, but not of silence;

knowledge of words, and ignorance of the Word.

...

where is the wisdom we have lost in knowledge?

where is the knowledge we have lost in the information?

(The Rock, TS. Elliot)

dedicated to:

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Ruspita, for demonstrating with ease perseverance and compromise

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List of abbreviations

BAPPEDA	: Badan Perencanaan dan Pembangunan Daerah (*Regional Planning and Developement Agency*)
BPS	: Biro Pusat Statistics (*Central Bureau of Statistics*)
CV	: Coefficient of Variation
ES	: Environmental Services
FALLOW	: Forest, Agroforest, Low-value Land or Wasteland?
IPCC	: Intergovernmental Panel on Climate Change
MRV	: Monitoring, Reporting and Verification
PES	: Payment for Environmental Services
REDD+	: Reducing Emission from Deforestation and Degradation
RTRW	: Rencana Tata Ruang dan Wilayah (*Spatial Land Use Plan*)
UNFCCC	: United Nation Framework Convention on Climate Change

Chapter 1

General Introduction

1.1 Foreword

Simulation models compile knowledge into tools that are increasingly being used in problem solving and in decision making. Such models also are used in applied situations for natural resource management by integrating multi-dimensional social and biophysical indicators. However, despite the various approaches in promoting use of simulation models as tools to support decision making in natural resource management, acceptance and use by decision makers and natural resource managers are still a challenging issue. One of the major concerns is related to the following questions "How good is model A?", "Is model A better than model B?", "'How valid are existing simulation models in addressing natural resource management issues?"

This dissertation is the result of PhD research on validation of simulation models for natural resource management. It includes studies of users' perspectives on the validity of simulation models, model application to assess trade-offs and uncertainty assessmenst for designing management intervention. This introductory chapter provides the background for the research comprising of challenges in natural resource management, in particular in the region of Southeast Asia and various tools that can be used to address these challenges. The conceptual framework, hypotheses and research questions that shaped this study are listed towards the end of this chapter.

1.2 Natural resource management: Challenges to date

Natural resource management entails integrating human needs, productivity enhancement and environmental services protection (Sayer and Campbell, 2001). Embedded within natural resource management is the concept of sustainable development and therefore sustainable natural resource management aims to manage natural resources in ways that '... meets the needs of the present without compromising the ability of future generations to meet their own needs' (World Commission on Environment and Development, 1987).

Globally, balancing land productivity, equality in resource access, and maintaining (and improving) ecosystems functions is a challenge due to development needs, population pressure and global market demand (Tilman et al., 2002; Robertson and Swinton, 2005). In Southeast Asia, population and pressures on resource use are particularly high[1] (United Nations Economic and Social Commision for Asia and Pacific, 2011). Trade-offs between sustainable natural resource management goals appear to be inevitable and have compelled many agricultural systems to operate in non sustainable ways. For example: farm-level soil degradation in rice-swidden systems in Vietnam (Lam et al., 2005), landscape-level loss of biodiversity in cocoa production in Indonesia

1 The population in Southeast Asia reached almost 600 million people in 2010, with a population density of 132 person.km^{-2}. South-East Asia lost 13% of its forest cover during the past 20 years – about 332,000 km^2, an area roughly equal to the size of Viet Nam. Indonesia alone lost around 241,000 km^2 (73% of total forest loss).

(Steffan-Dewenter et al., 2007) and biodiversity loss in global-market-pressured biofuel plantations in Indonesia and Malaysia (Danielsen et al., 2009).

Achieving sustainable natural resource management requires balancing trade-offs between sustainability goals at multiple scales. This can be achieved by regulating the use of land so that the development goal of production can be met while maintaining environmental services functions. At local level (district, provincial) or even at national scales, this activity is often carried out by policy makers/government through land use planning where, commonly, the outcomes are in form of land use zoning maps that designate the specific use of each zone. The challenges in producing such maps are to do it efficiently and ethically, allowing communities affected by the restrictions imposed to participate in development in other ways. Another challenge is ensuring that regional development will indeed be based on the land use plan and maps to achieve sustainable development while maintaining environmental integrity.

A complementary approach to manage natural services, in addition to regulation and spatial zoning, is to provide incentives to communities or regions that maintain environmental services provision, popularly termed as Payment for Environmental Services (PES) schemes. At global level, an example of such a financial incentive scheme is CDM (Clean Development Mechanism) aiming to reduce carbon emission from energy and waste sectors as well as increasing carbon sequestration through afforestation and reforestation. Another incentive scheme that is still under development is REDD (Reducing Emission from Deforestation and Degradation) mechanism. There are many challenges in developing and implementing PES (Muradian et al., 2010) which mainly concern with developing efficient schemes that can be accepted by all stakeholders and that can lead to sustainability of schemes based on real internalization of externalities (van Noordwijk et al., 2012).

To better manage natural resources in the landscape, natural resource managers and decision makers need to know the impacts and consequences of their policies on landscape dynamics, in particular productivity and environmental functions. Understanding the essential processes and behaviour of the land use systems can help in finding suitable policies and technological innovations that would allow progress towards balanced trade-offs. Thus, dynamic and efficient approaches are needed to help managers and policy makers in assessing trade-offs and choose viable options for meeting human needs and ensuring ecosystems functions (DeFries et al., 2004). Scenario analysis based on a credible simulation model is an efficient approach to assess the dynamic and complex interactions between components and their trade-offs (Carpenter et al., 2006). It can also provide plausible, challenging, and relevant projections about how the future might unfold given certain management strategies that can help decision makers consider positive and negative implications of alternative development pathways (Schneider et al., 2007).

1.3 Modelling approaches for natural resource management

There are many modelling approaches to understand, predict and manage natural resources and landscapes. However, they all have similar characteristics in that they include causal relationships, or at least an opportunity to extrapolate existing trends, in the human-environment relationship and that they conceptualize landscape as socio-ecological systems. The models need to be able to address and clarify some (if not all) of the following issues : (1) the dynamic nature of sustainability attributes, (2) the complex and nonlinear response of resources to management strategies, (3) the interactive and adaptive nature of management and landscape (4) the trade-offs involved when trying to optimize a set of linked critical indicators and (5) the need to deal with conflicts that arise between stakeholders with different and sometimes opposite interests (García-Barrios et al., 2008).

The following text describes the main modelling approaches currently used for natural resource management.

System dynamics

System dynamics (SD) is the most used modelling tool for complex systems. In a SD approach, systems are described as a set of modules or compartments (with aggregated stocks, variables, parameters) interlinked by flows which represents material fluxes of energy, matter or information. SD models can well describe macro-level processes and complexity. However, decisions and actions of multiple actors and potentially multiple spatial relationships are generally absent from SD models. The equations and feedbacks in SD are structural, and their ability to evolve is limited (Heckbert et al., 2010; Parker et al., 2003).

Cellular models

A cellular model (CM) considers landscape to be a set of cells where the future state of each cells depends on transition rules based on a local spatio-temporal neighbourhood. Cellular automata (CA) and Markov models use this approach. CM acknowledges that the actions of human agents are important as underlying causes of transition rules but do not explicitly model decisions. Transition rules are used as proxies to decision making.

Agent-based models

Agent-based models (ABM) focus on human actions, with agents as the crucial component. Agents are goal oriented and can interact with other agents and the external environment. In the past models generally assume that actors are perfectly rational optimizers with access to information. However, recent approaches recognize that perfect rationality may not be suitable for the complex environment and spatial inter-dependencies in which human decision making occurs. Thus, recent models employ some variant of bounded rationality (Villamor, 2012) or agents' learning capability (Schreinemachers and Berger, 2011).

Hybrid models

Hybrid models combine any of the above-mentioned techniques. These models mainly tried to combine the value of each approach, but may not be able to fully operate at the maximum utility of each approach. A new approach is soft coupling of highly specialized models to better represent biophysical-social-economics environments (Marohn et al., 2012).

Participatory/companion modelling

Participatory modelling is an approach to use models in multi-stakeholder settings, rather than to build a model as scientific activity *per se*. In this approach, end users and local stakeholders are directly involved in the modelling activity, ranging from model construction (Bots and Daalen, 2008), defining input parameter values (Ritzema et al., 2010) to evaluation of model results (Lusiana et al., 2011). The main aim is to stimulate discussion on what are the important components of the systems, and in particular about their future. Thus, the participatory process can contribute to decisions about complex landscapes (Sayer and Campbell, 2001). This approach is very often combined with role playing games (García-Barrios et al., 2008, Souchère et al., 2010).

The FALLOW model, extensively used in this PhD study, is an example of a model for natural resource management. It is spatially explicit land use change model that simulates consequences of farmers' land management decision to the overall landscape dynamics. FALLOW is a hybrid model that mixed the system dynamics approach with agent-based model approach (Villamor et al., 2011). Box 1 (page 11) provides further description of FALLOW model using 'Overview-Design concepts-Details (ODD)' protocol of Grimm et al. (2010).

1.4 Challenges for landscape based models

Landscape based models are complex, owing to the integration of human and environmental dynamics as well as the need to be spatially (and geographically) explicit. Landscape dynamics are the result of interactions between human actions and biophysical limits, which occur over a wide range of temporal and spatial scales, often in non-linear patterns. At each scale, there are different causes associated to change in the landscape, largely due to the different level of social organisations influencing decisions such as households, communities, nations, global companies and trade agreements. Developing models that are able to simulate the complexity of natural resource systems is a challenge. Given their complexity, data availability is also an issue, particularly in developing countries.

Uptake of models by natural resource managers and decision makers as a tool to manage natural resources is also a challenge. This may be because often results of models are too complex for direct use (Matthews et al., 2004). This can be remediated by better ways in communicating model results in terms of language and amount and form of information. Another reason is lack of trust and confidence in simulation models. Connected to this issue is the validation of models that would provide ways to assess the performance of a model. It would also enable to inform policy makers and other users of model on the uncertainties in the model outcomes and the suitability of the model for a particular situation.

1.5 Model validation

Application of simulation models requires trust in their (bounded) validity. Experts differ in their opinion, some consider model validation as essential, while others consider it as impossible (Oreskes et al., 1994) and that models can only be invalidated (Anderson and Papachristodoulou, 2009), just as hypotheses can be rejected but not 'proven'. However, according to Rykiel (1996), models can be validated and the process itself can be a useful model evaluation as well as for building model credibility in the users' community. He defined validity as relative to intended uses rather an absolute property: '*Validation means that a model is acceptable for its intended use because it meets specified performance requirements*'. Thus, assessment of model validity requires the perspective of potential users.

In a broader context, Gibbons (1999) stated that there are two validation steps in any scientific context of using information for complex decision. First, knowledge has to be reliable, meaning that it is considered by scientists themselves to be valid 'inside' (by intrapolation) as well as 'outside' (by extrapolation) of the environment it was developed. Second, knowledge has to be 'socially robust', involving validation by extended group of experts including lay experts. For landscape simulation models that aim to support policy makers and natural resource managers in managing their landscape, the first issue refers to technical model validation methods that are able to evaluate predicted spatial patterns relative to observed patterns (Turner et al., 1989; Loehle, 1997; Pontius et al., 2008). The second issue is associated with acceptance of simulation model by policy makers and natural resource managers. Participatory modelling is an approach that has often been used to enhance users' acceptance of a particular simulation model in which end users and local stakeholders directly involved in the modelling activity (Pahl-Wostl, 2002; Voinov and Gaddis, 2008).

Most experts agree with the concerns of model users that the purpose of model validation is to obtain an indicator of 'correctness' (qualitative or quantitative) of model results when compared to an observed reality[2]. This is challenging due to lack of independent data sets and also to the fact that most model outputs concern with the future where data is not available yet. By the time reality will have refuted or confirmed the models, the science may have moved on and the outcome of validation itself may no longer be perceived as relevant.

1.6 Concepts used in this study

1.6.1 Linking knowledge (science) into action (policy)

According to Cash et al. (2003), in linking knowledge into action (or science into policy), it is essential for any information targeting improved decision making on natural resource

2 Our observations are bound by the extent and resolution of measurements, thus each observation only provides partial description of the whole multi-scale land use system.

management to combine the three attributes of salience, credibility and legitimacy. Cash et al. (2002) and McNie (2007) defined salience, credibility and legitimacy as the following: (i) salience is the relevance of information for an actor's decision choices, or the relevance for

Table 1.1 Source and use of knowledge in boundary work (Source: Clark et al., 2011).

Boundary work		Use of knowledge for*		
		Enlightenment#	Decision	Negotiation
Source of knowledge from	Single community of expertise	Demarcation	Expert advice	Assessment
	Multiple communities of expertise	Integrative Research and Development	Participatory Research and Development	Political bargaining

*Knowledge users are varied. 'Enlightenment' refers to any users (not specific), 'Decision' refers to a single autonomous user while 'Negotiation' refers to multiple users.

Enlightenment refers to the advancement of general understanding that is not targeted at specific users but may influence decisions through a diffuse process.

the choices that affect a given stakeholder, (ii) credibility refers to whether or not information is perceived by the users to be accurate, valid, and of high quality, and (iii) legitimacy refers to users perception that information producer are free from bias and has the users' interests in mind. Understanding the importance of these attributes to the stakeholder will bridge the gap between knowledge and action. Activities that are carried out at the interface between communities of experts and communities of decision makers (linking knowledge into action) are known as 'boundary work' (Cash et al., 2003). Within this framework Clark et al. (2011) distinguished six situations where single or multiple knowledge paradigms are used for general enlightenment, for decision making or for negotiations about resource management (Table 1.1).

1.6.2 Uncertainty

<u>Accuracy</u> is often used to describe uncertainty. The common metric of accuracy is the root mean squared (r.m.s.) error; it is equal to the sum of the variance plus a squared bias term. <u>Bias</u> is a constant difference between the observed estimates and the true value (often defined as systematic error). <u>Precision</u> refers to variation around an estimate. Thus, an accurate estimate is one with low variance (high precision) and no bias (Figure 1.1). <u>Confidence intervals</u> is a range of values (or intervals) that act as good estimates of an unknown population parameter. The width of the confidence interval is often used as an indicator of uncertainty. Although in fact, it describes precision or variation only and does not include bias.

In a statistical and GIS (Geographic Information Systems) context uncertainty implies a quantifiable inexactness in a point (or aggregated) estimate. This inexactness may be quantified by the statistical distribution about a mean or expected value or degree of accuracy derived from a confusion matrix. Confusion matrix is a comparison between a classified map with ground truth data at a sample of locations, producing a table cross-tabulating map versus truth (what is observed in the landscape). It is commonly used to derive the following accuracy measures of a map:

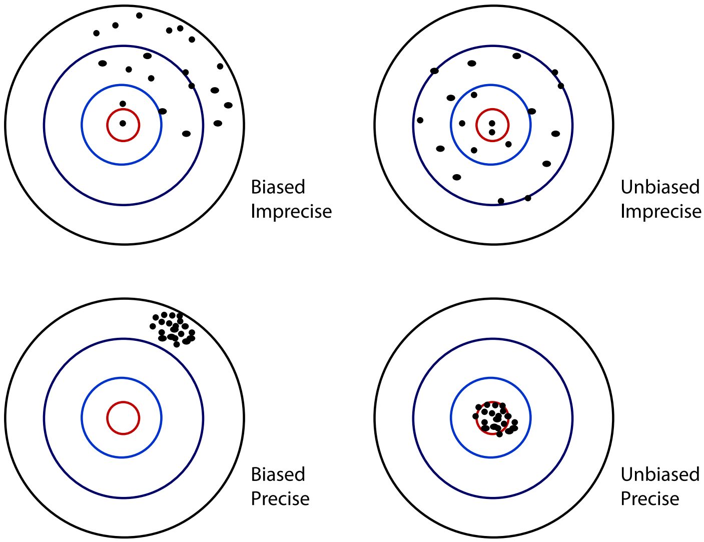

Figure 1.1 An illustration describing the concept of bias and precision.

- Overall accuracy, the probability of correctly classifying pixels[3] (or aggregates) across the entire map.
- Producer's accuracy, the probability that a pixel is classified as land use i in a map given that it is land use i in the landscape.
- User's accuracy, the probability that a pixel is land use i in the landscape map given that it is classified as land use i in the landscape.
- Kappa (or Cohen's Kappa), measures the agreement of land cover map with observation in the field and takes into account the fact that agreement may occur simply by chance. A Kappa of 1 indicates perfect agreement, whereas a kappa of 0 indicates agreement equivalent to random process.

1.6.3 Model users

Matthews et al. (2004) differentiated model users' into two main categories: target users and beneficiaries. Target users' are direct users of models such as researchers, consultants, educators and trainers. Beneficiaries are those that will benefit from the outcome of models that include policy makers, NGO (Non-Governmental Organization),

3 A pixel is the unit element of a picture or map; it can be a square, rectangle, hexagon or other shape that can be used for space-filling representations

extension staff and farmers. In the following chapters of this dissertation, the term model users' refers to both target users and beneficiaries.

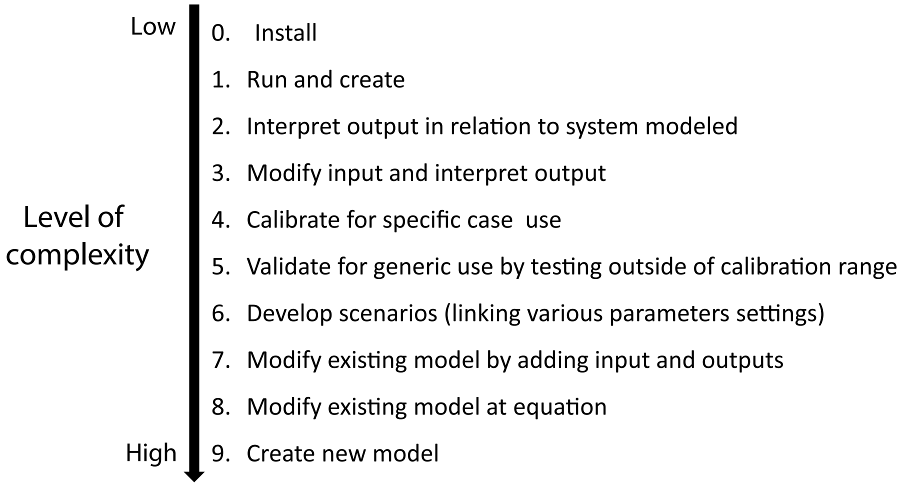

Figure 1.2 Ten level of model users, from novice (0) to advanced (9).

The concept of direct model users can further be differentiated into 10 levels (Figure 1.2), with a novice merely being able to install software as the lowest complexity to creating a new model based on an existing-model beingused as the highest complexity. As such, the participatory modelling study in Chapter 2 refers to model users of level 0–4.

1.7 Justification

With the increasing complexity of models and its functions, particularly in integrated landscape models for natural resource management, validation that is based on agreement between observed and simulated is not only impossible to do but also no longer relevant and sufficient. Model validation need to include the perspectives of model users and need to account the efficacy of the model for policy application. This study is scientifically relevant as it addresses issues and methods that can be used to improve model validation or evaluation processes. The study further shows the need to extend existing model concepts and outputs in order to make assessment of natural management options relevant not only for science but also for different stakeholders.

1.8 Hypothesis

The hypotheses of this research are:

1. Salience, credibility and legitimacy are equally important attributes in determining users' acceptance of a simulation model
2. There are synergistic opportunities in balancing land productivity, maintaining ecosystems functions that can be elucidated with modelling
3. Uncertainty is scale-dependent and environmental management institutions need a scale-dependent response to uncertainty in performance metrics.

The above hypotheses were further elaborated into specific research questions (Table 1.2).

1.9 Outline of the study

This dissertation is divided into 7 chapters. Chapter 1 introduces the objectives of the research and the overarching research questions as well as concepts underlying the research. Chapter 2 presents results of a survey carried out to explore potential model users' perspectives on the validity of simulation models for natural resource management. This chapter also presents a participatory modelling evaluation based on an application of FALLOW model in Aceh, Indonesia. Chapter 3 discusses the development of a new 'Livestock' module in the existing FALLOW model for an application to prospect the trade-offs of plausible land zoning policy options on farmers' welfare, fodder availability and carbon sequestration. Chapter 4 presents results of a study to estimate the uncertainty in estimating landscape carbon stocks where propagation of errors is an issue. The implications of the uncertainty study on designing a REDD scheme are elaborated in Chapter 5. This chapter also discusses both technical and social aspects of REDD+ in the broader context of validity of natural resource management models and management system. The dissertation continues with a general discussion (Chapter 6) and a section of references (Chapter 7). Summaries in English, German and Indonesian are included. The appendix includes the courses followed by the Ph.D. candidate at the beginning of her doctoral research, abstracts of additional articles published, seminar presentations and courses given during the doctoral time frame.

Table 1.2 List of hypothesis and research questions carried out in Ph.D. study.

	Hypotheses	Study title		Research question
1.	Salience, credibility and legitimacy are equally important attributes in determining users' acceptance of a simulation model	Users' perspectives on validity of a simulation model for natural resource management	a)	What do potential model users think about 'validity' of models?
			b)	Do occupation, prior modelling experience and interest in using a simulation model influence respondents' perspectives on attributes of a simulation model?
			c)	Would perceptions/concerns change when workshop participants start using a simulation model?
2.	There are synergistic opportunities in balancing land productivity and maintaining ecosystems functions that can be elucidated with modelling	Land sparing or sharing? Exploring livestock fodder options in combination with land use zoning and consequences for livelihoods and net carbon stocks using the FALLOW model	a)	How flexible and robust do models have to be? Is an 'add-on' approach efficient to reach saliency of a model?
			b)	How can livestock options be integrated in an existing land-use-change model?
			c)	How closely does the FALLOW model application match recorded historical land use change?
			d)	What landscape and economic dynamics can be expected under a 'business as usual' scenario?
			e)	What impacts are expected from land use zoning on trade-offs between fodder availability, carbon sequestration and livelihoods?
			f)	What does the case study suggest on a combination of land sharing and land sparing approaches
3.	Uncertainty is scale-dependent and environmental management institutions need a scale-dependent response to uncertainty in performance metrics	Uncertainty of net landscape carbon loss: error propagation from land cover classification and plot-level carbon stock	a)	Can we develop methods to assess uncertainty of landscape carbon stocks for different types of land cover maps and plot-level carbon data availability?
			b)	What is the implication of the uncertainty of carbon loss estimate and the distribution of carbon loss estimate for a potential incentive scheme for environmental service provision?
		Implication of uncertainty and scale in carbon emission estimates on locally appropriate designs to reduce emissions from deforestation and degradation (REDD+)	a)	Can we design a locally appropriate REDD+ scheme?
			b)	What is the implication of scale on the uncertainty of carbon emission?
			c)	Can we determine the appropriate scale for monitoring carbon emission that meet a given error tolerance level?
			d)	Is the suggested scale compatible with the local stakeholders proposed design of REDD+ scheme?

BOX 1

Box 1. The FALLOW model:

following the 'Overview-Design concepts-Details' protocol of Grimm et al. (2006)

1. Purpose

The purpose of FALLOW is to understand the consequences of farmers' land management decisions to the overall landscape dynamics and how the changes in the landscape impact on carbon sequestration, biodiversity and watershed functions. Farmers' decision is influenced by market dynamics, biophysical properties of the land, land zoning policy, farmers' knowledge and cultural preferences. When utilizing scenario-based analysis, FALLOW can be used to explore the impact of these changes.

FALLOW is particularly suited to simulate rural or peri-urban landscapes where land-based activities (i.e agriculture, forest extraction) are still the main livelihood option.

Scientists and students are the main target users of FALLOW. Parameterization of FALLOW requires the ability to work with maps. Resource managers, such as staff from land use planning agencies and watershed managers, can use a version that has been calibrated for their intended landscape. The model is built under PC-RASTER (http://pcraster.geo.uu.nl/), a spatially explicit environmental modelling freeware.

2. State variables and scales

The FALLOW model proceeds in annual time steps. FALLOW works with three main units:

Plot, represents the smallest landscape unit. The size of the plot equals to the size of the pixel of the maps used as input parameters. The default is 1 ha.

Livelihood options, can be (i) activities that are associated with a single type of land use system (e.g. cropping systems and agroforestry systems), (ii) activities that are associated with multiple types of land use systems (e.g. cattle rearing, rattan or firewood harvesting), or activities that are not associated with land use systems (e.g. off-farm activities).

Aggregated farmers as the main agents of land use change. An aggregated farmer can be described as an average farmer that represents a group of farmers with similar livelihood options (agriculture or non-agriculture) and similar learning style (see section 4 for detailed explanation). Thus, if each farmer has similar learning

BOX 1

style, the number of farmers in FALLOW model is equal to the number of livelihood options being simulated,

Farmers act as direct agents of land use change. The model also includes 'extension agents' as indirect agent that influences farmers' decision depending on their learning style.

FALLOW can be applied at various spatial scales.

3. Process overview and scheduling

The dynamic interactions between different modules in FALLOW (Figure Box 1) start from Module 'Plot level soil fertility' where soil fertility depletes during cropping periods and recovers during fallow periods, following the Trenbath model (Trenbath, 1989; van Noordwijk, 1999). Current fertility at plot-level determines the agricultural yield that can enhanced by adding fertilization.

The total agricultural production from the whole landscape together with the yield gained from other systems involving economic production (e.g. forest resource utilisation activities, off-farm activities) contributes to food sufficiency and/or household economic resources. This calculation is carried out in Module 'Aggregated household economics.

Population dynamics is based on local population growth rate that includes natural growth and migration. Population affects the magnitude of available labour force as well as the demand for food. Farmers conduct agricultural activities to meet food demand or their food-equivalent cost of living.

The strategic decision to open new land or to expand other economic production activities depends on available labour, financial capital and land (Module 'Farmers' decision making and learning'). This decision determines the magnitude of land use change in the model.

The model incorporates a simple optimisation approach where it is assumed that farmers/agents make a choice to undertake production activities (including planting crop or trees) with expectation of receiving the highest relative net labour or land return. The economic expectation starts with a certain initial knowledge and is able to change dynamically through learning from experience or from new information acquired during the simulation (e.g. from extension services and neighbouring farmers). The learning allows farmers/agents to change to other production activities. In Module 'Land use and land cover change', farmers will select suitable plots for clearing and planting based on their perceptions of plot attractiveness which is a function of relative soil fertility, land and market accessibility (i.e. slope, distance to a road/river, distance to market and distance to processing factory), land tenure status and spatially explicit rules on land zonation.

BOX 1

This decision determines location of the land use change. Activities related to agricultural land expansion will disturb natural succession as well as soil fertility recovery processes of the cleared plots. The overall landscape dynamics will lead to environmental consequences (changes in above-ground carbon stocks, biodiversity) at the landscape level.

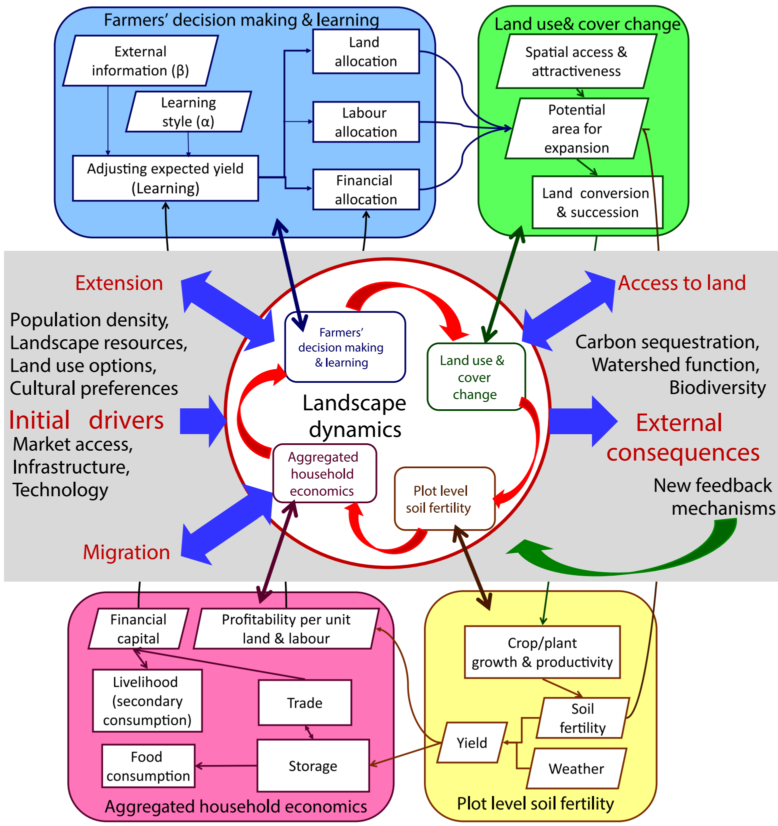

Figure Box 1. Schematic diagram of FALLOW model.

4. Design and concepts

4.1 Emergence

Land use patterns emerge as a consequence of change in farmers'decision in response to change in the relative expected profitability of land use systems as perceived by (aggregated) farmers.Farmers dynamic choice of production activity is emerging from the specific learning style.

BOX 1

4.2 Adaptation

Farmers can adjust their resource use (land, labour and financial capital) in response to: (a) changes in the relative expected profitability of livelihood options, and (b) changes in the relative scarcity of these resources; as affected by policies, technologies and markets. For example, if the relative expected profitability of horticulture systems increase, farmers will change some of their less profitable land use systems to horticulture. However, if the financial capital required is not available, farmers will not convert to horticulture systems. A reduction in cost of inputs (due to change in technology) may be able to support conversion to horticulture systems.

4.3 Objectives

Farmers decide to grow a crop/tree on a given plot or undertake other production activities with expectation of receiving the highest relative net labour or land return. For consumption, they aim to reach sufficiency (in staple food). Farmers select suitable new plots for planting that has the highest plot attractiveness, which is a function of relative soil fertility, land and market accessibility (i.e. slope, distance to a road/river, distance to market and distance to processing factory), land tenure status and spatially explicit rules on land zoning.

4.4 Learning and Prediction

Farmers form expectations about profitability based on past experience, following the theory of adaptive expectations. Profitability refers to expected return to labour and expected return to land. Farmers revise their expectation to profit each year in proportion to the difference between actual profitability (P_t) and expected profitability (E_t). The proportion of change depends on the farmers learning style that may range from conservative (dominated by long term trends, $\alpha = 0$) to creative (dominated by recent experience, $\alpha = 1$). Farmers expectation to profit can be also influenced by suggestion from others (*St*), such as extension officer or neighbouring farmers. Farmers adjust their expectation to profit proportional to their trust to agent providing suggestion (β), whereby farmers can completely abandon ($\beta = 0$) or adopt ($\beta = 1$) suggestion from others.

$$E_{t+1} = [E_t + \alpha(P_t - E_t)] + (\beta(S_t - [E_t + \alpha(P_t - E_t)]) \quad \text{.............} (1)$$

4.5 Sensing

FALLOW does not model the ability of farmers to 'sense'. Farmers are also assumed to know and obtain 'information'. The mechanism by which farmers obtain information is not modelled explicitly. For example, when farmers decide

BOX 1

where to open new plot based on plot attractiveness, farmers are assumed to be able to know which plot is attractive. The way the model derives plot attractiveness is a proxy to farmers' decision process and not an actual representation on how farmers form their decision.

4.6 Interaction

(a) Farmers – environment

Farmer decisions also have a direct impact on resource dynamics, through choice of livelihood options, investments in opening new plot and the use of fertilizers. Through reduction in yield from agricultural systems, farmers received feedback on changes in soil fertility conditions. This may lead to farmers to abandoned plot and open new land in more fertile plots.

(b) Agents– agents

FALLOW does not explicitly model agent to agent (or farmer to farmer) interactions. The interaction is implicitly implied in the learning process (see 4.4 Learning and Prediction) between extension agent to farmer or farmer to farmer (neighbouring farmer).

4.7 Stochasticity

The FALLOW model allows model users to impose stochasticity in input parameters (random numbers in combination with coefficient variation) to assign variation in individual plot and livelihood options/land use systems characteristics. The input parameters can be biophysical characteristics (e.g. soil fertility, carbon density) or economics (e.g. return to land, cost of input). For a complete list of input parameters, please refer to the FALLOW manual (Suyamto et al., 2009).

5. Initialization

At initial condition, FALLOW model requires all input parameters to produce initial condition of the landscape, in particular a land cover map and a soil fertility map. Plots are spatially explicit, while agents are not.

6. Inputs

FALLOW input data are categorized into 3 types: (i) spatial data (files with extension of .xxx), (ii) arrays (files with extension .par) and (iii) time series (files with extension .tss). The spatial data are required to be in the specific format for PC-RASTER (which can be derived from the ASCII format). The arrays and time series can be in TXT format.

BOX 1

The spatial data required by FALLOW are information on initial land cover, information to differentiate qualities, such as soil fertility, slope, distance to market, road and river; and if exist a suitability map for each agricultural system/ livelihood options.

FALLOW also requires information on profitability, input (labour and cash) and output (yield) for each livelihood option which can be based on a farm survey.

Landscape level information such as size of population, percentage of labour force and income per capita are initial information required to run the model.

FALLOW requires all data files to exist to be able to run the model, even though the application may not need the information. For example, biodiversity may not be of interest, but the data file related to that should exist. The data inside these files can contain zeroes or any numbers that can ensure the values will not affect the model to run the module.

7. Submodels

There are 4 main sub-models: (a) Plot –level soil fertility, (b) Aggregated household economics, (c) Farmers' decision making and learning and (d) Land use and land cover change.

See the FALLOW manual (http://www.worldagroforestry.org/sea/publication?do=view_pub_detail&pub_no=MN0044-09) for the complete description and equation of each module.

Chapter

2

Users' perspectives on validity of a simulation model for natural resource management[4]

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2.1 Abstract

Managers of agro-ecosystems trade off food production and livelihood strategies against environmental services. They need tools to prospect a wide range of external conditions. Integrated simulation models allow stakeholders to discuss plausible behaviour of agro-ecosystems and to evaluate dynamic trade-offs, as basis for planning and policy making in agriculture and natural resource management. However, simulation models need to gain stakeholders acceptance before they will be utilized. Gaining stakeholders' acceptance likely requires salience, credibility and legitimacy. We surveyed perceptions and expectations of 122 potential model users in four countries, prioritizing these model attributes. A possible shift in user perception was assessed during a participatory model evaluation of a resource management model (FALLOW) for post-Tsunami development in West Aceh (Indonesia). This provided insights into the representation of spatial patterns and of recognizable processes needed to gain acceptance in a model for local use. Potential model users, comprising of natural resource managers, policy makers, lecturers and scientists, ranked salience as most important characteristic for an integrated simulation model, followed by credibility and legitimacy. Model users' occupation, prior exposure and interest in using a simulation model did not have a statistically significant influence on users' perceptions of model attributes. Direct experience in using a simulation model in a known setting increased perceived credibility of the model results. The West Aceh study further highlighted the importance of gaining users' acceptance of a model as part of model validity tests, alongside existing quantitative validation tests.

2.2 Keywords

Land use change model, model validation, model users, participatory approach, salience-credibility-legitimacy.

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2.3 Introduction

Sustainable agro-ecosystems integrate three main goals; environmental health, economic profitability (providing income and food security), and social equity in resource access (Pearson, 2003) with the ability to sustain under short-term shocks and long-term stresses (Thompson and Scoones, 2009). Under population pressure and development needs, trade-offs between these goals appear to be inevitable and compelled many of the agricultural systems to operate in non-sustainable ways, e.g. farm-level soil degradation in rice-swidden systems (Lam et al., 2005), landscape level loss of biodiversity in cocoa-production (Steffan-Dewenter et al., 2007) and biodiversity loss in global-market-pressured biofuel plantations (Danielsen et al., 2009).

To attain sustainable agro-ecosystems requires balancing trade-offs of sustainability goals at multiple scales. Natural resource managers and decision makers need to understand the essential processes and behaviour of the systems they manage, in order to progress towards balanced trade-offs with the help of policies and technological innovations. They must be able to explain apparent trade-offs. Thus, dynamic and efficient approaches are needed to help managers and policy makers in assessing trade-offs in order to choose viable options for meeting human needs and ensuring ecosystems functions (DeFries et al., 2004). Scenario analysis based on a credible simulation model is an efficient approach to assess the dynamic and complex interactions between components and their trade-offs (Carpenter et al., 2006). Despite the potential of simulation models for decision support tools, acceptance and use by decision makers and natural resource managers are still a major challenge (Borowski and Hare, 2007), particularly in developing countries.

A 'good', credible model for sustainability analysis may need to be able to simulate a range of behaviours. Jackson et al. (2010) distinguished three levels of temporal scale in sustainability: efficiency, persistence and change. Efficiency mostly refers to agro-ecosystems role of provisioning at plot level scale where decision making aims at gaining resource sufficiency; while the main concern of persistence is on functional integrity to ensure agro-ecosystems services flows continuously (Thompson, 2007). Change issue is related to adaptive capacity or the resilience of the agro-ecosystems to recover from disturbances (Walker et al., 2010). Simulation models aiming at resource sufficiency only need to operate at the efficiency scale, while models used to assess global ecosystems behaviour in response to changes (Rockstrom et al., 2009) may need to include all the three temporal scales.

Simulation models for natural resource management involve quantifying landscape dynamics that entail non-linearity, multi-spatial and temporal interactions. Validating such complex model is not easy due to lack of independent data sets. Various statistical approaches have been developed for validating simulation model (Costanza, 1989; Pontius et al., 2004). Nevertheless, absolute validation aiming to obtain confirmation of 'truth' is considered impossible (Oreskes et al., 1994) particularly when the model is used for prospecting the future (Kok et al., 2001). However, Rykiel (1996) defined validity as relative to intended users rather an absolute property: *"Validation means that a model is acceptable for its intended use because it meets specified performance requirements"*. Thus, assessment of model validity requires the perspective of potential users.

Analysis of what stimulates use of new knowledge and models by stakeholders has recognized three main groups of attributes: salience, credibility and legitimacy (Cash et al., 2003; McNie, 2007). In the context of simulation models, salience is the relevancy of the model to lead to real changes in identified problems. Credibility entails perceptions by users that the concepts and processes in the model are acceptable as approximation of reality. Legitimacy refers to intention and agenda of the tools' developers as perceived by stakeholders. Further analysis on salience, credibility and legitimacy is needed to understand the gap between current model availability and use (Borowski and Hare, 2007). Evidence by (White et al., 2010) suggests that trade-offs between these attributes and that perceptions of 'model validity' differ between user categories. Participatory approaches to natural resource management models thus need to complement current statistical validation concepts.

In this paper we explore model acceptance and validity from the perspective of intended users, in the context of natural resource management in tropical landscape mosaics, through two studies: (1) a survey of the concerns of potential model users across multiple countries and backgrounds, (2) a participatory model evaluation of a specific model to ascertain how perceptions/concerns change following experience of using a simulation model tailored to the way their context and concerns had been understood by researchers.

2.4 Methods

2.4.1 Survey on simulation model attributes desired by stakeholders

2.4.1.1 Data collection

To explore potential model users' perceptions of a prospective model to be used, surveys were carried out during workshops on 'Tools, Methods and Approaches for Natural Resource Management' organized by the World Agroforestry Centre (ICRAF) in Indonesia, Kenya, Philippines and Vietnam in 2008. Respondents (122) included natural resource managers, policy makers, communicators (including extension workers), researchers and lecturers. Seven characteristics of simulation model were indentified that were simple and can be understood by users who may be new to modelling activities. The characteristics were based on common queries and comments from trainees at trainings by our research institutes in the use of specific simulation models over the past 10 years. Some of the characteristics were further categorized according to the salience, credibility and legitimacy attributes (Table 2-2). The questionnaire was pre-tested for readability and clarity at a workshop prior to this study case.

2.4.1.2 Statistical analysis

Statistical analysis was performed using Stata version 9.1 (StataCorp., 2005). The data were grouped by (i) respondents' occupation, forming 3 sub-groups: researchers/ lecturers, natural resource managers and policy makers/communicators; and (ii) respondents experience (first-time versus experienced) and with or without interest in using simulation models.

Rank-means were calculated for total respondents and different sub-groups. We applied the Skilling Mack test, a general Friedman test that can accommodate ties and missing data (Chatfield and Mander, 2009), to test the hypothesis that ***respondents ranked model characteristics equally***. Multiple comparison analysis was conducted when Skilling-Mack test was found to be significant.

To test the hypothesis that occupation, prior modelling experience and interest in using simulation model influence respondents' perspectives on attributes of a simulation model, we applied the Kendal Tau test (Gibbons, 1985) on rank-mean.

2.4.2 Participatory model evaluation

Would perceptions/concerns change when workshop participants start using a simulation model? We documented participatory model evaluation in two activities: (a) a prospective study using a resource management model (FALLOW) for post-Tsunami development in West Aceh, Indonesia and (b) an in-depth participatory discussion on modelling for natural resource management with respective potential users based on results obtained in activity (a).

2.4.2.1 Site Description

West Aceh District is geographically located in the western coast of Northern Sumatra island (Figure 1.1). It has an area of around 3,030 km^2 and is administratively part of Nanggroe Aceh Darussalam Province. The main land use systems in West Aceh are forest and tree based systems. Rubber, coconut and oil palm are the dominant crops in terms of area and production (BPS, 2005).

The four coastal sub-districts of Johan Pahlawan, Meurebo, Samatiga and Arongan Lambalek in West Aceh were severely damaged by the tsunami of 26th December 2004. The major sources of household income in these four sub-districts included fishery, paddy cultivation, tree-based agricultural systems, mixed systems (multiple products), off-farm labour and trading (Joshi and Nugraha, 2008). In Johan Pahlawan, a peri-urban sub-district where the district capital Meulaboh is located, off-farm labour and trading contributed almost 60% to household income. In the three rural sub-districts, tree-based agricultural systems contributed up to 45% of income.

The tsunami disaster dramatically changed livelihood options in the coastal sub-districts of West Aceh. A study conducted 6 months after the 2004 tsunami revealed that off-farm labour and trading had become more important particularly in Arongan Lambalek, increasing to 35% and 23%, respectively. In contrast, tree-based systems contribution to income was reduced by 35% (Joshi and Nugraha, 2008). The 2004 tsunami resulted also in increased pressure for natural resources in the area. The high demand for construction materials (sand, stone, brick and timber) for post-tsunami 'reconstruction', has led to intensified logging and sand/rock mining.

2.4.2.2 Prospective study using the FALLOW model

Similar to CLUE (Veldkamp et al., 2001)), FALLOW (van Noordwijk, 2002; Suyamto et al., 2003) simulates spatially explicit patterns and functioning of land use systems by analyzing drivers and consequences of land use change. FALLOW includes the dynamics of farmers' knowledge ('learning styles') as a factor that influences farmers' land management decision, based on their experience within the simulated landscape and external information obtained from outside agents such as extension workers. The biophysical responses at plot level lead to environmental consequences (carbon stocks, biodiversity and watershed function) at landscape level, allowing FALLOW as tool for assessing trade-offs between livelihood and environmental services as consequence of land use change.

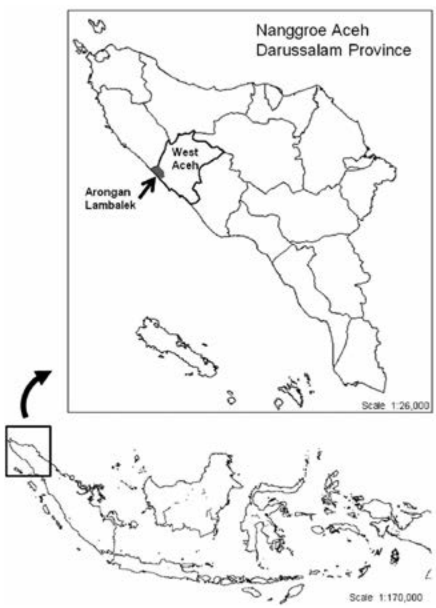


Figure 2.1 Location of Arongan Lambalek, a coastal sub-district in West Aceh District, Nanggroe Aceh Darussalam Province, Sumatra, used for the case FALLOW study.

We prospected landscape dynamics of Arongan Lambalek, a sub-district of 173 km^2 located in West Aceh, Nanggroe Aceh Darussalam, Indonesia using FALLOW model version 2.0 (Suyamto et al., 2009). The main objective of the simulation study was to provide potential users' with a general experience of what can be achieved and the type of results the model can produce, rather than providing accurate predictions of future land use and sustainability. These results were to be the starting point for the in-depth participatory model evaluation activity.

The prospective study involved simulating a baseline ('business as usual' condition) scenario of Arongan and seven development scenarios (Table 2.1). The scenarios were developed from existing livelihood options and government directives comprising of improvement to existing infrastructure, management practices and agricultural systems productivity, including development of tree-based systems. Post-tsunami, the government recommended a 200 m wide safety belt of trees along the coast. Instead of planting mangroves as initially suggested, cash-crop tree-based systems could also provide local income.

Economic and soil data (Budidarsono and Wulan, 2008; Wahyunto et al., 2008), and land cover maps (Dewi et al., 2008) were combined with biomass/carbon data from similar climatic conditions (Palm et al., 2005). The model was run for 25 years, starting in year 2002. The tsunami incidence was included in the simulation as an externally imposed natural disaster having an impact on land cover as well as labour availability.

Table 2.1 Scenarios of landscape dynamics in Arongan Lambalek, Meulaboh, developed for FALLOW model application.

A. Development of scenarios

No.	Scenarios	Description
1	Improving rubber-based systems	Improving systems productivity (better seedlings, management, technology, efficient labour), improving market and increasing subsidy for financial capital.
2	Improving oil-palm systems	
3	Improving cacao systems	
4	Improving coconut systems	
5	Strengthening off-farm activities	Improving access to off farm activities and its economic returns. Off-farm activities represented by off-farm labour and harvesting *Nypa* leaves for cigarettes
6	Conserving forest	Delineating 50% of existing forest area as protected forest prohibiting farmers' access for conversion
7	Integrated (all 1-7)	All of the above
8	Baseline – "business as usual"	No subsidy and no off –farm labour. For yield, return to labour, return to land and above ground biomass, see Table 1B.

B. Yield, return to labour, return to land and aboveground biomass of land use systems simulated in baseline scenario.

Livelihood options	Yield ($Mg.ha^{-1}$)	Return to labour ¤ (US $/man.days)	Return to land¤ (US $/ha)	Aboveground* biomass ($Mg.ha^{-1}$)
Forest	n.a.	n.a.	n.a.	430
Rice	2.5	2.6	293	2.5
Rubber	2.6	7.3	1389	215
Oil Palm	2.3	21.2	1322	144
Coconut	0.5	2	161	122

n.a. = not applicable, logging was not carried out in this part of area

¤Wage rate/land price at which the Net Present Value is zero, at an exchange rate of 1US$ = Rp. 9000 and 2007 prices. The values are for systems at productive stage. Based on Budidarsono and Wulan (2008)

*Based on Palm ***et al***. (2005)

To validate model results of the baseline condition, land cover resulting from model simulation was compared with reference land cover maps produced from LANDSAT-TM 2005 and 2006. Two criteria were tested: (1) area accuracy and (2) spatial accuracy. The accuracy of area was obtained from calculating relative area size differences, i.e. the difference between simulated and reference area relative to reference area. Spatial accuracy indicator was calculated as the ratio between total area of land cover ***i*** that was found exactly in the same position in both simulated and reference land cover maps (intersection) and total area of land cover ***i*** that was found in both simulated and reference land cover maps (union).

For model outputs we focused on parameters that enabled demonstrating participants how their landscape might evolve over time in future and how it could impact on farmers' livelihood and landscape function. We chose simple descriptive graphs to show land cover dynamics over the simulated period and the trade-offs between people's (farmers') livelihood and environmental services. Farmers' livelihood was represented by expenditure after meeting staple food requirements, while environmental services were represented by above-ground carbon stocks.

2.4.2.3 In-depth participatory discussion

Participatory approaches have been widely used in modelling to gain users' confidence and trust, e.g. stakeholders were solicited to involve in the modelling process ranging from the early stage of developing models at conceptual level (Newham et al., 2007), exploring appropriate input parameters (Becu et al., 2008, developing scenarios (VoVolkery et al., 2008), up to assisting in technology diffusion (Martin et al., 2005).

We used a similar approach for developing scenarios with a focus on assessing changes of perceptions/concerns of model users when they started using a simulation model. Target stakeholders were potential model users of West Aceh district who may directly work with the model in the future or collect input data in the field. There were fifteen (15) potential users, including two women, comprising officials from District Planning Agency, Natural Resources Management Department, local environmental NGOs and a local university. Several participants had previous experience in using simple simulation models, but none was experienced in using a landscape simulation model for integrated assessment.

An in-depth participatory discussion was held concurrently with FALLOW model training. The training objectives were to enable participants to use the model as well as to expose users to the underlying theory and concepts of the FALLOW model. Three approaches were used to obtain users' perspectives - by discussion, participatory modelling and questionnaire. Exploratory discussions were held in groups of 3-4 persons of different profession to maximize diversity. Group opinions on FALLOW model performance (resulting from prospective studies) particularly between simulation outputs and group understanding of what occurred on the ground were also highlighted. Each group had the opportunity to develop their own scenarios by modifying existing scenarios, giving attention to their expectation of simulation results and their actual results. Participants were asked to fill in a questionnaire individually at the end of the discussion. Topics for discussion and questionnaire included technical issues in using the model (ease in preparing input parameters and understanding the outputs), model accuracy, relevance and bias if any of scenarios developed and model outputs for prospecting landscape dynamics by the participants.

2.5 Results

2.5.1 Potential users' perceptions of attributes of simulation models

Respondents participated in the survey comprised of 50% lecturers and researchers, 43% practitioners (natural resource managers) and 18% policy makers and communicators (Table 2.2). Seventy two percent of the respondents had prior exposure/experience with modelling and 99% were interested in using models for their future work either as direct users or output users; the latter reflected that they were voluntary participants to a workshop about these topics.

Interestingly, the Kendall Tau test found a high degree of agreement in rank-mean of model attributes across occupation groups (Table 2.2). A similar result also persisted for the experience-interest groups, concluding occupation, prior exposure to simulation model and interest in using models did not significantly influence respondents' ranking of model attributes.

"Useful and applicable outputs for natural resource management" characterizes ***salience*** or ***relevancy*** of simulation model use and significantly has the lowest rank mean among attributes (most important) indicating that respondents considered relevancy as the most important attribute in a simulation model.

The ranked values of "Clear and understandable theory and processes", "Easy to use and parameterize" and "Outputs have similar pattern to what is observed in the field" were not significantly different from each other but significantly different to other model attributes. These attributes can be seen as representing how a model can be understood and operated by users (***operational***). Specifically, attribute "Outputs have similar pattern to what is observed in the field" equates 'goodness of fit' between observed data with model results which is often used in validation methods as indicator of a good simulation model. Thus it also characterizes model ***credibility***.

Attributes "Developed by well known scientist with stakeholders' involvement" and "Used by policy makers" had the largest rank-mean (lowest importance). Their ranks were not significantly different from each other, but significantly different from other model attributes. Both attributes can be seen as representing ***track record*** of the model and specifically attributes "Developed by well known scientist with stakeholders' involvement" characterizes ***legitimacy***.

Only 50% of the respondents further elaborated, in an open question, on reasons for their interest in using (or not using) simulation models in future. Expectations that a simulation model could help users to prospect and predict the future, help in decision making, as well as help to work more efficiently in a systematic way were the three main reasons stated for their interest in using a model. Difficulties in using a model and obtaining input parameters were the main reasons for their lack of interest to use models.

2.5.2 FALLOW model application: prospecting the landscape of Arongan Lambalek

The following section describes the results of the prospective study which later were shown as simple and descriptive graphs to the potential model users in West Aceh for in-depth discussion.

2.5.2.1 Baseline condition

FALLOW 'business as usual' scenario outputs for Arongan Lambalek prospected that between 2002 – 2026, forest area decreased by 20%, predominantly transformed into rubber and coconut systems (Figure 2.2 A). Consequently, aboveground carbon stock declined to 60% of its original amount at the end of the 25 year simulation period Figure 2.2 'Business-as-usual' scenario post 2004 tsunami results of case study of Arongan Lambalek running the FALLOW model for 25 years starting in 2002. (A) Landscape dynamics (% of total area), (B) Farmers' welfare (as non staple food consumption in '000 Rupiah) and (C) Aboveground carbon stocks (in Petagram = 1015 gram).(Figure 2.2C). The simulation result indicated that the tsunami did not directly affect surviving farmers' welfare per capita, due to the sudden (tsunami induced) decrease of Arongan's population by 50% and the stable income from coconut systems. This reflected observation in the field that during the first three years after tsunami coconut systems prevailed when other systems such as rubber, oil palm and fruit trees were more affected. The simulated farmers' welfare only started to decrease five years after the tsunami and started to improve when rubber systems came into production (Figure 2.2C).

Figure 2.3 shows comparison of simulated results in year 2005 and 2006 with reference maps produced from Landsat-TM images of the same year. Area accuracy (how good the model predicted area size) for tree based agricultural systems, except oil palm, was relatively good with area differences ranging between -14% to +11%. Oil palm systems showed high values of area difference, -51% and -66% for 2005 and 2006 respectively. Rice fields had the highest area difference of 90% and 98% in 2005 and 2006 respectively. The spatial accuracy (how good the model predicted location) of simulated results ranged from 24% to 73%. Forest had the highest spatial accuracy of 63% and 73% in 2005 and 2006 respectively, while grassland had the lowest spatial accuracy of 24% and 42% in 2005 and 2006 respectively.

Table 2.2 Rank-mean of model attributes (1 to 7; 1= most important, 7= least important) for total respondents and all sub-groups (by occupation and by experience and interest in using simulation models).

	Simulation model attributes	Total respondents	+OCCUPATION			#EXPERIENCE – INTEREST			
			Researchers/Lecturers	Natural Resource Managers	Policy makers/ Communicators	1st time user without interest	1st time user with interest	Experienced without interest	Experienced with interest
1	Useful and applicable outputs for natural resource management (SALIENCE)	2.4 a	2.4 a,	2.3 a	2.7 a	2.0 a	2.1 a	2.2 a	2.7 a
2	Clear and understandable theory and processes	2.9 b	3.0 a, b	2.8 a	2.7 a	2.1 a	3.5 b	2.8 a	2.7 a
3	Easy to use and parameterize	3.3 b	2.9 a	3.8 a, b	3.5 a, b	3.3 a, b	3.1 b	3.4 a, b	3.5 a
4	Outputs have similar patterns to what is observed in the field (CREDIBILITY)	3.6 b	3.5 b	3.6 a, b	3.6 a, b	3.5 a, b	3.6 b	3.4 a, b	3.6 a
5	Attractive and easy to understand outputs	3.7 b	3.7 b	3.8 a, b	3.6 a, b	3.6 a, b	3.7 b	3.7 a, b	3.7 a
6	Developed by well known scientist with stakeholders involvement (LEGITIMACY)	5.1 c	5.3 c	4.8 c	5.5 b	4.5 b, c	4.9 c	4.9 b	5.5 b
7	Has been used by policy makers	5.5 c	5.7 c	5.3 c	5.4 b	5.5 c	6.1 c	4.8 b	5.3 b
	Number of respondents	122	61	43	18	11	39	12	60
	% respondents	100	50	35	15	9	32	10	49

Note:

Numbers followed by different letters in the same column indicate significant differences ($p < 0.05$).+Statistically significant agreement across different occupation sub-groups (Kendal Tau statistics = 0.9, p-value < 0.05).

#Statistically significant agreement across different interest and experience in modelling sub-groups (Kendal Tau statistics = 0.95, p-value < 0.05).

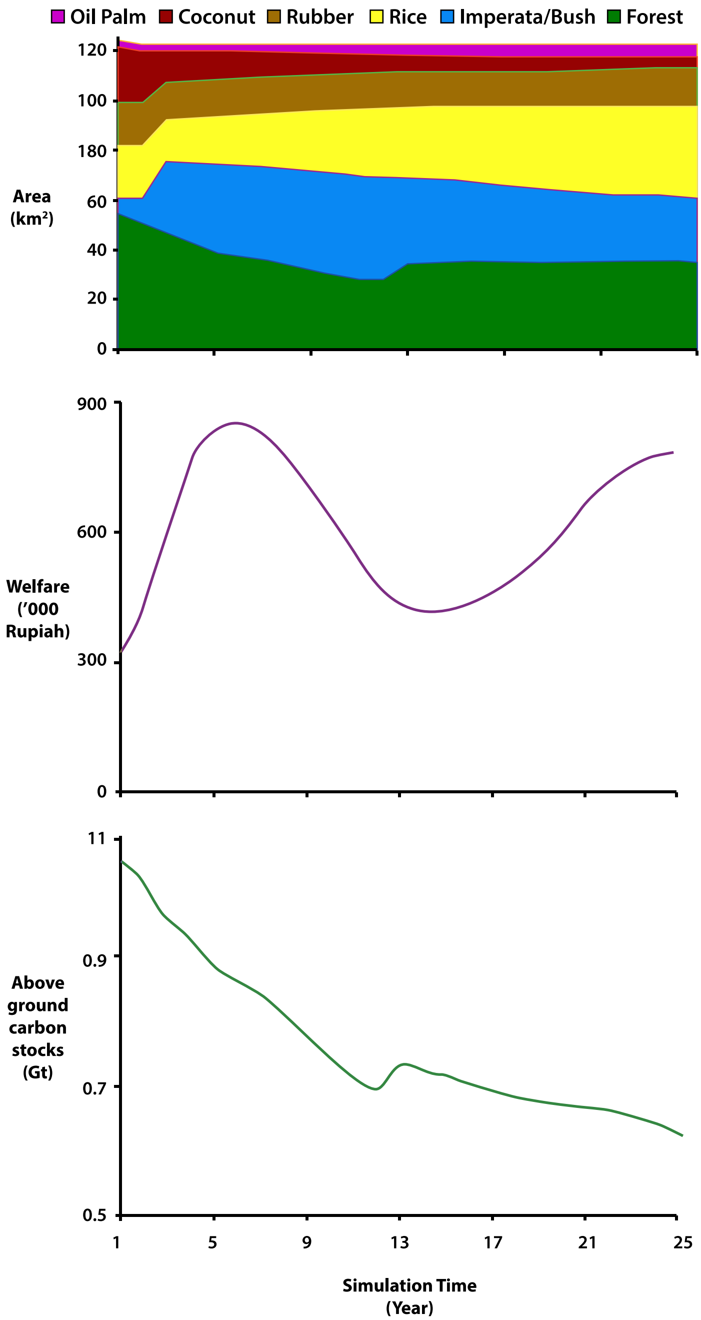
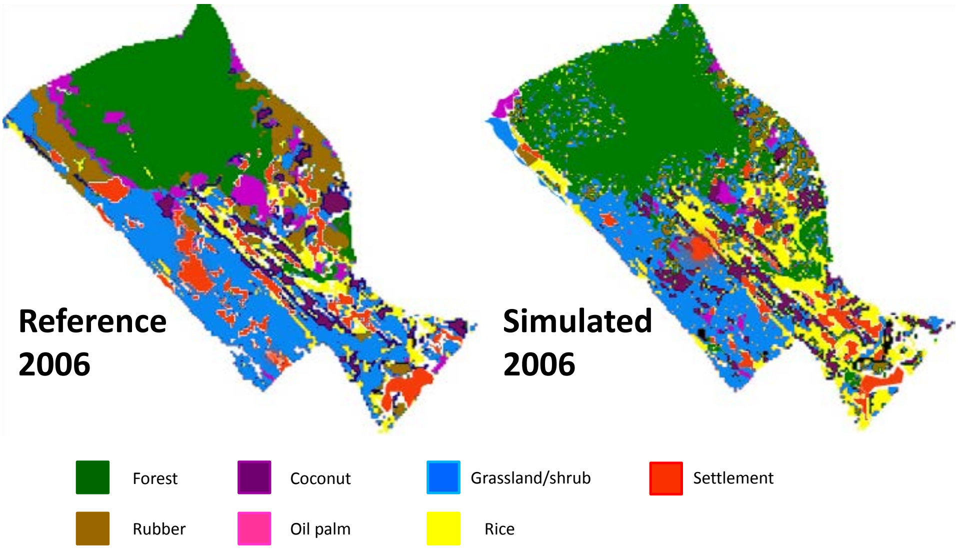


Figure 2.2 'Business-as-usual' scenario post 2004 tsunami results of case study of Arongan Lambalek running the FALLOW model for 25 years starting in 2002. (A) Landscape dynamics (% of total area), (B) Farmers' welfare (as non staple food consumption in '000 Rupiah) and (C) Aboveground carbon stocks (in Petagram = 10^{15} gram).



Land cover type	Relative area difference (%)		Spatial accuracy (%)	
	2005	2006	2005	2006
Forest	-10	-8	73	63
Rubber	-174	-11	38	33
Oil Palm	-51	-66	49	20
Coconut	2	11	59	49
Crop	90	98	45	40
Grassland	4.5	8	24	42

Figure 2.3 Comparison of land cover maps of Arongan Lambalek simulated by FALLOW model ('business-as-usual' scenario) versus reference (result of Landsat-TM interpretations) with results of quantitative validation for each land use systems. Figures shown to potential model users in West Aceh were in colour and not pattern as shown above.

2.5.2.2 Scenario analysis

The outputs of the seven development scenarios were compared to baseline outputs (Figure 2.4). We used relative additional carbon sequestration (%) as an indicator of environmental services provision and relative additional non-food expenses (%) as an indicator of farmers' welfare.

None of the scenarios produced negative additionalities relative to the baseline in terms of its landscape carbon stocks. Results of scenarios 5 (forest conservation) and 6 (improved off-farm activities) suggested an increase in carbon sequestration by 35% compared to the baseline. This is equivalent to maintaining 95% carbon stocks of its original level. Scenarios 1 and 2 (improved rubber and oil palm systems) resulted in large increase in welfare and ability to maintain carbon stocks at 'business as usual' level. The integrated scenario (scenario 7) increased both welfare and carbon sequestration.

2.5.3 Participatory model evaluation

2.5.3.1 Accuracy assessment of baseline condition: model credibility

All participants considered area and location (spatial) accuracy for model outputs as important aspects in evaluating model results. The groups had different opinion on FALLOW accuracy performance (Table 2.3A). Only two groups found FALLOW acceptable for application in the study area in its current form. One group required FALLOW to improve its accuracy, whereas another group recommended that all input parameters to be based on actual measurements in Arongan to improve its accuracy. The groups found the dynamics of land cover and farmers' welfare in terms of land use change and welfare to have plausible trends but unrealistic in terms of their magnitude. Several participants admitted, particularly for aboveground carbon stocks, they had little idea what would be a realistic magnitude.

2.5.3.2 Scenarios analysis: relevancy and efficacy

Most groups found a twenty-five-year simulation study too long for developing policy recommendations. Users felt a five-year simulation was deemed to be more appropriate and useful as it matched the government five-year development plan. However, the groups accepted that twenty-five to thirty years simulation was useful for strategic planning for future generations. Nevertheless, they challenged the idea of having 'static' parameters throughout the simulation period and were happy to find that most driving factors in FALLOW could be presented as time series.

Given the opportunity to develop their own scenarios, all participants chose rubber and oil palm scenarios (Table 2.3B). They felt these systems were the most relevant scenarios for their area as they were economically more attractive compared to other systems. Each group modified the input parameters related to systems productivity, technology and market to what they considered more realistic and stated their expectation of change in terms of welfare and carbon sequestration. After seeing the responsiveness of model parameters to changes, groups' confidence in FALLOW model performance ranged between 65 to 80%.

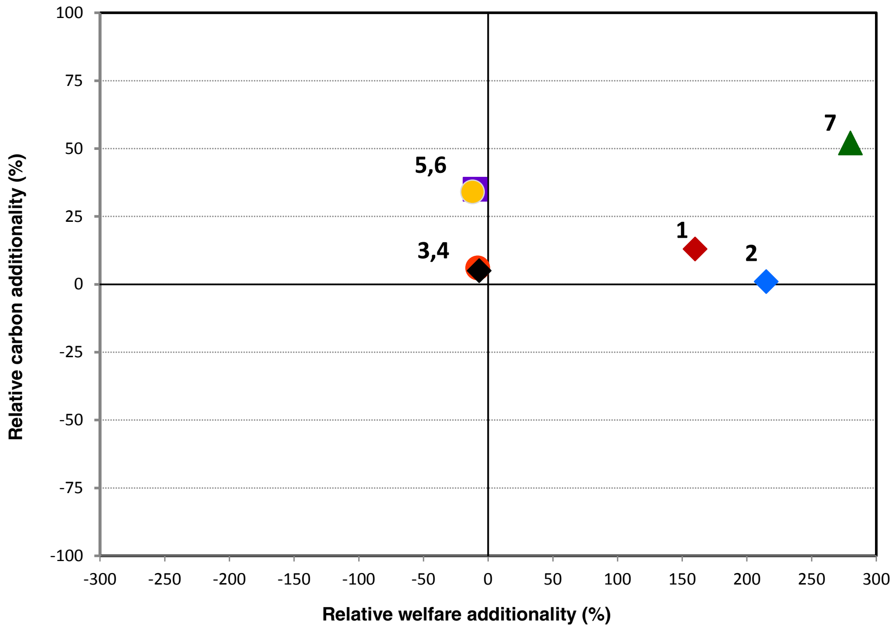


Figure 2.4 Farmers' welfare and carbon sequestration additionality relative to baseline conditions. These are the simulation results of prospective scenario applications of the FALLOW model for 25 years. Numbers refer to scenarios, where 1, 2, 3, 4 and 5 represent improvements in rubber, oil palm, cacao, coconut and off-farm systems respectively, 6 = forest conservation and 7 refers to the simultaneous integrated scenario of 1-6 (see also Table 2-1).

Table 2.3 Model users' responses to use of the FALLOW model in prospecting landscape dynamics of Arongan Lambalek, West Aceh.

A. Scenarios developed by model developers

	Model characteristics	Group 1	Group 2	Group 3	Group 4
1.	**Accuracy of model results (Fig. 2.3)**				
	Relative area difference	OK	Not OK	Not OK	OK
	Spatial accuracy	OK	Not OK	Not OK	OK
	Is the model valid and acceptable for further simulation study in Arongan Lambalek?	Acceptable	Acceptable for general understanding of landscape dynamics but not detail assessment	Acceptable after accuracy improvement	Acceptable only if input parameters used are actually measured in the site
	Expectation of accuracy level				
	Relative area difference	≤ 25%	≤ 10%	≤ 2%	≤ 1%
	Spatial accuracy	≥ 75%	≥ 85%	≥ 60%	≥ 100%
	Which accuracy is more important?	Spatial accuracy	Both equally important	Both equally important	Both equally important
2.	**Results of prospective study**				
	Land use change				
	Direction of change	Plausible	Unrealistic	Plausible	Plausible
	Magnitude of change	Plausible	Unrealistic	Unrealistic	plausible
	Livelihood/welfare				
	Direction of change	Plausible	Plausible	Plausible	Plausible
	Magnitude of change	Plausible	Don't know	Unrealistic	Unrealistic
	Carbon stocks				
	Direction of change	Plausible	Plausible	Plausible	Plausible

Model characteristics	Group 1	Group 2	Group 3	Group 4
Magnitude of change	Plausible	Don't know	Don't know	Unrealistic
What is the acceptable time length of a prospective study? Why?	30 years, long enough to prospect for the next generation based on current policies	Short term (5 years) for policy evaluation and long term (25 years) for strategic policy planning	5 years, as this matches current length of time for policy planning	5 years, as this matches current length of time for policy planning

B. Scenarios developed by model users

Group	Scenarios	Expected results		Actual results		Confidence in FALLOW model (%)
		Relative economic additionality (%)	Relative ecological additionality (%)	Relative economic additionality (%)	Relative ecological additionality (%)	
1	Improvement of rubber systems	+50	+50	+354	+21	70
2	Improvement of oil palm systems	+50	+50	+24	+1.6	75
3	Improvement of oil palm systems	+75	-10	+10	-20	60
4	Improvement of rubber systems	+60	+40	+417	+21	85

2.5.3.3 Operating FALLOW

Overall, participants found outputs of the FALLOW model were easy to understand (Table 2.4), particularly the dynamics of land cover maps. Moreover, they considered the outputs as moderately plausible and useful as input for land use planning and natural resource management of their area. However, the participants found that operating FALLOW, in particular the preparation of input parameters was challenging. They were concerned with data scarcity and technical difficulties in obtaining input parameters (such as aboveground biomass for various systems). Understanding the conceptual theory underlying the model was also considered difficult.

Table 2.4 Potential model users' evaluation of the FALLOW model, in percentage of respondents.

Model characteristics	Good (%)	Moderate (%)	Poor (%)
Ease in:			
• input preparation	27	9	64
• operating the model	27	36	36
• understanding the conceptual theories underlying the model	36	18	45
• understanding model results/outputs	82	8	0
Relevance of scenarios	64	36	0
Plausibility of results	45	55	0
Efficacy of model results for land use planning and agriculture and NRM	55	45	0

2.6 Discussion

Our first study focused on evaluating potential model users' concerns when given a hypothetical model and the West Aceh study case explored how these concerns change while working with an actual model. In assessing a hypothetical model, users' ranking of model characteristics put model salience (useful outputs) first, followed by how easy it is to operate the model and how credible it is (operational characteristics) and lastly what the model track record is. When users had direct experience in using a specific model, the emphasis shifted towards credibility. In a setting where basic salience issues had been confirmed, credibility became of importance. This indicate the incessantly importance of putting efforts to improve model outputs accuracy in model building to gain users acceptance.

The first scenario choice by model users' in the West Aceh case (increasing productivity of rubber and oil palm; not integrated scenarios or conservation) and a preference for a 5-year-model run suggested that the users' were operating at efficiency scale (Jackson et al., 2010), focusing on the economic profitability/ resource sufficiency goal of sustainability. One may infer that the recent experience of system shock may have triggered interest to prospect on a functional integrity goal of sustainability working at temporal scale of persistence or 'change. Their disinterest may be due to their current professional mandate or their ease and confidence in knowledge which lay both largely still at efficiency scale. The scale that users' operated may partly explain the shift of users' interest from salience towards credibility, i.e. the ability of the model results to depict reality on the ground (precision). For model users working at efficiency scale, precision of outputs would be of more important compared to users working at higher scale of persistence or change. At persistence or change scale, obtaining precision will be difficult as real field measurement will not be available, hence salience would be of more importance.

In a development context, our study in Aceh was timely as it provided the local government staff with knowledge and technical expertise, after being in a vacuum for the past three decades due to the separatist conflict. The prospective study using the FALLOW was their first experience in using a landscape simulation model. Recognizing interactions they had observed on the ground build confidence. Particularly, they liked graphs such as Figure 2.4 that were easy to understand and provided new insights into trade-offs.

Potential model users in West Aceh, when given a chance to work with the FALLOW model, directly understood that data-sparse conditions in their area could hinder their wish to work further with this model. This is a common problem in developing countries. Raising awareness towards use of model at higher scale (persistence or change) for prospective studies, exploring trade-offs with various scenarios that will be able to use qualitative data would be essential. Defining 'credibility' at a much broader concept that includes the ability of models to simulate various types of scenarios involving external changes and still producing understandable results would probably be more important than pixel 'goodness-of-fit'.

Our study intended to evaluate users' perception of simulation models referring to the salience-credibility-legitimacy framework. However, finding simple, unbiased model characteristics for the legitimacy attribute in the local language was not easy. A double negative construction of attribute ("…not having other agenda's") was easily misinterpreted in the pre-test phase of survey. White et al. (2010) reported a similar study exploring water managers' assessment of a hydrological model using the salience-credibility-legitimacy framework. They allowed participants to give open ended comments for salience and credibility attributes and post-hoc categorized responses in terms of positive, negative or neutral assessment. However, such an approach may not always be able to give sufficient feedback in the local cultural setting of our study site, where criticism tends to be concealed and expressed in understatements. A combined approach between White's study and our study, i.e. conducting an open-ended attribute assessment followed by ranking the summary of users' statements with additional characteristics from researchers may be interesting to explore.

2.7 Conclusion

Our paper explored potential model concerns and perspectives on what is considered to be an acceptable performance of a simulation model given a specific type of model use. The intended use of a model, as stated by most potential model users, is to help them to prospect the future, predict consequences of choices and assist in making decision, by working in a more systematic way. The findings of our study are not unexpected. Salience or relevancy is naturally the most important attribute as otherwise (academically) perfect models that only answer irrelevant questions in users' perspectives have limited utility. Credibility becomes important once salience is positively acknowledged and, once the model was used, became a critical aspect for users. However, our study elucidates the importance of involving model users' in evaluating a simulation model including its scenarios and results. This approach is as important as statistical validation tests, particularly if the main goal is to promote the use of simulation model for natural resource managers. Technical approaches in validation have their limit and may not always be feasible in data constrained situations. Assessments of salience-credibility-legitimacy dimensions can be used as a framework to evaluate utility of a simulation model.

Chapter

3

Land sparing or sharing? Exploring livestock fodder options in combination with land use zoning and consequences for livelihoods and net carbon stocks using the FALLOW model[5]

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3.1 Abstract

Livestock as an integral part of farming systems can increase resource use efficiency and land use intensity of agricultural systems, but can also be a driver of forest conversion and associated greenhouse gas emissions. Forest policies that limit land use options may be able to halt forest change, if strongly enforced, but concurrently may also reduce livestock carrying capacity. This study explored the use of the spatially explicit FALLOW model, with a new livestock module, to assess the impact of land use zoning strategies, in combination with access to fodder harvesting, on welfare, fodder availability and landscape carbon stocks in the Upper Konto catchment, Indonesia. The existing land zoning in Upper Konto catchment is in name 'land-sparing' but de facto combined with 'land sharing' approach with access to cut and carry fodder sources in watershed protection areas. Scenario analysis revealed that the existing land zoning approach is the most promising in terms of balancing fodder availability, farmers' welfare (total profits gained from production in the landscape minus products consumed by people living in the area) and ecosystem functions (with above-ground carbon stocks as indicator). A pure land sparing approach with agricultural intensification indicates increase in farmers' welfare but with a higher decrease (in percentage) of landscape above-ground carbon stocks. Hence, careful integration of livestock systems into zoned conservation areas can achieve multiple goals including enhancing peoples' livelihoods and protecting environmental services.

3.2 Keywords

Carbon stocks livelihood trade-offs, land sharing vs. sparing, land use zoning, model of ruminant cut-carry systems, scenario analysis.

3.3 Introduction

Throughout the world, a growing population increases the demand for food, fibre, feed and energy causing accelerated forest conversion to agricultural land (Tilman et al.,

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2011). During 1980-2000, 83% of new agricultural land in the tropics originated from forest conversion (Gibbs et al., 2010). The reduction of forest areas has raised concerns over its impact on the degradation of ecosystem services (Tomich et al., 2005), in particular carbon storage (West et al., 2010), biodiversity (Danielsen et al., 2009) and hydrological functions (Foley et al., 2005, Ellison et al., 2012).

Two strategies are commonly proposed to halt agricultural expansion into forest areas: 1) land sparing or segregation, and 2) land sharing or integration. The debate involves multiple scales of assessment, multiple perspectives on drivers and causality, and multiple interpretations of 'forest' or 'nature' as the complement to 'agriculture' in meeting current and future human needs. The 'land-sparing' or 'segregate' view of land use zoning and agricultural intensification asserts that in order to maximize the area for conservation and ecosystem services provision, the land allocated for agricultural production must be minimised by maximising its productivity (Phalan et al., 2011a). Phalan et al.(2011b) argued that 'land sparing' minimises the negative impacts of food production, but recognised the need to restore degraded land into productive land to reduce pressure on biodiversity rich (wild) areas.

Alternatively, 'land-sharing' or 'integrative' approaches, emphasize potential synergy and multi-functionality in intensification gradients from forest to intensive land use, converging agricultural production with nature conservation (Van Noordwijk et al., 1997). This approach known as the 'agroecological matrix' approach (Vandermeer and Perfecto, 2007) requires a shift from a purely agronomic goal of production towards sustainability, with 'sustainable intensification' as the pathway (Perfecto and Vandermeer, 2010). In the tropics, well managed agroforests (e.g. cacao or coffee multistrata systems) are known as systems that allow sustainable intensification (Tscharntke et al., 2011). These systems are currently under global market pressure for further intensification, shifting to less sustainable monoculture systems (Feintrenie and Levang, 2009).

Although the 'land sparing' versus 'land sharing' debate is often presented as a black-or-white choice, there are many shades of grey occurring in land-use planning influenced by geographical, ecological, economic, social and political factors. Fischer et al (2008) considered that both approaches offer different but complementary advantages and outlined a broad policy guideline for conservation in agricultural landscapes depending on the suitability of each piece of land and their trade-offs between conservation and production. Tscharntke et al. (2012) argued that multiple perspectives on agricultural intensification and its reliance on external inputs versus optimised use of ecological interactions are crucial. Economic and socio-political aspects that could hamper the effectiveness of the 'land-sparing' approach are being recognised. Examples are, the tendency of farmers to further convert forest when efficient agriculture systems are deemed profitable (Angelsen and Kaimowitz, 2001; Steffan-Dewenter et al., 2007) and Protected Areas schemes that often trigger conflicts between 'land protectors' (e.g. government) and local people particularly in land constrained areas (Kusters et al., 2007).

The existing literature on the land sparing versus sharing debate has mostly focused on biodiversity versus production. However, economic incentives for REDD (Reducing Emission from Deforestation and Forest Degradation) have raised similar issues within the forest and carbon community (Minang et al., 2011). Gockowski and Sonwa (2011) estimated that intensified cocoa technology (by using fertiliser and pesticides) could have saved 21,000 km^2 of deforestation and forest degradation and thus, avoided 1.4 billion Mg of CO_2 emission. They supported the use of agricultural intensification and land sparing approach in the REDD mechanism; however they also proposed plating native timber trees within the intensified cocoa plots to reduce forest degradation due to timber and fuel wood harvest, an approach also suggested in 'wildlife-friendly farming' (Fischer et al., 2008).

Jackson (2012) captured differences between tropical agroforest domains and intensively managed agricultural zones including systems that have followed degradation pathways where little agro-biodioversity is left. However, there is still little information on systems in between tropical agroforest domains and intensive agriculture that may be able to provide opportunities to maintain agro-biodiversity or other ecosystem services. Agricultural systems that include livestock such as agro-pastoral systems (combining crops and livestock) or agro-silvopastoral systems (combining tree species, crops, and livestock) provide options along an intensification pathway that may be able to halt further conversion of forest to cropping land. In such mixed systems, livestock plays an important role for livelihood by producing food, generating income, storing capital reserves and enhancing social status (Randolph et al., 2007). Livestock, particularly ruminants, are also important in providing manure for soil fertility and traction for land preparation and transportation (Herrero et al., 2010). Globally, intensive and large-scale livestock systems have been blamed for causing deforestation and the acceleration of agricultural intensification (Steinfeld et al., 2006), in particular pastoral systems that required vast areas of land (Nepstad et al., 2006). However, this may not be the case for cut and carry systems in mixed farming systems where fodder supply is mostly derived from existing land uses.

Herrero et al. (2009) highlighted the importance of understanding the trade-off between livestock, livelihood and the environment. Understanding how livestock management, particularly its feed production, can influence land use change, and consequently the environment, will ensure that the development of policies pertaining to livestock rearing will enable livestock to continue providing livelihood benefits while improving or maintaining agro-ecosystem and environmental services such as carbon sequestration. Well managed grassland systems are commonly considered to be favourable in terms of sequestering soil organic carbon (Fisher et al., 1994; Conant et al., 2001; Mutuo et al., 2005Bÿ6). Integrating forage production into tree-based systems, e.g. silvopastoral or agrosilvopastoral systems, will enhance carbon stocks in the landscape compared to pure pastoral/grassland systems or cropping systems while at the same time enhance economic productivity and farmers livelihoods. Thornton and Herrero (2001) developed a framework for developing generic integrated crop–livestock simulation models for scenario analysis and impact assessment. Since then, a number of integrated livestock system models have been developed, ranging from the farm-level (Castelán-Ortega et al., 2003; van Wijk et al., 2009) to the landscape level (Schreinemachers et al., 2007; Parsons et al., 2011) that simulate mixed farming systems including their biophysical and economic components. However, to date, there are few land use change models with low data requirements that simulate interactions between livelihood, livestock, forest and landscape. The development of such a model is pertinent for many tropical countries with low data availability.

This paper presents a case study of the Upper Konto catchment, an agricultural landscape of East Java, Indonesia where the question of land sparing or sharing is raised at the watershed scale. The watershed presents a landscape of mixed agroforest and rice with forest remnants, which is typically found in Southeast Asia, where dairy cattle and horticulture in a peri-urban setting lead to rapid land use change and forest conversion (Zhao et al., 2006). Conflicts over access to land have occurred in the past as two-thirds of the land was allocated to forest for production and conservation purposes. Therefore, it serves well as a relevant study case for stakeholders (e.g. local policy makers and scientists) to prospect the trajectory of current land use and the scenarios that would enhance farmers livelihood as well as the environmental services (above-ground carbon sequestration) of the landscape.

Using a spatially explicit dynamic landscape model (FALLOW, van Noordwijk et al., 2002) we explored: (i) the impacts of change in forest zone policy, (ii) the potential of further integration of fodder production in forest areas, and (iii) the impacts of open access of all land, on farmers' welfare and above-ground carbon sequestration in the entire landscape. Modelling and prospecting changes in a landscape that has livestock systems as livelihood options requires a land use model that can connect the landscape dynamics to the livestock systems with inter-linkages through household economics and land use systems. FALLOW has been successfully applied in prospecting trade-offs between carbon sequestration and local development benefits (van Noordwijk et al., 2008). However, the current FALLOW 2.0 does not explicitly include livestock options and thus a further objective was to develop a livestock (large ruminant) module to enhance the capacity of the model to assess trade-offs between different land sparing or sharing options on carbon sequestration and livelihoods.

Specific questions were considered: (i) How can livestock options be integrated in an existing land-change model? (ii) What landscape and economic dynamics can be expected under a 'business as usual' scenario? (iii) How closely does the FALLOW model application match recorded historical land use change? (iv) What impacts are expected from land use zoning on trade-offs between fodder availability, carbon sequestration and livelihoods and (v) What does the case study suggest on a combination of land sharing and land sparing approaches?

3.4 Methods

3.4.1 Study site

The Upper Konto catchment (237 km^2) is located in Malang, East Java, Indonesia; spanning elevations of 600 -2800 m above sea level with an average annual rainfall of 3000 mm.year^{-1} and a dry season that last two to four months. The landforms of the area consist of geologically young volcanic complexes, combined with eruption material and colluvial sediments, which have formed thick layers of highly permeable and relatively fertile soils, such as Andisols, Cambisols and Luvisols (Nibbering and Graaff, 1998; Rijsdijk et al., 2007).

Since the late 1980, one third of the area has consisted of privately owned farmland and settlements, which occupy the valleys, the lower and middle plateaus, and the foot slopes of the mountains. The farmland is used for intensive forms of agriculture, such as highland vegetable and maize cultivation in the upper parts, and maize, wet rice and perennial crop cultivation, notably coffee in the lower parts. Dairy farming (stall-fed cattle) is an important activity (Nibbering and Graaff, 1998; Rijsdijk et al., 2007). The other two thirds of the area consist of forest land covered with plantation forest and the remnants of natural forest in various stages of degradation. Most of the forest is found in the hilly and mountainous parts above 1400 m. The plantation forest is managed by Perum Perhutani, a state-owned forest corporation (SFC). Land cover types and livelihood options in the Upper Konto catchment have been relatively stable over the last 30 years. Vegetable, rice and coffee production as well as dairy farming are still the main livelihood options for people in the area. Farmers have started to introduce cacao to the area which in some plots is partially or fully replacing coffee. Over the years, the dairy cattle population has increased quite rapidly; the population of dairy cows has increased by 44% during the period 1990-2008. In 2010, milk production from the Upper Konto catchment reached approximately 194,000 litres day^{-1} or around 16% of milk production in Indonesia (Tribun Lampung, 2010).

Dairy cattle in the Upper Konto catchment are fed by manual feeding (or 'cut and carry') systems. Animals are usually fed in their stalls twice daily, with fodder cut and carried by farmers from the surrounding landscape. Napier grass (***Pennisetum purpureum***) is planted around houses and along the borders of arable land, banks and roadsides. For villages close to the plantation forest area, Napier grass under trees becomes an abundant source for fodder. Dairy cattle farmers spend considerable amount of time searching and gathering fodder. During extreme dry seasons, farmers need to buy fodder from sellers who come from outside Upper Konto catchment. All dairy cattle farmers are members of their local dairy cooperative, and obtain Friesian Holstein cattle (imported from Australia and New Zealand). The cooperatives provide all farmers with additional inputs such as concentrates for feeding. The cooperatives provide each village with well-equipped milk-storage facilities enabling dairy cattle farmers to have good access to the regional market.

3.4.2 Past land use and land cover change

There have been substantial changes in the land use of the area, particularly in the midst of the social and political instability that erupted in Indonesia following the onset of the Asian economic crisis in 1997 (Large, 2005). The government produced Designated Land Use[6] (Figure 3.1) map to provide guidelines in land use and spatial planning for local government. It contains three main categories of land use: 1) Forest Reserve; which is land allocated for forest/trees set aside for soil, land and biodiversity conservation, 2)

6 Land use is defined as a term describing how people utilize the land while land cover is the physical material on the surface of the earth (Comber et al., 2008).

Production Forest; which is plantation forest area for production purposes (e.g. wood, resin), and 3) Other Land Uses which includes settlement and agricultural activities. The Forest Reserve and Forest Production areas are owned and managed by the government while the Other Land Uses is a land use category for land that generally can be owned privately by individuals. In the Upper Konto catchment and elsewhere in Java, Production Forest corresponds to the plantation forest owned by the SFC.

The land cover map classified from LANDSAT-TM imagery (Hairiah et al., 2010) showed that between the period of 1990 to 2005 (Table 3.1) agricultural lands and settlement have increased from 36% to 55%, while the natural forest area has declined from 35% to 23%. However, from the point of view of the government a 1:2 ratio between non-forest and forest lands (as was the situation prior to the late 1980) is still the main target for the preferred landscape. This is depicted in the Designated Land Use Map produced by the Ministry of Forestry, where the total forest area (Forest Reserve and Production Forest) amounts to 70.9% of the area compared to 29.1 % of non-forest area.

Table 3.1 Comparison of land use areas form Upper Konto catchment Designated Land Use Map and land cover maps derived from Landsat-TM imagery of 1990, 2000 and 2005.

Designated land Use[a]		Land cover map[b]			
Land use category	Area (%)	Land cover type	Area (%)		
			1990	2000	2005
Forest Reserve	38.5	Degraded natural forest	34.9	27.1	22.8
Production Forest	32.4	Forest Plantation	28.8	29.9	22.3
Other Land Uses	29.1	Non-forest	36.2	43.1	54.9
		Settlement	0.8	1.6	1.6
		Bush fallow	3.4	1.1	0.7
		Agriculture	20.7	23.3	44.0
		Agroforestry	11.3	17.2	8.6

[a] Based on Designated Forest Area Map for East Java Province. Directorate General of Forest Inventory and Land Use Planning, Ministry of Forestry (2002)

[b]Image classification of LANDSAT-TM (Hairiah et al., 2010) with accuracy value (Kappa) of 73%.

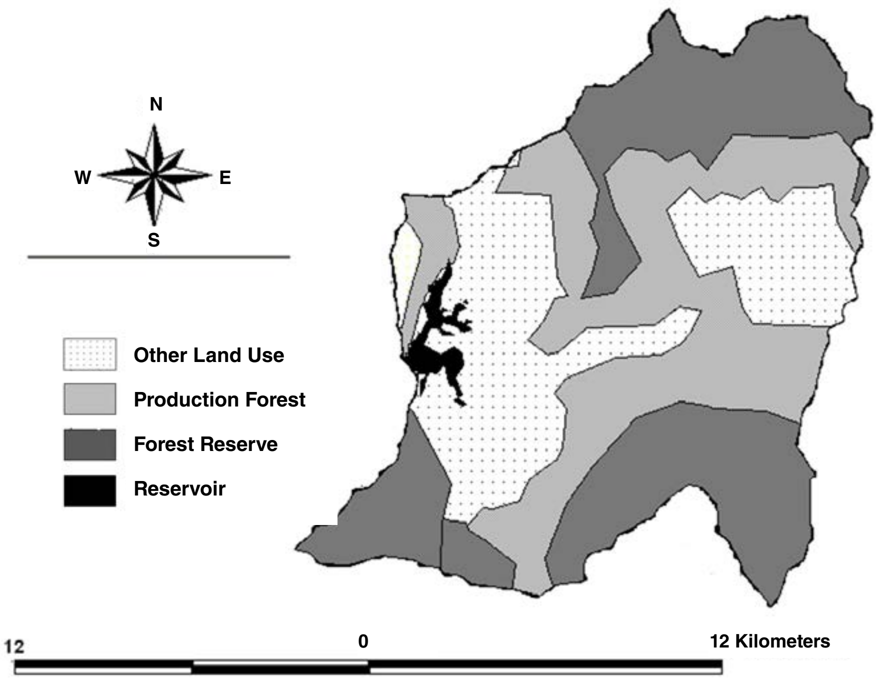


Figure 3.1 Map of designated land use in Upper Konto Catchment. Source: Designated Forest Area Map for East Java Province. Directorate General of Forest Inventory and Land Use Planning, Ministry of Forestry (2002).

3.4.3 FALLOW model: land use change impact assessment tool

Land use change models to understand the trade-offs between livestock, livelihood and environment need not necessarily be complex, in contrast to models that simulate detailed biophysical interactions between the crop, soil and livestock. However, it needs to include the important drivers involved in land use change processes, such as household economics and its influence on decision making and systems productivity. For such a study, ensuring saliency of a land use model – the ability of the model to perform the required simulation scenarios - is pertinent (Lusiana et al., 2011). Thus, desirable land use change models must be generic to enable their use at various sites and flexible enough to be modified for situation-specific processes and interactions (Rindfuss et al., 2008).

The FALLOW model (van Noordwijk et al., 2002) is a spatially explicit landscape dynamics model that analying drivers and consequences of land use change on a yearly basis at meso-scale. It was developed as an impact assessment tool to help integrate our understanding of landscape mosaics and resources. The FALLOW model treats land use and land cover land use and land cover simultaneously, assuming that land-use dynamics are a major determinant of land-cover change. FALLOW considers the roles of actors/farmers in transforming the landscape, biophysical responses from plot- to landscape-levels, and feedback from and actors'/stake-holders' on the changing landscape. Suyamto et al. (2003) and Suyamto et al. (2009) provide detailed descriptions of processes and inter linkages involved in the FALLOW model.

The dynamic interactions between different modules in FALLOW (Figure 3.2) start from the changes in soil fertility at plot-level based on the Trenbath model (Trenbath, 1989; van Noordwijk, 1999), where soil fertility is depleted during cropping periods and recovers during fallow periods. Current fertility at plot-scale determines the agricultural yield (for crop and/or tree based systems). The total agricultural production from the whole landscape together with the yield gained from other systems involving economic production (e.g. forest resource utilisation activities, off-farm activities) contributes to food sufficiency and/or household economic resources. Population dynamics (based on local population growth rate that includes natural growth and migration) affect the magnitude of available labour force as well as the demand for food. Farmers conduct agricultural activities to meet food demand or their food-equivalent cost of living. The strategic decision to open new land or to expand other economic production activities depends on the available labour, financial capital and land. This decision determines the magnitude of land use change in the model.

The model incorporate a simple optimization approach where it is assumed that farmers make a choice to grow a crop/tree on a given plot or undertake other production activities with expectation of receiving the highest relative net labour or land return. The economic expectation starts with a certain initial knowledge and is able to change dynamically through learning from experience or from new information acquired during the simulation (e.g. from extension services, neighbouring farmers). Farmers will select suitable plots for clearing and planting based on their perceptions of plot attractiveness which is a function of relative soil fertility, land and market accessibility (i.e slope, distance to a road/river, distance to market, distance to processing factory), land tenure status and spatially explicit rules on land zonation. This decision determines location of the land use change. Activities related to agricultural land expansion will disturb natural succession as well as soil fertility recovery processes of the cleared plots. The overall landscape dynamics will lead to environmental consequences (changes in above-ground carbon stocks, biodiversity) at the landscape level.

The FALLOW model has been applied for assessing trade-offs between livelihoods and the environment (van Noordwijk et al., 2008) and for determining changes in farmer's behaviour due to changes in soil fertility (Lippe et al., 2011). However, the model does not have a livestock systems component. Thus, we developed a livestock (large ruminant) component that interacts with household economics, farmers' decision making and land cover/use dynamics (Figure 3.2).

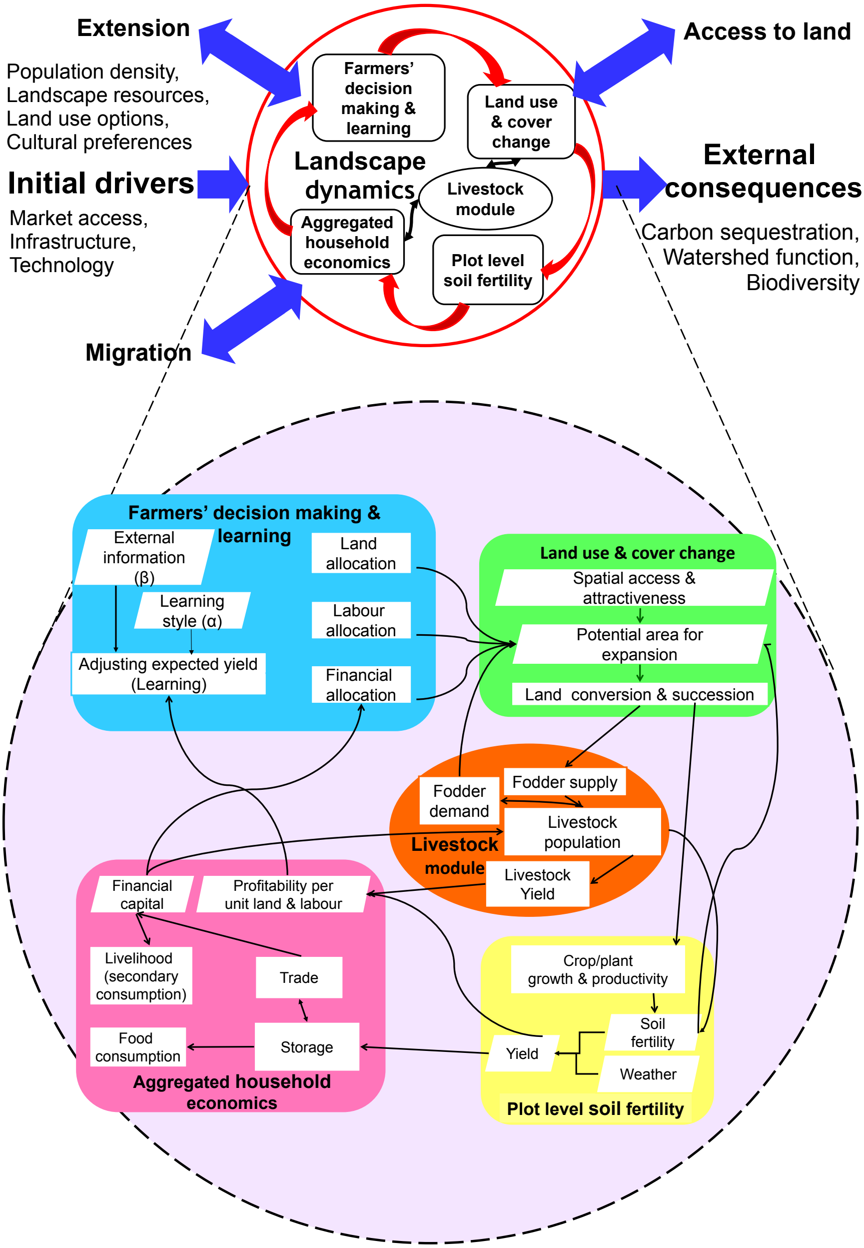


Figure 3.2 Schematic diagram of entire FALLOW model (upper part) and its modules (lower part of figure) including the Livestock module. The current Livestock module is directly linked to all other modules.

3.4.4 Development of livestock module for the FALLOW model

Seré and Steinfeld (1996) classified global livestock production systems into two main groups based on their feed sources: 1) solely livestock systems[7] and (2) mixed farming systems[8]. We developed a generic livestock module within FALLOW that allows assessment of both these systems. Based on the percentage of fodder obtained from the landscape and the proportion of time and space the grazing areas are utilised by livestock, we can differentiate four types of livestock systems (Figure 3.3): 1) free-range grazing, 2) confined pasture (ley, permanent), 3) zero grazing with cut and carry, and 4) landless livestock[9] (e.g. fattening in feedlots).

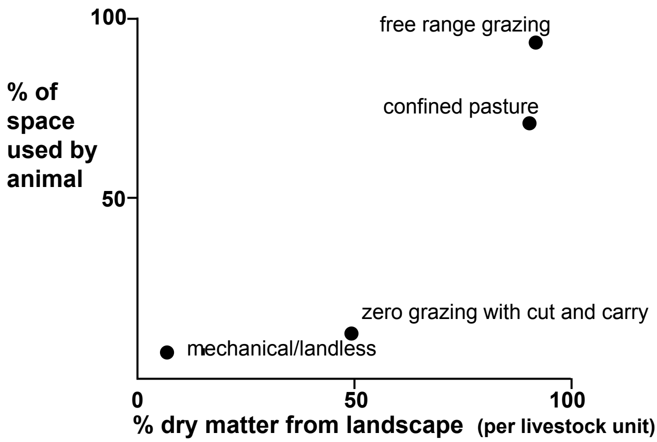


Figure 3.3 Types of livestock-based production systems based on percentage of dry matter/feed originated from landscape per livestock unit and percentage of space used by animals.

3.4.4.1 Livestock carrying capacity

Livestock carrying capacity is defined as the number of livestock units that can be sustained by the landscape (biophysically and financially). Here, the potential livestock carrying capacity is determined by the amount of fodder that is available (for grazing or harvest by farmers) from respective land use systems within the landscape (grass planted in the forest, monoculture systems of Napier grass, maize stover, rice straws or grasses along footpaths) and brought to the landscape from outside (if financial surplus is produced by the 'landscape'). Three factors influence the quantity of fodder produced

7 Livestock systems in which more than 90% of dry matter fed to animals comes from rangelands, pastures, annual forages and purchased feeds and less than 10% of the total value of production comes from non-livestock farming

8 Livestock systems in which more than 10% of the dry matter fed to animals comes from crop by-products and stubble, or more than 10% of the total value of production comes from non-livestock farming activities

9 A solely livestock system,where less than 10 % of the feed dry matter fed to the animals is farm produced. Feeding is mainly based on good quality high energy feeds with a maximum intake of concentrates and a minimum intake of roughage; these systems depend on an 'external footprint' beyond the landscape

in a particular land use systems: area, access and a specific livestock carrying capacity index for each land use type (Eq. 3.1).

$$L_{pot} = \Sigma_i A_i * C_i * g_i + F_{external} \quad (3.1)$$

where, L_{pot} = potential landscape livestock carrying capacity (number of livestock unit); A_i = area of land use i (ha); C_i = potential fodder yield or carrying capacity index of land use i (livestock units.ha^{-1}); g_i = a value 0 or 1 defining access to land use i (0 means no access, e.g. exclusion areas, protected forest and 1 means with access to these areas), and $F_{external}$ = fodder brought in from outside the landscape.

The livestock module in FALLOW can simulate a range of livestock systems (Figure 3.3) by parameterising different values of A, C and g. C_i is site specific depending on the site fertility and the type (and hence the quality) of fodder commonly found in each land use Table 3.2 shows examples of the livestock carrying capacity indexes for an application in the Upper Konto catchment. Specific for the Upper Konto catchment application, the index is defined as the number of tropical livestock units (300kg each) sustained per hectare of land use and refers to the amount of fodder harvested (cut and carry) by farmers. We treat monoculture fodder systems as other agricultural systems, i.e. economically compete with other livelihood options such as crop and agroforestry systems for available land as well as labour and financial capital.

3.4.4.2 Livestock dynamics

The livestock module includes a simplified livestock population dynamics approach based on the assumption that the ruminant population strongly depends on livestock carrying capacity and available capital to establish livestock systems. The annual growth of livestock is also constrained by the total number of livestock that can be managed by farmers.

In the FALLOW model, farmers in the landscape are represented by groups of farmers with the total number equal to the number of main livelihood options in the landscape. Each farmer represents an average farmer of a particular land use system/livelihood option. Decisions are simulated collectively for each livelihood option and not at the household level (Villamor et al., 2011), as opposed to the approach taken in multi-agent based systems (Berger and Schreinemachers, 2006). Henceforth, decisions related to livestock rearing including its inputs and outputs refers to aggregated scale over the total livestock in the landscape.

Table 3.2 Livestock-carrying-capacity index of different land use types in Upper Konto catchment, e.g. weeds, understory vegetation and Napier grass (***Pennisetum purpureum)***[a]

Land Use[b]	Carrying capacity index[c]
Napier monoculture (fertilized)[d]	10
Settlement (borders, homegarden)	1
Bush fallow	3
Young Secondary Forest	1
Old Secondary Forest	0
Primary Forest	0
Horticulture (weeds, borders)	0.5
Maize/Rice/Staple Food[e]	0.8
Forest plantation[f]: early stage	1
Forest plantation: young stage	2
Forest plantation: productive stage	0.5
Forest plantation: post-productive stage	4
Cacao[f]: early stage (weed control)	0
Cacao: young stage	0.5
Cacao: productive stage	0.5
Cacao: post-productive stage	0
Coffee: early stage (weed control)	0
Coffee[g]: young stage	0.5
Coffee: productive stage	1
Coffee: post-productive stage	0.2

[a]Based on farmers' interview, personal observation and Abdullah (2006)

[b]The land use types reflected the land use trajectories approach implemented in the FALLOW model where forest and tree-based systems are classified into four stages based on their ecological maturity and succession (for forest) and based on productivity and growth (for tree-based systems, Suyamto et al., 2009)

[c]Number of tropical livestock units (300kg) sustained per ha of land use

[d]Based on Parikesit et al. (2005) and potential dry matter provision and cattle requirements (i.e. 7 kg DM.300 kg animal unit^{-1}.day^{-1}) and farmers interviews

[e]Weeds, maize stover, border planting of Napier grass

[f]Napier planted in tree plantations, e.g. resin producing trees (***Pinus mercusii***), or ***Agathis alba*** and ***Swietenia mahogany***, with restricted access to plant Napier during early-mid productive phase. Tree spacing 3 x 4 m.

[g]Border planting of Napier grass and understorey vegetation

$$L_{act} = R \quad \text{...}(3.2)$$

$$\Delta L = L_{pot} - L_{act} \quad \text{...}(3.3)$$

where L_{act} = actual livestock population in a given landscape i.e. total livestock population (R, number of livestock units present). ΔL reflects the ability of the landscape to sustain additional livestock.

When ΔL is positive farmers increases livestock population using Eq. 3.4:

$$\text{If } \Delta L > 0 \text{ then } \Delta R = \min(L_R, M_R, R{*}R_f) \quad \text{...}(3.4)$$

where, Δ***R*** = changes in the livestock population, which can be positive (increasing) or negative (decreasing) given by the most limiting factor out of the following: L_R = potential additional livestock (livestock units) based on available labour; M_R = potential additional livestock (livestock units) based on available financial capital; and $R{*}R_f$ = additional livestock (livestock units) that can be managed by farmers each year, with R_f as fraction of the total livestock population.

When Δ*L* is negative, FALLOW assumes that farmers reduce the provision of feed before buying fodder. Hence, in cases of fodder scarcity, farmers accept suboptimal livestock performance until the ratio between supply and demand (***fodder ratio***) reaches a certain pre-defined threshold (Eq. 3.5). The reduction of fodder provision reduces milk yield and thereby income. When the ***fodder ratio*** reaches the defined threshold, farmers buy fodder from outside the landscape (Eq.3. 6). However, there is a user-defined limit on the amount of capital allocated for buying fodder. With lack of financial capital to buy fodder, farmers sell the livestock to reduce population and therefore reducing the fodder demand. A user-defined parameter constrains the number of ruminants that farmers can sell each year reflecting the farmers' gradual decision to reduce their source of income (Eq.3.7 and 3.8).

$$F_{rt} = L_{pot}/L_{act} \quad \text{...}(3.5)$$

where F_{rt} = fodder ratio, the ratio between potential fodder supply and actual fodder demand which reflects the relative amount of fodder obtained by each animal

$$\text{if } F_{rt} < F_{th} \text{ and } \Delta L < 0 \text{ then } F_{external} = \min(-\Delta L, F_{fb}{*}M/P_f) \quad \text{...}(3.6)$$

where , F_{th} = fodder ratio threshold, below which farmers will decide to buy external fodder as the yield production becomes too low, $F_{external}$ = amount of fodder bought (in livestock units), F_{fb} = fraction of financial capital that can be allocated to buying fodder, M = total available financial capital (US$), and P_f = fodder price (US$.livestock unit^{-1}).

$\Delta F = min(0, L_{act} - L_{pot})$ ....(3.7)

if $\Delta L < 0$ and $F_{external} < -\Delta F$ then $R_s = min(-\Delta F, R*R_{fs})$ ....(3.8)

where ΔF = fodder shortage (in livestock units, with the value always less than or equal to 0); R_s = the number of animals sold (in livestock units); and R_{fs} = the fraction of animal that can be sold in a year (as a fraction of the total ruminant population).

Supplement S.3.1 lists input parameters used in the livestock module. Other relevant input parameters for livestock dynamics are listed in Table 3.2, 3.3 and 3.4.

3.4.4.3 Impact of livestock (manure) on soil fertility

As discussed in Section 3.4.3, processes pertaining to soil fertility dynamics in the FALLOW model are based on the Trenbath model where soil fertility is assumed to decrease during cropping periods and recover during the fallow phase. Soil fertility is defined as a qualitative term of fertility units (van Noordwijk, 2002). Reduction of soil fertility is equivalent to an increase in plant yield (reflecting decomposition of soil organic matter and associated nutrient release). With the application of fertiliser, the impact of fertilisation on yield is differentiated from its impact on soil fertility. Comparatively, the impact of manure on soil fertility recovery is stronger than that of chemical fertilizer, whilst manure impact on yield is weaker than that of chemical fertilizer. Soil fertility dynamics and yield are expressed in the following equations:

$Y = f_d * E_c$ ....(3.9)

where, Y = actual yield ($Mg.ha^{-1}$); f_d = soil fertility depletion (fertility unit); and E_c = plant efficiency to convert released fertility (mineralized 'fertility') into yield during the planting season ($Mg.fertility\ unit^{-1}.ha^{-1}$).

$f_d = Fert * f_{dr}$ ....(3.10)

where, Fert = soil fertility (fertility unit, qualitative scale); and fdr = the fraction by which soil fertility decreases due to mineralisation during the planting season (dimensionless).

$f_{rec\text{-}fallow} = (Fert_{max} - Fert)^2/((1+hr)*Fert_{max} - Fert)$ ....(3.11)

where, $f_{rec\text{-}fallow}$ = soil fertility recovery due to fallowing land (fertility units); $Fert_{max}$ = the potential soil fertility value to which the soil returns after an infinitely long fallow period (fertility units); and hr = half recovery time for soil fertility, i.e time (years) needed to reach half $Fert_{max}$.

$$f_{rec\text{-}fert} = Fert^* f_{fert} \quad (3.12)$$

where, $f_{rec\text{-}fert}$ = soil fertility recovery due to fertiliser application (fertility units), and f_{fert} = the fraction by which soil fertility increases due to fertiliser application during planting season (dimensionless).

Finally, soil fertility at year ***t*** (*Fert$_t$)* is calculated using the following Eq. 3.13.

$$Fert_t = min(Fert_{max}, max(Fert_{t-1} + f_{rec\text{-}fallow\ t-1} + f_{rec\text{-}fert\ t-1} - f_{d\ t-1}, 0) \quad (3.13)$$

where $Fert_{t-1}$ = soil fertility at year t-1; $f_{rec\ t-1}$ = soil fertility recovery at year *t*-1; and $f_{d\ t-1}$ = soil fertility depletion at year *t*-1.

Supplement S.3.2 presents the plot level relationships between the yield and soil fertility for the two agricultural crops simulated in this study (non-intensive and intensive agriculture) including sensitivity analysis of the same relationship for various parameter values of f_{dr} and E_c. van Noordwijk (1999) has conducted sensitivity analysis for other soil parameters.

3.4.5 Sources of data: input parameters and validation of outputs

Data used for this study are a combination of actual field observations, farmers' interviews and secondary data and government statistics. Demographic information was obtained from the Statistical Bureau of Malang Regency, East Java. Dairy cattle population and production information was obtained from the two milk cooperatives in the area Koperasi Unit Desa Sumber Makmur, Ngantang and Koperasi Susu 'SAE', Pujon (Table 3.3).

A semi-structured, in-depth interview was carried out in 2008 to obtain productivity and profitability information of the main land use and livelihood options in the Upper Konto catchment area (Table 3.4). The five livelihood options were: (i) coffee and (ii) cacao agroforestry systems, (iii) intensive agriculture (horticulture), (iv) non-intensive agriculture (staple food, i.e. rice and maize) and (v) dairy cattle. For each livelihood option, five respondents were purposively chosen to represent the variety of crops (for horticulture) and the different stages of agroforestry systems. Additional socio-economic information was acquired from a household survey undertaken in 2008 to assess the impact of the Community-Based Forest Management Programme (*PHBM* = Pengelolaan ***Hutan Berbasis Masyarakat***) on farmers' income (Khususiyah et al., 2010). The household survey was carried out in four villages of the Upper Konto catchment with 120 respondents (30 randomly sampled respondents in each village).

The gross margin of each livelihood options was estimated using Eq. 3.14.

$$TotalGrossMargin_{tj} = \Sigma_i\,(Y_i * Py_i - \Sigma_k(I_{ik} * Pi_{ik})\,) \quad (3.14)$$

where $TotalGrossMargin_{tj}$ = profitability of livelihood ***j*** (US$); Y_i = yield of livelihood option ***i*** (yield units. e.g. litre, kg, m^3); Py_i = price of yield ***i*** (US$.yield unit^{-1}); I_{ik} = amount of input

k (e.g , labour, fertiliser, pesticides, seeds) required by livelihood option ***i*** (input unit, e.g. person days, litre, kg); ***k*** = number of inputs; and $\boldsymbol{Pi_{ik}}$ = price of input ***k f***or livelihood option ***i*** (US$.input unit^{-1}). The FALLOW model uses two types of profitability measure, namely, return to land (US$.ha^{-1}) and return to labour unit (US$.personday^{-1}) as shown in Table 3.4.

Information on time-averaged carbon stocks for the main land cover systems of the Upper Konto catchment were based on plot level carbon measurements carried out in 2006 and 2007 (Hairiah et al., 2010 in Table 3.4). The above-ground carbon stocks refer to the carbon in vegetation (trees, crops and understory), necromass (dead trees) and litter. Time-averaged carbon stocks correspond to the average carbon stored in the different land use systems during their rotation and are used to extrapolated carbon stocks from plot to landscape level (Palm et al., 2005). Hairiah et al. (2011) provide a detailed description of methods to measure carbon stocks, including Eq. 3.15.

$$AG\text{-}C(TA)_{landscape} = \sum_i Veg\text{-}C(TA)_i + Understory\text{-}C(TA)_i + Necromass\text{-}C(TA)_i + Litter\text{-}C(TA) \quad \text{.......} \quad (3.15)$$

where $\boldsymbol{AG\text{-}C_{landscape}}$ is the above-ground carbon stock in the entire landscape; $\boldsymbol{Veg\text{-}C_i}$ is carbon stock (Mg) derived from the aboveground vegetation components including living trees, crops and/or grassland found in land cover ***i***; ***Understory***- ***Ci*** is the carbon stock (Mg) derived from understory growing in land cover ***i***, including native and planted grasses, herbs and shrubs; $\boldsymbol{Necromass\text{-}C_i}$ is dead organic matter pool above the soil surface i.e. dead trees, coarse woody debris, litter and charcoal; and $\boldsymbol{Litter\text{-}C_i}$ is carbon stock (Mg) of standing litter. All carbon stocks estimates are time averaged (TA) values.

The land cover maps were based on image interpretation of Landsat-TM images of year 2000 and 2005 (Hairiah et al., 2010) using hierarchical classification procedure that includes use of ground truth data. The above data were used as input parameters, with exception of land cover map for 2005, cattle population and milk production data that were used to evaluate the model performance.

Table 3.3 Statistics on demography and dairy cattle population in Upper Konto catchment in 2000 and 2005.

Year	2000	2005
Population density [a] (person.km^{-2})	408	439
Labour force[b] (%)	68	72
Households working in agriculture [a] (%)	100	100
Number of dairy cow[c]	25,748	30,000

[a]Source: BPS (2001) and BPS (2006)

[b]Defined as population between 14-55 years old

[c]Source: 2000 and 2005 annual report of milk cooperatives Sumber Makmur-Ngantang and SAE-Pujon

3.4.6 Scenario analysis

The FALLOW model was used to simulate plausible land use zoning options (Table 3.5) and prospect their consequences on fodder availability, farmers' welfare and changes in the above-ground carbon stocks. Farmers' welfare in FALLOW is calculated at the landscape level and is defined as the total profits gained from production in the landscape minus the products consumed by the people living in the area (Eq. 16).

$$W = \sum_j (TotalGrossMargint_j * A_{j)} - (Dfrac_j * Pop * Py_j) \quad \text{.......................................} (3.16)$$

where W = welfare (US$); ***Profit***$_j$ = profitability (US$) for livelihood option ***j***; A_j = area of livelihood option ***j*** (for land-based livelihood option, or total cattle population for dairy cattle system); ***Dfrac***$_j$ = demand for product/yield (self consumption) of livelihood option ***j*** by people living in the landscape (yield unit per capita); ***Pop*** = total population in the landscape; and ***Py***$_j$ = yield of livelihood option ***j*** (US$.yield unit^{-1}).

The land use related policy options were derived from the Upper Konto Designated Land Use map (Figure 3.1). Scenario I ('Business-as-usual') simulated the current condition of the Upper Konto catchment, while Scenario II ('Agroforestry access') reflected the Upper Konto farmers' aspiration to have access to plantation forest areas and plant coffee or cacao in between the timber trees being grown for timber production. Scenario III ('No fodder harvest') represents a more restricted land policy that takes place in other parts of Java intended as pure conservation scheme. Scenario IV ('No monoculture Napier') simulates a hypothetical situation intended to illustrate the value of intensified monoculture of Napier grass systems for fodder availability in the landscape. Scenario V ('Open access') is another hypothetical scenario to prospect the effect of no restrictions on opening up the land. In Scenario I, II, III and V, intensified monoculture Napier grass (MNG) is allowed to grow in the non-conservation area (agricultural zone). We ran the model for 20 years starting from year 2000. For model outputs, we focused on the land cover dynamics over the simulated period; trade-offs between farmers' welfare and above-ground carbon sequestration; and fodder availability and its impact on welfare.

Table 3.4 Productivity, profitability and time-averaged carbon stocks of main livelihood options/land use in Upper Konto catchment.

Livelihood options/ Land cover	Return to labour[a] (US$.person.day^{-1})	Return to land[a] (US$.ha^{-1}.year^{-1})	Labour[b] (person.day.year^{1})	Cost [b] (US$.ha^{-1}.year^{-1})	Yield (Mg.ha^{-1})	Aboveground carbon stocks (Mg C.ha^{-1})[c]
Degraded forest	n.a.	n.a.	n.a.	n.a.	n.a.	40.4
Non-intensive agriculture - Staple food (irrigated rice, maize)	1.20	110.00	210	244.00	4	0.5
Intensive agriculture - Horticulture	9.50	2,450.00	350	900.00	11	0.5
Monoculture Napier grass	5.30	600.00	210	322.00	10	
Coffee systems[d]	5.00	225.00	270	***470.00***	4	57.4
Cacao systems[d]	7.50	1,390.00	315	1,530.00	2	24.9
Dairy cattle[e]	3.90	n.a.	90	933.00		n.a.

a Wage rate at which the NPV is zero, at an exchange rate of 1US$ = Rp. 9000 and 2008 prices, normal wage rate in 2008 was 3.9 US$.day-1

b Including for establishment and maintenance

c Hairiah et al. (2010)

d Return to labour and return to land values are for systems at productive stage

e For dairy cattle, the unit for return to land is per livestock unit instead of per land

n.a. = not applicable

Table 3.5 Scenarios (business-as-usual and prospective) of landscape dynamics in the Upper Konto catchment developed for FALLOW model application. Scenarios are based on fodder option as well as access to farm land and to harvest fodder on land zoning of the State Designated Land Use map (Figure 3-1), FR = Forest reserve, PF = Production Forest and OLU = Other Land Uses.

	Scenarios	Access to cultivate/to farm			Access to harvest fodder			Monoculture Napier Grass
		FR	PF	OLU	FR	PF	OLU	
I	Business as usual (BAU)	X	X	√	√	√	√	√
II	Agroforestry access in plantation forest	X	√ but restricted[a]	√	√	√	√	√
III	No fodder harvest in plantation forest	X	X	√	√	X	√	√
IV	No monoculture Napier grass	X	X	√	√	√	√	X[b]
V	Open access	√	√	√	√	√	√	√

[a]Farmers only permitted to plant tree-based systems (coffee or cacao systems)
[b]Farmers rely on fodder only from existing land use or from buying outside the landscape

3.4.7 Statistical analysis to evaluate model performance

To evaluate the performance of the FALLOW model in simulating land use change, we followed the methods developed by Pontius et al. (2011) for spatial validation and Costanza (1989) to compare the model goodness of fit with the 'null model'. Both methods used three sets of maps: (i) a reference map of the initial time, (ii) a reference map of the subsequent time and (iii) a prediction map of the subsequent time. In spatial validation, by overlaying the three maps we can distinguish two types of agreement: (i) pixels that are correct due to persistence, and (ii) pixels that are correct due to change. We can also distinguish three types of errors: (i) persistence predicted by model as change, (ii) change predicted as persistence, and (iii) correct prediction of change by the model but predicting transition to the wrong category. This method is particularly useful for model that predicts multi-category land use to distinguish errors due to prediction to wrong category. Pontius et al. (2011) suggested the use of 'figure of merit' (FOM) to measure the overall correspondence between observed and simulated changes (Eq. 17).

$$FOM = 100 * (A_o \cap A_m)/(A_o \cup A_m) \quad \text{.......} \quad (3.17)$$

where ***FOM*** is the figure of merit***;*** A_o is the observed change and A_m is the simulated change. The resulting value ranges from 0 to 1, where 0 indicates no intersection between the observed change and simulated change and 1 indicates a perfect intersection between the observed change and simulated change.

Costanza (1989) developed a goodness-of-fit indicator that compares the accuracy of the land use change model to the accuracy of its null model at multiple resolutions. The null model is defined as model that assumes complete persistence of land use across the simulated time period. The goodness-of-fit at a particular sampling window size is estimated by Eq. 18.

$$Gof_w = (\Sigma_t (1 - (\Sigma_p |a_{1p} - a_{2p}|)/2w^2)/t_w \quad \text{.......} \quad (3.18)$$

where Gof_w = the fit for one side of the (square) sampling window of linear dimension w; a_{ki} = the number of cells of category i in scene k in the sampling window; p = the number of different categories (i.e. land use types) in the sampling windows; s = the sampling window of dimension w; and t_w = the total number of sampling windows in the scene of window size w;

Gof_t (Eq. 19) is a weighted average of the fits over all window sizes that summarises the way the fit changes as the resolution of measurement changes.

$$Gof_t = (\Sigma_w Gof_w {}_* e^{-k(w-1)})/((\Sigma_w e^{-k(w-1)}) \quad \text{.......} \quad (3.19)$$

For the Upper Konto catchment study, we used the land cover map of year 2000 and 2005 (referred to as Reference 2000 and Reference 2005, respectively) and the FALLOW simulated result of year 2005 (referred to as FALLOW 2005).

3.5 Results

3.5.1 Landscape and economic dynamics of the Upper Konto catchment: business as usual

Exploration of the landscape dynamics within the 'Business as usual' (BAU) scenario showed that after 20 years of simulation, the landscape land cover was stabilised with 50% forest area (Figure 3.4A) and 50% agriculture and settlement, due to the restriction on opening up land in the forest area. The BAU scenario outcome suggested that intensive agriculture would be the main agricultural land use covering 60% of agricultural land in year 21, while cacao systems would replace most of the coffee systems. The shifts towards more intensive systems followed as a result of their favourable economic benefits, i.e. high returns to labour as well as to laTable 3.4 Productivity, profitability and time-averaged carbon stocks of main livelihood options/land use in Upper Konto catchment.nd (Table 3.4).

BAU scenario also suggested that dairy cattle was and remained a major welfare factor in the area, contributing 50-70% of the total gross income over the simulation period ((Figure 3.4B). The demand in fodder was partly met by initially increasing the area under MNG. The contribution of agroforestry systems (cacao, coffee) to gross income was predicted to remain relatively stable. Farmer's welfare per capita was prospected to increase initially due to the increase in the cattle population and the expansion of intensive agriculture (Figure 3.4C). However, subsequently welfare was reduced overall by 14% due to soil degradation. In comparison, the average landscape aboveground carbon stock (AVERAGE-C) was prospected to fall slightly, but compensated with an increase after year 7, hence ranging between 37 to 46 $Mg.ha^{-1}$. The decline of AVERAGE-C in the early years of the simulation was due to expansion of annual cropping systems (which showed an opposite trend to farmers' welfare). Meanwhile, the increase of AVERAGE-C towards the end of simulation was due to maturation of the additional agroforestry systems. Overall, the Upper Konto catchment under the BAU scenario sequestered 2 $Mg.ha^{-1}$ $year^{-1}$ of carbon over the 20 year simulation period.

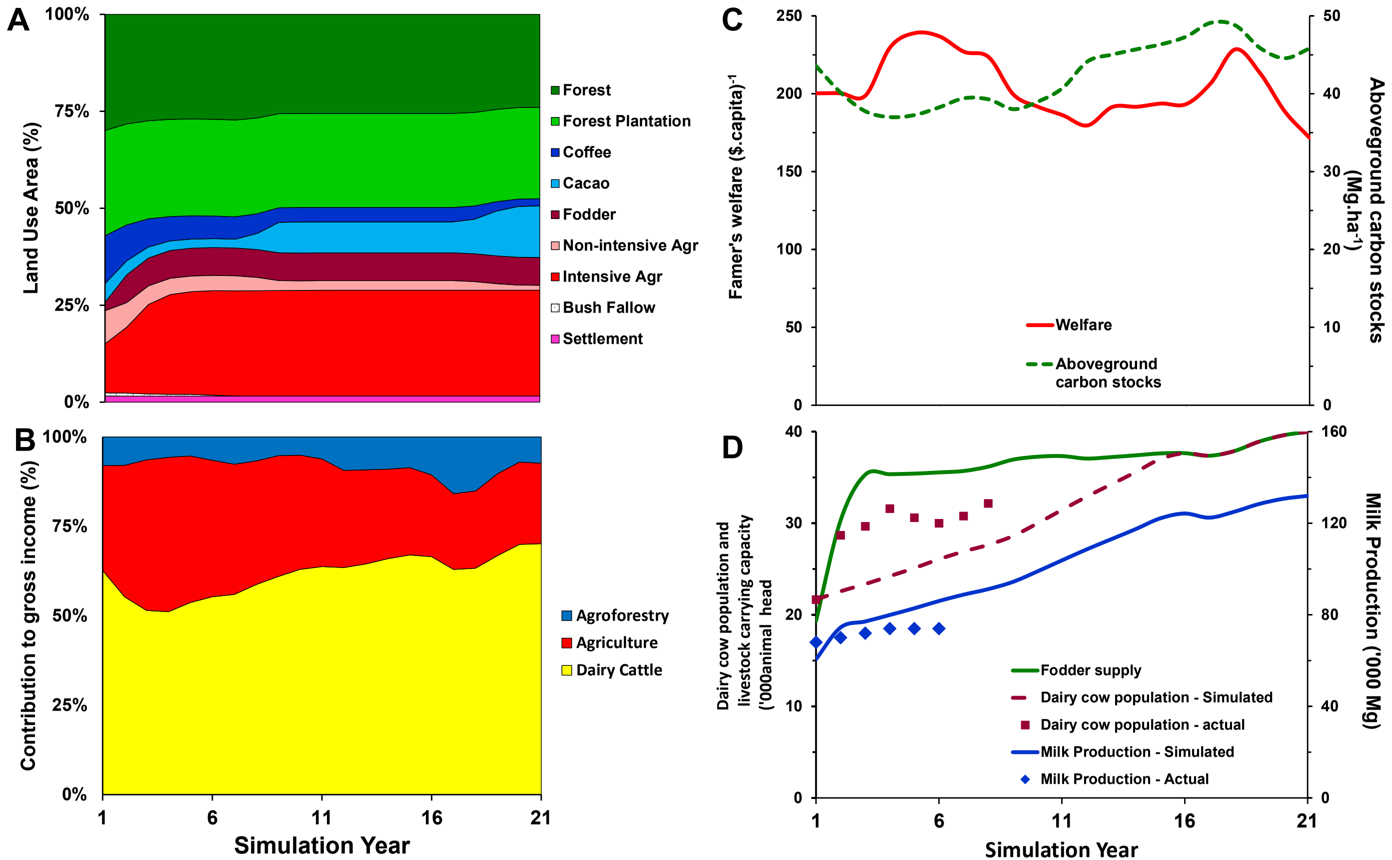


Figure 3.4 Baseline scenario results of FALLOW model run for the Upper Konto catchment: (A) landscape dynamics (% area); (B) contribution of main livelihood options on catchment gross income (%); (C) farmers' welfare ($ per capita per year) and average aboveground carbon stock (Mg.ha^{-1}); and (D) development of dairy cattle systems over time.

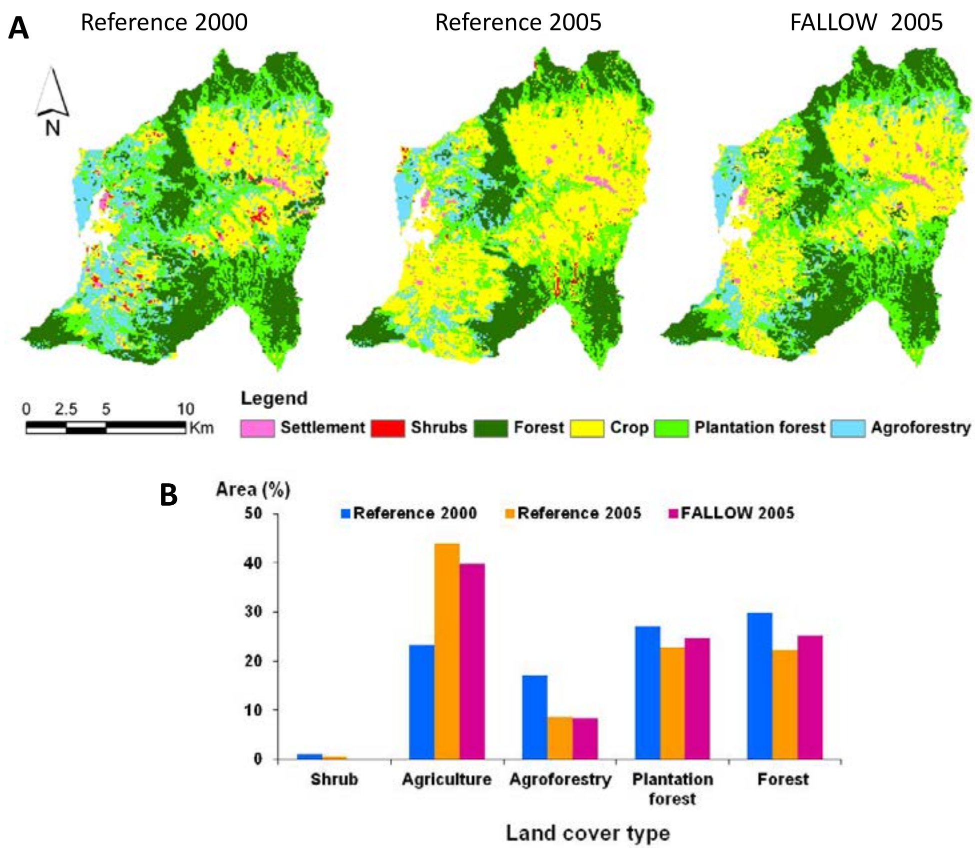


Figure 3.5 Comparison of land cover map of Upper Konto catchment simulated by the FALLOW model ('Business-as-usual' scenario) (FALLOW 2005) over five years (2000-2005) versus reference maps (results of Landsat-TM 2000 and 2005 interpretations): A) as maps and B) in percentage of study area for each land use category.

3.5.2 Performance analysis of FALLOW: land use change

To evaluate FALLOW performance in simulating spatial patterns of land use change in Upper Konto catchment we compared Reference 2000 and Reference 2005 maps with FALLOW 2005 (Figure 3.5A). Figure 3.5B shows the distribution of area of each land use categories for each map.

The result of FALLOW model baseline (BAU) scenario for year 2007 was more accurate then the null model for resolution between 100 – 1600 m (Figure 3.6A). The goodness of fit (*GOFt*) value of 0.82 represents a weighted average of the agreement over the pixel size varying between 1 (100 m) and 16 pixels (1600 m). The goodness of fit indicates the ability of FALLOW to simulate the overall land cover pattern in Upper Onto catchment. Further analysis revealed that the high accuracy of FALLOW application in Upper Konto catchment was largely due to the correctness in predicting persistence of land use (Figure 3.6B). At the original resolution of 100 m, FALLOW agreement was 81% with 72% due to simulating land use persistence. The 'Figure of Merit' (indicating the ability of FALLOW in simulating land use change) mostly stable between 31-39%, except for the highest of 46% that occurred at 1300 m resolution and the lowest of 15% at 1600 resolution. The relatively low value of 'Figure of Merit' is due to the error in predicting persistence when actually changes have occurred. The error is stable at around 15% and does not change with increase in resolution, indicating that error is not due to slight errors in positioning. However, the inaccuracy of FALLOW in predicting land category of land change decreased as the pixel resolution increases.

3.5.3 Prospecting the impact of land use zoning: trade-offs on fodder availability, carbon sequestration and livelihood

The goal of this modelling study was to prospect several plausible land use zoning policies as scenarios and to examine how they would affect fodder availability, carbon sequestration and farmers' welfare. Cattle rearing is an important source of income for the communities of the Upper Konto catchment (Figure 3.4B). Thus, fodder availability can be employed as an indicator of the ability of the Upper Konto landscape to sustains its people's livelihoods (environmental carrying capacity).

The model outputs suggested that the largest amount of fodder (approximately 30% more fodder than under BAU conditions), would occurred under the 'Free-access' scenario where farmers were free to cultivate any land. (Scenario V, Figure 3.7A). The lowest fodder production (roughly 40% of the BAU scenario) would happen in scenario VI where MNG were not adopted by farmers. A less severe fodder shortage occurred in the scenario where fodder harvesting was prohibited in the forest (Scenario III). Forest area is the main source of fodder in all scenarios (Figure 3.7B) except in Scenario III where agroforestry land becomes the main source of fodder. Providing farmers with access to some parts of forest areas (Production Forest) but restricting their farming systems to only agroforestry systems would produce slightly lower amounts of fodder compared to under the baseline situation (Scenario II, Figure 3.7A).

Providing access to open land in plantation forest areas would create new opportunities for farmers to cultivate agroforestry systems. Thus, contrary to the assumed positive linear relationship between fodder availability and welfare, Scenario II predicted an increase of welfare by 6% (Figure 3.8A). In other scenarios, a decreased in fodder availability led to a decrease of farmers' welfare. The highest impact is in scenario without MNG (Scenario IV) where the landscape lost 38% of its fodder with reduction of welfare per capita of 11% (Scenario IV). Providing farmers with access to all land within the landscape was predicted to increase welfare by 12% at the expense of losing 23% of aboveground landscape carbon stocks (Scenario V, Figure 3.8B). In a hypothetical situation where farmers did not cultivate MNG, landscape carbon stock was predicted to increase by 4%, but with a larger reduction in farmers' welfare of over 12% (Scenario IV).

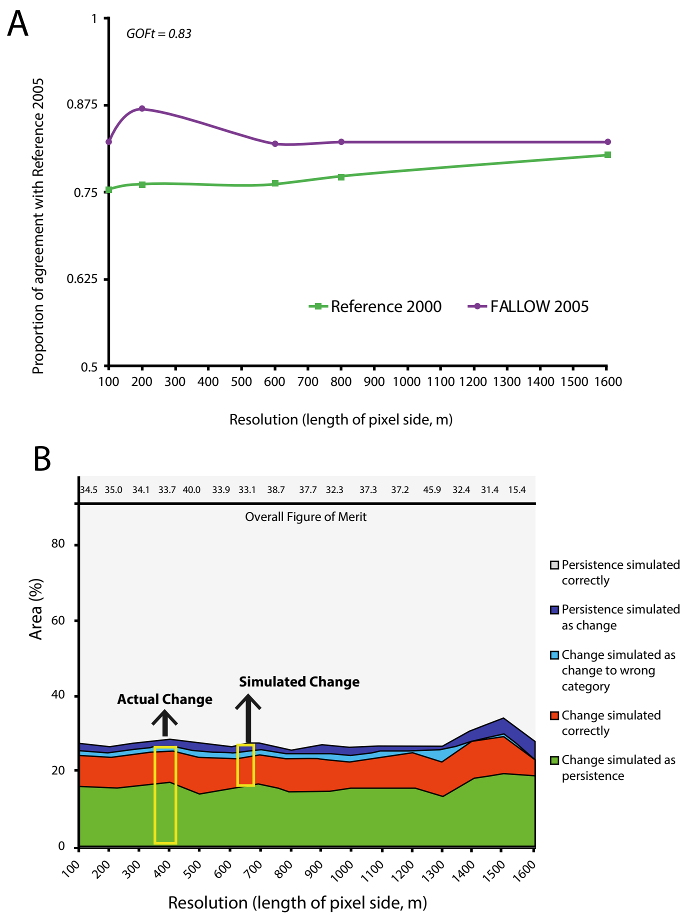


Figure 3.6 Result of spatial validation of FALLOW model outcome: (A) Goodness-of-fit of Upper Konto land cover map of year 2000 (as null model) and simulated FALLOW map of year 2005 to land cover map of year 2007 at multiple resolutions. GOFt refers to weighted average of goodness of fit between window size of 100-1600 m resolution (Costanza, 1989); (B) Components of agreement and disagreement for Upper Konto catchment at multiple resolutions with the associated overall figure of merit. Figure of merit is an indicator of model agreement in simulating land use change (Pontius et al., 2011).

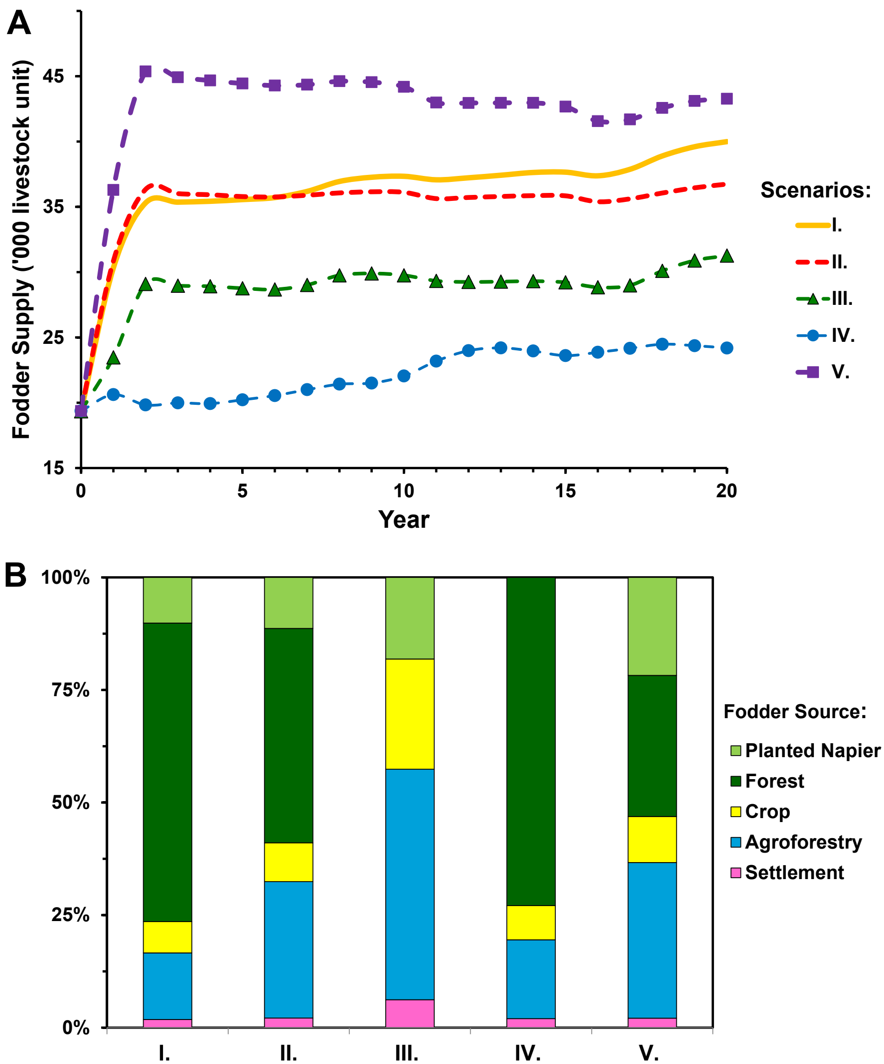


Figure 3.7 A. Dynamics of fodder supply at various landscape scenarios. B. Contribution of various types of land use systems to fodder availability in Upper Konto catchment under different scenarios, (I) Business as usual, (II) Agroforestry access in plantation forest, (III) No fodder harvesting in plantation forest, (IV) No monoculture Napier grass systems, and (V) Open access.

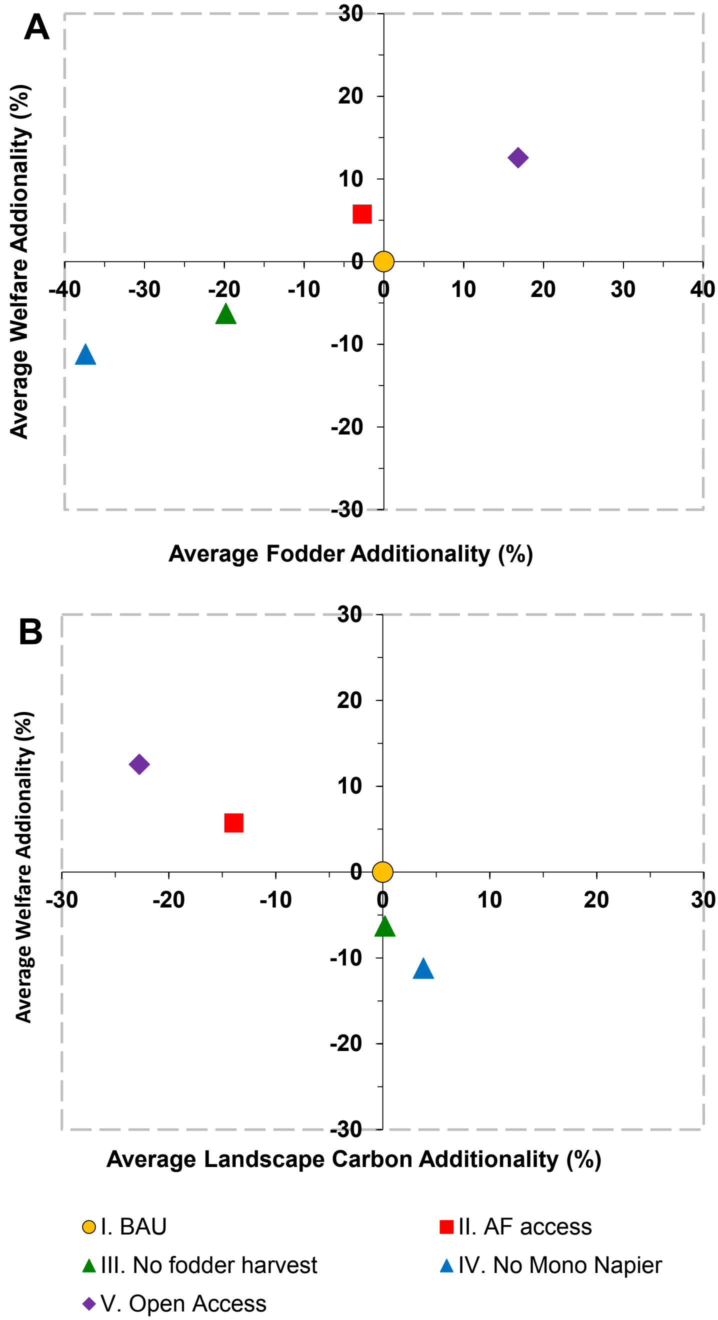


Figure 3.8 Trade-offs of A) average fodder additionality versus landscape above-ground carbon stocks, and B) farmers' welfare versus landscape above-ground carbon stocks relative to the 'business as usual' condition. Results of prospective scenarios (I-V) of the FALLOW model averaged over 20 years.

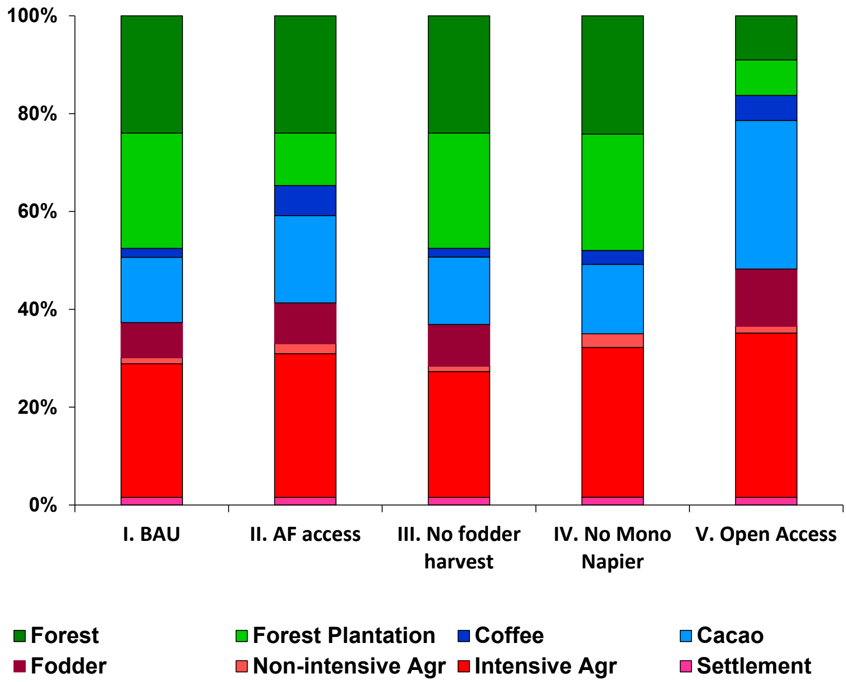


Figure 3.9 Land use distribution in Upper Konto catchment at year 20 under the different landscape scenarios as simulated by FALLOW model, (I) business as usual, (II) agroforestry access in plantation forest, (III) no fodder harvesting in plantation forest, (IV) no monoculture Napier systems, and (V) open access.and (V) open access.

Figure 3.9 depicts land use distribution in the Upper Konto catchment at the end of the simulation run (20 years). Allowing farmers to open up land (Scenario V) was simulated to reduce forest area by 31% and increase area under agroforestry (cacao and coffee) by 20%. Allowing farmers to cultivate agroforestry systems in plantation forest (Scenario II) was suggested to reduce the plantation forest by 16%, mainly converted into cacao systems. Overall, intensive agriculture dominated the agricultural landscape in all scenarios, except scenario V where agroforestry systems contributed equally.

3.6 Discussion

3.6.1 Value of integrating livestock into the land-use change model

We applied the FALLOW model with an added livestock module to different scenarios of access to forest examining the impact of land zoning on fodder availability and its consequences for farmers' livelihood and carbon sequestration. The overall FALLOW model structure was able to capture the differences in the land zoning policy between scenarios and reflected biophysical, labour, and economic outcomes. Thus, it is a potentially valuable tool to understand the inter-relationships between different system components for scenario analysis, and impact assessment. The additional livestock module was able to represent the two-way interactions between livestock and land use change – through fodder demand – as well as the interactions between livestock and livelihood – through milk production. The large contribution of dairy production systems to farmers' gross incomes in the Upper Konto catchment of 50-70% as simulated by the FALLOW model was in line with a previous study by Khususiyah (2009) who found that livestock contributed 45-80% to household income in the area.

The FALLOW model was able to capture the dynamics of agricultural expansion in the area as well as its impact on farmers' livelihood. For example, the increase of welfare at the start of the simulation coincided with the expansion of intensive agricultural systems and the second welfare peak occurred when cacao systems reached their full productive stage. The overall simulated decline of farmer's welfare (per capita, Figure 3.4) in the BAU scenario was largely due to the assumed increase in population (1.1% per year), while the area of cultivated land was relatively stable adding to the decline in the overall profitability of intensive agricultural systems (lower productivity due to the declining soil fertility). Fodder supply in the scenario I (BAU) increased rapidly during the early stage of the simulation due to the expansion of MNG (Figure 3.7A). It was farmers' strategy to cope with the short supply of fodder at the start of simulation. However, farmers did not fully translate the resulting large supply of fodder into an immediate increase in cattle population as the simulation imposed a restriction on additional numbers of livestock that can be managed by farmers in a given year. The currently used restriction was probably somewhat too strong as shown by the actual livestock population which initially increased more rapidly compared to simulated (Figure 3.4D). The above results indicated that the FALLOW model was flexible to modifications while at the same time being robust enough to be able to capture the essential processes of landscape dynamics with a minimum amount of modification and required data. The later is particularly important in the data-poor environment of tropical ecosystems.

The current livestock module was developed for a specific livestock system (cut-and-carry smallholder dairy cattle). However, the module has potential for wider application such as for beef production in pastoral systems. Application to other geographical regions, particularly in rural or peri-urban settings, is also possible. There are more specific simulation models develop for livestock production systems, such as APSIM-SRNS (Parsons et al., 2011) and NUANCES-FARMSIM (van Wijk et al., 2009) that operate at farm scale to assess the biophysical and economic consequences of farming practices. A broad-scale analysis at the continent-level has also been conducted to assess the impact of climate change on the productivity of cropping and consequently changes in farmers preference for livestock-production systems (Jones and Thornton, 2009). The FALLOW model is intended to operate in between the above two types of approach to assess livestock-production systems. The FALLOW model aims to conduct integrated assessment at the landscape/meso-scale level generating aggregated economic and biophysical results at the catchment level.

Within the global climate change debate it would be valuable to introduce further potential indicators of relevant greenhouse gas such as methane and N_2O into FALLOW to be able to assess the full trade-off balances of an integrated livestock production system. For example, the RUMINANT model predicts feed intake and nutrient supply in ruminants that is used as a basis to estimate methane emission from livestock systems (Herrero et al., 2008), however it requires an extensive dataset. For data poor environments, the calculation of methane and N_2O emissions from agricultural activities, including livestock systems and land use change, could be incorporated into FALLOW using a simple lookup table based on IPCC (Intergovernmental Panel on Climate Change) recommendation (IPCC, 2006). The values of the lookup table can be derived from models such as RUMINANT as well as from existing global databases (Werner, 2007).

3.6.2 Land sharing or sparing?

In this study, scenario II (Agroforestry access) represents the 'agroecological matrix' or land sharing approach where small-scale coffee and cacao systems (including multi-strata systems) are allowed to be cultivated within the plantation forest zone. Scenario III (No fodder harvest) represents a land sparing approach where zones for conservation are clearly distinguished from agricultural areas. Intensification of agricultural systems occurs both in the cropping systems (maize/rice to horticulture) as well as the tree based systems (coffee multistrata to small scale shaded cacao/monoculture cacao). Scenario I (BAU) is similar to scenario III, but with access to conservation area to extract fodder (mostly Napier grass but may also include other graminoid and herbaceous plants) growing as understory in the plantation forest area or as borders along paths. Consequently, access to fodder in the plantation forests allows farmers to non-intensively manage the understory in mature plantation forest areas. The understory allows growth of habitat features within the plantation forest area. Hence, scenario I represents a mix of land sparing and 'wildlife-friendly' farming approach.

Exploration of land zoning scenarios in the Upper Konto catchment revealed that neither a pure land sharing (Scenario II) nor sparing (Scenario III) approach alone was the best scenario' in terms of balancing the trade-offs associated with carbon sequestration, fodder availability and farmers' income. Scenarios that were able to increase farmers' welfare by providing access to forest areas had consequences leading to a reduction of substantial above-ground landscape carbon stocks. A more conservative policy to prohibit total access to forest, even for fodder harvesting, resulted in a decrease of farmers' welfare without having the benefit of increasing carbon stocks. A combination of land sparing with some sharing of the conservation areas for added value (fodder) prove to be the most promising land zoning approach to fulfil these multiple goals. Thus, a careful integration of livestock into forest systems, e.g. through a cut and carry approach can be beneficial for the state and farmers alike. This finding is along the views of Fischer et al. (2008) whereby a mix of land sparing and land sharing/'wild-life friendly' approach is recommended for frontier landscapes undergoing rapid conversion to agriculture. This finding also concur with the hypothesis that even under forest conservation, utilizing NTFP (Non Timber Forest Products, in this case fodder) could actually increase income of local communities providing buffer to further forest degradation or forest encroachment (Delacote, 2007). There are counterfactual cases where NTFP extraction led to forest degradation, in particular when market forces favoured commercialization of NTFP products (Arnold and Perez, 2001; Kusters et al., 2006). However, in the Upper Konto catchment, fodder extraction is carried out in mature plantation forest areas for household use and not for commercialization. Farmers also occasionally applied low amount of manure to fodder plants. Hence, farmers were able to minimized negative impact of fodder extraction on plantation forest condition and may even add benefit to timber growth. Currently, farmers carry out commercial production of fodder through intensified monoculture-napier-grass in the agricultural zone in competition with cropping systems.

The peri-urban situation of Upper Konto (easy access to market, fertilizer input and seedlings) makes intensive forms of agricultural systems (horticulture, cacao) that produce higher returns attractive to farmers. As shown in scenario II and scenario V (Figure 3.7) the substantial gain in farmers' income by opening currently restricted conservation areas led to substantial loss in landscape carbon stocks. Thus, if the objective is to maintain the current landscape carbon stocks land zoning that spared land for conservation areas is still a necessity in Upper Konto catchment.

3.6.3 Balancing production, livelihood and environmental benefits in Upper Konto catchment

The simulated increase in dairy production in Upper Konto catchment is highly probable as there is increasing meat and milk demand worldwide with the predicted changes in food habits along with the increasing welfare of the population (World Bank, 2009), particularly in developing countries (FAO, 2009). In Indonesia the demand for dairy milk is still higher than the supply, resulting in as much as 89% of total demand being imported (Amaliah and Fahmi, 2007). The fact that the livestock population may increase in the future hints at the importance of future policies to address the negative impact that may arise due to associated land and fodder demand and to find solutions that produce positive trade-offs.

In South America, livestock was blamed for the loss of natural forest due to conversion to pasture land (Nepstad et al., 2006). This appears not to have been the case in the Upper Konto catchment largely due to the fact that any pristine natural forests had already disappeared since the late 19th century (Nibbering and Graaff, 1998). In addition, fodder supply largely originated from non-forest areas and forest understry. Therefore, predicted livestock activities did not reduce landscape carbon stocks in the area. Recent observations in the field showed that monoculture Napier grass has replaced other agricultural systems, confirming the prospected global trend that intensification of livestock production due to the increasing global demand will drive competition for land between feed and food (Thornton, 2010).

Livestock, particularly in developing countries, is considered beneficial to the agro-ecosystem as an agent of nutrient recycling, provides additional labour and contributes to the household economy (Tarawali et al., 2011). In certain countries, livestock manure is a source of energy and bio-gas. The situation in the Upper Konto catchment partly reflects the above statement. The dairy cattle system provides a large contribution to farmers' household economy, providing stable and reliable income in comparison to vegetables (intensive agricultural system) that have higher risks related to weather, pests and diseases. Usage of manure as fertiliser is currently low in the area, with some of the liquid manure disposed to local drainage systems and to farmers' backyard polluting the environment and degrading reservoirs. Based on informal interviews with farmers, their lack of interest in using manure is largely due to the high transportation costs to bring the manure to the crop fields and the lack of space and labour to manage and store the manure properly.

The simulated decrease of farmers' income over time raises concerns that the current production options are unsustainable for farmers' livelihoods. Thus, the following alternatives may be useful to implement to balance livelihood and environmental services. For example, stronger dissemination of alternative feeds such as molasses in cooperation with the milk cooperatives may be able to prevent loss of food production in the area. Additionally, developing environmental schemes (in cooperation with the Public Water Services that managed local reservoir) that provide incentives for farmers to conduct good manure management could help halting environmental degradation.

3.6.4 Uncertainty in modelling natural resource management

To model the complex interactions between farmers, the socio-economic and biophysical components in a landscape must be simplified in the processes that are involved. For example, the soil fertility module in FALLOW uses semi-quantitative indicators for soil fertility and for fertiliser effect on yield to allow for easily accessible model inputs (Lippe et al., 2011). However, such an approach causes parametrical uncertainties and undermine the predictive ability of a simulation model (Oreskes, 1998). There are other types of uncertainties typically found in a complex model such as FALLOW that can further undermine the predictive ability of the model: 1) empirical uncertainties that arise from aspects of the system that are difficult (or impossible) to measure and 2) temporal uncertainties can be caused from the assumption that systems (input parameters) are stable over time (Oreskes, 1998).

Nonetheless, having good numerical predictability is not the main purpose of the FALLOW model or most conceptual models of natural systems. It is important that the model is not conceptually flawed and represents the main important processes. It is essential that users (and their stakeholders) know the limitations of the model that they utilized for scenario analysis purposes, to prospect plausible future trajectories.

The multiple-resolution spatial validation and goodness-of fit procedures such as have been carried out in this study are an approach that estimates the spatial uncertainty in the model prediction. The result from the application of FALLOW in the Upper Konto catchment showed FALLOW inclined to overestimate land use persistence, but still better than the 'null model' in the overall performance. Results also indicate that for the current study scale did not influence the overall spatial accuracy. However, this is not a general finding as another application showed that the accuracy changed with scale (Lippe et al., 2011). The relevance of spatially explicit model representations becomes an issue if the difference in accuracy changes drastically with scale. However, there is still no clear guidance on how scale influences the accuracy of the predictions. There are various internal feedback mechanisms (positive and negative) that influence the scaling of net results.

The validation procedures should also be seen as an approach to identify strengths and weaknesses of the model to provide guidance to the type of acceptable application for the model. For example, a model with low spatial accuracy but high accuracy in predicting magnitude of change may not be suitable for application in estimating trade-offs in watershed functions (erosion and sedimentation), but may still be suitable for estimating landscape carbon stocks. If higher accuracy of predictions are needed then more detailed simulation models will be needed, both in terms of biophysical as well as socio-economic capabilities, such as multi-agent models (Schreinemachers et al., 2007) preferably coupled to land-use change models (Marohn et al., 2012) but this comes at the expense of higher data requirements.

The existence of uncertainties in a complex simulation model such as FALLOW, require careful interpretation of simulation results. Over-interpretation must be avoided, particularly when the purpose of modelling is to develop policy recommendation. Model results should not be seen as a prescription on what to do, but rather should be used for understanding the range of possible outcomes. In our case, scenario analysis was used to test the consequences of land use zoning on fodder availability, welfare and above-ground landscape carbon stocks. The values of the outcomes should not be taken at face value rather should be interpreted in terms of the processes involved and the comparative trajectories that appear.

3.7 Conclusions

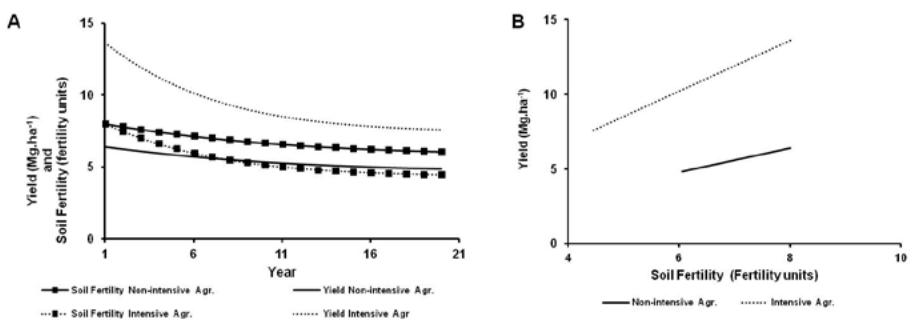
The extended FALLOW model with its livestock module is an effective tool to examine the interactions between livestock, cropping systems, household decision and natural resources in data poor environments. Through the application of the FALLOW model we assessed the impact of land zoning policy on farmers' welfare and landscape carbon sequestration in the Upper Konto catchment, Indonesia and demonstrated that a mix of a land sparing approach with restricted access for 'wild-friendly' farming in conservation areas maybe the best option. The current land zoning policy of establishing protected areas and allowing farmers access to fodder extraction in part of the protected areas was able to balance the trade-offs between fodder availability, farmers' livelihood and carbon sequestration as well as enhancing food (dairy) supply in the region and could serve as a model for other Southeast Asian countries. Hence, careful integration of livestock systems into conservation areas might be a useful approach to achieve multiple goals, although with respect to environmental services, an extended analysis of all relevant greenhouse gases and manure management options will be necessary. Improved FALLOW scenario results can serve as a basis for discussion with stakeholder drive further model and scenario development.

3.8 Supplements

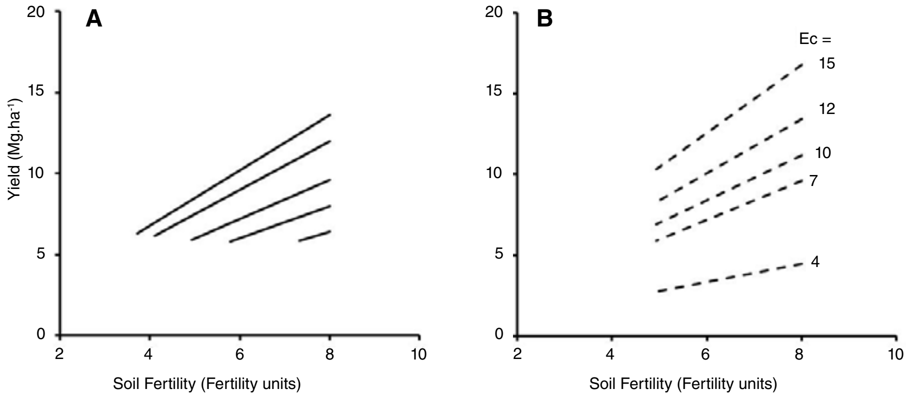
S 3.1 List of livestock parameters used in FALLOW model for the simulation of Upper Konto catchment.

Input parameters	Definition	Value	Unit
Pf	Fodder price	121	$\$.Mg^{-1}$
Rf	Fraction of new livestock	0.05	Dimensionless
Rfs	Fraction of livestock that can be sold	0.01	Dimensionless
Ffb	Fraction financial capital that can be used to buy fodder	0.001	Dimensionless
Fth	Fodder ratio threshold, below which farmers will decide to buy external fodder as the yield production becomes too low	0.6	Dimensionless
Lab_l	Labour required to establish and maintain additional livestock	90	Person days.year^{-1} livestock units^{-1}
$Cost_l$	Cost required to establish additional livestock	489	US\$. year^{-1} livestock units^{-1}

S 3.2 Yield and soil fertility dynamics in FALLOW model



S 3.8.2.1 Dynamics of plot level soil fertility and yield of two main crops in Upper Konto catchment generated by FALLOW soil input parameters and used as the basis for the 20-year simulation: A) yield and soil fertility as function of time; and B) yield as function of soil fertility. The parameter values used to generate this relationship were based on unstructured interviews with key farmers regarding the dynamic of crop yields over the last 5 years combined with experts' knowledge of the yield-soil fertility relationship of similar crops in similar geographical settings.



S 3.8.2.2 Sensitivity analysis for plot level soil fertility and yield dynamics; (A) at various f_{dr} (depletion rate) values and E_c = 10, (B) at various E_c (soil fertility-yield conversion factor) and f_{dr} = 0.07; and other soil parameters constant (f_{fert} = 0.02, $Fert_{max}$ =10, see Section 2.3.c for definition of parameters).

Chapter 4

Uncertainty of net landscape carbon loss: error propagation from land cover classification and plot-level carbon stocks[10]

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4.1 Abstract

This study developed a methodological framework to estimate the uncertainty of landscape carbon stocks and carbon loss estimates for various cases of data availability, including a minimum data situation often encountered in developing countries. The study was carried out within the framework of implementing a reward for ecosystem service (RES) scheme for natural resource management. Error propagation in land cover classification and/or variation in plot-level carbon were tested in a case study in Tanjabar, Indonesia representing a forest frontier region where extensive land use change is occurring from forest to more profitable intensified farming systems. Monte Carlo simulations based on propagation of errors in land cover classification and variation in plot-level carbon estimated a net carbon loss of 31.3 $Mg.ha^{-1}$ between year 2000 until 2009 with a coefficient of variation of 0.2%. Based on an estimated cumulative density function of carbon loss, the potential eligible area for an incentive carbon emission reduction program was 35%, using land cover maps with 100 m resolution. The assessment showed that excluding errors in land cover classification could lead to a biased estimate of an average landscape carbon emission, albeit small (maximum value of 2.7 $Mg.ha^{-1}$ or or 7.5% for net carbon loss) due to the tendency of errors in land use clasification to occur within land use of similar carbon values. An average landscape carbon is an aggregated indicator for carbon loss and thusrobust to error propagation. Further studies to explore how spatial aggregation may influence other indicator performances used for developing carbon incentive mechanisms such as probability distribution of carbon loss may be needed. The development of methods to assess uncertainty for various data availability situations can help in supporting initiatives to include local stakeholders and local planners in designing plans for resource management to reduce carbon loss from ecosystems.

4.2 Keywords

Uncertainty analysis, carbon loss, error propagation, land cover classification errors, Monte Carlo analysis.

10 A version of this chapter was submitted to the the Canadian Journal of Forest Research

4.3 Introduction

Increasing interest in the use of economic incentive schemes to mitigate carbon emission, such as CDM (Clean Development Mechanism) and REDD (Reducing Emission from Deforestation and Degradation), has highlighted the importance of uncertainty of landscape carbon stocks and carbon loss estimates (Sloan and Pelletier, 2012). A future REDD scheme may be required to comply with UNFCCC (United Nations Framework Convention on Climate Change) principles[11] for estimating and reporting emissions and removal of greenhouse gases (Grassi et al., 2008). Among the principles is 'accuracy': *"...Estimates should be accurate in the sense that they are systematically neither over or under true emissions or removals, as far as can be judged, and that uncertainties are reduced as far as practicable ..."* (UNFCCC, 2006, p. 5). Uncertainty studies are further required to ensure that carbon changes can be monitored and verified effectively, which includes ensuring that the level of uncertainty in landscape carbon loss/gain estimates is acceptable and in accordance with the agreed baseline (Persson and Azar, 2007, Pelletier et al., 2011).

The economic incentive schemes such as CDM and REDD are rooted in concepts of payment and/or rewards for ecosystem services (ES) where it is expected that ES, e.g. provisioning of fresh water, flood control, maintenance of biodiversity and climate regulation, can be conserved more efficiently compared to the costly command and control approach (Wunder, 2008,Van Noordwijk and Leimona, 2010), such as establishing protected areas or National Parks. The basic principle is that individuals and communities would be financially motivated to engage in mutually beneficial agreements regarding resource management (Daily and Matson, 2008). Efforts have been made to make reduction of uncertainty attractive in such schemes, by using the lower limit of the confidence interval of emission reduction as basis for payments (Costa et al., 2000). Under such rules, investment in higher data quality can lead to additional carbon credits for ES providers.

The uncertainty analyses carried out in this study were conducted within the framework of implementing a reward for ecosystem service (RES) scheme for natural resource management, in particular implementation of a carbon incentive scheme for reducing carbon emission (Figure 4.1). There are three main questions related to uncertainty that arises from the implementation of carbon incentive schemes:

1. How do we measure, across multiple scales, uncertainty of estimates of landscape carbon changes as the basis for RES?
2. What is the implication of uncertainty in landscape carbon loss estimates for designing an effective incentive scheme?
3. How will the recipients of RES (who are also the providers of ES) likely respond to errors of targeting that may arise from the uncertainty of landscape carbon loss?

These three questions relate to the three dimensions of science quality and model validity as discussed in Lusiana et al. (2011): i.e. question (i) addresses the credibility

11 The UNFCCC principles for estimating and reporting emissions and removals of greenhouse gases (GHGs) are: transparency, consistency, comparability, completeness, accuracy (UNFCCC, 2006)

of the resultant estimates and is a prime concern for scientists who develop methods to estimate carbon loss, while question (ii) is focusing on the salience or use of the estimates by the government/policy implementers and RES community (e.g, REDD+ community). Depending on the scale of RES, the last question, which is linked to the legitimacy of the knowledge used, is associated with land owners (farmers), community of farmers, district/provincial government or national government who are recipients of funds.

This paper addresses question (i) and part of question (ii) and was carried out in Tanjung Jabung Barat (Tanjabar) District, Jambi Province-Indonesia. A companion to this paper, Lusiana et al. (2013), discussed questions (ii and iii) on the implication of uncertainty for the design of potential REDD+ scheme in the district. Tanjabar is a typical frontier region where extensive land use change is occurring from forest (pristine or disturbed) to more profitable intensified farming systems. The specific objectives of this paper are: (1) to develop a step by step process in assessing uncertainty of landscape carbon stocks given different types of data: i.e. land cover maps and carbon density measurements, (2) to estimate aboveground landscape carbon stocks of Tanjabar for the year 2000 and 2009 including their uncertainty, (3) to estimate the corresponding landscape carbon loss and associated uncertainty in change during 2000-2009, and (4) to evaluate the implication of carbon uncertainty and distribution of carbon loss estimate for a potential reward or incentive design.

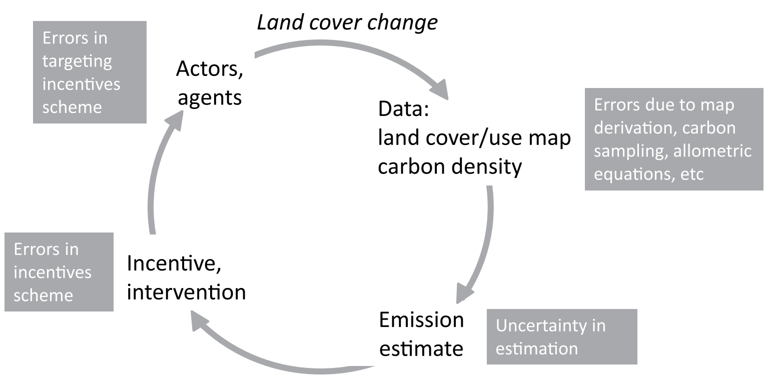

Figure 4.1 Feedback loop in implementation of an economic incentive scheme for reduction of carbon emissions with its associated uncertainties and errors.

4.4 Source of uncertainty in landscape carbon loss: case study data

Accounting approaches to estimate carbon emissions can be based on carbon stock change (Koh et al., 2012) and/or on quantifying, usually at annual time scale, all relevant carbon in- and outflow (Monni et al., 2007) at plot, landscape scale or higher levels of aggregation. This study used the carbon-stock change approach that entails using two types of data (Hairiah et al., 2011): (i) land cover maps to provide information on the area of existing land use types at different years, and (ii) carbon densities for each land use systems derived from plot measurement in the field. Both data sets have uncertainty or errors and thus, consequently the estimated landscape carbon stock also has a compounded uncertainty.

Land cover datasets derived from remotely sensed spectral data are not 100% accurate, even if developed from the most advanced satellite images (Friedl et al., 2002; Avitabile et al., 2012). The source of errors involves multiple steps in the data processing, including choice of ground-truth sampling points, landscape characteristics at ground-truth points and elsewhere, time of year, pre-processing steps, and classification procedures (Wang et al., 2005). Presently, the confusion matrix or error matrix is the most common approach to derive measures for accuracy of a land cover map. An error matrix is a cross-tabulation of the mapped land use class against that observed on the ground or reference data for a sample of cases at specified locations (Foody, 2002). A number of accuracy indicators can be derived from the error matrix. However, the most relevant for error propagation analysis is users' accuracy which measures the reliability of a land cover map generated from a classification scheme. It is defined as the percentage of a land use class on the map that matches the corresponding class on the ground and can be used to estimate error of commission (1 – users' accuracy).

Carbon density data is commonly obtained from measurement at plot level which involves the following steps (i) choosing representative sample plots that represent land use systems, preferably by randomization within the predetermined set of class members, (ii) measuring tree diameter (and/or other parameters of trees) and taking other biomass/necromass samples from the plot, (iii) converting tree parameters into biomass (commonly using allometric[12] equations), (iv) aggregating biomass at each plot and converting into carbon values and (v) deriving a time-averaged value using regression of carbon stock on age and information on the typical cycle length (Hairiah et al., 2011). Each step entails a potential source of bias and uncertainty that will eventually accumulate in the final estimates. Choice of sampling locations (Bradford et al., 2010) and persons measuring in the field can potentially affect errors of carbon-stock density (adding bias or systematic errors). Another potential source of uncertainty is the choice of allometric equations (Ketterings et al., 2001, Chave et al., 2004), particularly as not always site specific equations are available (van Breugel et al., 2011). The plot-level data need to be combined to a typical carbon stock density per land use type (see below).

The 'uncertainty' in landscape carbon stocks and net carbon loss analysis in this study refers to variation in estimates due to errors in the data inputs used. The basic carbon-stock change equation to estimate landscape net carbon emission/sequestration is:

$$\Delta\ Ct\rightarrow t+1 = \Sigma i(Ai,t+1\times Ci,t+1) - \Sigma i(Ai,t\ \ Ci,t) = \Sigma i((Ai,t+1 - Ai,t)\times Ci,t+1) + \Sigma i(Ai,t\times(Ci,t+1 - Ci,t)) \quad (4.1)$$

12 Allometic equation describes the relationship between a scalar, for example stem diameter (D) and other properties such as tree volume (T_v) or biomass (Y, in dry weight). A standard allometric equation follows a power-law form of: $Y=aD^b$ or $T_v = a\rho D^b$, where ρ is specific wood gravity.

where $\Delta C_{t \rightarrow t+1}$ is net change in landscape carbon from time ***t*** to time ***t***+1, $A_{i, t+1}$ is area of land use ***i*** at time ***t***+1, $A_{i, t}$ is area of land use ***i*** at time ***t***, $C_{i,t+1}$ is carbon density of land use ***i*** at time ***t***+1 and $C_{i,t}$ is carbon density of land use ***i*** at time ***t***. The change consists of an area change at subsequent (***t***+1) carbon stocks per class ***i***, plus a term for carbon stock change per class multiplied with the original (t) area. In some approaches (Tier 1 and Tier 2 accounting of IPCC) the latter term is ignored.

The following sections describe the data used for assessing the uncertainty of carbon loss in Tanjabar landscape using the carbon-stock change approach.

4.4.1 Ground truth points and land cover maps of Tanjabar

This case study is based on Tanjabar District, Jambi Province, Indonesia. Tanjabar (Tanjung Jabung Barat) district is situated in the north-eastern part of Jambi Province, Indonesia with a total area of 5010 km^2 (Figure 4.2).

To assess changes in landscape carbon stocks, existing land cover maps of Tanjabar district derived from Landsat-TM imageries of year 2000 and 2009 (Widayati et al., 2011) were used to determine land use change (Figure 4.3). Additionally, we used a classification error matrmatrix derived from the 2009 land cover map of Jambi Province (Supplement S.4.1) and 965 ground truth points obtained from field surveys in 2008 and 2009 (Widayati et al., 2011). We used all the ground truth-points of Jambi to obtain sufficient information on classification errors that can occur for the different types of land use in Tanjabar (Table 4.1). All maps were at 100 m resolution. For the purpose of this study we assumed that land cover maps with the current legend also represent land use maps. Thus, throughout this paper 'land cover' and 'land use' systems were used interchangeably. The land cover legend was chosen to align with such interpretation (Hairiah et al., 2011).

4.4.2 Plot level carbon stocks for main land use systems in Tanjabar

Carbon data of the main land use systems in Tanjabar were obtained from the carbon database compiled by and stored at the World Agroforestry Centre (ICRAF), which is based on the various research studies in Indonesia that have been carried out by ICRAF and its partners (Palm et al., 2005; Lusiana et al., 2005; Ekadinata et al., 2010; Hairiah et al., 2010; Khasanah et al., 2011; Widayati et al., 2011). Plot-level carbon data were based on tree diameter data which were measured using the protocol described in Hairiah et al. (2011). The tree diameter data were converted to above-ground biomass using allometric equations for moist forest developed by Chave et al (2005):

$$Y = \rho \times exp(-1.499 + 2.148 ln(D) + 0.207(ln(D))^2 - 0.0281(ln(D))^3) \quad \text{..........................} (4.2)$$

where ***Y*** is above-ground single tree biomass (kg), ρ is wood specific gravity (g.cm^{-3}) and ***D*** is tree diameter (cm) at breast height. This biomass can be converted to a carbon stock estimate (kg C.tree^{-1})by multiplication of a typical carbon concentration, e.g 0.45. Hairiah et al. (2011) provides detail procedure to scale up from tree to plot level carbon stocks.

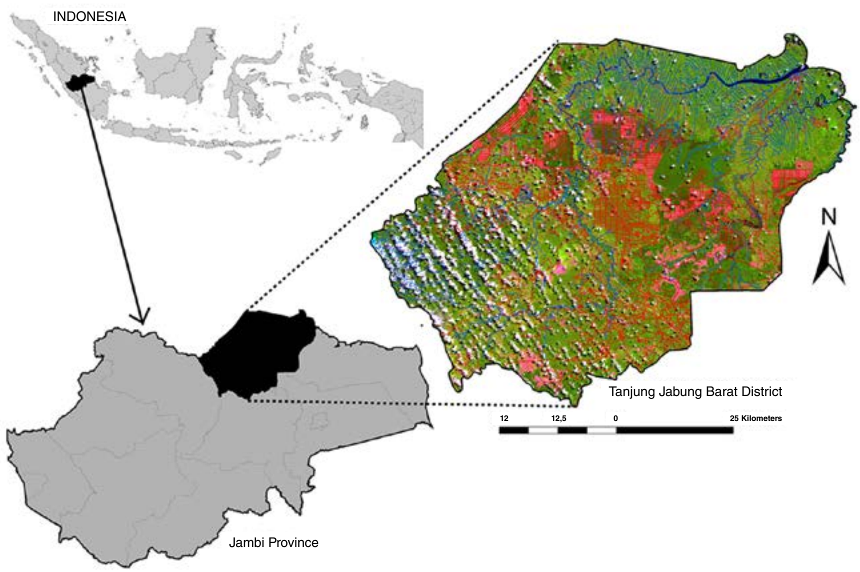

Figure 4.2 Location of Tanjung Jabung Barat District, Jambi, Indonesia. The coloured map showed the 'false colour' Landsat image with cloud cover.

Table 4.1 Land use systems in Tanjung Jabung Barat, distribution of data used in this study. Original map based on (Widayati et al., 2011).

No.	Land use category	Ground-truth points	Carbon plots
1	Undisturbed forest	20	73
2	Disturbed Forest	109	231
3	Rubber agroforest	138	86
4	Coffee agroforest	17	7
5	Coconut/Areca nut agroforest[#]	43	54
6	Acacia plantation	8	21
7	Rubber monoculture	207	27
8	Oil Palm	227	155
9	Shrub	55	52
10	Agriculture	34	23
11	Settlement, cleared land, grass	107	19

+ Estimated from accuracy assessment of ground truth points and land cover map (2009) of Jambi Province. Detailed calculation is available in Supplement 1.

[#]Areca nut (Areca catechu) is the seeed of areca palm. It is commonly referred to as betel nut as it is often, chewed with betel leaves.

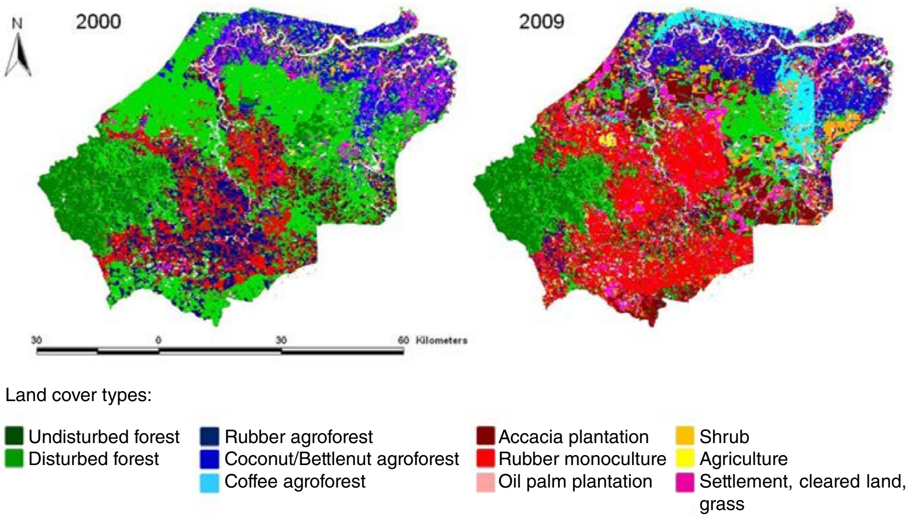

Figure 4.3 Land cover maps of Tanjabar in year 2000 and 2009 in 100 m resolution reclassified from Widayati, et al. (2011).

At the time of this study, the carbon database holds 1230 plot data from Indonesia, the Philippines and Thailand covering more than 20 land cover types. We selected only plot data that represented the 11 land cover types found in Tanjabar and resulted from measurement in Jambi province as well as other regions in Indonesia with similar climate and elevation to Jambi. This approach was chosen in order to have a better measure of variation of plot-level carbon data within each land use systems. A total of 648 plot data were used in the study to represent plot-carbon density of the main land use systems found in Tanjabar (Table 4.1). Data of each land cover systems were averaged using a simple mean equation and became the average above-ground carbon stocks that represented the associated land cover type:

$$C_{avgi} = \Sigma k\ C_{ik} / n_i \quad \text{.......................................} (4.3)$$

where C_{avgi} = average plot-level above-ground carbon stock for land cover type $\boldsymbol{i}$ (Mg ha^{-1}), C_{ik} = plot-level carbon stock (Mg ha^{-1}) of land use $\boldsymbol{i}$ and plot $\boldsymbol{k}$, where $\boldsymbol{k}$ = 1, ..., $\boldsymbol{n}$, n_i = total number of data of plot-level carbon stocks for land cover $\boldsymbol{i}$.

Variation of plot-level carbon stocks for each land cover type was estimated by the standard error of the mean:

$$SEM_i = ((\ \Sigma k\ (C_{ik} - C_i)2\ / n_i)^{½}\ ) / n_i \quad \text{.......................................} (4.4)$$

where SEM_i = standard error of plot-level carbon stocks mean for land cover type $\boldsymbol{i}$ (Mg ha^{-1}).

Forests had the highest average carbon stocks (C_{avg}), i.e. 214 Mg·ha^{-1} and 151 Mg·ha^{-1} for undisturbed and disturbed forest, respectively (Figure 4.4). The average carbon stocks of agroforestry systems were not significantly different to that of tree-based monoculture systems, except for rubber agroforest. Coconut and rubber agroforestry systems were other land use systems with relatively high average carbon stocks. The average carbon stocks of shrub systems were higher than one would expect from such systems due to existence of occasional trees with diameter larger than 20 cm in 40% of 'shrub' plots.

4.5 Land cover and land use change in Tanjabar

For the past 20 years, widespread conversion of land occurred in the area mainly converting forest to plantations of oil palm (***Eleais guineensis*** Jacg.), rubber (***Hevea brasiliensis*** Muell.Arg) and acacia (***Acacia crassicarpa*** Cunn. ex Benth. and ***A. mangium*** Willd.) (Widayati et al., 2011). In 2000, disturbed forest was the main land use system in Tanjabar comprising 40% of the area. By 2009, oil palm had become the main land use system in Tanjabar with 22.4% of the total landscape (Figure 4.5).

During the period 2000 to 2009, it was estimated that land use change occurred in approximately two-thirds of the Tanjabar area (Table 4.2) and around 70% of change involved conversion of disturbed forest and rubber agroforest areas. Almost half of the land use change area was converted to more intensified systems such as oil palm and acacia plantations or rubber monoculture. The most prominent land use change included establishment of oil palm (16% of total area) and acacia plantations (9% of total area).

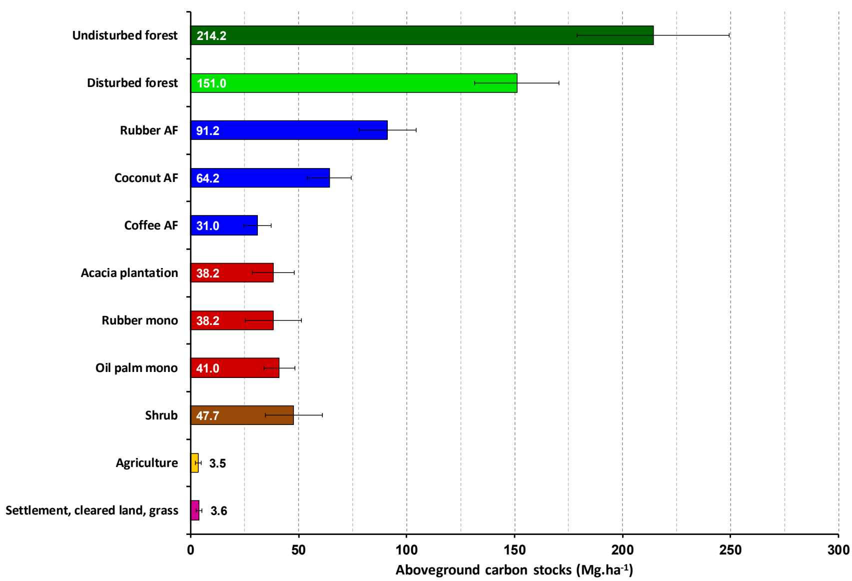

Figure 4.4 Average plot-level carbon stocks assigned for the main land cover systems in Tanjabar with corresponding error bars. The error bars refer to values within 2 × standard error of the mean). Source: ICRAF Carbon Database, http://db.worldagroforestry.org/.

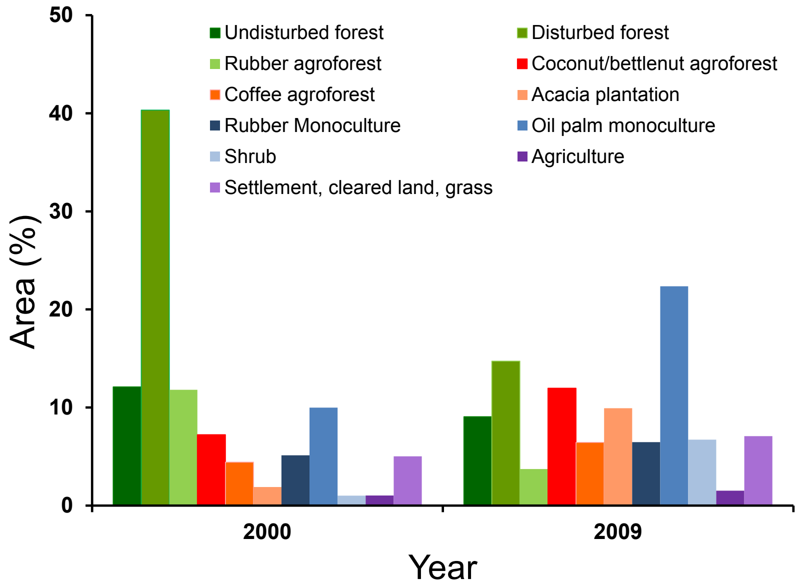

Figure 4.5 Distribution of land use in Tanjabar in 2000 and 2009. Calculated based on maps reclassified from Widayati et al. (2011).

Table 4.2 Distribution of main land use changes in Tanjabar from 2000 to 2009.

Land use in 2000 (in % area)	Land use in 2009 (in % area)				
	Acacia	Rubber monoculture	Oil palm	Others (8 land use systems)	Sub-total
Disturbed forest	6.3	2.4	6.6	15.2	30.4
Rubber agroforestry	1.7	1.7	5.4	3.0	11.8
Others (9 land use systems)	1.3	1.4	3.9	15.0	21.6
Sub-total	9.2	5.2	16.0	33.2	63.8
Landscape persistence	36.2				

4.6 Methods to estimate uncertainty in landscape carbon stocks and net carbon loss

The Intergovernmental Panel on Climate Change or IPCC (2006) proposed two methods for uncertainty analysis in carbon stock assessment: (1) error propagation equations, and (2) Monte Carlo simulations. The error propagation equations approach is based on the assumed additionality of the corresponding statistical variance of the factors involved in estimating carbon emissions[13] (Sluijs et al., 2004; Persson and Azar, 2007). The Monte Carlo techniques are a statistical based method using a probabilistic sampling procedure to select values of input data, then calculating deterministic results (realisations) for a large number of selected input data or parameters (Helton and Davis, 2003, Monni et al., 2007). This study used the Monte-Carlo approach to assess the error propagation in landscape carbon loss assessments.

In developing countries, data availability and accessibility are limited. This increases the likelihood of uncertainty and resulting errors, and also affects the choice of methods for subsequent approaches. Thus, we developed generic procedures for uncertainty analysis of aboveground carbon stocks that depend on the type of data available (Table 4.3). Such an analysis can provide an estimate of the overall uncertainty of carbon stocks and carbon loss at landscape level relevant for CDM and REDD schemes. Case I represents the situation where a minimum data set is available; i.e. existence of land use classification error matrix from a single land cover map, but data on the variation of plot-level carbon data is not available. In Case II, both error classification matrix and variation of plot-level carbon data are available. In Case III and IV, the additional availability of multi-temporal land cover maps allows an estimation of net carbon loss over a given time period.

Table 4.3 Four possible cases of data availability in estimating landscape carbon stocks, its methods and outputs.

Land cover	Plot level carbon-stock data	
	Single carbon-stock	Carbon-stocks from multiple plots
Single map	Case I: Expected carbon deviance, probability of possible carbon deviation	Case II: Monte Carlo simulation, coefficient of variation and bias of landscape carbon stocks
Multi temporal maps	Case III: Monte Carlo simulation, probability of error in land cover classification	Case IV: Monte Carlo simulation, coefficient of variation and bias of landscape carbon loss, probability of error in land cover classification

The following sections describe the approach used in this study to estimate landscape carbon stocks, net landscape carbon loss and their associated uncertainty.

4.6.1 Estimating landscape carbon stocks and net carbon loss

Using a spatial analysis approach, estimating landscape carbon stocks in Tanjabar entails: (i) developing a carbon map based on land cover maps of year 2000 and

13 For example, if emission (E) is calculated as Activity (A) multiplied with Emission factor (F), the error propagation equation can be written as: $\sigma_E^2 = \sigma_A^2 F^2 + \sigma_F^2 A^2$, if the covariance term is zero, where σ_E^2 is the emission variance, σ_A^2 is the variance of the activity data, σ_F^2 is the variance of the emission factor.

2009, and (ii) calculate sum of all pixels in the carbon map, or performing (once) step 3 in Figure 4.6. The net landscape carbon loss (Case III and IV) was estimated using the carbon stock change approach that is depicted in Equation 4.1. To estimate net landscape carbon loss, we subtracted carbon map of year 2000 with carbon map of year 2009 (subtraction pixel to pixel), or performing (once) step 3 and 4 in Figure 4.6.

4.6.2 Estimating uncertainty of landscape carbon-stocks and net carbon loss

4.6.2.1 Case I: Single land cover map without variation in carbon-stock data

Without data on the variation of plot-level carbon stocks, the uncertainty of landscape carbon stocks arises only from errors in land cover classification. Given the availability of data, the information that one can gain is the expected carbon stock difference between landscape carbon stocks estimated from land cover map and what is observed in the field ('expected-carbon- deviance'). Information on the expected-carbon-deviance allows estimation of the confidence level that the landscape carbon stock estimates are correct. Supplement S.4.2 provides procedure used in estimating the 'expected carbon deviance' (uncertainty estimate for Case I cases).

4.6.2.2 Case II, III and IV: The Monte Carlo approach

Uncertainty estimation in Case II, III and IV entail a similar approach that uses a land use classification error matrix and variation in plot-level carbon stocks. The main difference is that in Case II only the uncertainty of landscape carbon stocks can be estimated, while in Case III and IV, the uncertainty of net landscape carbon loss can also be estimated. The approach used is a Monte Carlo simulation analysis, which is a statistical technique that can be used to evaluate how errors propagate (Refsgaard et al., 2007). Monte Carlo (MC) techniques are based on the use of a probabilistic sampling procedure to select values of input data, then calculating deterministic results (realisations) for a large number of selected input data or parameters (Helton and Davis, 2003). How the samples are drawn efficiently, particularly for large number of parameters such as in a complex model, has been the subject of various methods developments resulting in approaches such as response surface methodology, Fourier amplitude sensitivity test, Sobol' variance decomposition, fast probability integration and with Latin hypercube sampling (Helton and Davis, 2003). In our study, the overall objective of the MC technique was to perturb (randomly vary) the land cover map with classification errors and plot-level carbon values with standard errors of mean, and then to produce many realisation of carbon maps. Figure 4.6 describes the basic steps involved in the MC analysis. Land use classification errors were generated following uniform distributions while variations of plot-level carbon stocks were generated using normal distribution. The method for Case II involves performing steps 1, 2 and 3; while for Case III, it entails conducting steps 1, 3 and 4. For Case IV, all steps (1, 2, 3 and 4) in Figure 4.6 were followed.

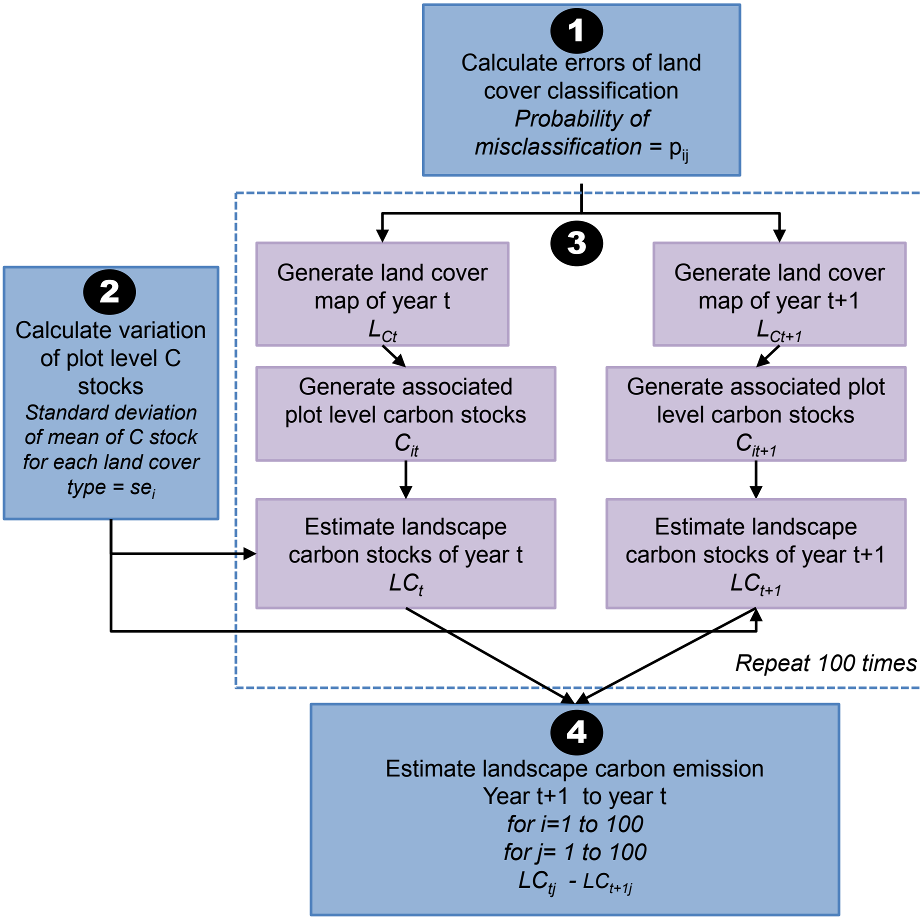

Figure 4.6 Steps in estimating uncertainty landscape carbon stocks and carbon emission using Monte Carlo simulations (Case II, III and IV). Case I, II, II and IV refers to type of data availability (Table 4.3).

The outputs of the uncertainty analysis were 100 maps of landscape carbon stocks perturbed with errors for each evaluation year (year 2000 and 2009) and 10,000 maps of net landscape carbon loss between 2000 and 2009 perturbed with errors. For each perturbed-map, we calculated average landscape carbon stocks (Mg ha^{-1}) and average carbon loss (Mg ha^{-1}). The coefficient of variation between simulated average landscape carbon stocks ($CV_{C\text{-stock}}$) and the coefficient of variation between the average net landscape carbon loss ($CV_{C\text{-loss}}$) derived from perturbed-maps were used as indictors of uncertainty.

4.6.2.3 Estimating distribution of carbon loss across landscape

A cumulative density function (cdf) was used to describe the distribution of carbon loss at pixel/patches level. The cdf describes the proportion of patches that have carbon loss values at lower or equal to the value in the X-axis (in $Mg.ha^{-1}$). The minimum value for a cdf is 0 and the maximum value is 1.

Standard deviation of pixel level carbon loss estimates was calculated based on the generated emission carbon loss maps, using the following equation:

$$STD = (\sum_k (E_k - (\sum_k E_k)/ n_k)^2/n_k)^{½}) \quad (4.5)$$

where E_k = estimated carbon emissions for pixel $\boldsymbol{k}$ (Mg ha^{-1}), where $\boldsymbol{k}$ = 1, ..., n, n = total number of pixels.

The coefficient of variation of carbon estimates (CV) was calculated using the following equation:

$$CV = (std/(\sum_k E_k /n_k)) \quad (4.6)$$

4.7 Results

4.7.1 Landscape carbon stocks and net landscape carbon loss in Tanjabar

Most of above-ground carbon of Tanjabar in 2000 as well as 2009 was stored in disturbed forest (Table 4.4). Between 2000-2009, the total average landscape carbon stock reduced from 111.2 Mg ha^{-1} to 73.8 Mg ha^{-1}, mainly due to conversion of forests (undisturbed and disturbed) and rubber agroforests to a lower carbon-containing-land-use type. Although the percentage of undisturbed forest area in 2009 was low (41.7 km^2) and had decreased compared to 2000 (56.1 km^2), the percentage it represents in total landscape carbon stock had increased by 1.2%. This reflects the importance of undisturbed forests as carbon sink, however small the area was. The expansion of oil palm plantations in the area resulted in a corresponding increase of 2.35 Tg carbon, making oil palm the third largest carbon storage system (12.5%) after disturbed (30.2%) and undisturbed (26.3%) forests. Furthermore, the increased areas of coconut/bettlenut agroforests and acacia plantations resulted in an increase of 1.39 and 1.44 Tg (1 Tg = 1×10^6 Mg) carbon respectively.

Land use conversion from systems with low carbon densities to systems with higher carbon densities will gain/sequester carbon (green colour in Figure 4.7) and, vice versa, conversion to systems with lower carbon densities will lose/emit carbon (yellow to red colour in Figure 4.7). During the period 2000-2009, these land conversions in Tanjabar induced gains of 3.06 Tg carbon as well as losses of 20.35 Tg carbon, resulting in an overall net carbon loss of around 37.5 Mg C ha^{-1} over 9 years or, if we assume all carbon loss was emitted as CO_2, this was equivalent to 15.3 Mg CO_2 $.ha^{-1}.year^{-1}$ (Table 4.5).

4.7.2 Expected carbon stock deviation: uncertainty of landscape carbon stocks without plot-level carbon variation

Based on the analysis of expected carbon deviance patterns for Tanjabar landscape the confidence level for correctly estimating the average landscape carbon stocks (or carbon deviation equals 0) was 70% and 63% for year 2000 and 2009, respectively (Figure 4.8). The expected-carbon-deviance reflects the probability of occurrence for the difference between the estimated average landscape carbon stock with its actual value. For example, in year 2000 the probability that carbon deviance is larger than |150| Mg ha^{-1} (that is lower than -150 and larger than 150) was 0.6% (Figure 4.8A).

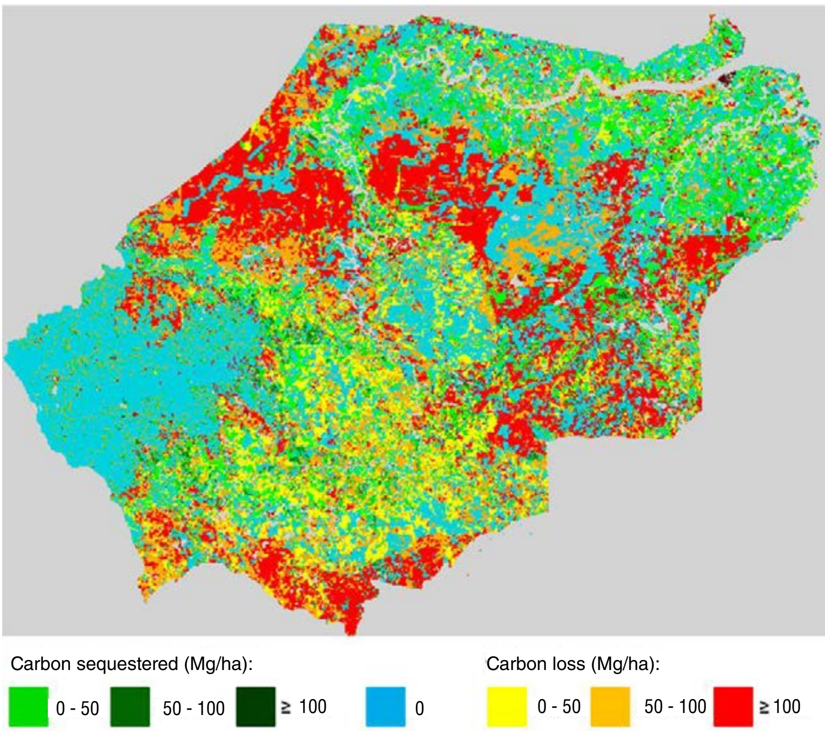

Figure 4.7 Net carbon loss map of Tanjabar, Jambi, Indonesia between 2000-2009. Pixel resolution is 100 m.

Table 4.4 Estimated aboveground landscape carbon stocks and areas under different land cover systems in Tanjabar in 2000 and 2009. Values in brackets refer to percentage values relative to total landscape carbon.

Land cover/use systems	Average plot-level carbon stock Δ (Mg. ha^{-1})	Area (km^2)		Estimated landscape carbon stock (Tg = 10^6Mg)	
		2000	2009	2000	2009
Undisturbed forest	214.2	56.1 (12.1)	41.7 (9.1)	12.01 (23.5)	8.93 (26.3)
Disturbed forest	151.0	185.5 (40.3)	68.0 (14.7)	28.06 (54.7)	10.26 (30.2)
Rubber agroforest	91.2	54.5 (11.8)	17.1 (3.7)	5.00 (9.7)	1.56 (4.6)
Coconut/bettlenut agroforest	64.2	33.1 (7.2)	54.7 (12.0)	2.13 (4.1)	3.52 (10.3)
Coffee agroforest	31.0	20.3 (4.4)	29.8 (6.4)	0.63 (1.2)	0.92 (2.7)
Acacia plantation	38.2	8.7 (1.9)	46.3 (9.9)	0.33 (0.6)	1.77 (5.2)
Rubber monoculture	38.2	23.5 (5.1)	29.7 (6.5)	0.90 (1.8)	1.13 (3.3)
Oil palm monoculture	41.0	46.1 (10.0)	103.3 (22.4)	1.29 (3.7)	4.24 (12.5)
Shrub	47.7	4.6 (1.0)	31.1 (6.7)	0.22 (0.4)	1.48 (4.4)
Agriculture	3.5	4.6 (1.0)	6.5 (1.5)	0.02 (0.03)	0.02 (0.07)
Settlement/cleared land/ grass	3.8	23.0 (5.0)	32.4 (7.1)	0.09 (0.2)	0.12 (0.4)
Estimated total landscape carbon stock (Tg)				51.19	33.96
Estimated average landscape carbon stock (Mg.ha^{-1})				111.2	73.8

ΔSource: http://db.worldagroforestry.org/, See section 4.4.2

Table 4.5 Estimated net carbon loss from Tanjabar landscape between 2000-2009.

Landscape carbon gain	3.395×10^6	Mg C
Landscape carbon loss	18.838×10^6	Mg C
Net carbon loss	15.443×10^6	Mg C
	1.72×10^6	Mg C.year^{-1}
	33.53	Mg C.ha^{-1}
	3.73	Mg C.ha^{-1}.year^{-1}
	13.66	Mg CO_2.ha^{-1}.year^{-1}

4.7.3 Uncertainty of landscape carbon stocks and carbon loss: Monte Carlo simulations

Using a Monte Carlo simulation approach, we can evaluate how uncertainty of land use classification and plot-level carbons stocks propagate and hence influence average landscape carbon estimates and net carbon loss estimates. Excluding uncertainty from land use classification and plot-level carbon density values produced biased estimates of average landscape carbon stocks and net carbon losses (WU versus LC+C, Figure 4.9, Table 4.6). The bias values varied across different runs; however, the maximum bias values were 1.20, -1.17 and 2.37 Mg.ha^{-1}, respectively for total average landscape carbon stocks 2000 and 2009, and average net carbon loss. The main source of bias was the exclusion of uncertainty in land use classification (WU versus LC, Figure 4.9), while excluding uncertainty in plot-level carbon density mainly influenced the variation in estimates (WU versus carbon, Figure 4.9, Table 4.6).

Overall the uncertainty of average landscape carbon and net carbon loss estimates of Tanjabar were small. The bias values were only 1% for both average landscape carbon stock estimates of 2000 and 2009; and 7.5% for the average net carbon loss. The coefficient variation values were 0.05% for both average landscape carbon stocks 2000 and 2009; and 0.2% for the average net carbon loss.

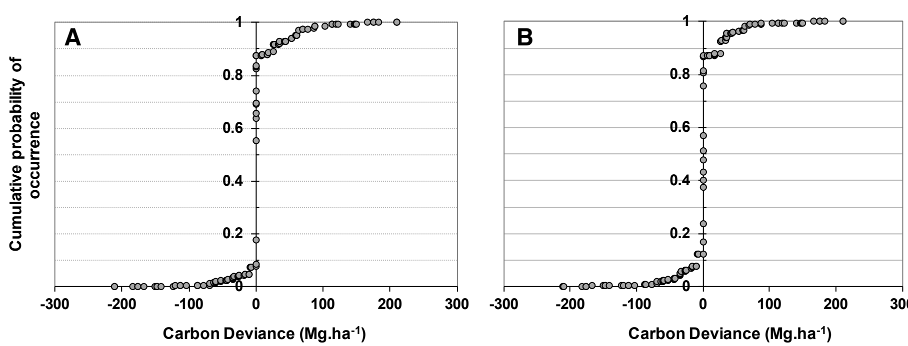

Figure 4.8 Expected carbon-stock-deviance patterns for Tanjabar landscape (Case I) for land cover maps of 2000 (A) and 2009 (B). The confidence level that the average landscape carbon stocks was estimated correctly (carbon deviation equals to 0) is 70% and 63% for year 2000 and 2009, respectively.

Table 4.6 Average landscape carbon stocks in 2000 and 2009 and corresponding average net carbon loss estimates: without uncertainty (WU) and with uncertainty (C, LC, and LC+C).

Estimate[a]	Mean ($Mg.ha^{-1}$)	Coefficient variation (%)	Confidence interval (2 x standard deviation, $Mg.ha^{-1}$)
Landscape carbon 2000			
WU	111.20	n.a[b]	n.a
C	112.27	0.01	112.22 – 112.30 (0.03)
LC	110.14	0.04	110.10 – 110.18 (0.10)
LC + C	110.14	0.05	110.10 – 110.18 (0.10)
Landscape carbon 2009			
WU	73.80	n.a	n.a
C	73.86	0.02	73. 84 – 73.88 (0.02)
LC	74.89	0.05	74. 82 – 74.96 (0.07)
LC + C	74.89	0.05	74. 82 – 74.96 (0.07)
Net Carbon loss 2000 - 2009			
WU	37.50	n.a	n.a
C	37.50	0.05	37.46 – 37.54 (0.04)
LC	35.35	0.17	35.18 – 35.52 (0.12)
LC + C	35.35	0.18	35.17 – 35.53 (0.13)

[a] WU = without errors, LC = with land use classification errors, C = with plot-level carbon variations, LC + C = with land use classification errors and plot-level carbon variations.

[b] n.a. = not applicable

4.7.4 Distribution of carbon loss estimates

The distribution of pixel level carbon loss estimates is depicted as cumulative distribution function (*cdf*). A useful indicator that can be derived from a ***cdf*** of carbon loss is the probability of carbon loss being equal or lower than 0, which describes the proportion of area that maintains or sequesters carbon. For the Tanjabar case, 35% of the area maintained or sequestered carbon (Figure 4.10). The ***cdf*** graph is a quantitative way of depicting a carbon loss map (Figure 4.7). The standard deviation for carbon loss was 25.5 Mg C ha^{-1} with a coefficient of variation value of 81.5%.

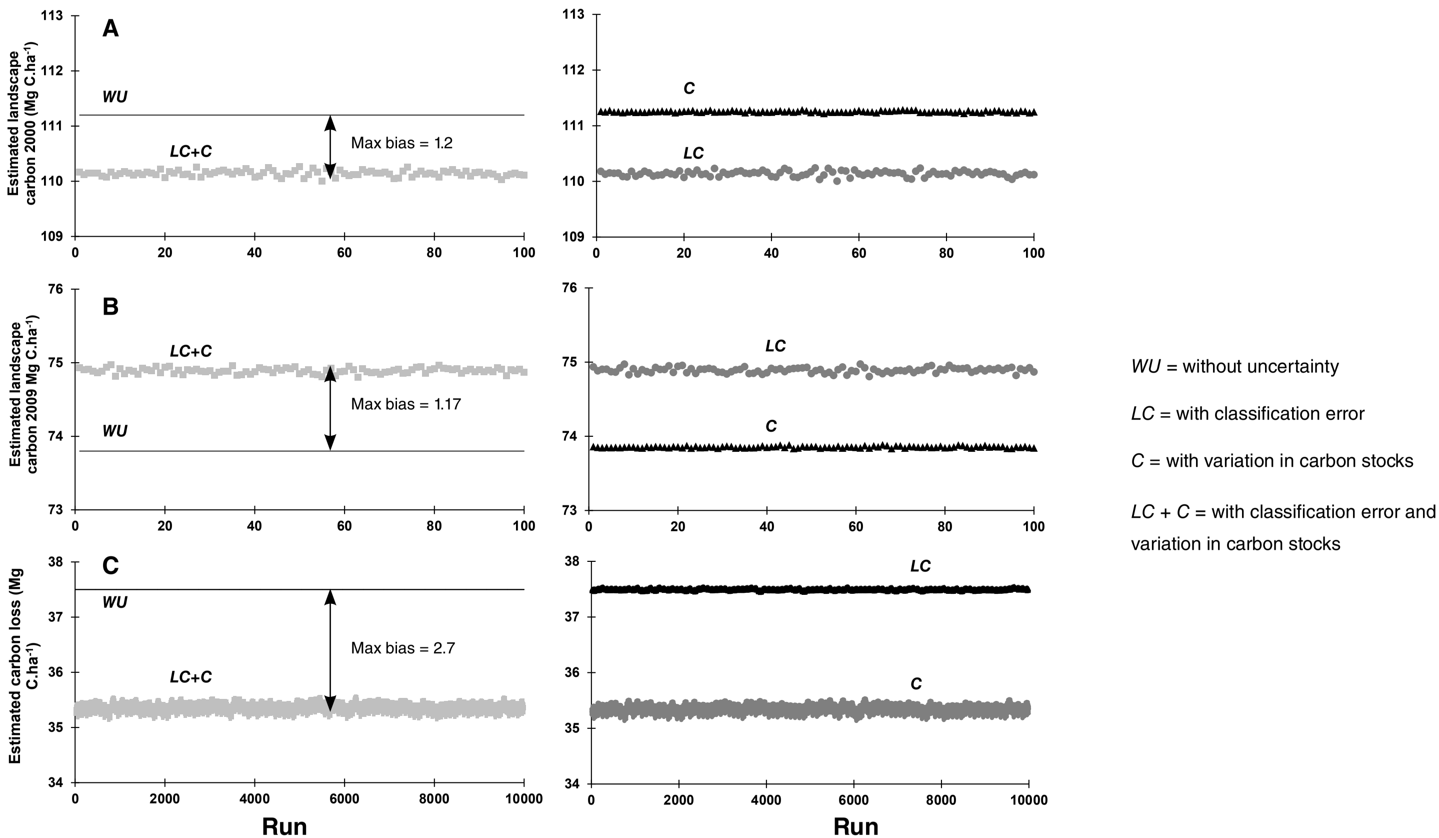

Figure 4.9 Results of Monte-Carlo simulation analysis for: A) average above-ground landscape carbon of year 2000; B) year 2009; and C) landscape carbon loss between year 2000-2009 for cases (Table 4.3) II (C), III (LC) and IV (LC+C).

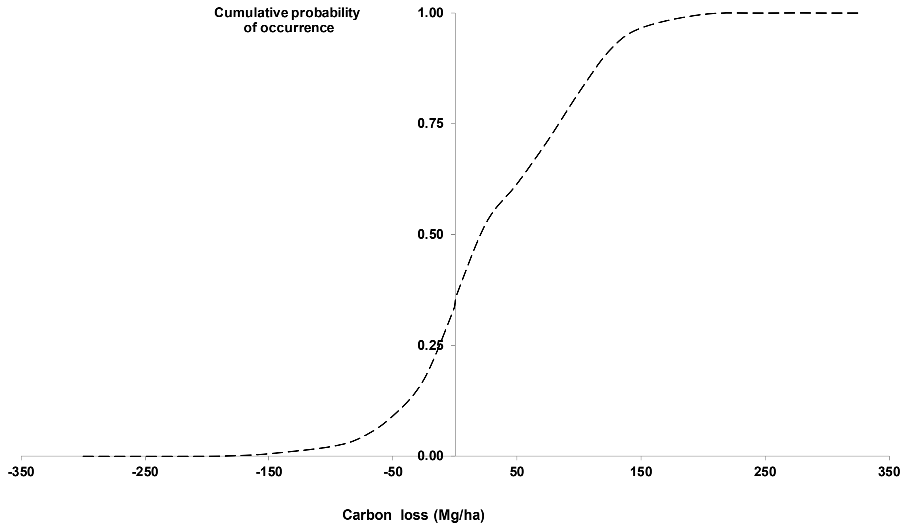

Figure 4.10 Cumulative density function of carbon loss at pixel level in Tanjabar during year 2000 – 2009 period.

4.8 Discussion

4.8.1 Representativeness of plot-level carbon density: simple average vs. time average carbon stocks

Palm et al. (2005) recommended the use of 'time-average carbon stocks' (TACS) to capture the dynamics of plot-level carbon of a particular land use systems over its various stages. TACS is the sum of the annual carbon stocks divided by the duration of the systems. TACS can also be estimated by regressing plot-level carbon with age of plot (or age of trees in the plot if age of trees or plot are not equal, van Noordwijk et al., 2002; Rahayu et al., 2005).

In this study, we assumed that the age distribution of sample points reflects their distribution in the landscape. Hence, we used a simple average carbon stocks for each land cover class to represent the carbon stocks of the main land use systems in Tanjabar. Comparison between simple-average versus time-average-carbon-stock values showed that the mean values were not significantly different. Thus, for the case of Tanajabar, the use of simple average may not give different results to the use of time–averaged carbon stocks.

4.8.2 Net carbon loss in Tanjabar

If we assume that a net carbon loss estimate is a proxy for carbon emission, then Tanjabar is categorized as an emitter district with an average carbon emission of 13.7

Mg CO_2 eq ha^{-1} $year^{-1}$ during the 2000-2009 period. This is higher than the average emission rate of Indonesia from 1990–2005, which was 2.14 Mg CO_2 eq ha^{-1} $year^{-1}$, and higher than the rate of 5.5 Mg CO_2 eq ha^{-1} $year^{-1}$ in the Riau province during 1990–2005 (Ekadinata and Dewi, 2011). The calculated carbon emission rate of Tanjabar is twice that of Chiapas, Mexico during an earlier period of 1975-1996 estimated to be 8.6 million C Mg.ha over 2.7 million hectare (Castillo-Santiago et al., 2007), or equivalent to 5.8 Mg CO_2 eq ha^{-1} $year^{-1}$.

The main source of carbon loss was establishment of large scale plantations and cash-crop oriented systems (oil palm, industrial forest of acacia, and rubber) in the area which contributed to almost half of land cover change (47.6%) and carbon loss (51.6%) in Tanjabar. Land use change from systems of low to higher carbon stock densities, such as establishment of agroforestry systems, only managed to mitigate 15% of total carbon loss. Thus, global market and trade relationships appears to be the main driver of carbon loss in the area similarly as elsewhere in Southeast Asia (Ziegler et al., 2012).

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4.8.3 Uncertainty assessment: bias in landscape carbon stocks and carbon loss estimates

The uncertainty of average landscape carbon stocks across the four cases (I, II, II and IV) understandably differed. In case I, the most simplistic situation, the approach relied only on information on probability of carbon-deviation or the possible difference in carbon values due to misclassification of land use classes. Case I method produced a wider confidence interval (less accuracy) compared to the Monte Carlo methods used in case II,III and IV that used more detailed information on variation in plot level carbons stocks or/and probability of error in land use classification. Based on the result of the Monte Carlo simulations, excluding errors in land cover classification (Case II) produced biased estimates of average landscape carbon stocks and carbon loss. The magnitude and direction (positive or negative) of the bias depended on the configuration of land cover distribution and its classification errors. For example, in the case of our study the estimated average landscape carbon stocks of year 2000 for Case II had a positive bias compared to the estimated value that included errors in land cover classification (Case III and IV). This occurred because in year 2000, 46% of the Tanjabar area was covered by high carbon containing systems of forests (undisturbed and disturbed) and rubber agroforests. Thus, excluding the error that some of the high carbon density systems areas actually could be low carbon values systems in reality (due to error in land use classification) led to over-estimations of carbon stocks. An opposite result of a negative bias occurred in year 2009. In that year, 73% of the Tanjabar area was mainly covered by low carbon containing systems (systems other than forests and rubber agroforest), combined with the relatively high probability that these low carbon containing systems in reality could be of higher carbon densities (e.g. classified as agriculture when the pixel could be undisturbed forest or coffee agroforestry) has led to under-estimation of carbon stocks. Bias due to land classification errors was discussed also by Riley et al. (1997) where they found that excluding classification errors may be underestimating the landscape carbon loss estimates by about 34 percent.

This study showed that the uncertainty assessed for the estimated average landscape carbon stocks and carbon loss was relatively low in our case study (narrow confidence interval, low bias). Methods that include uncertainty in both carbon density and land cover data are obviously more realistic compared to the other approaches. However, the land cover map used can still be further improved in terms of higher resolution and reduction of classification errors by increasing ground-truth samples. Persson and Azar (2007) similarly suggested the use of higher resolution images to improve the quality of land cover maps, or the use of advanced methods in image classification that uses high resolution imageries to calibrate lower ones. What is more important is to have land cover maps with resolution and given classes that can effectively characterize land cover changes at field scale (Zhao et al., 2010) which may differ between landscapes.

The challenge faced in the current study, that employed the carbon-stock-change approach, was to include changes (degradation or increment) within a land cover class that may occur over-time. The variation of plot level carbon density stocks that was used for uncertainty analysis and the use of separate categories for 'disturbed' and 'undisturbed' forest class may partially address this issue. Based on her study in Panama using a modelling approach, Pelletier et al. (2011) suggested to use narrow time- intervals of land cover maps to capture the dynamics of land cover, e.g. development of fallow systems in between time periods. For the Tanjabar study, this may not be an issue as the missing changes may still be related to low carbon values of almost similar magnitude and range (Figure 4.4).

Overall, given the data that currently exist, the result form Tanjabar showed a feasible approach to monitor carbon changes and their uncertainties in the landscape. To further improve opportunities by developing countries to use the developed approaches, it would be necessary to make accurate land cover maps that have an effective resolution with sufficient land use classes and with their associated information of carbon densities freely available.

4.8.4 Uncertainty assessment: reality check for the efficacy of carbon incentive scheme implementation

This study developed a methodological framework to estimate landscape carbon stocks and losses over time and their uncertainty for various cases of data availability, including the minimum data situation of having only a single land cover map and a single carbon estimate for each land use class that is often encountered in developing countries. The development of such a range of methods can help in supporting initiatives to increase the capacity of local stakeholders and local planners in designing development plans that synergize with plans to reduce carbon emissions (Angelsen et al., 2012; Dewi et al., 2011). The proposed approaches can assist in increasing awareness on uncertainty surrounding carbon emission estimates and steps that can be taken to reduce such uncertainty for better development of appropriate emission reduction policies.

The approach used in this study is intended to improve the Monitoring, Verification and Reporting (MRV) required for carbon incentive schemes. It was aimed to calculate the historical carbon loss in Tanjabar that would be useful for determining the baseline value. Thus, the ability to maintain or sequester carbon in an agreed period of time in the future could be monitored and be the basis for rewards. In this study, we used the definition of uncertainty as a deficit of our knowledge (van der Sluijs, 2008) and thus it was assumed that improving our knowledge, in this case better land cover maps and plot level carbon data, can improve the uncertainty of carbon loss estimates.

For developing a carbon incentive scheme, that would be of interest for local stakeholders, it would be important to know the eligible area for rewards, i.e. areas that are able to maintain or sequester carbon. If we use the results of Tanjabar as an example, during 2000 – 2009 approximately 35% of the total area would be eligible for such rewards, with carbon sequestration values ranging from 0 to 163 Mg.C ha^{-1}.$year^{-1}$. The coefficient of variation (representing uncertainty) of the carbon loss maps was 81.5% (Fig. 4.10). Lusiana et al. (2013) further explored this analysis to determine the appropriate scale (pixel resolution) for monitoring carbon change that would meet an appropriate error threshold. Their analysis showed that changes in pixel resolution would not only change the uncertainty of estimate but also the target area for rewards. The spatial certainty in terms of magnitude and location is important to consider for developing and implementation carbon incentive schemes, particularly at local level.

The issue of spatial uncertainty was analyzed by Sloan and Pelletier (2012) with a modelling approach to project a forward looking baseline. Comparing the projected map with actual maps for accuracy showed 14.8% disagreement mostly due to location disagreement. They concluded that the accuracy of a spatially projected baseline is unlikely to be acceptable for the purposes of a REDD+ scheme.

4.8.5 Limits of the study and possible future research

The study was carried out using only above-ground tree biomass as an estimate of plot-level carbon, excluding the component from understorey, necromass (dead wood), litter layer and soil. An analysis that takes into account the full component of carbon change would be desirable. In particular, understanding how land use change affects soil carbon contents merits further exploration. From the data set of Sumatra derived for mineral soils (Rahayu et al., pers. comm.) total carbon stock differences involving all pools (and soil until 30 cm depth) were 31% larger than differences in aboveground carbon stocks, with most of the difference related to root biomass. Other pools tended to have compensatory effects, e.g. the decrease in litter and soil carbon was associated with increase in understorey biomass. The primary additional uncertainty in the calculation of total below-ground carbon is likely to be the plant shoot:root ratio values, for which little empirical data is available. Below-ground changes on peat soils due to land use can be large. However, it has different determinants of uncertainty than what was considered in the current study.

The uncertainty analysis using a Monte Carlo approach was using an assumption that classification errors and plot level carbon were independent from each other. An analysis that includes correlation and/or spatial correlation in the classification errors would be the next step.

4.9 Conclusions

During the 2000 – 2009 period, around 13.7 Mg CO_2 eq ha^{-1} $year^{-1}$ were emitted in Tanjabar district. The main source of carbon emission was establishment of large scale plantation and cash-crop oriented systems (oil palm, industrial forest of acacia, and rubber) in the area. An uncertainty assement for the different cases of data availability (land cover maps and plot-level carbon density data) showed that excluding errors in land cover classification could lead to a biased estimate for the total average landscape carbon and average carbon emission. The magnitudes of the bias were relatively low: 1.17, 1.2 and 2.7 Mg CO_2 eq ha^{-1} $year^{-1}$ for landscape carbon of 2000, 2009 and landscape carbon loss from 2000-2009, respectively. The relatively small values were due to the tendency of errors in land use clasification to occur within land use of similar carbon values. Based on an established cumulative density function of carbon loss, the potential eligible area for an incentive carbon emission reduction scheme was 35%, based on land cover maps with 100 m resolution.

The development of a range of methods to estimate average landscape carbon stocks and carbon loss including their uncertainties can help in supporting initiatives to increase the capacity of local stakeholders and local planners in designing development plans that synergize with plans to reduce carbon emissions.

4.10 Supplements

S 4.10.1. Error matrix: probability that pixel in classified land cover map of Jambi 2009 is classified as land use i when it is land use j on the ground.

Classified Map+	Ground truth points+											Users' accuracy*
	Fu	Fd	Rf	Co	Cf	Ac	Rm	Op	Sh	Ag	Set	
Fu	0.78	0.17	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0.00	0.00	0.78
Fd	0.01	0.93	0.03	0.01	0.00	0.00	0.02	0.00	0.00	0.00	0.00	0.93
Rf	0.01	0.03	0.73	0.01	0.01	0.01	0.08	0.09	0.01	0.01	0.04	0.73
Co	0.00	0.00	0.00	0.25	0.00	0.00	0.38	0.38	0.00	0.00	0.00	0.25
Cf	0.00	0.02	0.02	0.11	0.74	0.00	0.00	0.04	0.04	0.00	0.02	0.74
Ac	0.00	0.00	0.00	0.00	0.00	0.33	0.44	0.11	0.00	0.00	0.11	0.33
Rm	0.00	0.01	0.05	0.01	0.00	0.01	0.85	0.06	0.01	0.00	0.02	0.85
Op	0.00	0.00	0.03	0.00	0.00	0.00	0.06	0.85	0.01	0.00	0.02	0.85
Sh	0.04	0.00	0.02	0.06	0.02	0.00	0.02	0.04	0.75	0.00	0.06	0.75
Ag	0.03	0.09	0.04	0.01	0.06	0.03	0.03	0.16	0.07	0.39	0.10	0.39
Set	0.00	0.00	0.00	0.02	0.02	0.00	0.04	0.11	0.01	0.05	0.75	0.75

+ Fu: Undisturbed forest, Fd:Disturbed forest, Rf: Rubber agroforestry, Co: coconut/bettlenut agrofrestry, Cf: coffee agrofoerstry,

Ac: acacia plantation, Rm: rubber monoculture, Op: oil palm, Sh: shrub, Ag: agriculture; Set: settlement, cleared land, grass.

* User' accuracy refers to probability that classified land cover map of Jambi 2009 matches gorundtruth data.

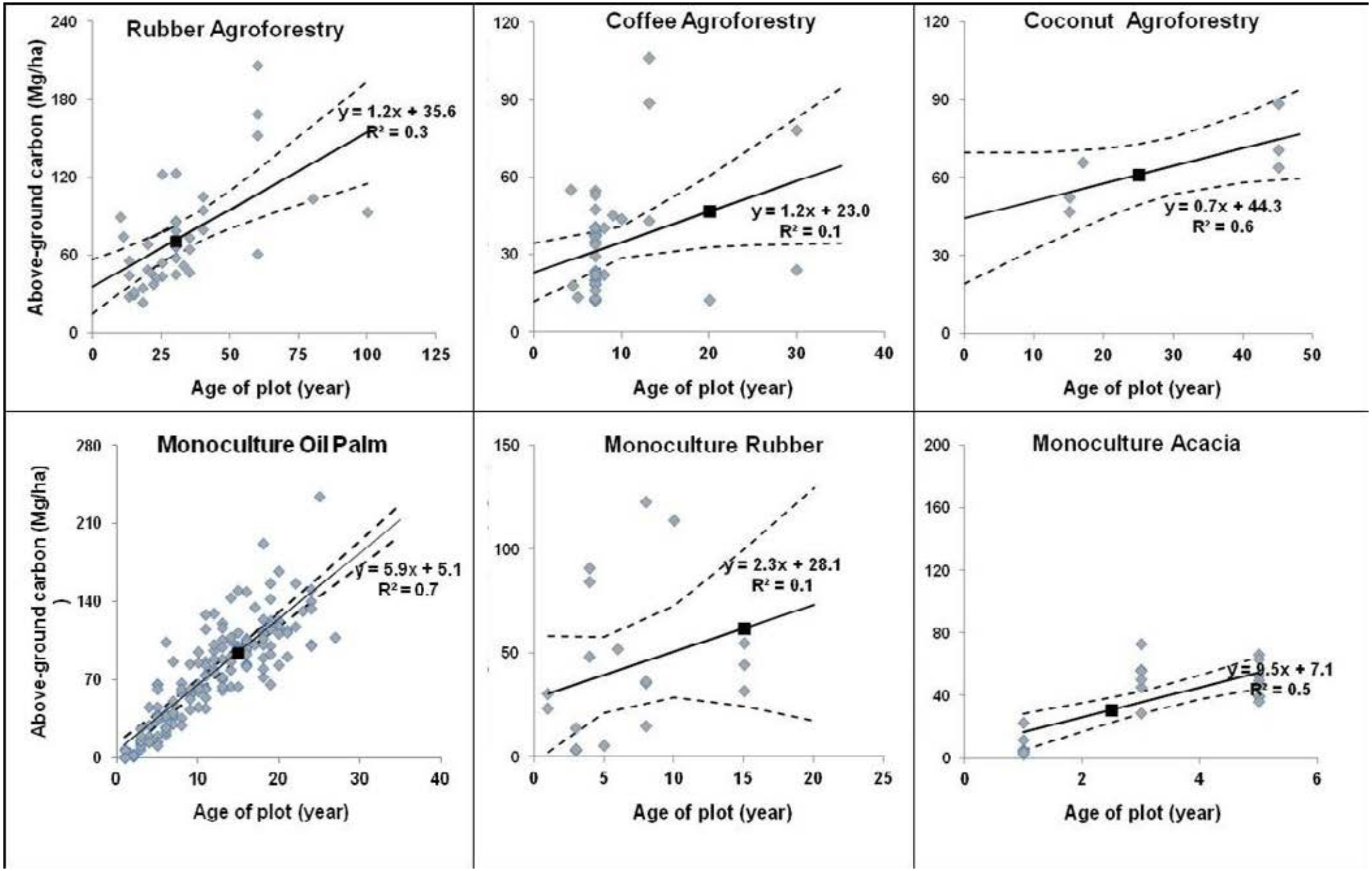
S 4.10.2 Estimated carbon average for main land use types in Tanjabar, Jambi: simple carbon average versus time average carbon stocks

Table S 4.10.2.1 Plot-level carbon average versus time average carbon stocks.

Land cover/ use systems	Average carbon stocks [a]		Time average carbon stocks[b]		95% Confidence interval of estimated time average carbon stock	Assumed rotational period of systems
Agroforestry systems						
Rubber	91.2	(9.8)	71.2	(10.7)	32.7	60
Coconut	64.2	(24.3)	61.2	(11.6)	11.0	50
Coffee	31.0	(4.2)	46.7	(14.0)	19.6	40
Monoculture tree based systems						
Acacia	38.2	(8.3)	30.7	(29.0)	7.7	5
Rubber	38.2	(7.4)	62.2	(37.1)	36.2	30
Oil Palm	41.0	(5.9)	94.2	(8.6)	23.4	20

[a]Value in brackets refer to standard error of mean

[b]Value in brackets refer to root mean square error



S 4.10.2.1 Plot of tree biomass as a function of plot age for managed tree-based systems found in Tanjabar. Solid line represents the regression line used to estimate time average. Dash-line represents confidence interval of the regression line. Black square represents the estimated time average carbons stocks. Acacia plantation is a rotational systems that is harvested every 5 years.

S 4.10.3 Methods to estimate uncertainty of landscape carbon stocks in cases where data of variation in plot-level carbon stocks is not availale.

The following describes detail description of calculation involved in estimating 'expected carbon-deviance':

1. Estimating errors of land cover map

Errors of land cover map can be derived from confusion matrix (Kohavi and Provost, 1998) that is commonly used to assess the accuracy of land use map classified from spatial imageries. In this study, we derived the confusion matrix (error matrix) by comparing ground truth points with land cover map 2009 of Jambi Province. Table 3 depicts an error matrix where $\boldsymbol{p_{ij}}$ is the probability of a pixel is land use category ***i*** (as observed in the field) when it is classified as land use category ***j***. For ***i*** = ***j***, it is the probability of correct classification, and the opposite applies for ***i*** ≠ ***i***.

S. 4.10.3.1 Matrix of errors in land cover classification.

Error Matrix		Land cover type j (land cover map)			
1		2	...	n	
Land cover type i (ground truth points)	1	p_{11}	p_{12}	...	p_{1n}
	2	p_{21}	p_{22}	...	p_{2n}
	...	...	...	...	...
	n	p_{n1}	p_{n2}	...	p_{nn}

2. Calculating deviation of carbon stock estimates

We defined 'Carbon deviation' or $\boldsymbol{E_{ij}}$ as the difference of plot-level carbon (ton/ha) of land use category ***i*** when it is classified as land use category ***j***.

For ***i*** = ***j***, the value of $\boldsymbol{E_{ij}}$ is 0.

S 4.10.3.2 Matrix of 'carbon deviation'.

Carbon deviation		*Land cover type* j *(land cover map)*			
		1	*2*	*...*	*n*
Land cover type *i* (ground truth points)	*1*	E_{11} = $(C_1 - C_1)$	E_{12} = $(C_1 - C_2)$	...	E_{1n} = $(C_1 - C_3)$
	2	E_{21} = $(C_2 - C_1)$	E_{22} = $(C_2 - C_2)$	...	E_{2n} = $(C_2 - C_3)$
	...	...	...	...	...
	n	E_{n1} = $(C_n - C_1)$	E_{n2} = $(C_n - C_2)$	...	E_{nn} = $(C_n - C_n)$

3. Calculating 'Weighted land cover error'

We defined 'Weighted land cover error' or $\boldsymbol{p_{w11}}$ as error of land cover type ***i*** weighted by the proportion of its area in the whole landscape ($\boldsymbol{L_i}$).

S 4.10.3.3 Matrix of 'Weighted land cover error'.

Weighted land cover error		Land use category j (land cover map)			
		1	2	...	n
Land use category *i* (ground truth points)	1	$p_{w11} =$ $L_1 * p_{11}$	$p_{w12} =$ $L_1 * p_{12}$	...	$p_{w1n} =$ $L_1 * p_{1n}$
	2	$p_{w21} =$ $L2 * p_{21}$	$p_{w22} =$ $L_2 * p_{22}$	...	$p_{w11} =$ $L_2 * p_{2n}$
	...	...	...	...	...
	n	$p_{wn1} =$ $L_n * p_{n1}$	$p_{wn2} =$ $L_n * p_{n2}$	...	$p_{wn3} =$ $L_n * p_{nn}$

4. Estimating expected carbon stocks deviation

The expected carbon stock deviation is a plot between carbon stock deviations with weighted land use error ($\boldsymbol{p_{w11}}$).

Chapter 5

Implication of uncertainty and scale in carbon emission estimates on locally appropriate designs to reduce emissions from deforestation and degradation (REDD+)[15]

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5.1 Abstract

This study combined uncertainty analysis of carbon emissions with local stakeholders' perspectives to develop an effective REDD+ scheme at the district level. Uncertainty of carbon emission estimates depends on scale while local stakeholders' views on plausible REDD+ schemes influence and limit transaction costs. The uncertainty analysis formed the basis for determining an appropriate scale for monitoring carbon emission estimates as performance measures for REDD+ incentives. Our analysis of stakeholder' perspectives explored (i) potential location and activities for lower emission development pathways, and (ii) perceived fair allocation of REDD+ incentives. Our case study focused on frontier forest in Tanjung Jabung Barat District, Jambi, Indonesia. The uncertainty analysis used Monte Carlo simulation techniques using known inaccuracy of land cover classification and variation in carbon stocks assessment per land cover type. With decreasing spatial resolution of carbon emission maps, uncertainty in carbon estimates decreased. At 1 km^2 resolution uncertainty was dropped below 5%, retaining most of the coarser spatial variation in the district. Fairness, efficiency and transaction cost issues in the design of REDD+ mechanisms were readily recognized by local stakeholders, who converged on an equal allocation to short-term efficiency (emission reduction activities) and long-term fairness (alternative livelihood development). A striking difference occurred in desirable transaction costs (which include monitoring, reporting and verification), with NGOs aiming for 8%, while government and researchers accepted transaction costs of 40%. Feasible measures for emission reduction in the district, derived from a participatory planning process, are compatible with the 1 km^2 spatial resolution of performance measures.

5.2 Keywords

Effective REDD+ design; fairness and efficiency; low-emission development; scale dependence; uncertainty of carbon emission.

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5.3 Introduction

Land use change, in particular tropical deforestation, is a major source of carbon emissions. From the 1960's until now, the amount of land use based carbon emissions has been relatively stable at about 1.1 Pg C/year. However, its contribution to global carbon emissions and the location of hotspots has changed. In the 1960's, land use based emissions represented about 30% of the total anthropogenic emissions, while in the 1990's this was 18% and in 2010 only 9% of the total. This is due to the large increase in fossil fuel emissions (Canadell et al., 2007,Peters et al., 2012). Globally terrestrial ecosystems are still a net sink for carbon, sequestering 26% of carbon emitted (Le Quere et al., 2009). Thus, maintaining forest and other tree-based systems remains highly desirable.

A scheme called Reducing Emissions from Deforestation and Forest Degradation (REDD) has been proposed by the international community to assist developing nations to reduce their GHG emissions that arise from forest conversion and degradation. Under this scheme, countries will receive incentives or compensation for slowing down or avoiding forest conversion and degradation (Agrawal et al., 2011). An extension of the REDD scheme, called REDD+, includes activities that promote forest management and carbon sequestration.

Finding an effective design to implement REDD+ that strikes a good balance between fair and efficient objectives has been a challenge to date (Hoang et al., 2013, Minang and van Noordwijk, 2013). Angelsen et al. (2008) and Pedroni et al. (2009) asserted the importance of designing a nested, multi-scale REDD+ scheme that includes developing programs at national, sub-national and project level. A well designed nested approach allows the REDD+ scheme to be sensitively designed according to the local forest and tree cover conditions. It would also allow the scheme to match the capabilities and the demands of its local stakeholders thus meeting the REDD safeguard criteria of '*full and effective participation of relevant stakeholders*' (Murphy, 2011). However, it is important to realize that the ability of a REDD+ policy to meet its national target requires understanding how its processes are linked across scales. An approach to reduce deforestation that is effective at the project level may not be as effective at an aggregated level such as at the district level. Therefore, the issue of scale must be addressed when designing REDD+ activities, such as designing a monitoring and crediting framework, implementation, ownership of credits as well as approval and verification of credits (Cattaneo, 2011). Cattaneo (2011) further differentiates the issue of scale as a resolution problem (dependency on pixel resolution) as well as an aggregation problem (dependency on how pixels are aggregated).

Angelsen, et al (2008) proposed the '3E' criteria in designing a REDD+ scheme: carbon ***effectiveness***, cost ***efficiency***, and ***equity***. Carbon effectiveness refers to the magnitude of the additional emission reductions achieved and inclusion of significant emission sources. Cost efficiency indicates whether the given emission reduction is achieved at minimum cost; including costs for starting up and running the emission reduction scheme as well as compensation for opportunity costs of foregoing legally allowed activities that would lead to higher emissions. Equity entails, but is not limited to, ensuring that all countries have equal footing in terms of implementation (international level equity) and in terms of sharing the benefits of a REDD+ scheme (national level equity). Similar and complementary criteria have been developed in the realm of Payment for Environmental Services (PES) design: fairness and efficiency (van Noordwijk et al., 2012. ***Efficiency*** refers to both carbon effectiveness and cost efficiency, while ***fairness*** includes stakeholder perceptions of fairness as well as quantifiable equity.

The main objective of this study was to design effective carbon emission reduction activities at sub-national (district) level that can be carried out by stakeholders in the district, with the district government spearheading and coordinating the activities. This study draws upon three parallel activities carried out in Tanjung Jabung Barat (Tanjabar), a high-emission district in Jambi province, which involved: (i) developing district planners' design for carbon emission reduction activities that also allow economic growth in the area, (ii) assessing the level of spatial scale/resolution for measuring carbon emissions that meet a given error tolerance level, and (iii) compiling local stakeholders' views on fair distribution of potential benefits from REDD+ schemes. Hence, this paper discusses two of the '3 Es' criteria: carbon ***effectivenes***s and ***equity***. The analysis on the cost ***efficiency*** for the proposed emission reduction scheme for Tanjabar is discussed in Mulia et al. (2013), a companion to this paper that calculated the opportunity cost of several emission reduction pathways.

5.4 Material and methods

5.4.1 The study site: Tanjung Jabung Barat (Tanjabar) district.

The district of Tanjabar is situated in the north eastern part of Jambi Province, Indonesia with a total area of 5010 km^2 (Figure 4.2). Tanjabar is a coastal area with the geographic location of 7.35 S–102.64 E and 1.45 S–103.58 E. The site represents a typical forest frontier situation in the topics, where forest conversion is ongoing and carried out by large-scale operators as well as smallholder farmers.

For the past 20 years, widespread conversion of land occurred in the area mainly converting forest to plantations of oil palm (***Eleais guineensis*** Jacg.), rubber (***Hevea brasiliensis*** Muell.Arg) and acacia (***Acacia mangium*** Willd.). Ekadinata, et al. (2011) showed that conversion from forest to oil palm and acacia plantations alone contributed to 33% of total emissions in the area during 2000–2009. This calculation was based on estimated losses of aboveground biomass and did not take into account the potentially large carbon emissions from drained peat areas (Wösten et al., 1997) which constitutes 40% of the land. Mulia et al. (2014) provide a further detailed description of the study area.

5.4.2 Planning for a low-emission development pathway in Tanjabar

The development of a plan for low, carbon, emission trajectories in Tanjabar was carried out by the Tanjabar District Planning and Development Agency (Badan Perencanaan Pembangunan Daerah/Bappeda) in collaboration with the World Agroforestry Centre.

Preparing the development plan encompassed several stakeholders meetings, in depth discussions and joint analysis in order to carry out the following activities: (i) development of a land use allocation zone map based on existing land use related maps, i.e. the District Spatial Land Use Plan (Rencana Tata Ruang Wilayah – RTRW), mining areas, oil palm plantations and timber plantation concessions; (ii) creating detailed development and management plans for each land use allocation zone based on discussions with relevant governmental agencies, e.g. forestry, mining, and agriculture; (iii) projection of landscape carbon emission estimates based on the existing Tanjabar development plan; (iv) constructing low carbon emission development pathways based on the projected carbon emission estimate; (v) calculating landscape carbon emission estimates based on the low carbon emission development pathways (decided in the previous step). The steps followed a framework described in Dewi et al. (2011), aiming to build a platform for negotiation between different stakeholders in planning development pathways that can reduce carbon emissions and, in general, enhance ecosystem services (ES) provisions.

The results of these evaluations formed the background of a recently published policy brief (Ekadinata et al. 2011). In the current paper, we combine these results with additional estimations of the errors of carbon estimates at different spatial resolutions (described in the next section) in order to identify an appropriate scale for monitoring carbon emissions as performance measure for a REDD+ scheme.

5.4.3 Estimating the errors of aboveground carbon emission estimates at different spatial resolutions

This study aimed to identify an appropriate spatial resolution for monitoring carbon emission estimates for the Tanjabar district that meet an error/accuracy threshold based on local conditions and acceptability of error. The study had two main steps:

1. Developing carbon emission maps for Tanjabar from measured and observed changes in aboveground carbon stocks between 2000 and 2009. The carbon emission maps include uncertainty originating from errors in land cover map classifications and variations of carbon assessments (variations associated with land use/cover), and
2. Based on the carbon emission maps developed in step (i), carbon emission estimates at various resolutions were calculated.

Chapter 4, in particular Section 4.6 and Figure 4.6, provides detailed description of the methods used to generate the 2000–2009 carbon emission maps for Tanjabar and the associated effects of scale on estimated carbon emissions.The basis of the methods is that the uncertainty of carbon stocks of a carbon emission map can be quantified and generated using a Monte Carlo simulation approach, which is a statistical technique used to evaluate how errors propagate (Refsgaard et al., 2007). The carbon emission maps have 100 m resolution, similar to the resolution of the 2000 and 2009 land cover maps used in the analysis.

Based on the propagated carbon emission maps, we then carried out the spatial aggregation analysis that recalculated patch level carbon emission estimates using a moving window approach. The finest spatial resolution used was 100 m, which was subsequently increased to emulate land holdings of individual households, local communities and villages within the district. Hence, the spatial resolution was varied from 100 m to 30,000 m. Next, we evaluated the effect of resolution of the corresponding carbon emission map on the following attributes:

1. Cumulative probability distribution of carbon-emission estimates, that can provide information on percent of area sequestering or emitting carbon;
2. uncertainty of patch level carbon emissions indicating potential errors of carbon emission estimates relevant for determining an appropriate resolution for monitoring the performance of a REDD+ scheme; and
3. ability to identify carbon emission hotspots that are useful for stakeholders' negotiations preceding REDD+ project development and implementation.

The methods for these evaluations were (a) development of cumulative probability functions of carbon emissions, (b) estimation of standard deviation of carbon emissions and coefficient variation, and (c) carbon emission maps at various spatial resolutions.

A cumulative density function (***cdf***) was used to describe the cumulative probability of carbon emission estimates. It is basically a cumulative frequency distribution rescaled by the total frequency and thus its value spans between 0 and 1. In this study, the ***cdf*** specifically describes the fraction of area that has carbon emission values at lower or equal to the associated value on the X-axis.

Standard deviation of carbon estimates at resolution $\boldsymbol{p}$ ($\boldsymbol{std_p}$) is calculated based on the aggregated emission carbon maps developed for each spatial resolution, using the following equation:

$$std_p = \left(\Sigma_k \left(E_k - (\Sigma_k E_k)/ n_k\right)^2/n_k\right)^{½} \quad \text{.......}(5.1)$$

where $\boldsymbol{E_k}$ = estimated carbon emissions for patch $\boldsymbol{k}$ (Mg.ha^{-1}), where $\boldsymbol{k}$ = 1, ..., $\boldsymbol{n}$, $\boldsymbol{n}$ = total number of patches

The coefficient variation of carbon estimates at resolution $\boldsymbol{p}$ ($\boldsymbol{CV_p}$) is calculated using the following equation:

$$CV_p = \left(stdp_p/(\Sigma_k E_k /n_k)\right) \quad \text{.......}(5.2)$$

5.4.4 Exploring stakeholders' views on fair distribution of benefits to be gained from REDD+ schemes

To explore stakeholders' perceptions and expectations for 'fair and efficient' REDD+ schemes we used a framework named FERVA (Fair and Efficient REDD Value-chain Allocation) described by van Noordwijk (2008). The FERVA approach claimed that any future gain (payment) from REDD+ schemes is derived from a 'value-chain' of four main implementation efforts: (i) direct reduction of emissions, (ii) reorientation of development pathways and livelihood alternatives towards the maintenance of high carbon stock landscapes (as opposed to lucrative economic gains from deforestation), (iii) transaction costs incurred for participating in emission reduction schemes, including for Monitoring, Reporting and Verification (MRV) processes, and (iv) activities to connect potential buyers with sellers, including raising awareness of the REDD+ project to potential buyers (costs to secure buyers). Activities (i) and (ii) occur at the local level, while most of activities (iii) and (iv) occur at the international level.

We held a focus group discussion (FGD) attended by 30 participants from Jambi province and Tanjabar: 10 people from local NGOs working on environmental issues and community empowerment, 10 local university staff and 10 government officials working on development planning, forestry, agriculture, environment and socio-economic issues. Stakeholders were asked to qualitatively allocate financial units over the four value-chain elements of REDD+ benefits (payment): (i) emission reductions, (ii) livelihood alternatives, (iii) transaction costs, and (iv) costs to secure buyers. The main outputs of this activity was a stakeholders' perspective on the expected current situation and desirable future distribution of benefits gained from the proposed REDD+ scheme(s), which is important and of relevance in the implementation of REDD+ in the local context.

5.5 Results

5.5.1 Low emission development plan in Tanjabar: local planners' views

The existing Tanjabar development plan categorized Tanjabar district into twelve land use allocation zones (Table 5.1). The categorization was based on the District Spatial Land Use Plan (***RTRW)*** in combination with other land zoning maps developed by various governmental agencies such as the Department of Forestry, Mining and National Land Bureau. In the existing development plan for Tanjabar District for 2010–2030, more than half of the area (54%) is allocated to large-scale oil palm and industrial forest plantation companies (Figure 5.1A, Table 5.1). It was projected that implementing the existing development plan would emit 36 CO_2 Mg ha^{-1} $year^{-1}$ during the period 2009–2025 with the main sources of emissions from conversion of forest to oil palm and industrial forest plantation, including deforestation in peat forests (Ekadinata et al. 2011). Therefore to reduce carbon emissions, the local planning agency focused on modifying the existing development plan in zones allocated for industrial forest plantations, oil palm plantations, production forest and protected (Figure 5.1B, Table 5.1) encompassing 63.7% of the total landscape as a potential area. The proposed low emission plan did not modify activities in the other allocation zones, thus activities in these areas would follow the existing plan. Hence, mining and conversion of forest to rubber systems would still be allowed to take place. Similarly, logging would continue within the areas already given to concessionaires.

Table 5.1 Land use allocations in Tanjabar showing current/existing development and proposed 'low carbon emission' development plans.

Development zones[1]	Area (km^2)	Existing development plan	Low emission development plan[2]
Mining concession	12.5	To operate all current mining concession areas existing land use will be converted to open areas. Mining operations will adhere to land reclamation and land restoration policies	n.a
Production forest[3]	75.6	Non-concession areas will be developed as buffer zones in the form of community forests or ecotourism forests.	Maintain undisturbed forest area Establish rubber systems in non-forested areas
Limited production forest[4]	340.1	Non-concession areas will be developed as buffer zones in the form of community forests or ecotourism forest.	
Industrial forest plantations	1563.0	All land will be converted to Acacia mangium[5] plantations except for settlements, oil palm systems and tree-based systems that already exist in the area	Avoid conversion of primary forest to acacia, maintain existing smallholders' tree-based systems and expedite planting acacia in shrub and grassland areas within the concession zone (5% of total landscape area)
Protected area	109.7	The protected area will be maintained as it is	n.a
Peatland protected forest	120.2	Forested area will be protected and oil palm systems will be converted to mixed tree-based systems by planting jelutung (Dyera costulata[6]).	Increase effort to maintain existing forest area and establish Dyera costulata systems.
Big scale oil palm concession	906. 6	Establishing large-scale oil palm plantations	Prohibit conversion of forest to oil palm (2% of total landscape area)
Settlement	21.0	Expand and develop as needed	n.a
Irrigated agricultural land	231.3	Establishing paddy rice systems will be a priority	n.a
Dryland agriculture	734.0	Establishing tree-based systems such as oil palm, rubber, fruit and coconut	n.a

Development zones[1]	Area (km^2)	Existing development plan	Low emission development plan[2]
Other land uses	448.7	Establishing tree-based systems such as oil palm, rubber, fruit and coconut	n.a
Forest park[7]	18.8	All land will be converted to rubber systems	n.a

(Source: Ekadinata et al. 2011).

[1] Based on Tanjabar district Spatial Land Use Plan (***Rencana Tata Ruang Wilayah – RTRW)*** 2010 – 2030 combined with other land zoning maps

[2]Proposed by Tanjabar District Planning Agency during participatory land use planning (LUWES) exercise

[3] Production forest is state forestland designated for production purposes

[4] Limited production forest is state forestland designated for limited production purposes

[5] ***Acacia mangium*** is a major tree species in plantations owned by large pulp and paper industries. Other uses include fuelwood, timber for building and furniture and particle boards.

[6]***Dyera costulata*** (local name: jelutung) is an endangered tree species mostly found in the rainforest of Indonesia, Malaysia and Thailand. Its latex is tapped for chewing gum and is mainly exported.

[7]Locally known as Taman Hutan Rakyat (Tahura) is an area designated for conservation in particular to preserve endemic and non-endemic flora and fauna for the purpose of research, science, education, cultivation, cultural, tourism and recreational purposes.

n.a. = not applicable, the low emission development plan proposed by local planners did not include these areas.

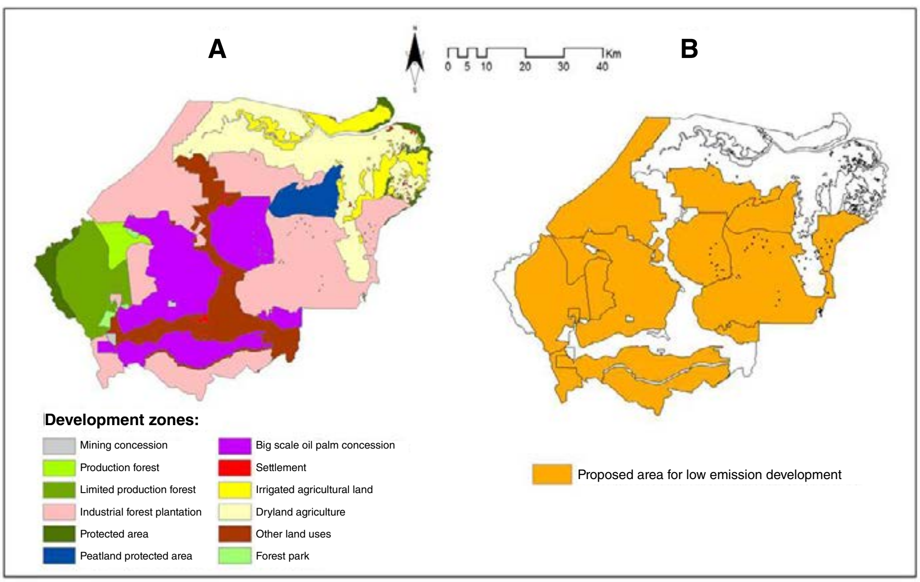

Figure 5.1 Map of Tanjung Jabung Barat district, A) current land use allocation zones of Tanjabar district as the basis for the new (low carbon emission) development plan and B) potential low carbon emission development activity areas (highlighted in orange) as planned by local stakeholders (modified from: Ekadinata et al., 2011). Table 5.1 provides description of each allocation zones.

Projection results showed that the proposed activities in low emission development could potentially reduce emissions by 27% by 2025 (Ekadinata et al. 2011). The highest potential reduction could be obtained from implementing low emission plans in areas allocated for oil palm plantations by prohibiting conversion of natural forests to oil palm and allocating new oil palm plantations to be established on degraded or abandoned land with lower carbon stocks.

5.5.2 Scale effect on the distribution of carbon emission estimates

The next three sections discuss the results from the uncertainty and scale study using the actual Tanjabar land cover maps of 2000 and 2009, but not the land zoning map (as described in the previous section).

The evaluated distribution of carbon emission estimates over the period 2000 – 2009 (9 years) at different spatial resolutions presented in Figure 5.2 represents the percentage area that has carbon emission values at lower or equal to the value on the X-axis. At the original spatial resolution of 100 m, the lowest carbon emission estimate was -550 Mg CO_2 eq. ha^{-1} (or equal to 61.1 Mg CO_2 eq. ha^{-1} $year^{-1}$ sequestration), while at 200 m spatial resolution the lowest carbon emission was -275 Mg CO_2 eq. ha^{-1} (or equal to 30.6 Mg CO_2 eq. ha^{-1} $year^{-1}$ sequestration). Thus, the possible lowest carbon emission increased as spatial resolution decreased. An opposite trend occurred for the highest carbon emission, whereby the value decreased as spatial resolution decreased. Overall, as the spatial resolution decreased the carbon emission value shifted towards the average landscape carbon emission (at approximately 130 Mg CO_2 eq. ha^{-1}), which was also the value where all the ***cdf***'s converged.

Using the developed ***cdf,*** we can derive the proportion of area that has zero or lower carbon emissions between year 2000 and 2009 for different spatial resolutions (Figure 5.3). Thus, this function denotes the ***proportion of carbon sequestration area*** (P_{Cseq}) in the landscape and can then provide an indication of the potential area eligible for receiving emission reduction incentives via the REDD+ scheme. For the Tanjabar landscape, at 100 m spatial resolution the P_{Cseq} value was 34.8%. The P_{Cseq} value decreased along with the decrease of spatial resolution reaching 0% at a pixel length of 30,000 m (or equal to a pixel size of 900 km^2). Consequently, decreasing the spatial resolution has led to loss of information on potential patches/areas that sequestered carbon; hence also a loss of information about potential areas eligible for receiving incentives from the REDD+ scheme(s) occurred.

5.5.3 Scale effects on uncertainty of carbon estimates

The uncertainty of carbon estimates (represented by the coefficient of variation) decreased with the decrease of spatial resolution (Table 5.2). As spatial resolution decreased, neighboring pixels were aggregated and averaged forming a new value. Thus, variations that may have existed between neighboring pixels were reduced or lost, while the overall average value was more or less intact. Consequently, the coefficient variation values followed the trajectory of spatial resolution.

Depending on a given threshold value for acceptable uncertainty, we can propose an appropriate scale for monitoring carbon emissions to be used in the proposed REDD+ scheme (Figure 5.3). For example, using a threshold of 5% uncertainty (or 95% accuracy) the corresponding unit for performance measure is carbon emission map with pixel resolution of 1000 m or equal to a pixel size of 1 km^2. A lower uncertainty threshold implies the need for a larger spatial resolution of patch size as a performance measure in the REDD+ MRV scheme, and vice versa a higher uncertainty threshold implies a smaller patch size can be used for performance measures.

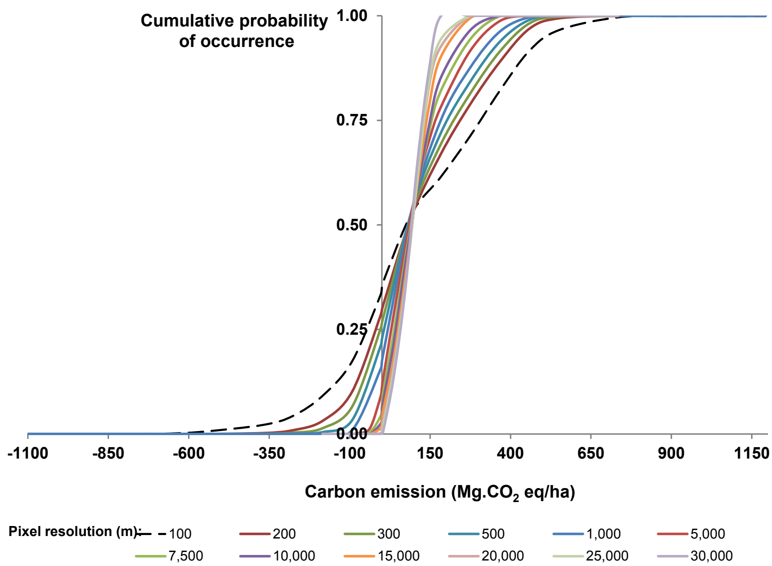

Figure 5.2 Cumulative distribution functions of carbon emission estimates for Tanjung Jabung Barat, Jambi during the period 2000 – 2009. Pixel resolution of 100 m equals to 1 ha pixel area and pixel resolution of 1,000 m equals to 1 km^2 pixel area.

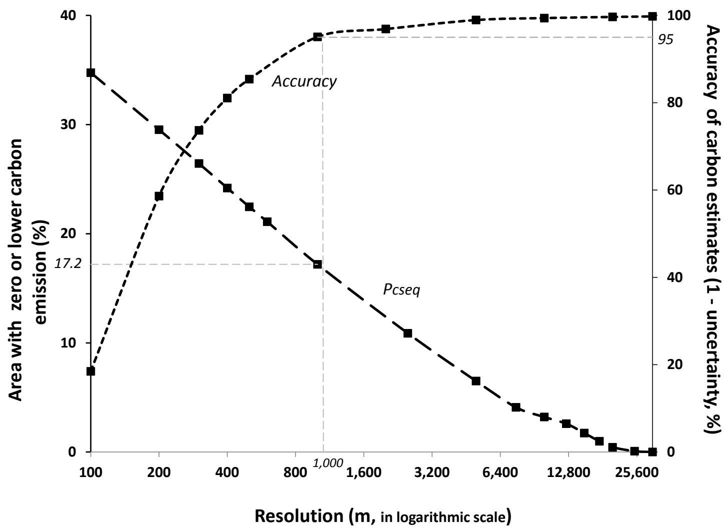

Figure 5.3 Potential carbon sequestration area (P_{Cseq}) and accuracy (1- uncertainty) of carbon estimates at different spatial resolutions for Tanjung Jabung Barat, Jambi during the period 2000 - 2009. Pixel resolution of 100 m equals to pixel area of 1 ha and pixel resolution of 1,000 m equals to pixel area of 1 km^2.

Table 5.2 Uncertainty of carbon emisions for Tanjung Jabung Barat, Jambi from 2000 and 2009.

Spatial resolution (m)	Standard deviation[a] ($Mg.CO_2$ eq.ha^{-1})	Coefficient of variation (uncertainty)[a] (%)
100	93.5	81.54
200	47.3	41.37
300	30.1	26.31
400	21.6	18.91
500	16.8	14.59
1,000	5.7	4.91
2,000	3.6	3.11
5,000	1.2	1.05
10,000	0.7	0.60
20,000	0.4	0.34
30,000	0.3	0.22

[a] Based on 100,000 Monte Carlo generated carbon emission maps

5.5.4 Spatial pattern of carbon emission hot spots

Spatial information on hot spots of carbon emissions and sequestration is useful in targeting areas for REDD+ implementation. Figure 5.4 provides information on the location of net carbon emissions areas (red) and carbon sequestration areas (green). Figure 5.4 is basically a spatial image representation of the ***cdf*** graph shown in Figure 5.2. It shows how carbon emissions are distributed spatially across the landscape. Carbon emission maps with 1000 m and 2500 m resolution are compatible with the location for the proposed low emission activities depicted in Figure 5.1. It shows the area where carbon sequestration will be maintained (forest peat land in green, Figure 5.4) and carbon emissions will be reduced (zone allocated for large scale plantations in Figure 5.1).

Areas of the greatest source of carbon emissions were identified as oil palm and industrial forest plantations converted from disturbed forest, while areas of greatest carbon sink were rubber and coconut/bettlenut agroforesty systems converted from agriculture and shrub areas (Ekadinata et al., 2011).

The carbon emission maps with decreasing spatial resolutions provide different perspectives on the location of hotspot areas. An extreme comparison showed that carbon sequestration areas identified at 100 m resolution had completely disappeared at 30,000 m resolution

5.5.5 Fair distribution of emission reduction incentives scheme: stakeholders' perspectives

In evaluating the allocation of economic benefits of a certain emission reduction scheme, the stakeholders were grouped into NGO (n=10), government (n=10) and researchers (n=10). Each group were asked to assess the expected distribution of the allocation of REDD+ benefits and what the desirable distribution should be, with respect to the four elements involved in implementation of REDD+ schemes: (i) direct emission reduction, (ii) finding sustainable livelihood alternatives, (iii) transaction costs (including cost of MRV), and (iv) cost to secure buyers. The result of the FGD indicated that all stakeholders expected that benefits would mostly be allocated for 'transaction costs and Monitoring, Reporting and Verification (MRV) procedures', i.e. 48%, 35% and 50% by NGO, government and researchers respectively (Figure 5.5). However, most stakeholders desired the transaction costs to receive lower financial allocation, except for the government who wished an increase in allocation by 10% from what they expected it would be.

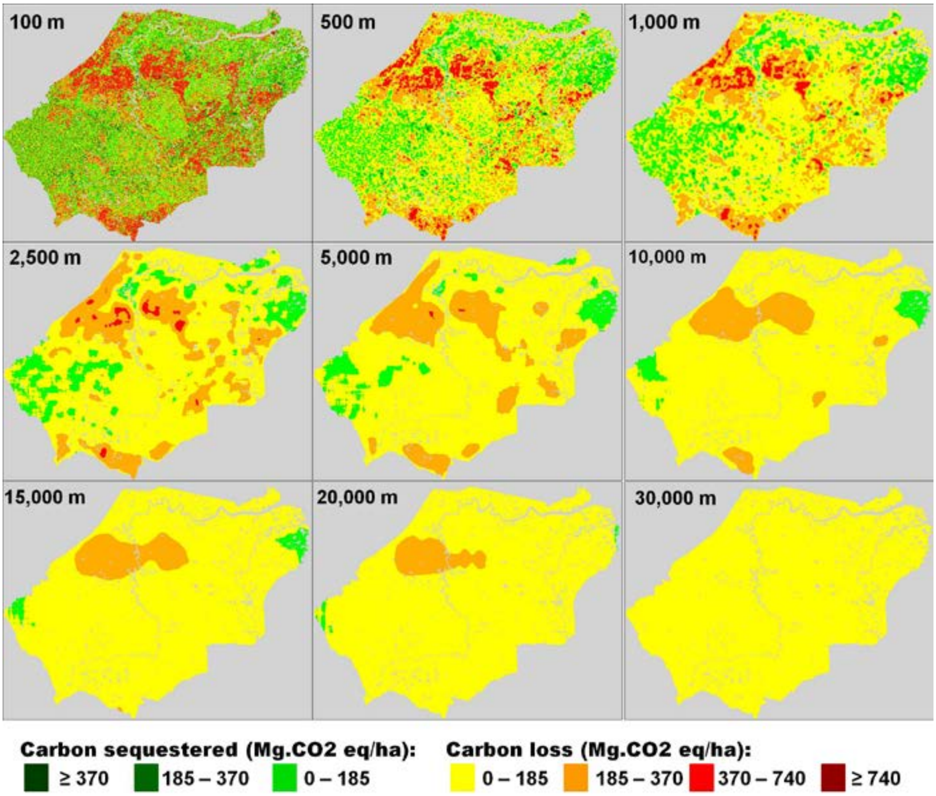

Figure 5.4 The effect of scale on hot spots of carbon emissions in Tanjabar, Jambi, Indonesia between 2000 and 2009. Pixel resolution of 100 m equals to pixel area of 1 ha and pixel resolution of 1,000 m equals to pixel area of 1 km^2.

Compared to the other stakeholders, NGOs expected the allocation for 'direct emission reduction' and 'finding sustainable livelihood alternatives' components would be higher compared to transaction costs and cost to secure buyers. They also desired that 87% of the benefits derived from the REDD+ scheme should be allocated to this component. This was strikingly different to the desire of the government and researchers with 35% and 20%, respectively. Nevertheless, all stakeholders agreed that ideally allocation for 'emission reduction' and 'sustainable livelihood' components should be higher than currently expected, while allocation 'to secure buyers' should be lower. Government expected that a large part of the benefits be allocated to 'secure buyers' (40%) compared to the 10% and 30% by NGOs and researchers.

5.6 Discussion

5.6.1 What level of spatial aggregation is appropriate for an incentive scheme for emission reduction in Tanjabar?

Carbon emission reduction estimates at any scale combine signal (actual carbon emission) and noise (uncertainty in measurement of land use change and associated carbon stocks). An incentive scheme for emission reduction requires performance measures for monitoring changes at the landscape scale with a clear signal and low noise. Setting the threshold for an acceptable uncertainty at 5%, a decrease in spatial resolution from 100 m (1 ha pixel size) of an interpreted land cover change map to a 1000 m (1 km^2 pixel size) would be required in the Tanjabar landscape for monitoring emission changes. If a 1 km^2 grid size were applied, the major differentiations of high and low emission areas within the district could still be maintained (Figure 5.4). However, at the 1 km^2 performance scale, only 17.2 % of the Tanjabar area had a zero or negative carbon emission estimate over the observation period and would therefore be eligible to receive incentives under a hypothetical emission reduction incentive scheme, while at a 1 ha scale twice this fraction of pixels would appear to be eligible if a higher uncertainty was tolerated (Figure 5.3). The average landholding of farmers in Tanjabar ranged from 4 – 8 ha, depending on where they were operating (peat or mineral soils) and who they were (migrant or local farmers) (Khususiyah N, 2012). Thus, at a 1 km^2 performance scale the emission reduction scheme will not be targeting individual farmers, but more likely villages or farmer groups.

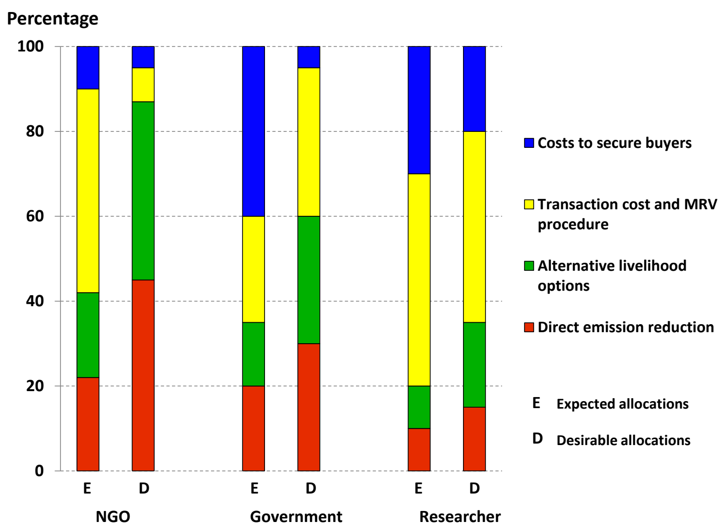

Figure 5.5 Proposed allocation of benefits (proportion of total carbon payment) to the REDD+ value chain in both expected and desired scenarios according to the stakeholders of Tanjabar, Sumatra. The values are based on n=10 for each group.

The 5% threshold is rather arbitrary in the absence of specific research on tolerance to uncertainty of REDD+ or PES implementation schemes in general. Further empirical data on such tolerance levels will be needed in order to better justify necessary designs. The threshold uncertainty would function as an indicator of how much uncertainty can be tolerated in the case where 'carbon emission reduction payments' would be off target, i.e. eligible ES providers did not get any payment or ineligible ES providers got paid. A welfare aid program in the US used two separate tolerance levels, 5% for payment for ineligible recipients and 3% for overpayment or underpayment to eligible (Griswold and Spurrier, 1975).

The current assessment was based on aboveground carbon stock estimates, but evidence for Jambi suggests that land use effects on belowground carbon stocks on mineral soils are strongly related to aboveground changes (van Noordwijk et al., this issue). The spatial distribution of peat and peat depth in Tanjabar district, however, adds further uncertainty, as transitions between mineral and peat soils tend to be gradual. Future spatial aggregation and uncertainty studies that combine above and below ground carbon stock change in the district would increase its local relevance, but would make it less applicable elsewhere.

5.6.2 How can efficiency, fairness and transaction costs be balanced in the design of emission reduction or REDD+ mechanism?

As suggested from the experience PES implementation (Wunder et al., 2008), spatial aggregation (and hence larger area) is likely to reduce transaction costs, particularly for carrying out MRV procedures. All stakeholders, both in expected and desired configurations, agreed on an approximately 50-50 % split between emission reduction developing alternatives for livelihood options. However, they differed in expectations about the share that 'costs-to-secure-buyers' plus transaction costs would take. They agreed that 'costs-to-secure-buyers' and transaction costs should be reduced (65, 40 or 15 % of the total benefit for academicians, government officials and NGO participants, respectively), but differed in the proportion they would be willing to allocate to transaction costs (including MRV): this ranged from 40% for academicians to 8% for NGO-participants. A more detailed costing of the components of MRV in relation to scale will be needed. However, the use of a 1 km^2 performance measure, of lower resolution than most land cover maps, is likely to shift transaction costs to a lower value, which is a desirable direction from the stakeholders' perspectives (Börner and Wunder, 2008).

In addition to the spatial scale of efficiency discussed above, a temporal perspective on efficiency is also relevant. Efforts towards 'direct emission reduction' are aiming to gain short-term efficiency, while efforts for 'finding livelihood alternatives' encompass finding options to avoid potential emission reduction. This in turn is aiming for long-term efficiency as well as finding options to support sustainable livelihoods as part of gaining fairness. Thus, at intermediate or long-term temporal scale, efficiency and fairness converged.

5.6.3 How do local planners prioritize carbon emission reduction with economic growth in the district?

Potential REDD+ activities developed by local planners are compatible with the 1 km^2 aggregated carbon emission 'hot-spot' map (Figure 5.2 and Figure 5.4). Local planners recommended that emission reduction schemes in Tanjabar should focus on three allocation zones that were the main source of carbon emissions: Industrial Tree Plantations (HTI), Oil Palm Plantations (HGU) and Peatland Forest Management Units (KPHLG). However, these areas were also the main land use systems contributing to economic growth in the area (Sofiyuddin et al., 2012). Thus, in the modified Tanjabar development plan, local planners proposed activities that maintain the use of these areas for productive purposes; while at the same time increasing carbon sequestration by encouraging optimized use of abandoned and degraded land through the establishment of agroforestry systems that potentially can be managed by local farmers. Therefore, the establishment of agroforestry systems could also provide alternative livelihood options for farmers. Local planners also proposed allocation of conservation zones within the allocated oil palm plantation zone, albeit at a small fraction of 2%.

Other emission reduction activities include encouraging the use of raw materials (for timber or plywood) from planted trees and reduce (or even forego) the use of wood from natural forests (Ekadinata and Agung, 2011).The potential emission reduction activities developed and proposed by local planners avoid drastic change that solely aimed for emission reduction as they understood the importance of maintaining economic growth as well. This approach appropriately matched the desires of the other stakeholders participating in the FGD when asked to allocate benefits from a hypothetical emission reduction scheme, which is to maximize the allocation for livelihood benefits.

5.6.4 Optimizing efficiency and stakeholders' perspectives for developing the REDD+ scheme in Tanjabar

Many stakeholders are involved in natural resource management, hence sustainable natural resource management requires a reconciliation of multiple knowledge systems: local, government, and science-based (Clark et al., 2011). In such a context, quality criteria for the application of science in natural resource management involve salience (actionable conclusions), credibility (evidence-based and empirically tested theoretical frameworks, explicitness of assumptions, and analysis of confidence intervals) and legitimacy (matching multiple stakeholder perceptions of representing their perspectives) (Lusiana et al., 2011).The current study met these requirements; whereby the actionable conclusion is a set of REDD+ plan activities derived from land use planning and carbon emission maps (at various scales with estimated uncertainty). This became a basis for performance based rewards and stakeholders' evaluation on what they perceived as fair and efficient benefit allocation of a future REDD+ scheme.

Tanjabar carbon emission maps were derived from 1 ha scale land cover maps, while a stakeholder REDD+ activities plan was based on land allocation maps with its zone allocation as the scale of its unit activities. These different units of scale were able to be reconciled at a 1 km scale. Matching the institutional scale and the scale at which ES performance were based is important to support the formulation or implementation of ES management (Hein et al., 2006).

A quantitative approach to the issue of scale in REDD+ or PES designs based on the tolerance for uncertainty in the implementation stage has, to our knowledge, not been previously attempted. It can add perspectives to the spatial analysis of co-benefits that makes use of the spatial correlation of determinants of biodiversity and carbon stock (Strassburg et al., 2010), or biodiversity and watershed functions (Douglas et al., 2007). The 1 km^2 scale identified in this study, applies to its specific spatial properties of land cover and land use change, as well as the reliability of the land use change detection method and carbon stock uncertainty. However, replication of the approach in other landscapes may yield different results, depending on the quality of carbon and land cover data, as well as type of land use change activities (Pelletier et al. 2011). For the Tanjabar case, improved methods in the future could be aimed at increasing the spatial resolution feasible within the tolerated uncertainty range.

REDD+ designs should not only take care of the uncertainty of MRV, but need to also look into the social uncertainty related to land tenure security in the area (Galudra et al., this issue). Harmonization between district (Tanjabar) and provincial (Jambi) spatial planning as well as with large-scale operators (industrial plantations and oil palm) and local people/farmers are necessary for developing effective and equitable implementation of national REDD+ programs.

5.7 Conclusion

An uncertainty assessment of carbon emissions combined with spatial aggregation analysis can provide insights into how carbon emissions are distributed within the landscape. The outcome can provide recommendations on units for performance measures that can support efficient implementation of the REDD+ scheme. For Tanjabar, using 1 km^2 spatial aggregation the uncertaintys in carbon emission estimates dropped below 5%, while much of the spatial distribution of patch level carbon (high and low emissions) in the area is retained. Fairness and transaction cost issues in the design of REDD+ mechanisms are also recognized by local stakeholders, who converge on an equal allocation to short-term efficiency and long-term fairness aspects. Feasible measures for emission reduction in the district, as derived from a participatory planning process, are compatible with the 1-km^2 aggregation level of spatial performance data. Efforts should be made to improve methods that allow reducing uncertainty/variability in carbon estimates as these could increase the potential area/beneficiaries from 17.2% at 1 km^2 patch size to 34.8% at 1 ha patch size. The uncertainty analysis combined with spatial approach has the potential to support REDD+ activities, in particular identifying the right scale for MRV activities.

Chapter

6

General discussion

6.1 Salience, credibility and legitimacy in land use change modeling: model validation as product or process?

No model, other than trivial ones, is universally valid. Model validation in essence is a statement about the validity of a specific type of use of a model for the purpose of a user (or group of users) for a particular given context. Most reported model validation tests refer to the degree of correspondence (goodness of fit) between a 'predicted' and 'observed' set of changes in specified properties of a system. For complex socio-ecological models, such model validation may not be feasible due to lack of independent data, or in ex-ante analysis, data for the future is reasonably not available. In addition to 'goodness of fit', another indicator of a 'valid' model is the ability of model to perform under a wide of range of situation (robustness). For many models, robustness and precision (goodness of fit for a specific circumstance) may be negatively correlated, and this points to an approach for model validation that applies only for specific circumstance.

In the introduction (Chapter 1) three overarching hypotheses were framed for this thesis that explores the concept of model validity:

1. Salience, credibility and legitimacy are equally important attributes in determining users' acceptance of a simulation model (Chapter 2)
2. There are synergistic opportunities in balancing land productivity and maintaining ecosystems functions that can be elucidated with modelling (Chapter 3)
3. Uncertainty is scale-dependent and environmental management institutions need a scale-dependent response to uncertainty in performance metrics (Chapters 4 and 5).

In this chapter the findings of these preceding chapters will be discussed in a wider context aiming to relate the various groups of (potential) model users through their primary needs for information, guidance and understanding, to the concepts of model validity that are most relevant to them. Table 6.1 summarises the various model evaluation/validation approaches that were carried out within the PhD study. The approaches implemented in this study ranges from a simple calculation of errors, goodness of fit to Monte-Carlo simulation approach.

6.2 Is it possible to have synergies instead of negative trade-offs between farmers' welfare and environmental services?

Throughout the world, a growing population increases the demand for food, fiber, feed and energy as well as settlement areas, causing accelerated forest conversion to agricultural land and conversion of existing agricultural land for settlement and urban development. The accelerated land use change has raised concerns over its impact on the degradation of ecosystems services. Although a win-win solution is desired, trade-offs between economic growth and environmental services are inevitable.

The scenario analysis study in the Upper Konto catchment showed that farmers' welfare and fodder availability were increasing at the expense of carbon stocks (Chapter 3). This is in line with the global trend of negative trade-offs between economic growth/ farmers' welfare and environmental services (West et al., 2010, Green et al., 2005) and the rare occurrence of synergies (DeFries and Rosenzweig, 2010). Farmer' welfare in terms of food security or income is of direct benefit to farmers and thus, in the Upper Konto catchment as well as in many other places in the world, farmers will generally prioritize welfare compared to environmental services.

Table 6.1 Validation/model evaluation approaches carried out in the preceding chapters.

	Methods	Indicator	Chapter
1.	Sensitivity analysis	The simulated range of responses for reasonable parameter values reflect what is expected in reality	3
2	Two map comparison: comparing reference map with simulated maps at subsequent time	Area accuracy, location accuracy	2
3	Comparing the accuracy of the land use change model to the accuracy of its null model at multiple resolutions, where null model is defined as model that assumes complete persistence of land use across the simulated time period.[@]	Goodness of fit: summarises the way the fit/accuracy changes as the resolution of measurement changes	3
4	Three map comparisons: comparing reference maps (initial and subsequent times) and simulated map (subsequent time).[#]	Figure of merit: measure the overall correspondence between observed and simulated changes based on partitioning of errors	3
5	Comparing predicted value or trend with secondary data (survey results, statistics data)	Plausibility of model results against expert opinion.	3
6	Uncertainty assesment: assess the propagation of errors in input values	Confidence interval of parameter estimates	4, 5
7	Participatory model evaluation: in-depth particpatory discussion with potential model users on model results	Model performance from the perspective of model users, e.g. accuracy of model results and future projection.	2
8	Evaluation of NRM recommendation derived from uncertainty assesment with results obtain from stakeholders view based participatory approach	Plausibility of model results against stakeholder opinion.	3

[@] Based on methods developed by Costanza (1989)

[#] Based on methods developed by Pontius et al. (2011)

Both the land sparing and land sharing approaches for natural resource management (NRM) aims to synergize farmers' welfare and environmental service, but use different approaches in reaching these goals. The land sparing approach suggests that by segregating areas for protecting environmental services from areas for production, conceptually at landscape level both goals will be maximized. However, in developing countries such a synergy rarely occurs. Protected areas for conservation are often blamed for cutting the access of the adjacent community to their main source of livelihood (Kusters et al., 2007). In contrast, our results showed that a modified land sparing approach (e.g. fodder production in forest plantations) can at least partially address these issues. Without having to convert existing forest, thus not reducing carbon stocks, the farmers were able to increase their livelihood by approximately 10%.

Incentive or compensation schemes such as Payment for Environmental Services (PES) and REDD+ are also approaches to synergize the goal of livelihood and environmental services. Benefits earned from PES or REDD+ can potentially compensate foregone income/production due to choosing a certain land management that maintains environmental services. Nevertheless, realization in the field has not yet reached beyond trials that still require ample help from donors or research institutes. Some of the main challenges in implementing ES schemes include: effective and accurate approach to monitor performance, e.g. carbon emission reduction (Pelletier et al., 2013) and effective and fair mechanism for sharing benefits derived from ES incentive schemes (Pascual et al., 2010). Developing approaches and methods as carried out in Chapter 5 could help in ensuring an effective implementation of ES incentive scheme.

6.3 What are the factors influencing users' acceptance of simulation models for natural resource management?

Following Matthews et al. (2004) definition of model users' we defined potential model users as both the target users and beneficiaries. Target users' are direct users of models such as researchers, consultants, educators and trainers. Beneficiaries are those that will benefit from the outcome of models that include policy makers, NGO (Non-Governmental Organization), extension staff and farmers.

Result from a survey carried out with potential model users for NRM showed that model users' considered salience (the relevancy of the model to address identified problems) and credibility (the concepts, processes and results of the model are acceptable as an approximation of reality) as the most important factor to accept a model for NRM, both in hypothetical situation. Similar findings were also gained when model users were confronted with an actual model. A process to increase legitimacy (perceptions by stakeholders that the model developers have unbiased intention and agenda) through jointly developing model scenarios for model application has led to increase in users' perception of model credibility (Chapter 2).

Users' knowledge and understanding of the dynamics and processes involved in NRM are also influencing model user' acceptance of a simulation model. Issues and concerns in NRM are operating beyond plot and efficiency[15] scale and mostly at landscape and persistence[1] scale; e.g. establishing land zoning policy for watershed function that can balance carbon sequestration and local income (Chapter 2) or designing a development plan at district level that can sustain economic growth while at the same time lower carbon emission rate (Chapter 4). Thus, when users' main goal and interest of using models is still operating at efficiency scale, i.e. focusing on the economic profitability/ resource sufficiency; their acceptance and perceived credibility of a model may largely be influenced by precision of model results.

Consequently, increasing users' acceptance of dynamic land use change models requires parallel efforts to increase model users' ability to understand processes at persistence or change scale. This is crucial, particularly if the aim is to help natural resource managers and policy makers to use model as a tool for ex-ante analysis on the impact of any NRM policies. To help bridging this gap of knowledge, activities such as land use planning with local land use planners and natural resource managers using a simple model can help in bridging the gap of knowledge (Chapter 2 and 5).

Efforts to increase model salience and credibility can also increase overall users' acceptance. Increasing salience of model can be through adding or modifying sub-modules or through participatory modelling activities to involve local stakeholders in developing model scenarios (Ritzema et al., 2010;Bots and Daalen, 2008), as was carried out in the Upper Konto catchment study that added a livestock module into the FALLOW model (Chapter 3). Sensitivity analysis, evaluation of model output and uncertainty analysis can provide quantitative measures to users to assess model's credibility (Chapter 3).

6.4 Is FALLOW a suitable model tool for data poor environments?

FALLOW is a dynamic spatially explicit model that requires limited detailed inputs, where most of the inputs easily accessible or obtainable using surveys and participatory approaches (Lippe et al., 2011, Chapter 3). The model was developed to carry out ex-ante analysis of landscape dynamics and its consequences for various landscape indicators at aggregated level (Chapter 1, Box 1). Thus, the model is potentially suitable for data poor environment.

From the experience of using FALLOW in Aceh, Indonesia, using limited data that were available (land cover, profitability of land use systems) with results that were actually far from accurate compared to reality with only 24 – 73% spatial accuracy (for each land use type), the model was able to demonstrate the efficacy of a simulation model for land-use planning and natural resource management of the landscape in Aceh. Hence, model users accepted the model. However, model users in Aceh still considered lack

15 Jackson et al. (2010) distinguished three levels of temporal scale in sustainability: efficiency, persistence and change. Efficiency refers to agro-ecosystems role of provisioning at plot level scale where decision making aims at gaining resource sufficiency. Persistence involves functional integrity to ensure agro-ecosystems services flows continuously. Change refers to human capacity to deal with change.

of data inputs as the main obstacles for further use of FALLOW. This may be a general issue for developing countries in general where data collection are often considered of less priorities compared to other activities that have direct benefits, e.g. building infrastructures. Thus, in addition to have credible parsimonious simulation models, efforts to increase model users awareness on the importance of data collection as well as efforts to make databases accessible at no cost are important, e.g. allometric equations (Henry et al., 2013), GHG inventory data (UNFCCC, 2013) and wood density (Chave et al., 2009b, Chave et al., 2009a). The caveat of using model such as FALLOW is that one may lose the details related to household level innovation and adaptation.

Today much effort is made to develop improved methods to acquire data, e.g. participatory approaches (Lippe et al., 2011), or new technologies such as infrared technology to rapidly assess soil quality (Demyan et al., 2013). As more data becomes available also other or more complex models could be used, e.g. LUCIA (Marohn et al., 2012). A complex model, however, need further understanding how the processes interact and thus has higher requirements for establishing quantitative thresholds.

Nevertheless, the essential criterion for choosing the appropriate level of model complexity is that a simulation model should be salient to its intended use, in terms of temporal and spatial scale, level of aggregation and abstraction of processes. Another factor to consider is the robustness of the model to adapt to new application, i.e. the ease in adding additional module or coupling the model to existing models.

6.5 How does scale influence uncertainty and what is the implication of uncertainty for model application?

The uncertainty analysis carried out in Chapter 4 aimed to determine the confidence interval (variation) of average landscape carbon and carbon loss estimates, while the study in Chapter 5 aimed to evaluate the effect of scale/resolution on the variation of patch-level carbon emission estimates. The overall aim was to assess the implication of uncertainty and scale in carbon emission estimates on the design of an incentive scheme for carbon emission reduction. The result showed that exclusion of uncertainty in the estimation of carbon emission could lead to a biased estimate for an average landscape carbon emission, albeit small and insignificant due to the tendency of errors in land use clasification to occur within land use of similar carbon values (Chapter 4). In Chapter 5 we showed that uncertainty analysis can formed the basis for determining an appropriate scale for monitoring carbon emission estimates as performance measures of a REDD+ scheme. The resolution level also determined the magnitude of potential area eligible for carbon payment (carbon emission equal or less than zero). In the case of our study area in Tanjung Jabung Barat district, Indonesia, we found that at 1 km^2 pixel size (1000 m resolution) was the appropriate scale for monitoring carbon emission as the errors of carbon emission estimates dropped below 5%. At this scale, 17.2% of Tanjabar landscape was potentially eligible for a REDD+ scheme. At larger scale (lower resolution) the potential eligible area was lower, whereas at finer scale (higher scale) the potential eligible area was higher.

The uncertainty analysis carried out in the study is based on the inclusion of known quantifiable/ measureable uncertainty from land cover classification and variation of carbon. stocks at plot level. However, in reality, unquantifiable uncertainties exist, e.g. landscape level dynamics of soil carbon that highly influence the overall carbon emission in Tanjabar. In complex systems these unquantifiable uncertainties are often the most relevant and salient ones. Although the modelling activities may not be able to include all these uncertainties, acknowledging the existence of such uncertainties and the ability to identify what information of uncertainty is relevant and meaningful for the modelling objectives will enhance the use of modelling for NRM.

6.6 What is the role of modelling in supporting decision making for natural resource management?

Argent (2004) distinguish 4 levels of model development and application, ranging from Level 1 that aims for own use by researchers for specific research questions to Level 4 that aims for policy making where model very often is packaged as black box. FALLOW, the model used in this thesis, is in between level 3 and 4, being flexible for a range of applications and usefully describes some natural phenomena at a moderate level of detail, and with manageable data requirement that are operationally useful. The results from FALLOW can be used to provide policy recommendations; however, its user interface has not yet reached the maturity to be used directly by policy makers themselves.

Our finding in Chapter 2 showed that simple model like FALLOW is useful to attract policy makers and natural resource managers to use a model. The model is useful for teaching the processes and interactions that take place in the landscape, aiming to increase policy makers understanding of temporal scale from efficiency to persistence. At the same time, it is essential to make policy makers understand that the objective of a model like FALLOW is not to obtain prescriptive results but rather on finding options and tradeoffs. In many cases of NRM win-win outcomes are rarely feasible, at least in the short to medium term, what is possible is to understand and negotiate trade-offs among stakeholders. Hence, providing support to develop policy that can provide innovative solution towards trade-offs is the ultimate goal for modelling for NRM. Establishing a modeling consortia where scientists along side with policy makers and natural resource managers apply a model could be of assistance, as complex interactions at landscape scale even in a simple model might be challenging that could lead to wrong interpretations.

The processes and knowledge that must exist in a simulation model depend on the purpose of the simulation model (Table 6.2). A simple erosion model such as EPIC (Sharpley and Williams, 1990) or crop model such as CERES (Ritchie, 1998) were built for 'scientific curiousity' and intended to provide coherent account of obervable and predictive powers.The main requirement for such a model is credibility. Models such as FALLOW (van Noordwijk, 2002) and LUCIA (Marohn et al, 2013) are aim at clarifying trade-offs between goals and consequences of choices foreseen. Ultimately, simulation model for NRM intends to support negotiation between stakeholders by providing common platform for understanding issues.

Table 6.2 Characteristics of model according to its purpose and knowledge sources.

Actors/Agents	None	Single	Multiple
Attributes of knowledge	Credibility	Salience and credibility	Salience, credibility and legitimacy
Source of knowledge / Aim of model use	Curiousity	Decision making	Negotiation support
Science	Providing coherent account of observables and predictive powers		
Science and policy makers' knowledge		Clarifying tradeoffs between goals and consequences of choices foreseen#	
All stakeholders' knowledge			Providing common platform for understanding issues, and choices, stakes involved, space for new solutions*

Note.

#The model need to include the plausible responses of land users to new rules, incentives and opportunities.

*This may require linking/coupling several existing models.

The overall table is inspired by Clark et al. (2011). See Table 1-1.

6.7 Conclusions and Outlook

It is no longer enough that models are "syntheses of existing information and guides or maps to direct future work". A model must be able to prospect plausible scenarios and provide a range of options to policy issues. Hence model can become the framework for decision making. Increasing salience, credibility and legitimacy of a simulation, through technical model improvement, improving communication of model results and building trust, can increase the use of simulation model for natural resource management.

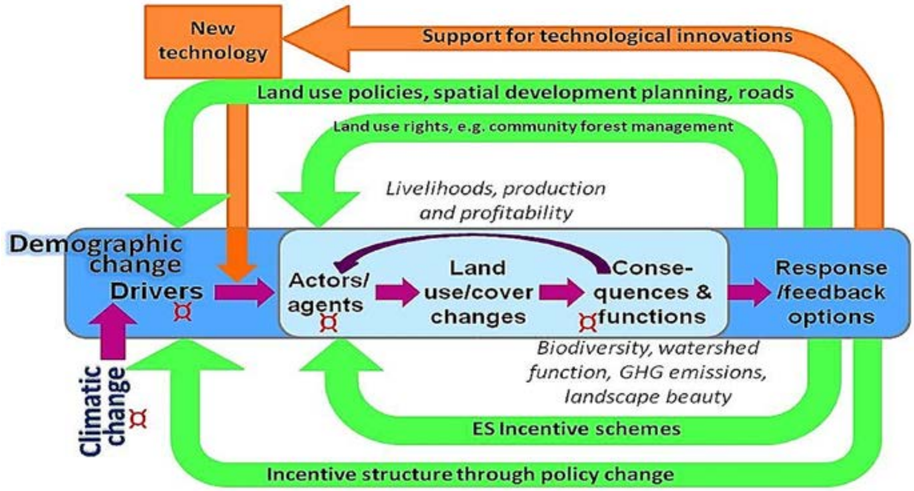

Figure 6.1 A conceptual framework of a simulation model for natural resource management. Components in blue-square refer to processes included within a simulation model. Green and orange arrows refer to feedback loops that allow scenario analysis to explore the effect of NRM policies or interventions. Symbol ¤ showed the component where uncertainty can be incorporated. Modified and extended from van Noordwijk et al. (2011).

In developing countries, development of simulation models for NRM at landscape/ watershed level that can provide a platform for negotiation among stakeholders with different interest and goals is pertinent. These models must embrace the existence of uncertainties including the varying level of uncertainties. The model must also have the ability to allow direct model users to, first, feel confident about the salience of the model, then gradually learn trade-offs and the consequence of a particular model scenario and ultimately explore possible options.

For effective modelling we will need increasingly:

- Intuitive GUI's
- Databases
- Better knowledge of feedbacks
- Ability to easily link or couple models
- Framework to bridge model developers with users, policy makers and public

Summary/ Zusammenfassung

Summary

Sustainable resource management requires balancing trade-offs between land productivity and environmental integrity while maintaining equality in resource access. Scenario analysis based on a credible simulation model can help to efficiently assess the dynamic and complex interactions in between components and their trade-offs. However, despite the potential of simulation models as decision support tools, acceptance and use by decision makers and natural resource managers are still major challenges, particularly in developing countries. This study was carried out to address issues related to validation of simulation models that includes users' perspectives on validity of simulation models, scenario-based trade-offs analysis and uncertainty assessment for designing management intervention.

Firstly, the current study analyzed users' perspectives on validity of a simulation model for natural resource management based on two activities. The first activity is based on surveys in four countries (Indonesia, Kenya, Philippines and Vietnam). It explored the perceptions and expectations of potential model users (researchers, lecturers, natural resource managers, policy makers, communicators) on a hypothetical model. The second activity was a participatory model evaluation in Aceh, Indonesia involving use of the spatially explicit FALLOW model and evaluation of its outputs. When assessing a hypothetical model, potential model users' considered salience (relevance) as the most important attribute in a simulation model followed by credibility. Once a model was used, the ability of the model results to depict reality on the ground (credibility) became a critical and most important aspect for users. Nevertheless, even in cases where model performance was poor, users considered the scenario approach in evaluating their landscape a novelty. Potential model users' profession, prior exposure to a simulation model and interest in using models did not significantly influence respondents' ranking of model attributes (salience, credibility, legitimacy).

In the second study, to improve salience of a FALLOW model application, a livestock module was developed and tested for a peri-urban situation in the Upper Konto catchment, East Java, Indonesia. This study aimed to explore the impact of land use zoning strategies on farmers' welfare, fodder availability and landscape carbon stocks. Scenario analysis revealed that the current land zoning policy of establishing protected areas and allowing farmers' access to fodder extraction in part of the protected areas is the most promising strategy in balancing the trade-offs of production (farmers' welfare) and environment (represented by above-ground carbon sequestration). Compared to the scenario reflecting current policy, the 'open-access' scenario that allows opening land in protected areas, was simulated to increase farmers' welfare by 13% at the expense of losing 23% of landscape carbon. The extended FALLOW model with its livestock module proved an effective tool to examine the interactions between livestock, cropping systems, household decision and natural resources in data poor environments. The FALLOW model was able to simulate the land cover spatial pattern in the catchment (2002-2005) with a goodness of fit of 81% while the ability of predicting land change was 34.5% at a pixel resolution of 1 ha.

In the third study, to understand the effect of uncertainty in input parameters influencing model outcome, an uncertainty analysis of landscape C stock and emissions was carried out using several approaches that can cater for different situations of data availability (plot level carbon stocks and land cover maps). The analysis used data collected during a study assessing opportunities for REDD+ (Reducing Emission from Deforestation and Degradation) in a forest frontier region in Jambi, Indonesia, during 2000-2009. In a minimum data set situation (only single plot carbon estimates and a single land cover map available) the average landscape C stock estimates were 114.5 Mg.ha-1 and 81.0 Mg.ha-1 for 2000 and 2009, respectively. Based on an 'expected-carbon-deviance' curve, the confidence levels that the landscape C estimates were correct were 70% and 63% for 2000 and 2009, respectively. For other cases of enhanced data availability, Monte Carlo simulations were carried out to evaluate the propagation of land use classification errors and plot-level carbon stocks variation, jointly influencing landscape C stock and emission estimates. Results showed that excluding errors in land use classification resulted in biased estimates of landscape C stock and emissions. However, the bias over the whole area was estimated to be less than 7.5% (or 2.8 Mg.ha-1) with a coefficient variation of less than 0.2%.

In the last study, we combined spatial aggregation analysis on the error-perturbed C emission maps (resulting from Monte Carlo analysis in the third study) with local stakeholders' perspectives to develop an effective REDD+ scheme at the district level. The uncertainty analysis formed the basis for determining an appropriate scale for monitoring carbon emission estimates as performance measures of a REDD+ scheme. Changes in spatial resolution of C emission maps influenced the magnitude of potential area eligible for carbon payment and the uncertainty in carbon emission estimates. At 100 m resolution, 34.8% of the area would be eligible for REDD+ with an uncertainty of 82% , while at 5000 m resolution only 6.5% of the area would be eligible with a 1% error. At 1 km2 pixel size (1000 m resolution), the errors dropped below 5%, retaining most of the coarser spatial variation in the district. Hence, feasible measures for emission reduction in the district, derived from a participatory planning process, are compatible with the 1000 m spatial resolution of the C emission map.

Overall, the research elucidates the importance of involving model users in evaluating a simulation model, including scenario development and subsequent results analysis and interpretation. The study also indicates the importance of making efforts to improve model output accuracy to gain users' acceptance as users consider spatial accuracy is an important aspect of landscape-based models. In data-scarce situations, model users considered model 'robustness' in responding to new situations to be more important than 'precision'. Scenario analysis proved to be an effective tool to examine interactions in a complex landscape, including their consequences for trade-offs (e.g. farmer's welfare versus landscape carbon stocks) and synergies (e.g. fodder availability and farmers' welfare). Analysis of uncertainty of landscape C emission during land use changes can provide guidance in developing appropriate natural resource management interventions. Although model users may perceive model validation as a product, it is in fact a process.

Zusammenfassung

Nachhaltiges Ressourcenmanagement erfordert eine Balance der Austauschbeziehungen (trade-offs) zwischen Landproduktivität und ökologischer Integrität, sowie der Aufrechterhaltung eines gleichberechtigten Ressourcenzugangs. Szenarienanalyse basierend auf glaubwürdigen Modelsimulationen kann dabei die dynamischen und komplexen Interaktionen zwischen beteiligten Komponenten und deren Austauschbeziehungen unterstützen. Ungeachtet des Potentials von Simulationsmodellen als unterstützendes Mittel im Entscheidungsprozess des Landmanagements, stellen Akzeptanz und Verwendung von Entscheidungsträgern und Naturschutzmanagern eine noch immer große Herausforderung, vorallem im Kontext der Entwicklungsländer, dar. Diese Studie wurde erstellt, um relevante Aspekte im Bezug der Validierug von Simulationsmodellen aus Sichtweise der Nutzer-Perspektive sowie der Anwendung von Simulationsmodellen zur Bewertung von Austauschbeziehungen und deren Unsicherheitsabschätzungen zur Erstellung von Managementinterventionen zu ergründen.

Zunächst wurde in der vorliegenden Studie die Validität von Simulationsmodellen aus Sichtweise der Modelnutzer basiered auf zwei unterschiedlichen Aktivitäten des Naturschutzmanagements untersucht. In der ersten Aktivität wurde dabei die Wahrnehmung und Erwartungshaltung von potentiellen Modelnutzern (Wissenschaftlern, Dozenten, Naturschutzmanagern, Entscheidungsträgern und Kommunikatoren) aus Indonesien, Kenia, Philippinen und Vietnam mit Hilfe von Fragebögen betrachtet. Die zweite Aktivität beinhaltete eine partizipative Validierung von Modelergebnissen generiert mit dem räumlich-expliziten FALLOW Model in Aceh, Indonesien. Die Ergebnisse ergaben, dass potentielle Modelnutzer Relevanz als das wichtigste Attribut vor Glaubwürdigkeit als Grundlage zur Bewertung eines hypothetischen Models verwendeten. Nachdem ein Model bereits eingesetzt wurde, wurden Glaubwürdigkeit der Modelergebnisse im Vergleich der Realität vor-Ort als ein kritisches und zugleich als das wichtigste Nutzerkriterium angesehen. Auch in Fällen schlechter Simulatiosnergebnisse wurde von den Modelnutzern der Szenarienansatz zur Bewertung Ihrer Landschaft als Novum angesehen. Der potentielle Modeliererhintergrund, eine vorherige Erfahrung mit Simulatiosnmodellen und das Interesse zur Verwendung von Simulationsmodellen beeinflußten die genannte Rangfolge der Modelattibute (Relevanz, Glaubwürdigkeit, Legitimiät) der befragten Personen dabei nicht signifikant.

In der zweiten Studie wurde zur Verbesserung der Relevanz einer FALLOW Modelanwendung ein Modul zur Viehbestandsbewertung entwickelt und anhand eines peri-urbanen Fallbeispiels im Upper Konto Wassereinzugsgebiet, Ost-Java, Indonesien getestet. Die Studie untersuchte dabei den Einfluss von Landnutzungszonenstrategien auf die Wohlstandssituation von Bauern, Futtermittelverfügbarkeit, und der Kohlenstoffbestände auf Landschaftsebene. Eine Szenarienanalyse machte deutlich, dass die gegenwärtigen Richlinien der Landzonierung zur Etablierung von Schutzgebieten und die erlaubte Nutzung von Futtermittelextraktion aus Teilen der Schutzgebiete die bestgeeigneteste Strategie ist, um ein ausgeglichene Bilanz der Austauschbeziehungen von Produktion (Wohlstand der Bauern) und Umwelt (Kohlenstoffsequestrierung) zu erreichen. Im Vergleich zur bestehenden Richtlinie hatte das „Open-access“ Szenario, das eine vollständige Nutzung der Schutzgebiete erlauben würde, eine simulierte Wohlstandsteigerung der Bauern von 13% zur Folge, im Gegenzug dazu müße mit einem Verlust von 23% der Kohlenstoffbestände auf Landschaftsebene gerechnet werden. Das erweiterte FALLOW Model mit dem Tierbestandmodul bewies dass es als effektives Werkzeug, zur Bewertung der Interaktionen zwischen Viehbestand, Anbausystemen, Haushaltsentscheidungen und Naturschutzmanagement, in einer datenarmen Umgebung einsetzbar ist. Das FALLOW Model war in der Lage, die räumlichen Landbedeckungsgrade im Wassereinzugsgebiet mit einer Genauigkeit (Goodness-of-fit) von 81% und einer Ladnutzungsänderung von 34,5% auf der Pixelebene von 1 ha im Zeitraum von 2002-2005 vorherzusagen.

In der dritten Studie wurden die Effekte untersucht, die durch die Unsicherheit von Eingabeparamtern auf Seiten der Modelergebnisse entstehen können. Unter Zuhilfenahme einer Unsicherheitsanalyse von Kohlenstoffbeständen und Emissionen auf Landschaftsebene wurden dabei verschiedene Situationen der Datenverfügbarkeit (Schlagspezifische Kohlenstoffbestände, Landbedeckungskarten) untersucht. Die Analyse verwendete Daten die während einer Studie zur Abschätzung der REDD+ (Reducing Emission from Deforestation and Degradation) Verwendungsmöglichkeiten in einer Waldgrenzregion in Jambi, Indonesien während der Jahre 2000 bis 2009 erhoben wurden. In einer Minimumdatensatzsituation (nur schlagspezifische Datensätze, und einzelne Landbedeckungskarte verfügbar) wurde der Durchschnittskohlenstoffbestand auf Landschaftsebene auf 114,5 Mg ha^{-1} in 2000 und 81,0 Mg ha^{-1} für 2009 geschätzt. Basierend einer „zu erwartenden Kohlenstoff-Abweichungs“ Kurve, wurde der Vertrauensgrad der Kohlenstoffschätzung auf Landschaftsebene im Jahr 2000 mit 70% und in 2009 auf 63% beziffert. Für Fälle mit erweiterterter Datenverfügbarkeit wurde ein Monte-Carlo Simulation ausgeführt, um die Fehlerfortpflanzung bedingt durch Landnutzungsklassifizierung und Variation der schlagspezifischen Kohlenstoffbestände auf den Einfluss der Kohlenstoffbestände auf Landschaftsebene und deren Emission gemeinsam zu untersuchen. Die Ergebnisse zeigten, das ein Fehlerauschluss innerhalb der Landnutzungs-klassifizierungen, sowie die Variationen der schlagspezifischen Kohlenstoffbestände einen systematischen Fehler in den Abschätzungen des Kohlenstoffbestandes und der Emission auf Landschaftsebene erzeugten. Allerdings betrug der Fehler im Landschaftskontext weniger als 7,5% (oder 2,8 Mg ha^{-1}) mit einem Varianzskoeffiezient von weniger als 0,2%.

In der letzten Studie wurde eine räumliche Aggregierungsanalyse der fehlerbehafteten Kohlenstoddemissionskarten (basierend auf den Monte-Carlo Simulationen) dazu verwendet, eine räumlich-skalierte Abhängigkei, der Durschnitts- und Varianzabschätzungen der Kohlenstoffemissionen auf Schlagebene aufzeigen. Die Skalenabhängigkeit beinflusste das Ausmaß des potentiellen Gebietes für Kohlenstoffausgleichzahlungen und daher die Gerechtigkeit im Kontext des Ausgleichszahlungplans. Auf Schlagebenengröße von 1 ha (Einheitsgröße Kohlenstoffzahlungen), wären 34,8 % des Untersuchungsgebietes für einen REDD+ Plan geeignet, während bei einer 2500 ha großen Schlaggröße nur 6,5 % geeignet wären.

Ein Entwurf eines leistungsbasierten Zahlungsschemas sollte diese Skalenabhängigkeit berücksichtigen.

Im Gesamtüberblick konnte die vorgelegte Dissertation die Bedeutung der mit einzubeziehenden Modelnutzer zur Evaluierung eines Simulationsmodels, der Ergebnis- und Szenarienanalyse aufzeigen. Die Studie konnte ebenso deutlich machen, dass die Steigerung der Genauigkeit von Modelergebnissen im Bezug von Landschaftsmodelle vorallem der räumlichen Genauigkeit, zu einer höheren Modelnutzerakzeptanz führt. In datenarmen Situationen zeigte sich, dass Modelnutzern die „Robustheit" eines Models sich an eine neue Situation anzupasen wichtiger erschien als „Genauigkeit". Szenarienanalyse erwies sich als ein effektives Werkzeug, um Interaktionen in einer komplexen Landschaft, inklusive deren Konsequenzen (z.B. Wohlstand der Bauern im Vergleich von Kohlenstoffbeständen auf Landschaftsebene) und deren Synergien (z.B. Futtermittel-verfügbarkeit im Vergleich zum Wohlstand der Bauern) untersuchen zu können. Die Analyse von Unsicherheiten von Kohlenstoffemission auf Landschaftsebene im Verlauf eines Landnutzungswandels bietet eine Orientierung zur Entwicklung von angemessenen Interventionen im Naturschutzmanagement. Auch wenn Anwender Modelvalidität als ein Produkt wahrnehmen ist es in Wirklichkeit ein Prozess

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Simulation models compile knowledge into tools that are increasingly being used in problem solving and in decision making. Such models also are used in applied situations for natural resource management by integrating multi-dimensional social and biophysical indicators. However, despite the various approaches in promoting use of simulation models as tools to support decision making in natural resource management, acceptance and use by decision makers and natural resource managers are still a challenging issue. This thesis is the result of PhD research on validation of simulation models for natural resource management. It includes studies of users' perspectives on the validity of simulation models, model application to assess trade-offs and uncertainty assessment for designing management intervention.

Betha Lusiana *is a scientist with the World Agroforestry Centre (ICRAF) Indonesia program, where she leads the Ecological Modelling Unit. Her past work includes co-developing simulation models on tree, crop and soil interactions (WaNuLCAS), semi-agent-based land-use-change dynamics (FALLOW), semi-distributed river flow (GenRiver) and, more recently, a model for assessing farmers' livelihoods and food (ENLIFT). Her current work focuses on assessing the trade-offs between agricultural development, farmers' livelihoods and ecosystem services through the use of participatory, quantitative and/or model simulation approaches. In particular, she is interested in exploring and analyzing use of such assessment for natural resource management.*

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